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Rehabilitating mangrove ecosystem services: A case study on the relative benefits of abandoned pond reversion from Panay Island, Philippines

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ABSTRACT

Mangroves provide vital climate change mitigation and adaptation (CCMA) ecosystem services (ES), yet have suffered extensive tropics-wide declines. To mitigate losses, rehabilitation is high on the conservation agenda. However, the relative functionality and ES delivery of rehabilitated mangroves in different intertidal locations is rarely assessed. In a case study from Panay Island, Philippines, using field- and satellite-derived methods, we assess carbon stocks and coastal protection potential of rehabilitated low-intertidal seafront and mid- to upper-intertidal abandoned (leased) fishpond areas, against reference natural mangroves. Due to large sizes and appropriate site conditions, targeted abandoned fishpond reversion to former mangrove was found to be favourable for enhancing CCMA in the coastal zone. In a municipality-specific case study, 96.7% of abandoned fishponds with high potential for effective greenbelt rehabilitation had favourable tenure status for reversion. These findings have implications for coastal zone management in Asia in the face of climate change.

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1. Introduction

Environmental management is placing increasing emphasis on the services provided by the world's ecosystems (Cardinale et al., 2012). Mangrove forests deliver numerous important ecosystem services (ES) to humans, valued at \$194,000 ha⁻¹ yr⁻¹ (Costanza et al., 2014): food and fuel, nursery habitat, recreation (Barbier et al., 2008, 2011). Mangroves are of particular significance in the context of climate change (Duarte et al., 2013), affording among the largest per hectare global carbon stores and coastal protection from regular waves and frequent tropical storms (Dahdouh-Guebas et al., 2005; Donato et al., 2011). Growing global policy emphasis on both emissions reduction and climate impact mitigation in vulnerable countries (UNFCCC, 2015) places ever higher significance on the climate change mitigation and adaptation (CCMA) properties of mangroves. High susceptibility to anthropogenic activities and climate change impacts (Primavera, 2005; Duke et al., 2007; Lovelock et al., 2015) has, however, led to mangrove areal declines of 30–50% globally (Field et al., 1998; Valiela et al., 2001), with continued losses of 0.16–0.39% per annum (Hamilton and

Casey, 2016; Richards and Friess, 2016). 16% of mangrove species are now threatened with global extinction (Polidoro et al., 2010). Extensive loss has left degraded and highly fragmented mangroves in many parts of their global distribution (Giri et al., 2011; Hamilton and Casey, 2016) that may have limited potential to deliver CCMA services into the future (Koch et al., 2009; Barbier et al., 2011; Lee et al., 2014).

To combat mangrove losses, and to enhance CCMA efforts in the tropics, rehabilitation is an essential management tool (Ellison, 2000; Kairo et al., 2001; Lewis, 2005; Primavera and Esteban, 2008; Primavera et al., 2012a). Rehabilitated mangrove blue carbon-based Payments for Ecosystem Services (PES) projects are emerging (Wylie et al., 2016), and governments are increasingly recognising the significance of mangrove coastal protection (Marois and Mitsch, 2015), with national coastal greenbelt replanting programmes now widespread following recent natural disasters (Dahdouh-Guebas et al., 2005; Primavera et al., 2014). Recent studies on potential blue carbon PES schemes have concluded that projects would benefit from inclusion of “bundled services” to offset low voluntary carbon market prices (Locatelli et al., 2014; Thompson et al., 2014), with particular reference to coastal protection (Kairo et al., 2009; Locatelli et al., 2014). However, mangrove rehabilitation efforts have historically seen low successes (e.g. Primavera and Esteban, 2008; Dale et al., 2014; Bayraktarov et al., 2015; but see Arnaud-Haond et al., 2009; Goessens et al., 2014), and where established, longer-term monitoring

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of functionality has been minimal (Bosire et al., 2008). Where this has been monitored, the structure and specific functionality of rehabilitated mangroves can be comparable to adjacent natural stands (Kairo et al., 2001; Bosire et al., 2008; Ren et al., 2010; Salmo et al., 2013; Nam et al., 2016); however, their relative potential for high multiple ES delivery is mostly unknown (but see Rönnbäck et al., 2007 and Nam et al., 2016). We thus currently lack quantitative information on the combined CCMA potential of current mangrove rehabilitation efforts.

There are two major potential sources of variation in the ability of rehabilitated mangroves to deliver high multiple CCMA ES. First, ignorance of or noncompliance to scientific guidelines has driven many rehabilitation efforts to take place in low-intertidal seafront areas where sub-optimal hydrological conditions limit survival and growth of replanted mangroves (Iftekhhar, 2008; Primavera and Esteban, 2008; Primavera et al., 2012a, 2012b, 2014). Rehabilitation in such areas may result in low relative mangrove biomass and density, and associated carbon stocks and coastal protection potential, particularly where rehabilitation failure has historically been high. Second, site areal configuration may heavily impact the potential ES delivery of rehabilitated mangroves. Rehabilitated mangrove carbon stocks may be expected to increase linearly with site area, while coastal protection rapidly increases with mangrove greenbelt width (Koch et al., 2009). Indeed, low-intertidal rehabilitated mangroves exist primarily in monospecific narrow-fringing stands (Ellison, 2000; Iftekhhar, 2008; Primavera et al., 2012a, 2012b), with potentially severely limited ability to deliver effective coastal protection (Ewel et al., 1998; Barbier et al., 2008, 2011; Koch et al., 2009). Larger rehabilitation sites in the mid- to upper-intertidal zone may thus be expected to deliver much higher multiple CCMA benefits than narrow, low-intertidal rehabilitated mangroves. However, the spatial configuration and area of suitable land for mangrove rehabilitation is often constrained by land tenure conflicts and complexities in the coastal zone (e.g. agri- and aquaculture); often the major driver for prioritizing low-intertidal zone rehabilitation (Iftekhhar, 2008; Primavera and Esteban, 2008; Primavera et al., 2012b, 2014). CCMA arguments for rehabilitation actions may be key in future decision-making and spatial planning. To guide effective coastal zone management in the face of climate change, there is thus a need to identify and prioritise rehabilitation locations in which high CCMA gains may co-occur with minimal tenure issues (Locatelli et al., 2014; Primavera et al., 2014; Thompson et al., 2014).

This study examines the CCMA potential of mangrove rehabilitation in abandoned aquaculture ponds relative to low-intertidal, seafront areas across Panay Island, Philippines. We first quantify the relative vegetation and sediment carbon stocks, and coastal protection potential of rehabilitated mangrove areas (mid- to upper-intertidal abandoned fishpond and low-intertidal seafront areas), against mature natural reference mangrove stands, to explore the ES potential of these different rehabilitation strategies. We then conduct a municipality-specific case study to model the potential CCMA benefits of targeted abandoned fishpond reversion, with specific reference to current coastal greenbelt rehabilitation efforts. We conclude by examining the feasibility of prioritising abandoned fishpond reversion for CCMA goals under current fishpond tenure status across the case study.

2. Materials & methods

2.1. Study areas: Panay Island, Philippines

The Philippines is among the most typhoon-ravaged countries in the world (Peduzzi et al., 2012; UNU, 2014). High typhoon-exposure of coastal areas, and the infrastructural and institutional vulnerability to typhoon events (UNU, 2014), has been recently evidenced by devastating impacts suffered during super-typhoon Haiyan (Soria et al., 2015). The Philippines has experienced substantial mangrove loss: approximately 50% of the former 500,000 ha (Spalding et al., 2010) disappeared over the last century, due primarily to shallow brackish-water fishpond

aquaculture development in former estuarine, basin and riverine mangroves (Primavera, 2005). Some of the highest fishpond densities occur in the West Visayas Region; e.g. on Panay Island (Primavera and Esteban, 2008). Development is largely unregulated, and despite laws mandating 50–100 m of mangrove greenbelt (Primavera et al., 2012b), fishponds are often built to the shoreline. Abandonment is high (estimates in the thousands of hectares; see Samson and Rollon, 2011; Primavera et al., 2012b), due primarily to bank breaches in sea-facing fishponds over low productivity (Primavera et al., 2014). Fishponds are tenured by among the wealthiest in society, and operated by the poorest. Reversion of abandoned fishponds to former mangroves for greenbelt resurrection could thus benefit coastal community livelihoods through associated fisheries enhancement (Walton et al., 2006).

Philippines' public mangrove land is released by the Department of Environment and Natural Resources (DENR) for aquaculture under multiple tenure arrangements: from titled ownership, to temporary leaseholds under Fishpond Lease Agreements (FLAs) granted under the jurisdiction of the Bureau of Fisheries and Aquatic Resources of the Department of Agriculture (DA-BFAR). Under Philippine law, failure to adhere to FLA terms should preclude FLA cancellation by DA-BFAR, and reversion of jurisdiction to the Forest Management Bureau of DENR for subsequent mangrove rehabilitation. This includes Abandoned (no operational activity, subleasing, or neglect of payments), Underutilised (no commercial production within three years), and Undeveloped (pond infrastructure absent) (AUU) FLA fishponds (see Primavera et al., 2014). Herein, the term 'abandoned fishpond' refers to all AUU fishponds. However, non-coordination between government departments (DA-BFAR and DENR), low institutional capacity, exclusion of local government units (LGUs) and coastal communities from decision-making, and a lack of political will means FLA monitoring is minimal, and cancellation and reversion rarely occurs: large areas of former mangrove lie fallow. Furthermore, cancelled abandoned FLAs are often absorbed and re-tenured under new FLA leases or operated illegally (Primavera et al., 2014).

Due to the challenges of abandoned fishpond reversion, national greenbelt rehabilitation programmes continue to focus on low- and sub-intertidal planting seaward of coastal infrastructure and fishponds ('seafront rehabilitation'). High mortality in plantations of inappropriate species wastes public and international funds, while threatening other intertidal systems (seagrasses, mudflats; Primavera and Esteban, 2008; Samson and Rollon, 2008, 2011). Surviving seafront rehabilitated mangroves are often small areas growing at the limits of their physiological tolerance ranges (Tomlinson, 1986). In contrast, some Non-Governmental Organisation-led projects, in partnership with specific LGUs, have begun to target rehabilitation of abandoned fishponds in the mid-upper intertidal zone where more natural hydrological conditions largely remain (Primavera et al., 2012b, 2014).

This study investigated the relative CCMA ES delivery by rehabilitated low-intertidal seafront and abandoned fishpond areas across Panay Island, with reference to adjacent natural stands. Six mangrove stands from four sites in Iloilo and Aklan Provinces were used (Fig. 1):

1. Bakhawan ecopark, Buswang, Kalibo, a remnant area of a former deltaic mangrove at the mouth of Aklan River (Cadaweng and Aguirre, 2005; Walton et al., 2006). Following over-exploitation of mangrove timber, large portions of the seaward area have been replanted with *Rhizophora* spp. since the early 1990s. A wide band of mature natural *Avicennia marina* and *Sonneratia* spp.-dominated mangrove remains behind the rehabilitated areas. This study focused on (1) a seafront area replanted in 2006 with *Rhizophora apiculata*, and subsequently naturally recolonised by *A. marina*, *Nypa fruticans* and *Sonneratia alba* individuals ("Bakhawan rehab"); and (2) the inland natural mangrove area ("Bakhawan natural").
2. Ermita, Dumangas. A remnant now-fringing area of a former deltaic mangrove cleared inland for fishpond aquaculture, bordered in the landward direction by active fishponds and a coastal road. The site

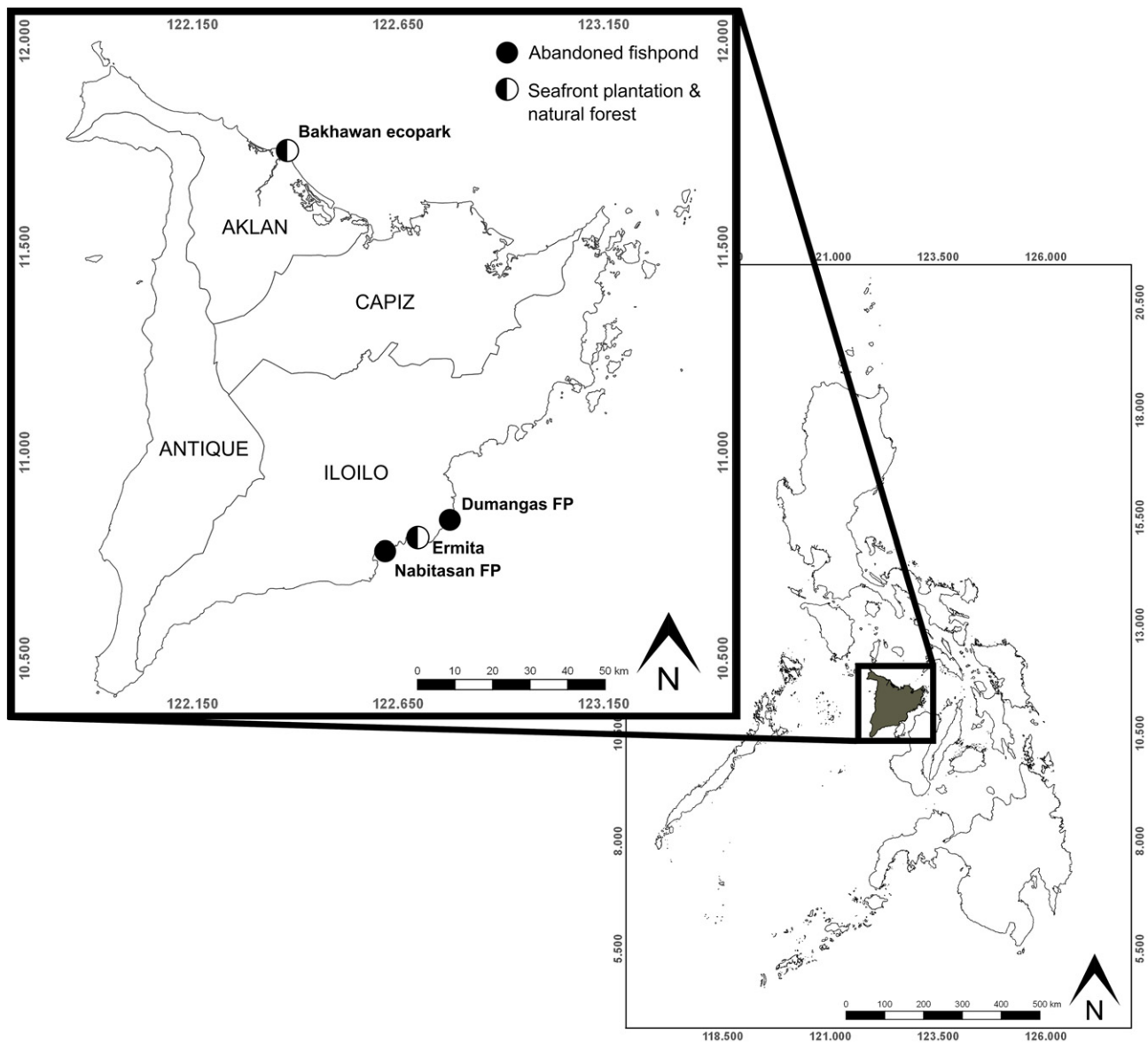


Fig. 1. Location of rehabilitated and natural mangrove sites considered in this study, and location of Panay Island in West Visayas (Region VI), Philippines. Filled circles donate rehabilitated abandoned fishpond sites, while half-filled circles donate replanted seafront sites and adjacent natural areas. Details of all sites are provided in [Table 1](#).

contains a narrow band of remaining natural mangrove (*A. marina* dominant; “Ermita natural”). Seaward of the natural mangrove is a low-intertidal stand planted in 2007 (“Ermita rehab”). The area was originally planted with *S. alba*, *A. marina* and *Rhizophora* spp. seedlings; however, only *S. alba* survived algal (*A. marina*) and barnacle (*Rhizophora* spp.) infestation.

3. A sea-facing abandoned fishpond (FP) in Nabitasan, Leganes (“Nabitasan FP”), which was reverted and replanted under a partnership between the Zoological Society of London (ZSL) and Leganes LGU in 2009 (Primavera et al., 2012a). *A. marina* is dominant, with *Rhizophora* spp., *S. alba* and *A. rumphiana* also present. Prior to assisted rehabilitation, very low mangrove cover and seaward bank destruction drove erosion of former fishpond sediments at the seaward edge. The area drains directly into the sea and is flushed twice-daily by the tide.
4. A sea-facing abandoned fishpond in Dumangas municipality (“Dumangas FP”) which was abandoned following a seaward bank breach in 2005–2006. The site has subsequently been naturally recolonised. Vegetation is dense and dominated by *A. marina*, with *S. alba* and *Rhizophora* spp. also present. The area drains directly

into the sea; however, much of the seaward bank remains. The site is not presently flushed completely with the tide; in areas organic matter is trapped in waterlogged sediments. The area is currently leased under an FLA, and the leaseholder is in breach of terms following abandonment (Primavera et al., 2014).

2.2. Field data collection

Field data collection was conducted from 2014–2015. Temporary circular plots (radius = 7 m) were established via stratified sampling with variation in distance from the shoreline (Kauffman and Donato, 2012) within a 15 × 15 m grid ($N = 8$ per site). For each plot, tree species, diameter at breast height (DBH; at 1.37 m height, or above the highest prop root for *Rhizophora* spp.; Kauffman and Donato, 2012), height, and maximum canopy width (m; Kauffman and Donato, 2012) of all trees (>1.37 m height) were measured. Small trees were defined as those with ≤1.5 cm DBH, in order to avoid underestimation of biomass in areas with a high abundance of small trees (e.g. rehabilitated sites; Kauffman and Donato, 2012), and were measured within a 3 m radius sub-plot (from the plot centre). All larger trees (>1.5 cm DBH)

were measured throughout the plot. Low, shrubby, heterogeneous canopies in younger rehabilitated sites restricted traditional methods of rapid canopy closure estimation (e.g. spherical densitometer or vertical position digital photography; Korhonen et al., 2006). To avoid error from ocular estimates (Korhonen et al., 2006), we estimated average plot-level canopy closure (%) at all sites from horizontal position digital photographs ($N = 4$ per plot; at the plot centre facing toward the plot 'corners'). Average plot-level canopy closure was estimated as the average percentage of non-sky, -water or -sediment image pixels, classified for all plot photographs from the ratio of green to red light. These estimates were then averaged across all plots for each site ($N = 8$) to estimate site-specific average canopy closure (%). This approach enabled us to index both large canopy gaps (where present) and canopy penetration across plots.

Sediment cores were taken in a triangular configuration within the 3 m sub-plot ($N = 3$ per plot) with an Eijkelpamp gouge auger (30 mm diameter, 50 cm sampling length). Within each core, six 5 cm samples were taken at specified depths and aggregated to represent the 0–50 cm ($N = 3$ depth subsamples \times 3 cores = 9) and 50–150 cm ($N = 3$ depth subsamples \times 3 cores = 9) sediment horizons (5–15 cm, 15–30 cm and 30–50 cm, and 50–100 cm, 100–150 cm and 150 cm respectively; Kauffman and Donato, 2012). Sediments were sampled to 150 cm depth due to laboratory constraints (rather than the entire sediment profile; Donato et al., 2011; Kauffman and Donato, 2012). Average sediment profile depth was measured ($N = 3$ per plot) in a triangular configuration within 3.5 m of the plot centre, avoiding areas close to large trees (aerial roots), by inserting an iron rod (diameter = 1.5 cm) vertically into the sediment by hand until it could no longer be pushed.

2.3. Ecosystem services quantification

2.3.1. Vegetation carbon stocks

Vegetation biomass was calculated via allometric equations. For single-stemmed trees, biomass was estimated via the general mangrove allometric equations derived by Komiyama et al. (2005) (Eqs. (1) & (2)), in order to ensure continuity in biomass estimation across plots and sites. Individual tree above- (B_{AG} ; kg) and belowground biomass (B_{TB} ; kg) were estimated by:

$$B_{AG} = 0.251 \times p \times D^{2.46} \quad (1)$$

$$B_{TB} = 0.199 \times p^{0.899} \times D^{2.22} \quad (2)$$

where p is species-specific wood density (g cm^{-3}) (Komiyama et al., 2005). Two individual trees in the natural area at Ermita, Dumangas were beyond the DBH limit of these equations and their biomass was overestimated (Komiyama et al., 2005; Kauffman and Donato, 2012). In the absence of guidelines to accommodate this situation (see Kauffman and Donato, 2012; Thompson et al., 2014), these individuals were here treated as multiple trees. Their DBH was split to create two separate 'individuals' (e.g. DBH1 = 49 cm, DBH2 = 12 cm) and their biomass calculated. While this method likely still overestimates biomass, the resulting overestimates were substantially smaller than those caused by calculation via actual DBH. Estimates of p were taken from the Global Wood Density Database (Chave et al., 2009; Zanne et al., 2009). Species-specific averages of p were taken across all Southeast Asian estimates (Supplementary Information), in the absence of Philippines-specific estimates for all species in this study.

For multi-stemmed trees, aboveground biomass was estimated via the method of Fu and Wu (2011). B_{AG} (kg) was calculated as:

$$B_{AG} = CD^2 \times H \quad (3)$$

where CD is the maximum canopy diameter (m) and H is tree height (m). This method has been found to be applicable to species with

variable growth form (e.g. *A. marina*; Fu and Wu, 2011). Belowground biomass of multi-stemmed trees was calculated via the general equation of Komiyama et al. (2005) (Eq. (2)), using an artificial DBH calculated from the DBH of the largest stem and the number of stems per individual.

All sites were highly heterogeneous in biomass distribution; remote sensing methods were thus employed to index spatial variability. We employed a technique similar to that derived by Simard et al. (2006) and Fatoyinbo et al. (2008), using Shuttle Radar Topography Mission (SRTM) near-global Digital Elevation Model (DEM) data (30 m resolution; Rodriguez et al., 2006). All analyses were conducted in R v. 3.2.1 (R Development Core Team, 2015). SRTM pixel values (SRTM-derived height; m) were first extracted for each temporary field plot location. C- and X-band SRTM radar scattering is influenced by vegetation density, biomass and canopy closure, and error exists between observed SRTM DEM height and true canopy height (Simard et al., 2006; Fatoyinbo et al., 2008). We thus established a linear regression to predict field plot mean height (m) from corresponding SRTM DEM pixel height (m). Plot-level mean height data was used to derive (1) a mean height-aboveground biomass relationship, and (2) a mean height-belowground biomass (plot level Mg [tonnes], extrapolated to Mg 900 m²) relationship with our allometrically-estimated biomass values (Komiyama et al., 2005), using linear regression. These equations were then applied across all SRTM DEM pixels at each site, and summed (biomass value \times proportion of pixel within the site boundary) to estimate site-level biomass. Site-level above- and belowground biomass estimates were then multiplied by carbon concentration values of 0.464 (Donato et al., 2011) and 0.39 (Kauffman and Donato, 2012) respectively to obtain vegetation carbon stock estimates. Mean per hectare above- and belowground vegetation carbon stocks were calculated across all pixels with $\geq 50\%$ of their area within the site boundary.

2.3.2. Sediment carbon stocks

Sediment samples were analysed at the Bureau of Soils and Water Management, Cebu. Bulk density (BD ; g cm^{-3}) was determined against the sample volume as:

$$BD = \frac{DM}{SV} \quad (4)$$

where DM is oven-dried mass (g) and SV is the sample volume (cm^3). Air-dried subsamples were homogenised and sieved (2 mm) (Donato et al., 2011; Thompson et al., 2014). Organic carbon content (OC; %) was determined gravimetrically for each subsample (0.25 g) via the dry combustion method (Schumacher, 2002). Sediment horizon carbon stock was calculated by summing the carbon stock for each of the two sampled depth intervals (Kauffman and Donato, 2012). The 50–150 cm depth samples were used to extrapolate sediment carbon to the entire sediment profile depth for each plot, as OC does not vary greatly below 30 cm (Thompson et al., 2014). Sediment carbon (SC; Mg ha^{-1}) was calculated for each depth interval as:

$$SC_i = BD \times D \times OC \quad (5)$$

where D is the sediment depth interval (cm; e.g. here 50 cm and maximum depth–50 cm for the 0–50 cm and 50–sediment profile depth intervals respectively) and OC is the sample OC (%; Kauffman and Donato, 2012). Linear regressions (lms) were applied to model differences in vegetation and sediment carbon estimates (Mg ha^{-1}) across sites (with Site Name as a predictor variable). Site-level sediment carbon stock estimates were calculated from the plot-level estimates from each site (Mg 153.94 m^2). The area of each site was divided by plot area (153.94 m^2) to create 'sub-site' areas, and an estimate of sediment carbon stock for each 'sub-site' was drawn from a normal distribution created from the mean and standard deviation of plot-level estimates for each depth interval. These 'sub-site' estimates were then summed to create total site

sediment carbon storage estimates for each depth interval at each site (mean over 1,000,000 runs).

2.3.3. Coastal protection potential

The model of wave attenuation derived by Bao (2011) was employed to assess the coastal protection potential of each site. This model estimates the width of mangrove greenbelt required to attenuate a regular wave of three metres height to a 'safe height' of 0.3 m behind the forest (Bao, 2011). A Forest Structure Index (FSI), calculated from field-derived mangrove structural parameters, is calculated and characterises different forests into protection level categories (I–V). FSI was calculated from mean plot-level data for each site as:

$$FSI = -0.048 + 0.0016 H + 0.00178 \log(SD) + 0.0077 \log(CC) \quad (6)$$

where H is mean mangrove height (m), SD is mean stem density ($N \text{ ha}^{-1}$), and CC is mean canopy closure (%). The required greenbelt width (B_W ; m) required at each site was calculated as:

$$B_W = \frac{\log(H_{safe}) - \log(0.9899 H_0 + 0.3526)}{0.048 - 0.0016 H - 0.00178 \log(SD) - \log(CC)} \quad (7)$$

where H_{safe} is the 'safe wave height' (30 cm), and H_0 is initial wave height (300 cm). Required mangrove greenbelt width (B_W) was compared to the median actual landward width (B_{Actual} ; m) to determine the current coastal protection potential of each site. Landward and seaward boundaries of the polygon outline of each site were extracted in QGIS (QGIS Development Team, 2015) and the median distance between seaward and landward boundary vertices ($N \approx 2000$) were calculated for each site using the function 'gDistance' (Bivand et al., 2015).

2.4. Case study: Dumangas municipality

Fishpond density in Dumangas, Iloilo is among the highest in West Visayas (4282.7 ha; Primavera et al., 2014). We mapped the distribution of abandoned fishponds (mangrove vegetation present) from high-resolution satellite imagery (Google Earth, 2015). SRTM-derived vegetation biomass estimates (Mg ha^{-1}) from the two abandoned fishpond sites (Nabitanan and Dumangas) were used to project potential vegetation carbon stock accumulation across Dumangas' abandoned fishponds over a period of 6.5 years (mean abandoned fishpond site age) via random sampling from the observed distribution of vegetation carbon stock (Mg ha^{-1}) estimates (mean over 1,000,000 runs). This method was employed to produce a conservative estimate, as some abandoned fishponds are currently in relatively advanced stages of mangrove recolonisation (Primavera et al., 2014). Potential sediment carbon accretion was estimated using a conservative accretion rate of 25 mm y^{-1} for the Indo-Pacific Region (Lovell et al., 2015; 1.63 cm surface accretion over 6.5 years) and random sampling from the observed distribution of sediment carbon stock (Mg ha^{-1}) for the 0–50 cm depth interval (mean over 1,000,000 runs). We compared the mapped abandoned fishponds against a recent survey of tenure status in Dumangas (e.g. breached FLA leases; Primavera et al., 2014).

We then subset mapped sea-facing abandoned fishponds and calculated median distance between landward and seaward boundary vertices as above. Vertices with median distance to landward boundaries greater than estimated required greenbelt widths to attenuate three metre initial waves (B_W ; estimated for abandoned fishpond sites as above; Bao, 2011) were retained to calculate the length of Dumangas' coastline with potential for effective coastal protection from regular wind waves following abandoned fishpond reversion. We then compared this against tenure status for identified sea-facing abandoned fishponds (Primavera et al., 2014).

3. Results

3.1. Vegetation structure

Mean vegetation basal area, DBH, height and biomass were greatest in the natural areas, followed by the rehabilitated seafront areas, and abandoned fishponds (Table 1). Stem density was highly variable across natural and rehabilitated sites, being greatest at the rehabilitated seafront area at Bakhawan ($11,839.18 \text{ stems ha}^{-1}$) and lowest at the rehabilitated seafront area at Ermita (1916.36 ha^{-1}) (Table 1).

3.2. SRTM vegetation biomass modelling

A positive linear relationship was found between mean plot-level vegetation height (m; log-transformed) and SRTM-derived height (m; square root-transformed): $R^2 = 0.43$; root-mean-square error (RMSE) = 0.41 (Fig. 2). Allometrically-estimated (Section 2.3) plot-level above- (linear; $R^2 = 0.73$; RMSE = 0.50; Fig. 3a) and belowground biomass (quadratic; $R^2 = 0.89$; RMSE = 0.44; Fig. 3b), both extrapolated to $\text{Mg } 900 \text{ m}^2$, showed a positive relationship with mean plot-level vegetation height (m; log-transformed). We applied these equations across SRTM tiles for Panay to predict above- (Fig. 4) and belowground biomass across sites.

3.3. Carbon stocks

Mean above- and belowground vegetation carbon stocks (Fig. 5) were greatest at the two natural areas (Ermita: 50.41 Mg ha^{-1} ; Bakhawan: 46.95 Mg ha^{-1}), followed by the rehabilitated seafront areas (Ermita: 41.01 Mg ha^{-1} ; Bakhawan: 30.42 Mg ha^{-1}), and were lowest in the abandoned fishponds (Dumangas: 25.68 Mg ha^{-1} ; Nabitanan: 5.17 Mg ha^{-1}). Both rehabilitated seafront sites had significantly lower SRTM-derived vegetation carbon stocks (Mg ha^{-1}) than adjacent natural areas ($\log(\text{vegetation carbon})$; intercept = 4.46 ± 0.10 (1 s.e.); Bakhawan rehab: -1.08 ± 0.18 (1 s.e.), $p < 0.001$; Ermita rehab: -1.35 ± 0.18 (1 s.e.), $p < 0.001$). Both abandoned fishponds had significantly lower SRTM-derived vegetation carbon stocks (Mg ha^{-1}) than rehabilitated seafront areas ($\log(\text{vegetation carbon})$; intercept = 3.25 ± 0.13 (1 s.e.); Dumangas: -0.58 ± 0.23 (1 s.e.), $p = 0.02$; Nabitanan: -1.42 ± 0.23 (1 s.e.), $p < 0.001$).

Sediment OC estimates for the 0–50 cm depth interval ranged from 0.71% (Ermita Rehab) to 4.88% (Dumangas FP), and for the 50–150 cm depth interval ranged from 0.82% (Bakhawan Rehab) to 4.59% (Dumangas FP) (Table 2). Bulk density ranged from $0.37\text{--}0.38 \text{ g cm}^{-3}$ at the Dumangas abandoned fishpond to $0.85\text{--}0.91 \text{ g cm}^{-3}$ at the Ermita seafront rehabilitation site (Table 2). Mean total sediment carbon stocks were similarly variable (Fig. 5), being lowest at the two seafront rehabilitated areas (Bakhawan: 120.85 ; Ermita: $131.14 \text{ Mg ha}^{-1}$), followed by the natural area at Bakhawan ($204.47 \text{ Mg ha}^{-1}$), the Nabitanan abandoned fishpond ($207.04 \text{ Mg ha}^{-1}$), and the natural area at Ermita ($324.85 \text{ Mg ha}^{-1}$). The largest mean total sediment carbon stock occurred at the Dumangas abandoned fishpond ($684.67 \text{ Mg ha}^{-1}$) (Fig. 5). Both rehabilitated seafront sites had significantly lower plot-level total sediment carbon stocks (Mg ha^{-1}) than adjacent natural areas ($\log(\text{total sediment carbon})$; intercept = 5.52 ± 0.10 (1 s.e.); Bakhawan rehab: -0.86 ± 0.17 (1 s.e.), $p < 0.001$; Ermita rehab: -0.67 ± 0.17 (1 s.e.), $p < 0.001$). Plot-level total sediment carbon stocks at the Nabitanan abandoned fishpond were not significantly different than at the two natural areas ($\log(\text{total sediment carbon})$; intercept = 5.52 ± 0.09 (1 s.e.); Nabitanan: -0.23 ± 0.16 (1 s.e.), $p = 0.16$), while those at the Dumangas abandoned fishpond were significantly larger (Dumangas: 0.92 ± 0.16 (1 s.e.), $p < 0.001$).

Table 1

Mean field-derived structural parameters across plots ($N = 8$) in all surveyed sites. Biomass refers to mean total (above- and belowground) biomass.

Site	Basal area ($\text{m}^2 \text{ha}^{-1}$)	Stem density (N ha^{-1})	Canopy closure (%)	DBH (cm)	Mean height (m)	Biomass (Mg ha^{-1})
Bakhawan Natural	27.68	6496.12	88.10	5.76	6.16	177.88
Bakhawan Rehab	16.52	11,839.18	84.45	2.71	3.78	79.27
Ermita Natural	33.16	2151.84	84.85	18.0	6.67	241.62
Ermita Rehab	10.75	1916.36	87.19	6.86	4.63	52.88
Nabitasan FP	3.38	2379.21	63.60	3.27	1.99	15.25
Dumangas FP	8.17	6950.85	84.00	3.98	2.89	37.96

3.4. Site-level carbon storage

The largest site-level total carbon stock occurred at the Dumangas abandoned fishpond (vegetation carbon = 1176.12 Mg; sediment carbon = 31,482.69 Mg), followed by the natural area at Bakhawan (vegetation carbon = 1813.37 Mg; sediment carbon = 7888.11 Mg), the seafront rehabilitated area at Bakhawan (vegetation carbon = 599.06 Mg; sediment carbon = 2358.92 Mg), and the Nabitasan abandoned fishpond (vegetation carbon = 46.82 Mg; sediment carbon = 1870.91 Mg). Owing to small areal coverage (2.34 ha and 0.50 ha respectively), total site carbon stock was low at both the natural (vegetation carbon = 114.03 Mg; sediment carbon = 760.10 Mg) and seafront rehabilitated areas at Ermita (vegetation carbon = 7.19 Mg; sediment carbon = 64.60 Mg) (Table 3).

3.5. Site-level coastal protection

Variable vegetation structure (Table 1) produced variable potential coastal protection and required greenbelt width (B_W) across sites (Table 4). Both natural and rehabilitated areas at the Bakhawan site provide relatively strong protection (protection categories III and II respectively) and require narrow greenbelt widths ($B_W = 201$ and 270 m respectively). This is within the median actual greenbelt width ($B_{Actual} = 842$ m) (Table 4). At the Ermita site, despite high FSI and protection category (III and II respectively), the required B_W for adequate coastal protection potential (229–331 m) is not achieved, due to a narrow fringing area available for rehabilitation (median $B_{Actual} = 81$ m; Table 4). Both abandoned fishpond sites had high median actual greenbelt width (B_{Actual} ; Nabitatan = 268 m and Dumangas = 827 m). High stem density and vertical growth (Table 1) translated to a higher

protection category for the Dumangas abandoned fishpond site (II) and adequate current coastal protection potential ($B_W = 370$ m). However, low vegetation density and height (Table 1) at the Nabitatan

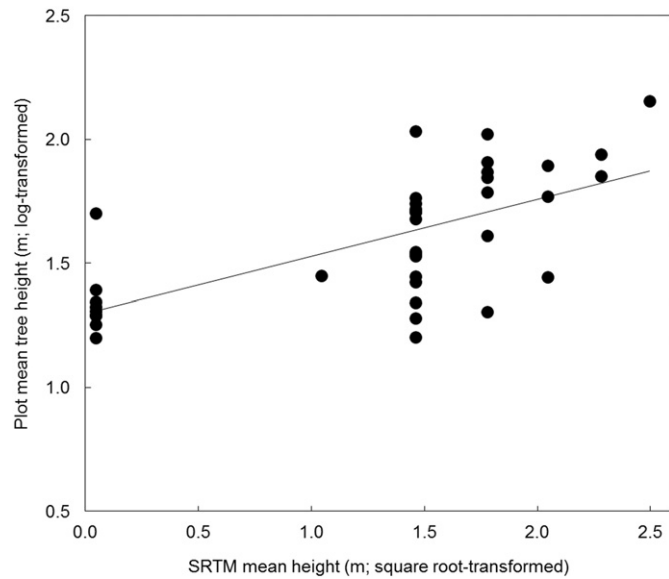


Fig. 2. Regression of observed plot-level mean mangrove height (m; log-transformed) against SRTM DEM-derived elevation (m; square root-transformed). $\log(\text{mean height}) = 1.31 + 0.23\sqrt{\text{SRTM elevation}}$; $R^2 = 0.43$; RMSE = 0.41.

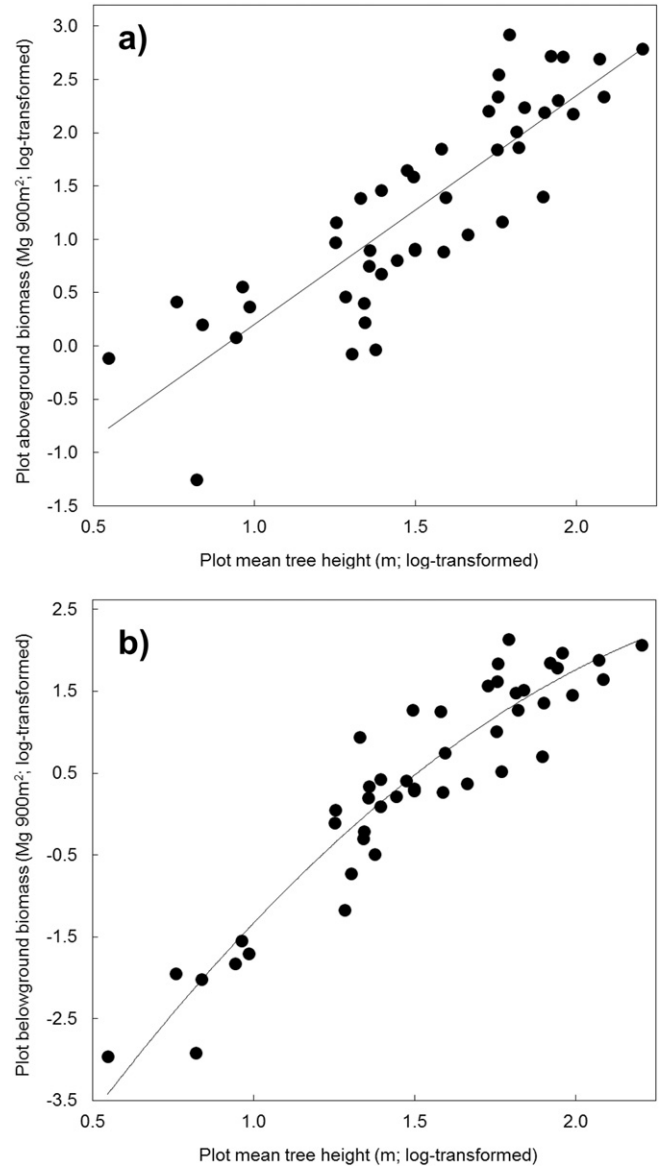


Fig. 3. Best-fitting regressions (highest variance explained; R^2) of (a) above- and (b) belowground biomass (allometric estimation from field DBH data; Komiyama et al., 2005) against observed mean mangrove height (m; log-transformed). Allometrically-estimated above- and belowground biomass has been scaled from plot- to pixel-level estimates (900m^2 ; log-transformed). The relationship between allometrically-estimated aboveground biomass and observed mean mangrove height was linear in form $\log(\text{aboveground biomass}) = 2.14\log(\text{mean height}) - 1.84$; $R^2 = 0.73$; RMSE = 0.50), while the relationship for allometrically-estimated below-ground biomass was a quadratic relationship $\log(\text{belowground biomass}) = -1.05\log(\text{mean height})^2 + 6.13\log(\text{mean height}) - 6.25$; $R^2 = 0.89$; RMSE = 0.44).

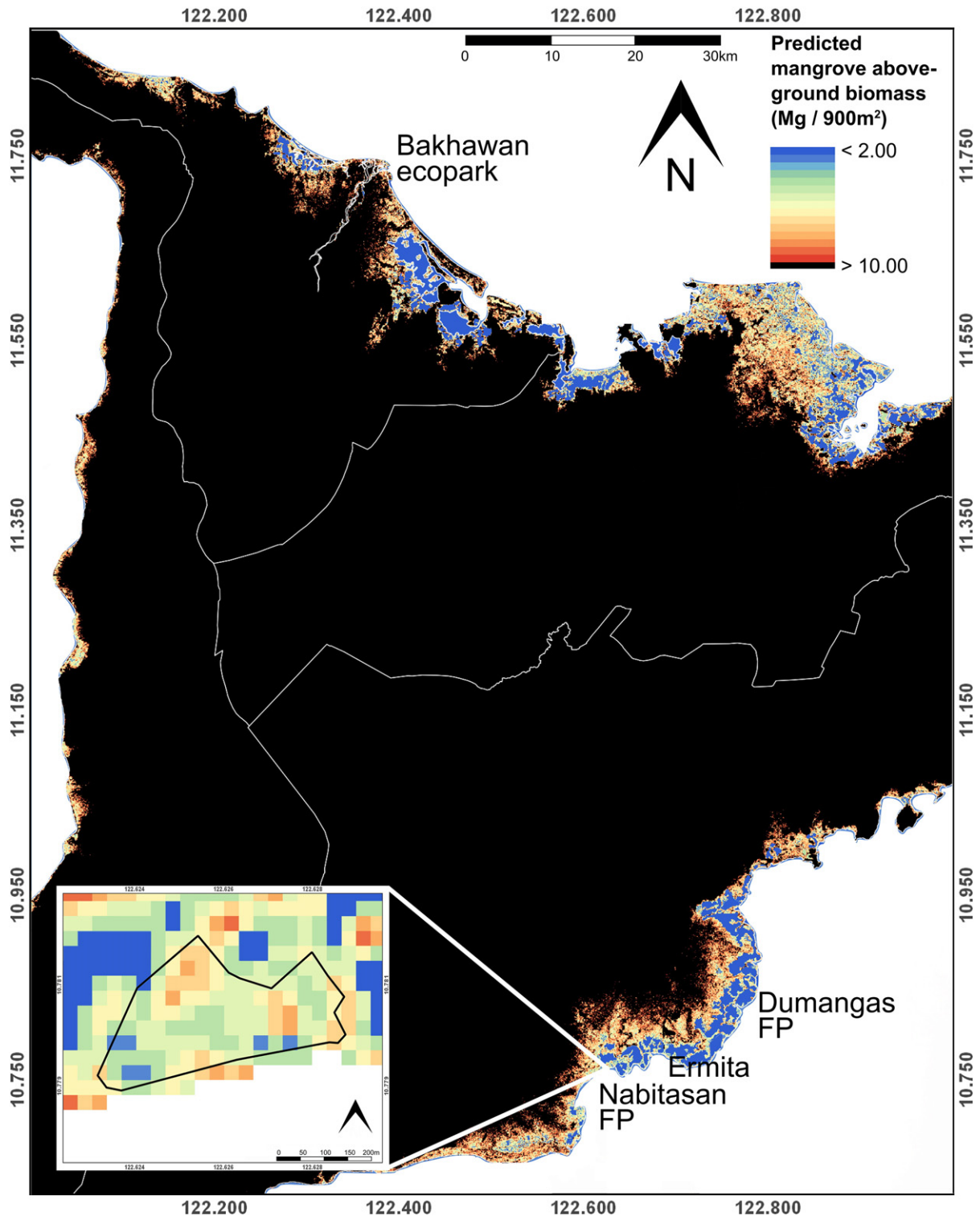


Fig. 4. Predicted aboveground mangrove biomass ($\text{Mg } 900\text{m}^2$) across the two SRTM DEM tiles on Panay Island. Blue pixels denote areas of low biomass, while red areas denote higher biomass areas. Dark blue pixels indicate active aquaculture pond areas, and black pixels denote areas with biomass $> 10 \text{ Mg } 900 \text{ m}^2$. N.B. This figure illustrates predictions of areas outside of the distribution of mangroves on Panay Island (e.g. beach forest and terrestrial forest and plantation areas), which were not included in the analyses of this study.

abandoned fishpond site resulted in a low protection category (I) and high required greenbelt width ($B_W = 2405 \text{ m}$).

3.6. Case study: Dumangas municipality

We delimited an area of 377.25 ha of abandoned fishponds in Dumangas (8.8% of total current aquaculture area; Primavera et al.,

2014). Potential total vegetation carbon stock accumulation across this area over 6.5 years was estimated at 4691.73 Mg. We estimated potential sediment carbon accretion (1.63 cm ; 0.25 cm yr^{-1}) over 6.5 years at 911.32 Mg. We further undertook this analysis based on only vegetation and sediment carbon stock estimates (Mg ha^{-1}) from the partially-banked Dumangas abandoned fishpond site in order to model potential abandoned fishpond reversion carbon gains under seaward dike

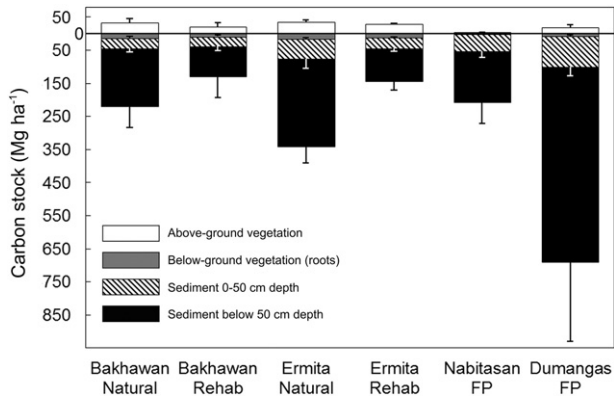


Fig. 5. Mean mangrove vegetation and sediment carbon stocks across all study sites (Mg ha^{-1}). Mean plot-level sediment carbon stock to 30 cm depth is depicted in hashed bars, and from 30 cm to soil profile depth in filled black bars. Mean aboveground vegetation carbon stock is depicted in white bars, and below-ground (roots) carbon stock in grey bars. Mean above- and below-ground vegetation carbon stocks (Mg ha^{-1}) were calculated as the mean SRTM DEM-predicted above- and below-ground carbon storage across all pixels with >50% of their area within each given study site (see Methods). Mean sediment carbon stock estimates (Mg ha^{-1}) were calculated across all plot-level measurements ($N = 8$ per site). Error bars depict one standard deviation from the mean of each estimate at each site.

management for high propagule and organic matter retention (see Section 2.1). Under this scenario, we estimated potential total vegetation carbon accumulation of 6564.49 Mg and sediment carbon accretion of 1166.41 Mg in Dumangas municipality's abandoned fishponds over 6.5 years. Of the 377.25 ha of abandoned fishponds identified in Dumangas, 167.59 ha (44.4%) are currently leased under FLAs (Primavera et al., 2014).

We estimated a length of 8.59 km of Dumangas' coastline (30.5% of total coastline) currently fringed by abandoned fishponds. All abandoned fishponds within this 8.59 km had an estimated median landward width > 100 m (current greenbelt mandate). 3.66 km of coastline had an estimated median landward width > 370 m (estimated required bandwidth (B_w) based on the Dumangas abandoned fishpond site), translating to 13.0% of the Dumangas municipality coastline with potential adequate coastal protection from abandoned fishpond mangroves over eight years from reversion (Dumangas abandoned fishpond site age; Table 4). Of this, 3.45 km (96.7%) is fringed by abandoned fishponds in breach of FLAs (Primavera et al., 2014).

4. Discussion

This study provides a quantitative analysis of relative CCMA ES delivery by different mangrove rehabilitation areas and adjacent natural

stands. While per hectare carbon stocks were variable across both rehabilitated and natural areas, rehabilitation for enhanced CCMA goals appears more promising in abandoned fishponds. Despite currently lower per hectare biomass production, carbon-rich sediments and large areal coverage enhanced the overall carbon stocks and coastal protection potential of rehabilitated abandoned fishponds. Our municipality-specific case study revealed that overlap may exist between areas of high rehabilitation potential for CCMA goals and low competing opportunity costs, with 96.7% of the identified wide sea-facing abandoned fishponds currently in breach of lease agreements (FLAs) on public lands. Our results may also have implications regarding reverted abandoned fishpond management for high carbon stocks.

All rehabilitated stands exhibited structural parameters (stem density, DBH, biomass) within observed trajectories to maturity (15–20 years; Bosire et al., 2008; Alongi, 2011; Table 1), suggesting good rehabilitation status to date. However, contrary to expectations according to their mid- to upper-intertidal position, both abandoned fishponds had comparatively low per hectare vegetation carbon stocks (Table 1; Fig. 5). This may in part reflect methodological limitations that could have underestimated true biomass and canopy height: first, shrubby *A. marina* was dominant at both sites, and biomass calculation for multi-stemmed individuals (Fu and Wu, 2011) may have underestimated biomass; second, SRTM-DEM data were acquired prior to rehabilitation of these areas (Rodriguez et al., 2006). Application of new high resolution TanDEM-X global DEM data (2014) to be released for scientific use in late 2016–2017 will enhance the application of SAR-derived DEM data to recent mangrove rehabilitation monitoring (Zink et al., 2015). However, the SRTM-DEM related limitation also applied to the seafront rehabilitated areas, where estimated vegetation carbon stocks remained high (Fig. 5). High heterogeneity across both abandoned fishponds, due to on-going natural recolonization at the Dumangas site and high pre-rehabilitation erosion at the Nabitasan site, likely contributed to low average per hectare vegetation carbon stocks. However, lower relative biomass production may also indicate possible hydrological or fishpond effluent constraints to biomass production in abandoned fishponds (Lewis, 2005; Matsui et al., 2010; Primavera et al., 2014). These results suggest further active rehabilitation may be necessary to enhance mangrove functioning in reverted abandoned fishponds. Indeed, the Bakhawan seafront site had the highest per hectare vegetation carbon stocks of all rehabilitated sites (Fig. 5), which may reflect a combination of active replanting and high natural re-colonisation as an effective strategy in low-mid intertidal areas (sensu Matsui et al., 2010), and a possible positive influence of multi-species rehabilitation (Lang'at et al., 2011).

As observed in other studies from the region (Thompson et al., 2014), carbon stocks in natural areas (Fig. 5) were at the lower end of estimates for Indo-Pacific mangroves (Donato et al., 2011). The abandoned fishponds were either not different (Nabitasan) or had significantly greater (Dumangas) plot-level sediment carbon stocks than natural areas, and greater than seafront rehabilitated areas (Fig. 5). This reflects their position on former mangrove sediments, and highlights their greater potential over seafront rehabilitation sites in reforestation PES schemes that recognise existing stocks (Locatelli et al., 2014). The large sediment carbon stocks at the Dumangas fishpond are notable (mean $684.67 \pm 263.39 \text{ Mg ha}^{-1}$), as these are among the highest recorded in Panay. This includes a large ancient basin mangrove (mean sediment carbon stock $372.60 \pm 128.61 \text{ Mg ha}^{-1}$; C. Duncan unpublished data). A possible driver is site configuration: much of the seaward bank is retained and reduces complete tidal flushing, trapping organic matter in partially-waterlogged sediments. Conversely, near-complete loss of the Nabitasan fishpond seaward bank before recolonization has resulted in extensive erosion at the seaward margin. While our observations are limited to only two abandoned fishponds, these results suggest that retaining partial seaward banks may be important for preventing erosion, retaining sediment carbon stocks and improving propagule establishment in recovering abandoned fishponds (similar to breakwater

Table 2
Data summary from soil analysis, organised by site and soil depth interval. Values represent mean values across all plots at each site ($N = 8$) \pm one standard deviation. Sediment depth values denote the range of depths across all plots.

Site	Depth interval (cm)	Bulk density (g cm^{-3})	OC (%)	Sediment depth (cm)
Bakhawan Natural	0–50	0.60 ± 0.07	1.01 ± 0.29	207–238
	50–150	0.67 ± 0.08	1.61 ± 0.80	
Bakhawan Rehab	0–50	0.65 ± 0.16	0.85 ± 0.33	181–226
	50–150	0.68 ± 0.16	0.82 ± 0.55	
Ermita Natural	0–50	0.70 ± 0.11	1.84 ± 1.11	163–391
	50–150	0.63 ± 0.07	2.33 ± 0.83	
Ermita Rehab	0–50	0.91 ± 0.07	0.71 ± 0.24	159–172
	50–150	0.85 ± 0.06	1.02 ± 0.32	
Nabitasan FP	0–50	0.62 ± 0.04	1.72 ± 0.60	146–250
	50–150	0.63 ± 0.03	1.70 ± 0.51	
Dumangas FP	0–50	0.38 ± 0.08	4.88 ± 0.97	380–400
	50–150	0.37 ± 0.12	4.59 ± 1.41	

Table 3
Results of site-level carbon stock analysis.

Site	Aboveground vegetation C (Mg)	Belowground vegetation C (Mg)	Sediment C 0–50 cm (Mg)	Sediment C below 50 cm (Mg)	Total (Mg)	Site area (ha)
Bakhawan Natural	1267.27	546.10	1176.94	6711.17	9701.48	38.59
Bakhawan Rehab	430.18	168.88	563.18	1795.74	2957.98	19.52
Ermita Natural	79.30	34.73	140.31	619.79	908.86	2.34
Ermita Rehab	5.05	2.14	15.56	49.04	71.79	0.50
Nabitasan FP	41.35	5.47	477.97	1392.94	1917.73	9.04
Dumangas FP	856.90	319.22	4315.52	27,167.17	32,658.81	45.99

interventions: Hashim et al., 2010; Primavera et al., 2012a). Relatively high bulk densities and low OC at seafront rehabilitation sites compared to adjacent natural areas (Table 2) suggest low rates of soft sediment accretion, and slow gains in potential sequestration-oriented PES schemes (Locatelli et al., 2014). Further surveying of abandoned fishpond areas, and study of accretion rates in abandoned fishponds and seafront plantations, will be required to investigate these findings further and establish relative sediment carbon sequestration rates.

Overall, the comparatively large size, mid- to upper-intertidal position and high sediment carbon of the abandoned fishponds translated to high relative total ecosystem carbon stocks (Table 3). We moreover predicted high vegetation and sediment carbon sequestration gains over 6.5 years of potential future rehabilitation of abandoned fishponds in Dumangas municipality (5809.95–7995.75 Mg). It is important to note that these estimates are preliminary and likely underestimates; they employ basic (but conservative) accretion rates and do not account for important processes such as carbon burial (Lee et al., 2014). Large area translates to landward widths available for rehabilitation at abandoned fishpond sites (Table 4) that are considerably greater than at rehabilitated seafront areas (Bakhawan: 186 m; Ermita: 41 m). Seafront rehabilitation sites were bordered at their landward margins by existing natural mangrove, increasing effective greenbelt width (Table 4). In the unique case of Bakhawan ecopark (see Section 2.1), the combination of rehabilitated and natural forest make the total greenbelt well above that estimated for effective coastal protection potential (Table 4). However, for Ermita, a site more typical of remaining seafront mangroves in the Philippines, the combined greenbelt width of both rehabilitated and natural areas is far below that required (Table 4). Repeated LGU planting in adjacent areas to the Ermita site since the early 2000s have seen no survival. The existence of a coastal road between these now fringing areas and the former deltaic mangrove (now active fishponds) inland further prohibits development of an effective greenbelt in the area.

In contrast, both abandoned fishponds had large median landward widths (Nabitasan: 268 m; Dumangas: 827 m). Low vegetation density caused very large estimates of required greenbelt width for Nabitasan abandoned fishpond mangroves (Table 4). However, saplings (high density at Nabitasan) were not considered in these analyses, and can contribute to protection from regular wind waves (Mazda et al., 1997). Furthermore, rehabilitated mangroves at this site are comparatively young (Table 4), and assisted rehabilitation by the Leganes LGU

is on-going. The landward greenbelt width of the Dumangas abandoned fishpond may provide ample coastal protection, even while in relatively young stages of development (Table 4; Alongi, 2011). In direct conflict with current rehabilitation programmes in Dumangas, we found a large length of coastline (3.66 km) with adequate future protection potential from abandoned fishpond mangroves (>370 m wide) eight years post-rehabilitation. While this wide greenbelt requirement will reduce as rehabilitated abandoned fishpond mangroves mature (Alongi 2008; Bao, 2011; Mclvor et al., 2012; Lee et al., 2014), it is important to note that: (1) these estimates are derived for protection from regular waves (Bao, 2011), while greenbelt requirements for storm surge attenuation may be substantially greater (Koch et al., 2009; Mclvor et al., 2012), and (2) natural stands were estimated here to have required greenbelt widths of >200 m (Table 4). This suggests that current Philippines greenbelt laws (50–100 m) may be inadequate for coastal protection in typhoon-prone areas. Reallocation of Philippines' National Greening Program (NGP) funds and capacity away from seafront planting toward targeted reversion and assisted rehabilitation of large sea-facing abandoned fishponds is thus advisable if effective mangrove greenbelt reestablishment and integrated CCMA goals are to be realised (Blankespoor et al., 2016). Our municipality-specific case study revealed that 96.7% of coastline identified as having rehabilitation potential for effective coastal protection within eight years is currently tenured under and in breach of FLA terms. As such, reversion (DA-BFAR) and rehabilitation (DENR) of these areas is a legal requirement, is not in direct conflict with opportunity costs of fishpond leaseholders, and could provide substantial benefits to inland coastal communities from coastal protection and fisheries enhancement (Walton et al., 2006; Primavera et al., 2012a).

Integrated spatial planning and coastal zone management approaches incorporating ES goals are increasingly on the agenda (Lester et al., 2013; Arkema et al., 2015) to prioritise areas for management to reduce ecosystem degradation (e.g. Atkinson et al., 2016). Similarly, such approaches, including multiple CCMA ES and considering their value to the full cohort of stakeholders, could be an important tool for identifying key areas for whole ecosystem rehabilitation and restoration in converted lands. However, existing formally-recognised and local tenure structures provide a major challenge to spatial planning and prioritisation approaches (e.g. Adger et al., 2005; Brown et al., 2014), and may impact the effectiveness of management decisions (Weeks et al., 2010). In contrast, our case study identified substantial areas with minimal tenure conflict within priority areas for CCMA mangrove rehabilitation (mid- to upper-intertidal zone and large areal coverage). Spatial planning exercises to prioritise multiple CCMA ES greenbelt rehabilitation (based on e.g. coastal vulnerability assessment) should first aim to evaluate the existence of formally-recognised tenure gaps (e.g. tenure breaches or unproductive lands) to evaluate the rehabilitation potential of currently-tenured areas. In cases such as that of mangrove rehabilitation in the Philippines, this may maximise benefit:cost ratios of CCMA efforts for coastal communities and avoid ineffective and wasteful allocation of limited conservation funds. Such spatial planning approaches for mangrove rehabilitation may have particular relevance elsewhere in South and Southeast Asia where fishpond abandonment is similarly high: e.g. Malaysia (60%; Choo, 1996),

Table 4
Results of coastal greenbelt protection analysis. FSI = Forest Structure Index and the associated protection category (min = I; max = V), B_W = required greenbelt width (Bao, 2011), B_{Actual} = median actual greenbelt width calculated from GIS distance analysis (Bivand et al., 2015).

Site	FSI	Protection category	B_W (m)	B_{Actual} (m)	Age (years)
Bakhawan Natural	0.012	III	201	842	–
Bakhawan Rehab	0.009	II	270	842	8
Ermita Natural	0.011	III	229	81	–
Ermita Rehab	0.007	II	331	81	7
Nabitasan FP	0.001	I	2405	268	5
Dumangas FP	0.006	II	370	827	8

Thailand (50–80%; Hossain and Lin, 2001) and Sri Lanka (60–90%; Jayakody et al., 2012; Bournazel et al., 2015). In other, more typhoon-prone countries believed to have high rates of fishpond abandonment (e.g. Vietnam, Taiwan; Stevenson, 1997), the potential CCMA benefits of abandoned pond identification and rehabilitation may be of particular consequence. In many parts of Asia, much unproductive abandoned fishpond area is rapidly being converted to alternative uses, cementing mangrove loss (e.g. salt pans, agriculture; Stevenson, 1997; Hossain and Lin, 2001; Jayakody et al., 2012). Monitoring and evaluation of aquaculture productivity within these coastal areas and their inclusion within spatial planning for mangrove rehabilitation may provide a means to halt such loss. However, the relevance of such approaches to wider regional areas may be comparatively limited where (1) wider continental shelves enhance the suitability of low-intertidal seafloor rehabilitation, or (2) titled ownership is the predominant form of coastal zone land tenure.

In recent decades, evidence has been mounting on the relative potential of abandoned fishpond rehabilitation for conserving mangrove forests (e.g. Lewis, 2005; Matsui et al., 2010; Primavera et al., 2012a, 2012b, 2014; Brown et al., 2014). Our case study highlights a high contribution of ES strings to the bow of fishpond reversion, with a high relative potential for coastal greenbelt rehabilitation for integrated CCMA goals in the Philippines. While our case study revealed favourable tenure status for most potential greenbelt abandoned fishpond areas, strong political will and capacity shifts will be required to cancel leaseholds and rehabilitate former mangrove areas (Primavera et al., 2014). However, mid-upper intertidal mangrove rehabilitation in abandoned fishpond areas may hold one of the many keys to safeguarding mangrove forests and their CCMA ES delivery, particularly important in the context of the increasing severity of storms and sea level rise associated with climate change (IPCC, 2013; Lovelock et al., 2015; Blankespoor et al., 2016).

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Appendix A. Supplementary data

Supplementary data to this article can be found online at <http://dx.doi.org/10.1016/j.marpolbul.2016.05.049>.

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