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# Soil carbon stocks and accumulation in young mangrove forests

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ABSTRACT

Mangrove reforestation and afforestation programs have been initiated in many countries recently to compensate for historical losses. At the same time, awareness of the high carbon (C) sink potential of mangrove forests is growing, and C sequestration is beginning to be considered among forestation goals. To assess whether and at what rate C accumulates in the soil of young mangrove forests following afforestation, we conducted a field study at an afforestation project in southeast China, including repeated measures taken over six years at two young forests (consisting of *Kandelia obovata* and *Sonneratia apetala*, aged 0–6 years old), and also a chronosequence of forests aged 0 (mudflat), 6 (both species), 20 (*S. apetala*), and 70 (*K. obovata*) years old. In the repeated measures, surface (0–10 cm) soil C concentration (%C of dry soil mass) increased significantly over six years, from 1.14% to 1.52% (*K. obovata*) and 1.23% to 1.68% (*S. apetala*). The rates of increase did not differ significantly between the two species, despite much greater biomass of *S. apetala*. In the chronosequence, soil C also increased with age across sites, but only the 70-year-old forest was statistically different, suggesting that localized environmental differences may obscure age-related patterns in soil C. At all sites, soil C concentration for 1-m soil depth (0.62%–2.43%) was low compared to published global averages, yet the estimated soil C accumulation rate (155 g C $m^{-2}$ $y^{-1}$) was comparable to published averages for mature forests. We supported this field study with a literature review of similar studies containing soil C concentration data from young mangrove forests: data compiled from 15 studies, comprising 31 sites, showed consistent, positive changes in soil C concentration with forest age, even in the youngest (<5 years old) forests, supporting our field observation that soil C increases over time following mangrove afforestation.


## 1. Introduction

Recent studies have shown that coastal wetlands such as mangroves, salt marshes, and seagrass beds are among the most efficient carbon (C) sinks on the planet (Donato et al., 2011). Reviews of the available data estimate that carbon sequestration in these systems (popularly termed “blue carbon”) per unit area is on average more than an order of magnitude higher than terrestrial forests. These blue carbon systems are also among the most threatened ecosystems on the planet. For mangroves, the most recent areal estimates range from 13.8 million to 16.7 million ha (Valiela et al., 2001; Giri et al., 2011), which is a 20% reduction since 1980, and an estimated 35% loss compared to historical extents (Valiela et al., 2001; FAO, 2007).

To address this trend, mangrove reforestation and afforestation programs (collectively referred to as forestation in this study, unless otherwise specified) have been initiated in many countries over the past few decades (Field, 1998; Ellison, 2000). The earliest forestation efforts were essentially silviculture-oriented, while more recent projects have included ecological objectives, such as shoreline stabilization and environmental remediation (Ellison, 2000). With increasing concerns related to climate change, carbon sequestration is also beginning to be considered among conservation and forestation goals (Murray et al., 2011). An example of this trend is the recent release of guidelines for C sequestration accounting in mangrove restoration projects certified under the Cleaner Development Mechanism (CDM) program of the United Nations Framework Convention on Climate Change (UNFCCC, 2011).

In mangrove forests, a large proportion—from half to over 90%—of total ecosystem C is found in soil organic matter as opposed to

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biomass (Alongi et al., 2003; Khan et al., 2007; Donato et al., 2011). Direct measurements of soil C accumulation in young forestation projects are scarce, however. Several reviews of C sequestration in mangrove soil have been published recently (e.g. Bouillon et al., 2008; Mcleod et al., 2011; Alongi, 2012), but such reviews focus on mature forests and the existing data appear to be biased toward highly organic systems (Kristensen et al., 2008). Similarly, reviews of mangrove restoration projects (e.g. Bosire et al., 2008) provide data on post-restoration biomass accumulation, but not soil properties. Along these lines, CDM approval of restoration projects does not require soil C monitoring before or during the project lifetime. Of the published data that do exist, only a few studies provide direct measurements of change in soil C using repeated measures over time (Ren et al., 2010; Chen et al., 2012). Several more have used forestation chronosequences (a.k.a. space for time substitution) (e.g. Bosire et al., 2003; Alongi et al., 2004; Ren et al., 2008), and additional insight has been gained from similar studies of natural chronosequences, in which forest age varies in relation to naturally changing coastal conditions (Marchand et al., 2003; Lovelock et al., 2010).

In contrast, much more is known about soil C accumulation in terrestrial forestation projects, and several reviews have been written on this topic (Paul et al., 2002; Berthrong et al., 2009; Laganiere et al., 2010). Such studies have found that on average across all projects, soil C initially decreases following afforestation, and may maintain a deficit for up to 30 years (Paul et al., 2002). However, the results vary significantly depending on forestation species, previous land use, soil clay content, and climate. Mangroves allocate a large proportion of biomass belowground compared to many terrestrial tree species (Lovelock, 2008), so we hypothesize that soil C would increase relatively sooner following forestation. Given the high proportion of C stored in soil in mangrove ecosystems, accurately quantifying this change is especially important for assessing total ecosystem C during forest development.

China lies at the northern edge of global mangrove distribution, and similar to other countries with mangroves, has experienced drastic losses in forest cover in recent history. The most recent estimate of mangrove cover is 22,700 ha (Chen et al., 2009), less than a third of the historical extent (Li and Lee, 1997). To address this loss, the Chinese government began investing in mangrove reforestation in the early 1990s (Zheng et al., 2003), and by 2002 more than 2600 ha (more than 10% of the existing area) were replanted (Chen et al., 2009). This reforestation program presents an opportunity to study soil C accumulation in young forests following afforestation, including results across different species. Of particular interest is *Sonneratia apetala*, a non-native mangrove species from Bangladesh introduced to China in 1985, which has been used for approximately half the country's forestation projects so far (Chen et al., 2009). *Kandelia obovata*, a native mangrove species distributed along the southeast Chinese coast, has also been widely used in mangrove forestation (Chen et al., 2009). Previous research has shown that *S. apetala* has much higher rates of biomass C accumulation than most native species (Ren et al., 2008, 2010). Ren et al.'s (2008) study also found higher soil C values in *S. apetala* forests compared to neighboring native forests, suggesting that *S. apetala* may have higher rates of C accumulation in soil as well as biomass. According to the soil development model proposed by Chen and Twilley (1999), soil C is correlated with biomass inputs to soil, particularly root biomass. Thus, it is reasonable to expect that soil C will accumulate faster in *S. apetala* forests. However, comparative data of direct measurements is only available from a single study of 2.5-year-old forests (Chen et al., 2012).

In this study, we aimed to assess whether, and at what rate, soil C content increases over time in young, afforested mangrove forests. To achieve this, we collected two distinct data sets at four young forests in a mangrove forestation project in southeast China. The first data set consisted of repeated measures of surface soil C concentration taken in two young forests (*K. obovata* and *S. apetala*) from age 0 to 6 years old. The second data set consisted of a chronosequence (one-time measurements) of 1-m soil cores taken at the two 6-year-old forests, a 20-year-old *S. apetala* forest, a 70-year-old *K. obovata* forest, and a nearby mudflat. Using these data sets, we tested the hypotheses 1) that soil C increases with forest age following mangrove forestation, even in very young forests; and 2) that soil C accumulates more quickly in forested *S. apetala*, compared to *K. obovata* plots, due to higher rates of above- and belowground biomass growth in the former species. To support our first hypothesis, we also compiled all available data (published and unpublished) on soil C change in young mangrove forests, aiming to assess whether trends in the literature support or refute the results of this field study.

## 2. Materials and methods

### 2.1. Study sites

The Futian National Nature Reserve is a small, 300-ha reserve on Shenzhen Bay (also known as Deep Bay), in Shenzhen, Guangdong, China (Fig. 1). The reserve is contiguous with the Mai Po Reserve in Hong Kong. A large levee borders the reserve on the landward side, beyond which are aquaculture ponds, and then a high-density urban area. Urban, industrial, and sewage inputs contribute to high nutrients, heavy metals, and other pollutants in the mangrove area (Tam, 2006). Access to the Futian reserve is strictly regulated, however, and thus human physical disturbance is minimal. The area has a monsoon-influenced, humid, sub-tropical climate with a mean annual temperature of 22.5 °C. The mangroves in the reserve are located in a 50–150 m wide strip along the coast, with both forestation areas and naturally colonized forests. The primary species are *K. obovata*, *Aegicerias corniculatum*, and *Avicennia marina*, with smaller areas of *S. apetala* and *Sonneratia caseolaris*. *S. apetala* (non-native) and *S. caseolaris* (native to Hainan Province of China) have been planted in patches near the native forests beginning in 1993 (Zhang, 1997).

For the repeated measures study, we studied two young forests in the reserve, both planted in May, 2006 using 6-month-old *S. apetala* seedlings and *K. obovata* hypocotyls. The forests were planted into mudflat areas in $8 \times 8$ m monoculture quadrats ($n = 4$), at a density of one tree m$^{-2}$. The quadrats were randomly distributed in a line parallel to the levee, while retaining a buffer of >10 m between each quadrat. These forestation sites were approximately 250 m southeast from, and at similar elevations to the older forests (described below).

For the chronosequence, we sampled five distinct sites within the reserve: the two young forests described above, two older forests, and a reference mudflat. The oldest forest consisted of a 70-year-old forest composed predominantly of *K. obovata*, with occasional *A. corniculatum* and *A. marina*. A 20-year-old forest composed primarily of *S. apetala* with occasional *A. corniculatum* was sampled 100 m seaward of the *K. obovata* forest, and a mudflat was sampled immediately adjacent and seaward of this forest. The bed slope of the research area is approximately 0.003°, and elevation of the sites range from approximately 1.3 to 1.7 m above the Hong Kong Observatory chart datum.

### 2.2. Biomass and forest structure

We estimated above- and below-ground biomass of the two young (6-year-old) forests annually from 2006 to 2011. For years