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Sonneratia apetala Buch.Ham in the mangrove ecosystems of China: An invasive species or restoration species?

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ABSTRACT

By the end of 1990s when China initiated a 10-year mangrove reforestation project, the mangrove forest area had decreased from 250,000 to 15,000 ha. Over 80% of current Chinese mangroves are degraded secondary forests or plantations. As an initial restoration and reforestation effort, *Sonneratia apetala*, a native of India, Bengal and Sri Lanka, was introduced in 1985 to Dong Zhaigang Mangrove Nature Reserve in Hainan Island from Bengal. It has then been introduced into other places since 1991. However, the further use of the species is becoming increasingly controversial as there are emerging signs that it may become invasive in certain locations. A comprehensive evaluation of the species' condition in China regarding benefits and risks is critically needed. Here, we map the introduction and dispersal routes and monitor the growth of *S. apetala* in China from 1985 to 2006. *S. apetala* grows fast and performs well in the introduced 2300 ha muddy beaches area. It greatly improves the soil fertility and shows a suite of suitable characteristics as a pioneer restoration species. Currently, no natural invasion of *S. apetala* has been observed in the northern mangrove area. However, invasion into natural forests does occur in southerly locations such as Shenzhen, Zhanjiang and Dong Zhaigang. In these locations, *S. apetala* exhibits invasive characteristics such as overgrowth and high spreading ability that evidently affects local mangrove ecosystem structure and function. While the species clearly offers some benefits at some locations where it cannot naturally invade, it appears harmful to other native mangrove species, posing a major practical problem to both ecologists and land managers. This situation will be similar to previously imported non-native and invasive intertidal wetland species, *Spartina alterniflora* (smooth cordgrass), with similar results and problems.

1. Introduction

There are 84 mangrove species around the world including 12 varieties, belonging to 16 families and 24 genera and distributed in a total area of about 17,000,000 ha (Wang and Wang, 2007). Mangrove is a typical ecosystem in the intertidal area from tropical to subtropical muddy beaches worldwide. It has critical ecosystem functions such as coastal protection, land stabilization, water purification, and CO_2 fixation. It also provides important ecosystem services such as food, wood, chemical, pharmaceutical production, and other aesthetic values. However, mangroves are often located in fragile habitats and very few mangrove systems could be restored to the original state once destroyed (Saenger et al., 1983; Tomlinson, 1999). Because of its under-estimated value, increasing human activities in these systems, and immediate economic needs, large mangrove areas have been converted in the past to agriculture, aquaculture and even industrial use (Robertsen and Alongl, 1992). Currently, many mangrove ecosystems are still in degrading status worldwide.

China has 26 mangrove species (including 1 variety, and 4 endemic species) of 12 families and 15 genera. They are mainly distributed in Guangdong, Guangxi, Fujian, Hainan, Taiwan, Hong Kong and Macao (Wang et al., 2003). Their northern natural distribution boundary in China is in Fuding County, Fujian (27°20′N; mean temperature of air and water in January is 9.8 and 10.9 °C, respectively). For introduced mangrove species and a native species, *Kandelia candel* (Linn.) Druce, the northern distribution boundary is in Leqing County, Zhejiang where Kuroshio often has influences (28°25′N, mean temperature of air and water in January is 9.3 and 10.6 °C, respectively) (FAO Forestry Department, 1994; Lin, 1997; Wang et al., 2003).

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The mangrove area in China once occupied 250,000 ha, but was reduced to 42,000 ha by 1956 according to the National Investigation of Forest Resource (Wang and Wang, 2007). As a result of land reclamation during 1970 to 1980, the mangrove area in China further shrank to 21,000 ha by the end of 1980s and to 15,000 ha by the end of 1990s. Causes included the development of aquaculture production, urbanization, and port construction. In the early 1990s, China initiated a 10-year mangrove reforestation project, through which the mangrove forests in China have since increased to 22,000 ha (Wang and Wang, 2007). However, over 80% of these mangroves are degraded secondary forests. Over 90% of the mangrove systems in China are in protected areas (Zhang and Sui, 2001).

China has a great potential for the rehabilitation of mangrove ecosystems. Based on a preliminary estimation, about 21,531 km coastline and 300,000 ha beaches are suitable for restoration. In early 1882, some mangrove seedlings were introduced from Southeast Asia to Longhai County, Fujian. Since then, a total of eight mangrove species have been introduced and mangrove ecosystem restoration has been undertaken on some beaches. After more than a century of mangrove introduction in China, only some cold resistant and widely distributed species such as *K. candel*, *Avicennia marina* (Forsk). Vierh, and some thermophilic and widely distributed species (e.g., *Bruguiera* species) have been successfully introduced. Among the eight mangrove species used in mangrove reforestation in China, *Sonneratia apetala* Buch.Ham occupies about 95% of the restoration area (Liao et al., 2004; Wang and Wang, 2007). In the next 10 years, the Chinese government intends to plant an additional 60,000 ha of mangroves and *S. apetala* is listed as one of the leading choices.

S. apetala is one of the woody mangrove species with high adaptability and seed production. The height of the species ranges from 15 to 20 m, and diameter at breast height (DBH) varies between 20 and 30 cm. The species is naturally distributed in India, Bengal and Sri Lanka as a dominant species in local mangrove communities (Jayatissa et al., 2002). The species was introduced to China in 1985 for restoration purposes but becomes invasive in some places, and yet no other species have been identified as a good substitute. This paper assesses a critical issue regarding whether and/or where *S. apetala* Buch.Ham in China is an invasive or restoration species for mangrove ecosystems. We examine the introduction routes, its impacts, and assess the invasion risks. At the end, we discuss several issues and viewpoints about the controversial use of the species.

2. Methods

The Chinese Mangrove Survey Group introduced *S. apetala* from Bengal to Dong Zhaigang Mangrove Nature Reserve (DZMNR, 19°56′N and 110°34′E; tropical monsoon climate, annual mean temperature 23.8 °C, annual mean precipitation 1685 mm) in Hainan Island in 1985 (Wang and Wang, 2007). The species produced seeds in the following three years (Chen et al., 2003; Liao et al., 2004). The species was then introduced to Zhanjiang City in western Guangdong (21°30′N and 109°41′E; subtropical monsoon climate, annual temperature 22.8 °C, annual precipitation 1757 mm), and Shenzhen Bay in southern Guangdong during 1991–1995. It grew well in all those new places and its distribution boundary extended 2.34° northward (Liao et al., 2004; Wang and Wang, 2007). During 1996–2000, the species was further introduced to Chenghai City in eastern Guangdong (23°21′N and 116°41′E; subtropical monsoon climate, annual temperature 21.2 °C, annual precipitation 1555 mm), and the estuary of Jiulong River in Fujian (24°24′N and 117°55′E; subtropical monsoon climate, annual mean temperature 21 °C, annual mean precipitation 1365 mm) where the species seeded in only two years and its distribution further extended 4.28° northward. No disease or pest was observed for the species in these introduced locations (Chen et al., 2003; Liao et al., 2004).

Since 1993, we have been monitoring the subsequent introductions (time and routes) and spread of *S. apetala* in China. In 2006, we established and investigated 81 sites of *S. apetala* in 24 urbanized coasts where all *S. apetala* plantations and distributions were located. We established three research sites in Zhanjiang, Shenzhen and Qiongshan to examine the detailed biological and ecological characteristics of the species. In 1998, we established 27 quadrats at the three sites to study the biomass and community structure. We also initiated an ecological restoration project in 2004 and established permanent transects in the restoration sites. In order to compare the average growth between the introduced *S. apetala* and a local mangrove species, *K. candel*, we established the *S. apetala* and *K. candel* plantations (seedlings) in 2003 in Qiongshan. We planted a total of 2500 individuals per ha for each species with 2-m spacing among them. We established three transects across the whole land–sea interface zone for each monoculture in each study area, and various numbers of 10 m × 10 m quadrats were placed along each transect depending on the length of the transects. For each transect, three quadrats were selected for further analysis. The name of species, plant height, DBH-trees only, basal diameter, crown size, and growth status (alive or dead) of each individual were recorded (Snedaker, 1984; Krebs, 1985).

Three 60-cm soil cores were taken at random locations from each of the 10 m × 10 m quadrats to form a mixed sample, using a 3.7-cm diameter coring tube. Three replicates were taken from each quadrat at low tide. Salinity and pH were measured using an optical refractometer and a pH meter, respectively. Samples were oven-dried at 80 °C for about 3 days until constant dry weight was obtained. Analysis of organic matter, total N, total P, Sanity and clay followed Ren et al. (2008), Robert et al. (1997), Kairo et al. (2002), and Bosire et al. (2003). The sediment characteristics were assessed using a Tukey-test after one-way ANOVA ($P < 0.05$) to indicate significant differences. All statistics were analyzed using SPSS11.5 software (Duan et al., 2008).

To examine the species introduction history in China, we conducted a literature search using the Weipu Database and Chinese Journal Database which contained almost all related Chinese publications. We used the word "*S. apetala*" or "mangrove invasion/restoration" in the title, abstract or keywords. We also examined major books published since 1991 dealing with mangroves in China (Richardson et al., 2000). After eliminating the publications that were not appropriate for our survey, our final database comprised 58 papers. In addition, we searched local government forestation planning projects in an attempt to understand the background of previous planting and management policies.

3. Results and discussion

3.1. Introduction and spread of *S. apetala* in China

The introduction routes and spread over time are shown in Fig. 1. The estimated *S. apetala* area in 2006 was about 2300 ha. A continuous colonization of over 200 ha of *S. apetala* was observed in Zhanjiang City, Jiangman City and Shantou City in Guangdong.

3.2. Ecological impacts of *S. apetala* in China

Climate data in all of the introduced sites indicate that *S. apetala* is a cold resistant species and can maintain normal growth in habitats with 14.1 °C mean temperature in the coldest month, and 0.2 °C extreme low temperature. *S. apetala* blooms and seeds in both spring and autumn but seeds mainly in autumn. The species had

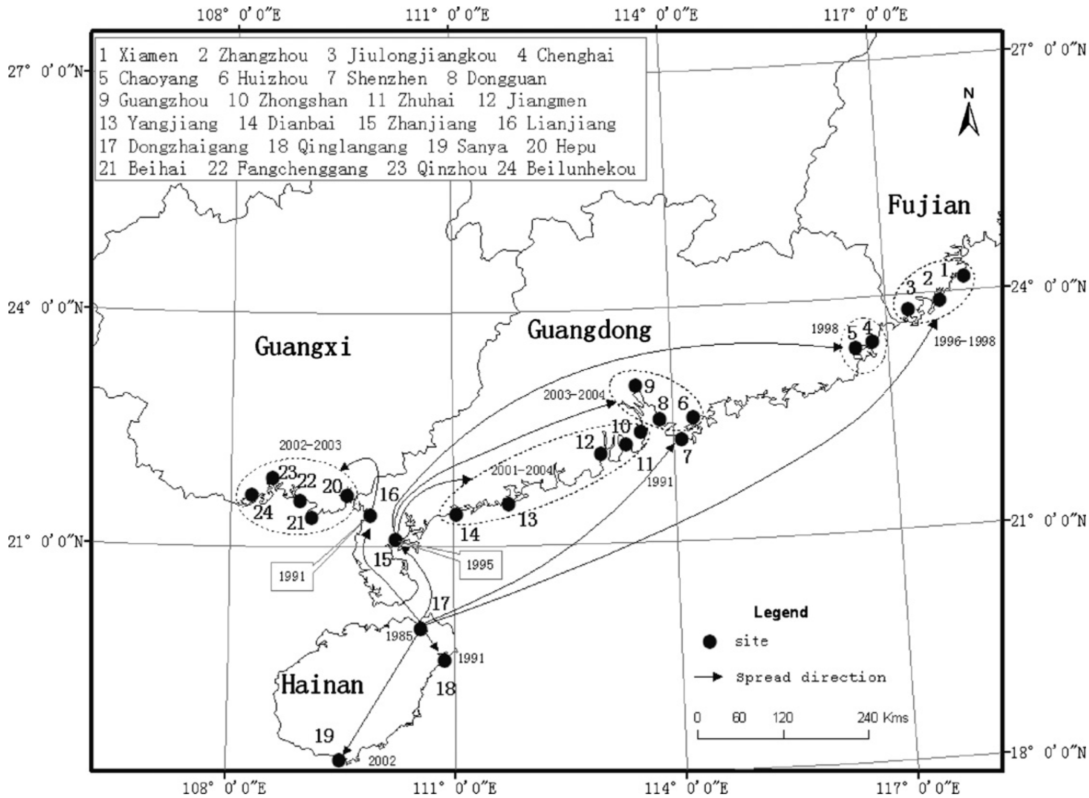

Fig. 1. The introduction, spread timeline and direction, and year of plantation of *S. apetala* in southern China.

great adaptability in tidal areas, and can grow in middle to low tide areas. It normally grows in silty soil to clay soil, but grows best in thick and soft muddy soils from the middle to low tidal zone. In general, *S. apetala* prefers muddy beaches with low salinity (0–15‰) (Huang and Zhang, 2002; Chen et al., 2003; Liao et al., 2004).

Relative to local native species, *S. apetala* performs better on barren muddy beaches with higher survival and growth rates. Pioneer *S. apetala* plantations usually help recolonization of local mangrove species, such as *K. candel*, *A. marina*, and *Aegiceras corniculatum* (Linn.) Blanc (Lin, 2003). As companions with *S. apetala* in the species-poor barren flat, waterfowl and other birds were found incubating in the plantation, and eels also moved into the mud. As a result, the biodiversity on the original barren flat increased (Wang and Wang, 2007). At the Zhanjiang site, following the growth of *S. apetala*, the pH of soil quickly decreased, while the content of clay particle, organic matter, total N, and Fe content increased, suggesting substantial improvements in soil conditions (Table 1).

Chen et al. (2003) and our previous studies (Table 2) showed a suite of adaptive characteristics of *S. apetala* as a pioneer species for restoration such as easy establishment, fast growth, and high adaptability in three coastal provinces in China.

3.3. Risk assessment of the spread of S. apetala

Natural dispersal and spread of *S. apetala* occurred in nearly all the plantations close to the introduced sites in Guangdong, Guangxi, Hainan, and Fujian. Because the species also invaded native ecosystems and in 2004 the species was consequently listed as an invasive exotic species (Wang et al., 2004), land managers were concerned about the ecological risks of further introductions of *S. apetala* to other locations.

First, *S. apetala* seeds may persist in the soil seed banks for a long time. The inter-simple sequence repeats analysis (ISSR) by Li and Lin (2005) showed high population genetic diversity and high adaptability of *S. apetala* from the secondary seed resources, an indication of species invasiveness. Second, the species has high survival and growth rates (Table 2) and its dense canopy and high stems usually shade out native mangrove species. For example, Ren et al. (2008)

Table 1
Soil chemical properties in *S. apetala* plantations of different age classes.

Age	Depth (cm)	pH	Sanity	Organic matter (g/kg)	Total P (g/kg)	Total N (g/kg)	Fe (g/kg)	C/N
1	0–20	7.33 ± 0.24a	5.74 ± 0.21b	16.98 ± 0.57a	0.36 ± 0.04b	0.54 ± 0.01a	13.02 ± 1.09a	18.2 ± 0.7
	20–40	7.06 ± 0.05b	4.32 ± 0.07a	15.96 ± 0.18a	0.35 ± 0.03b	0.73 ± 0.13b	12.99 ± 1.02a	12.7 ± 0.2
	40–60	6.93 ± 0.14b	5.46 ± 0.13b	34.62 ± 0.03c	0.37 ± 0.08b	0.48 ± 0.05a	15.27 ± 0.89b	41.8 ± 0.1
4	0–20	6.10 ± 0.12a	6.24 ± 0.24a	37.63 ± 0.18c	0.40 ± 0.05b	1.83 ± 0.08c	17.61 ± 0.48b	10.4 ± 0.1
	20–40	5.33 ± 0.03c	6.43 ± 0.05a	29.70 ± 0.26c	0.34 ± 0.06b	0.49 ± 0.02a	16.92 ± 0.64b	35.2 ± 1.8
	40–60	6.82 ± 0.04a	6.61 ± 0.26a	34.70 ± 0.08c	0.31 ± 0.12b	0.83 ± 0.18c	17.24 ± 0.09b	24.3 ± 0.1
6	0–20	5.82 ± 0.14b	7.89 ± 0.35c	49.33 ± 0.41c	0.27 ± 0.02a	1.66 ± 0.04c	18.30 ± 0.34c	17.2 ± 0.8
	20–40	4.48 ± 0.03c	5.74 ± 0.18b	32.76 ± 2.15c	0.21 ± 0.07a	1.37 ± 0.07c	15.42 ± 0.87b	13.9 ± 2.1
	40–60	5.42 ± 0.07c	4.45 ± 0.31a	37.04 ± 0.57c	0.17 ± 0.03c	0.82 ± 0.06b	15.18 ± 1.27b	26.2 ± 0.9

Note: Different lowercase letters (a, b, and c) indicate that difference in chemical property of different ages is significant.

Table 2
A comparison of average growth status of the planted *S. apetala* and local species *K. candel* in Longhai County, Fujian (following Chen et al., 2003) and Chenghai City, Guangdong (following Chen et al., 2003), and Qiongshan City, Hainan (this study).

Location	Species	Age	Survival Rate (%)	Height (m)	Diameter at basal height (cm)	Diameter at breath height (cm)	Crown size (m^2)
Longhai, Fujian	*S. apetala*	3.5	87.5	7.5	10.6	7.4	2.3
	K. candel (Linn.) Druce	10.5	70.0	3.5	3.7	1.7	1.5
Chenghai, Guangdong	*S. apetala*	2.2	70.0	3.6	11.3	5.6	0.3
	K. candel (Linn.) Druce	2.5	61.0	0.8	2.7	–	0.3
Qiongshan, Hainan	*S. apetala*	2.5	91.0	5.4 ± 0.8a	13.2 ± 1.2a	9.6 ± 0.6b	0.8 ± 0.2a
	K. candel (Linn.) Druce	2.5	72.0	0.9 ± 0.3a	3.8 ± 0.5a	–	0.5 ± 0.2a

Note: The data without standard deviation at Longhai and Chenghai were from Chen et al. (2003), and the quadrats were about 2 ha in size. Different lowercase letters (a and b) indicate that difference in growth of different species is significant.

reported that the survival rate of planted local native species on the bare muddy beach was lower than 20%, while that for *S. apetala* was about 95% in Leizhou Bay. Fast-growing *S. apetala* could close the canopy in 4 years and accumulate soil up to 50 cm in depth in 10 years, and dramatically increase soil fertility (Ren et al., 2008). Third, as Zan et al.'s (2003) study found out, *S. apetala* could easily adapt to the climate, salinity, tidal flat, and soil characteristics in Shenzhen Bay. Although the extreme low temperature may limit the settlement and growth of the species, the cold resistance of its young generation steadily increases. Consequently, the species spreads further north and improves the environment condition of the beach muddy sites. Local people at the Maipo Mangrove Reserve in Hong Kong (on the other side of the bay to Shenzhen Futian Mangrove Reserve) consistently remove seedlings and seeds of *S. apetala* floating from the sea, since the species might colonize the beach and out-compete the native species (Lin, 2003).

Invasions of the exotic plant *S. apetala* have caused significant changes to the native ecosystems, including alteration of habitat structure, extinction of native species, and altered ecosystem productivity. Liao et al. (2004) observed that *S. apetala* mainly appeared on the muddy beach at the low tidal boundary of *K. candel* or *A. marina* natural communities in Dong Zhaigang, Hainan. Liao et al. (2004) also observed that the introduced *S. apetala* could affect the population development of *A. corniculatum* (Linn.) Blanc and *A. marina* which was the secondary *A. corniculatum* (Linn.) Blanc community. They also found that one individual of *S. apetala* c may out-compete 21 native mangrove individuals). Li et al. (2004) reported that *S. apetala* had strong allelopathic/inhibitory effects on germination and seedling growth of four native mangrove species. The bare beach where *S. apetala* naturally invaded changed in the depth of soft mud (about 25 cm depth sediment deposition during 8 years), salinity, water flow velocity (reduced about 20% during 8 years), water surface elevation (increased about 22 cm during 8 years) and lee (Liao et al., 2004; Ren et al., 2008).

However, available evidence shows and some researchers still believe that *S. apetala* does not have high invasion ability in certain locations. For example, after 10-year observation of *S. apetala* introduction, Pan et al. (2006) reported that the species would not turn to be invasive. Instead, they believed that *S. apetala* would be an ideal species for artificial reforestation only in the

Fig. 2. An example of *S. apetala* invaded into natural mangrove forests in Zhanjiang National Mangrove Nature Reserve.

estuary of Jiulong River in Fujian. Zan et al. (2003) also suggested that *S. apetala* was unlikely to become invasive in Shenzhen Bay.

At present, no invasion of *S. apetala* was observed in the north boundary of the mangrove forests in China. However, invasion into natural forests did occur in Shenzhen, Zhanjiang (Fig. 2) and Dong Zhaigang where the conditions were more favorable to the species. In 2006, the density of individuals with basal diameter >2.5 cm in the natural mangrove sites in Shenzhen, Zhanjiang and Qiongshan sites was 2812 ± 76, 2473 ± 164, and 2899 ± 32 individuals per ha, respectively. Within the three sites invaded by *S. apetala*, the density was 37 ± 12, 51 ± 8, and 111 ± 21 individuals per ha, respectively.

Considering the characteristics of naturalization and invasion of exotic species (Richardson et al., 2000; Nentwig, 2007), all three regions mentioned above might be at risk of invasion when the species approaches. Considering *S. apetala*'s natural temperature range in its original South Asia distribution (Jayatissa et al., 2002), it is also possible for *S. apetala* to invade all of the bay areas south of Xiamen City (Fig. 1). Since this species has potential to become invasive in certain parts of the coastal China, careful monitoring of the development and dispersal of the species is critically needed.

4. Conclusion

China previously imported a non-native and an invasive intertidal wetland species, *Spartina alterniflora* (smooth cordgrass), which represents similar results and problems. *S. alterniflora* is a perennial salt marsh grass and is native to the Atlantic and Gulf coasts of North American. It was introduced to the eastern coasts of China for the purpose of land reclamation in 1970s and 1980s (Li and Zhang, 2008). This species has a number of biological traits which are similar to *S. apetala* (e.g., fast growth, high salt tolerance, great reproductive capacity, and alteration of habitat structure). These traits made it a suitable species for ecological restoration. However, it has undergone rapid expansion and may out-compete native plants, threaten the native ecosystems and coastal aquaculture, and cause declines in native species richness. The Chinese have developed physical, chemical, and biological methods to control this species, but physical control is the only method typically chosen (Li and Zhang, 2008; Wang et al., 2008). In order to prevent the same problem created by smooth cordgrass, we should pay attention to *S. apetala*'s invasion from now on.

S. apetala was initially introduced to China in 1985 as a candidate species for restoration. However, the species is now becoming invasive in some locations, having caused substantial changes to the native ecosystems. As satisfactory native mangrove species for restoration have not been identified because of the high cost, slow growth and low survival, *S. apetala* continues to be used (State Marine Administration of China, 1996; Zheng et al., 1999). Nevertheless, this species may be considered as invasive in some places where it should not be used further but in other locations where it can grow only by plantation and cannot naturally spread aggressively, it helps restoration projects at least during the initial stages.

Determining the success of mangrove restoration has long been challenging and sometimes contentious (Kentula, 2000). Mangrove restoration undergoes three phases including persistent vegetative cover, functional equivalency and ecosystem restoration. The establishment and early development of mangrove seedlings in the intertidal zone has eco-physiological and growth constraints (Lewis, 2000). Various mangrove restoration pathways in a functional framework dependent on site conditions and emphasizes community and ecosystem level. In above three stages, the hydrological process restoration is most important. The hydrology such as flooding depth, duration and frequency are critical factors in the survival of both mangrove seedings and mature trees (Mitsch et al., 2002; Lewis, 2005).

Our overall assessments of *S. apetala* in China suggest that, to maintain the integrity of natural mangrove communities, *S. apetala* should not be further introduced to natural or semi-natural mangrove reserves or where native species can perform reasonably well. In extremely degraded tidal areas where native species cannot recover even with plantation, however, *S. apetala* could be used as pioneer or nurse species with caution. It may be feasible to use introduced species that already exist in the region for initial restoration but close monitoring of the species is needed. Comprehensive ecological, economic, and social effects should be taken into account in developing restoration and management strategies. In many situations, more attention should be paid to the ecological benefits of mangrove reforestation, instead of direct economic cost (Ellison, 2000; Ren et al., 2004).

There is a practical dilemma facing ecologists and managers. Currently, the mangrove forests and areas suitable for mangrove restoration suffer from severe environmental problems such as water pollution and human disturbance associated with rapid economic growth. The key issue for mangrove restoration is to select the sites carefully. Flexible and adaptive management strategies should be in place as early as possible in conservation and restoration through selecting more suitable habitats and native species for plantation. Reserving the suitable mud flat for mangrove restoration, improving water quality, and maintaining the integrity of hydrological processes are critical components in mangrove ecology and management in China (Teas, 1983; Wu and Richard, 2002; Biswas et al., 2007; Dahadough-Guebas et al., 2005; Guo, 2005).

Our study suggests that current widespread planting of *S. apetala* in China needs to be reconsidered. We should only restore mangroves at the historical mangrove distribution areas, using only native species. Exotic species should not be planted even in area currently devoid of mangroves. When restoring mangroves, we should get the hydrology right first. Of course, the final course of action will depend on government will, investment, and local culture (Stevenson et al., 1999; Samson and Rollon, 2008).

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