

Conservation of Chinese Plant Diversity: An Overview

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1. Introduction

1.1 The significance of plant diversity in China

China is one of the richest countries in plant diversity, ranking third in the world (after Brazil and Colombia) in number of species, and one of the world's 17 'mega-diversity' countries (Mittermeier et al., 1997). The estimated number of vascular plant species may approach 33,000, with 30,000 angiosperms, 250 gymnosperms, and 2,600 pteridophytes (up to 12%, 27% and 20% of world's total, respectively). Furthermore, approximately 2,200 bryophytes can be found in China (López-Pujol et al., 2006; Table 1). There are more than 3,000 genera and *ca.* 350 families of vascular plants (Li et al., 2003; MacKinnon & Wang, 2008). Nevertheless, these figures refer to mainland China and do not include either Taiwan or Hong Kong. Taiwan alone harbors more than 4,000 vascular plants (over 3,300 angiosperms, about 30 gymnosperms, and about 600 pteridophytes; Hsieh, 2002). With an area of only about 1,100 km², Hong Kong still retains a very rich plant diversity, with more than 2,100 higher plants (Wu, 2002).

China encompasses enormous diversity in geographical, climatological and topographical features, in addition to a complex and ancient geological history (with most of its lands formed as early as the end of the Mesozoic era; Wang, 1985). The country spans five major climatic zones (cold-temperate, temperate, warm-temperate, subtropical, and tropical), and is home to the highest mountain range on Earth (the Himalayas) and perhaps the most rugged one (the Hengduan Mountains), vast plateaux such as the Tibetan (*Qinghai-Xizang*) Plateau, deserts (e.g. Taklamakan), deep depressions (e.g. Turpan Depression), large flat areas (e.g. Sichuan Basin and North China Plains), and some of Asia's largest rivers, including the Mekong (*Lancang*), Brahmaputra (*Yarlung Zangbo*), Yangtze (*Changjiang*), and Yellow (*Huanghe*) rivers. All of these features contribute to the enormous diversity of ecosystems,

including almost all types of forests, grasslands, shrublands, deserts (which cover more than 25% of the Chinese territory), marshes, savannas, tundras, or alpine meadows (Hou, 1983; Enright & Cao, 2010). Thus, it is not surprising that 19 of the 238 WWF global priority ecoregions (Olson & Dinerstein, 2002) are located totally or partially within China.

Diversity of cultivated plants is equally rich. China is one of the eight original centers of crop plants in the world (Vavilov, 1951), with more than 200 originating and differentiating there (Gu, 1998), resulting from more than 7,000 years of agricultural activities. Notable examples include rice (*Oryza sativa*), which consists of some 50,000 cultivars and three wild relatives (*O. granulata*, *O. officinalis*, and *O. rufipogon*) and soybean (*Glycine max*), embracing about 20,000 cultivars (SEPA, 1998; MacKinnon & Wang, 2008). It is also estimated that as many as 2,200 ornamental species originated in China (SEPA, 1998), and about 15,000 vascular plant species native to China have been cultivated around the world (Zhao & Zhang, 2003). Other economic plants as well show significant figures in China: over 1,300 taxa of food plants have been recorded (Hu, 2005), as well as up to 1,200 fiber plants and about 2,000 species of timber plants (CSPCEC, 2008); in addition, nearly 11,000 species of medicinal plants have been in use since the Palaeolithic period (Hamilton, 2004; CSPCEC, 2008).

2. Patterns of species richness and endemism: Evolutionary issues

China possesses the richest flora of the North Temperate Zone (Axelrod et al., 1996; López-Pujol et al., 2006, 2011); however, more relevant are the rates of endemism: fifty to sixty percent of the total number of species (i.e. 15,000 to 18,000) might be endemic to China (SEPA, 1998; CSPCEC, 2008). This wealth of species diversity and endemism is attributable to a series of factors largely related to the biogeography, tectonics and geological history of the country, including: (i) a complex and extended geological history, with many tectonic events, (ii) a large proportion of the area of China within tropical and subtropical latitudes, (iii) the wide and persistent connection of China to tropical regions of Southeast Asia as well as with other regions, (iv) an unbroken connectivity between tropical, subtropical, temperate, and boreal forests, (v) a highly rugged and dissected topography (especially in southern China), and (vi) perhaps the most significant, reduced extinction rates during the late Cenozoic global cooling (e.g. Tiffney, 1985; Latham & Ricklefs, 1993; Axelrod et al., 1996; Guo et al., 1998; Guo, 1999; Qian & Ricklefs, 1999; Qian, 2001, 2002; Ying, 2001; López-Pujol et al., 2006, 2011; Qian et al., 2006; Wu et al., 2007; Li, 2008).

Taxa	Species in China (SC)	Species in the world (SW)	SC/SW (%)
Lichen	2,000	10,000	20.0
Fungi	8,000	70,000	11.4
Algae	8,979 - 12,500	40,000	22.4 - 31.2
Bryophyta	2,200	15,000	14.7
Pteridophyta	2,300 - 2,600	13,025	17.6 - 20.0
Gymnospermae	192 - 270	980	19.6 - 27.5
Angiospermae	25,000 - 30,000	258,650	9.7 - 11.6

Table 1. Richness of Chinese flora (data are only for mainland China). All the figures have been taken from López-Pujol et al. (2006); original sources can be found there

One of the main features of Chinese vascular flora is its ancient origin; the modern flora of China is still showing a strong relictual character (Thorne, 1999; Qian & Ricklefs, 1999; Qian, 2001; López-Pujol & Ren, 2010; López-Pujol et al., 2011). This is directly linked to the geologic history of China's landmass, with some of the events influencing the flora composition tracing back to the early Mesozoic. Taking into account that the origin of vascular plants is often situated in Gondwana (Graham, 1993; Steemans et al., 2009), the collision of Gondwanan terranes (including southern China, Indochina and Sibumasu) with what is now northern China in the Triassic-early Jurassic period, and the further impact of the Indian subcontinent during the late Paleocene/early Eocene (Metcalfe, 1988; Şengör, 1997) would have extensively supplied China with pteridophytes and gymnosperms. This may help to explain the disproportionate number of lineages in China belonging to the two most ancient groups of vascular plants (Qian & Ricklefs, 1999; Qian, 2001).

In addition, China could have been close to the center of origin of flowering plants, which for some authors was located somewhere in SE Asia (Thorne, 1963; Takhtajan, 1969, 1987; Smith, 1973). The continuous land connection with southern Asian tropical areas (dating back at least 200 million years; Hallam, 1994) would have enabled a straightforward transferring of the most primitive angiosperm lineages to China. More recent studies suggest that angiosperms might have originated instead in Gondwana, but still in palaeotropical latitudes (Morley, 2000; Barrett & Willis, 2001). With the finding of the early angiosperm *Archaeofructus* fossils (Fig. 1) in Liaoning Province (north-east China) in addition to other ancient angiosperms in neighboring areas (e.g. Sun et al., 2011), Sun et al. (2008) are postulating, however, for an 'Eastern Asian origin of angiosperms'. In any case, there is general agreement that China constituted an important area for the early diversification of angiosperms (Qian & Ricklefs, 1999; Barrett & Willis, 2001; Qian, 2001; López-Pujol et al., 2006; Sun et al., 2008, 2011).

As explained below, the collision of first the Gondwanan terranes and the further impact of the Indian subcontinent are responsible for China having inherited elements of Gondwanan floras in addition to its own Laurasian ones. However, despite China's (and East Asia's) initial separation from western Eurasia by the Turgai Sea, a land bridge linking eastern and western Eurasia ensured their connection since *ca.* 35 million years ago (Hallam, 1994; Qian, 2001), permitting a floristic exchange with Central Asia (and subsequently with the Mediterranean basin and even North Africa; Qian & Ricklefs, 1999), thus providing China with Tethyan floristic lineages, in addition to other Laurasian and Gondwanan elements (Wu & Wu, 1996; Wu, 1998; Qian et al., 2006). Moreover, the connection with North America via the Bering Sea land bridge allowed the migration of numerous species from this area (Tiffney & Manchester, 2001; Qian et al., 2006).

The most important force shaping the contemporary floristic richness of China is, nevertheless, the climatic change that occurred during the Neogene period. During the late Tertiary and the whole Quaternary, that is, from the middle Miocene climatic optimum (*ca.* 15 Ma; Zachos et al., 2001) onwards, a progressive global cooling produced numerous plant extinctions in most of the Northern Hemisphere, especially Europe and North America (Tiffney, 1985; Sauer, 1988; Axelrod et al., 1996; Jackson & Weng, 1999), and these were centered on the relictual, thermophilic elements of the 'boreotropical flora' (Tiffney, 1985; Latham & Ricklefs, 1993). In contrast, the limited ice coverage in East Asia during the Quaternary glaciations (significantly milder than in Europe and North America, and limited only to northern areas north of 60° N; Qian & Ricklefs 1999), coupled with the numerous

mountain ranges in the southern section of China (which would have provided long-term stable habitats through local buffering of the extreme climatic oscillations) and the lack of barriers for southwards migration enabled the country to serve as a refuge for many Tertiary plant lineages, ensuring their survival to the present day (Axelrod et al., 1996; Qian & Ricklefs, 2000; López-Pujol et al., 2011). The presence of numerous and extensive refugia is likely to be the cause of the overrepresentation of relict elements in the modern flora of China and the much larger overall taxonomical richness of the Asian country with respect to other territories from the Northern Hemisphere (Europe and the United States, with comparable areas, harbor a much poorer flora: 11,500 and 18,000 species, respectively; Axelrod et al., 1996).



Fig. 1. Fossil of *Archaeoфраuctus liaoningensis*, one of the earliest known genera of flowering plants, exposed in the Beijing Museum of Natural History (photo by Jordi López-Pujol)

The role of the Neogene plant refugia as places for plant survival (i.e. 'plant museums') has been long recognized (e.g. Ying & Zhang, 1984; Wang, 1989; Wu & Wu, 1996; Li, 2008), and this is reflected in the exceptional richness of several phylogenetically primitive vascular plant groups in China; e.g., pteridophytes, gymnosperms, magnolids, and ranunculids (Qian & Ricklefs, 1999; Qian, 2001). China has a significantly higher number of ancient endemic genera and families than the United States (Qian, 2001), and most of the endemic genera of spermatophytes of China (about 240) are of ancient origin (Ying & Zhang, 1984; Ying et al., 1993; Wu et al., 2007). Moreover, a more recent study (López-Pujol et al., 2011) has revealed that the relict (pre-Quaternary) component of the modern endemic seed flora at the infrageneric (species and subspecies) level is considerable (around 40%; the remaining 60% are taxa of Pleistocene origin). For the Mediterranean endemic flora, the weight of the relict taxa is much lower (ca. 25%; Thompson, 2005). Noteworthy is the occurrence,

especially in the central and southern regions of China, of numerous palaeoendemic lineages whose distribution was much wider and today is restricted to a few, often disjunct refugia in East Asia. The fossil record shows that some of these lineages once existed in Europe or North America, but were extirpated from there during the Neogene as a consequence of climate deterioration (Latham & Ricklefs, 1993; Axelrod et al., 1996; Manchester, 1999; Manchester et al., 2009). Representative examples within the gymnosperms include the monotypic and/or oligotypic genera *Amentotaxus*, *Cathaya* (Fig. 2), *Cunninghamia*, *Ginkgo* (which is the unique representative of the monotypic family Ginkgoaceae but also of the entire order Ginkgoales; Fig. 2), *Glyptostrobus*, *Keteleeria*, *Metasequoia* (Fig. 2), *Pseudolarix*, *Pseudotaxus*, and *Taiwania*. From the angiosperms, several examples merit citation here: *Craigia*, *Cyclocarya*, *Davidia*, *Dipelta*, *Diplopanax*, *Emmenopterys*, *Eucommia* (the only representative of the Eucommiaceae), *Fortunearia*, *Pteroceltis*, *Sargentodoxa*, *Tapiscia*, *Tetracentron*, and *Trochodendron*. Some 'living fossils' (taxa which have remained superficially unchanged for millions of years) such as *Metasequoia glyptostroboides* or *Glyptostrobus pensilis* constitute the most conspicuous examples of these relict lineages, generally remnants of the boreotropical flora that once spanned most of the Northern Hemisphere (Kubitzki & Krutzsch, 1996).



Fig. 2. Some 'living fossils' of China. Left, type tree of *Metasequoia glyptostroboides* at Moudao Town (Hubei Province) (photo by Qin Leng); top right, leaves and cones of a wild individual of *Cathaya argyrophylla* (photo by Zhao-Shan Wang); bottom right, yellowish leaves of a planted individual of *Ginkgo biloba* in Beijing (photo by Jordi López-Pujol)

Most of these plant refugia generally also served as areas for plant evolution and speciation ('plant cradles') in addition to being places for their persistence during the Neogene cooling (Axelrod et al., 1996; López-Pujol et al., 2011). Interestingly, the main centres of evolution were located in the southwestern part of China, that is, the eastern fringe of the Tibetan Plateau. The Hengduan Mountains (Fig. 3) and the neighboring ranges (Daxue Mountains, Min Mountains) constitute the 'evolutionary front' of China, because they are home to numerous neoendemisms (Ying et al., 1993; Wu & Wu, 1996; Li & Li, 1997). Recent orogenic processes in western China (the major uplift of the Tibetan Plateau took place during the Pliocene and the Pleistocene, and it is still active; Li & Fang, 1999; Zhang D. et al., 2000; Zheng et al., 2000) created a vast array of new habitats across wide altitudinal ranges (up to 5,000 m in the Hengduan Mountains; Fig. 3), which stimulated plant differentiation and speciation (Chapman & Wang, 2002; Qian, 2002), including adaptative radiations (Liu & Tian, 2007). This can be the case of *Nannoglottis*, a relic genus which probably arrived at the Tibetan Plateau not later than the Oligocene and suffered a rapid re-diversification during the Pliocene-Pleistocene (Liu et al., 2002). In contrast, the relative tectonic stability in central, south-central and southeastern China seems to have favoured the survival of relict lineages (López-Pujol et al., 2011).

Since refugia are areas that offered many opportunities for persistence and speciation, these stand out as harboring high rates of endemism as well as overall species richness. Centres of species richness and centres of endemism are, thus, generally coincident (Ying, 2001; Tang et al., 2006; Xu et al., 2008) and almost exclusively located in the mountainous regions of central and southern China, at latitudes below 35° N, including the two main Chinese islands, Hainan and Taiwan (Fig. 4). One of the most significant areas for plant diversity are the Hengduan Mountains (Fig. 3), which span north-west Yunnan, western Sichuan and south-east Tibet, and which are also considered one of the main world's biodiversity hotspots (with a total flora of about 12,000 species, of which ca. 3,500 are endemic; Myers et al., 2000). It is widely acknowledged that its extremely varied topography supporting a wide array of vegetation zones (from subtropical low mountain evergreen rainforests in the deep valleys to alpine communities on the summits), have greatly contributed both to the appearance of many new species and the conservation (although in a lower scale) of relict elements (Chapman & Wang, 2002; Qian, 2002). Some examples of this 'paradox' are the genera *Primula* and *Rhododendron*, present in the region before the Himalayan uplift and which became highly diversified through allopatry with the creation of numerous new habitats (Chapman & Wang 2002); for example, more than one-quarter of the world's *Rhododendron* species (276 of ca. one thousand) and ca. 30% of *Primula* (143 out of 500) can be found in these mountains (Zhang D.-C. et al., 2009).

Central China Mountains is also one of the richest areas in plant diversity of China, with about 6,400 plant species and more than 1,500 endemics (Ying, 2001). This region, in contrast to the Hengduan Mountains, contains more relict than recently-evolved taxa (López-Pujol et al., 2011). One of the most interesting areas within this hotspot is the so-called 'metasequoia area', a region of about 800 km² in the juncture of Hubei and Chongqing, where there are still natural populations of *Metasequoia glyptostroboides* (Hu, 1980). In this very small land extension, at least 550 species of vascular plants occur; the most interesting, however, is that we can find many 'living fossils' here in addition to *M. glyptostroboides*, most of them belonging to monotypic or oligotypic genera, e.g. *Cunninghamia lanceolata*, *Eucommia ulmoides*, *Keteleeria davidiana*, *Pseudolarix amabilis*, *Taiwania cryptomerioides*, *Tapiscia sinensis*, and *Tetracentron sinense* (Hu, 1980).

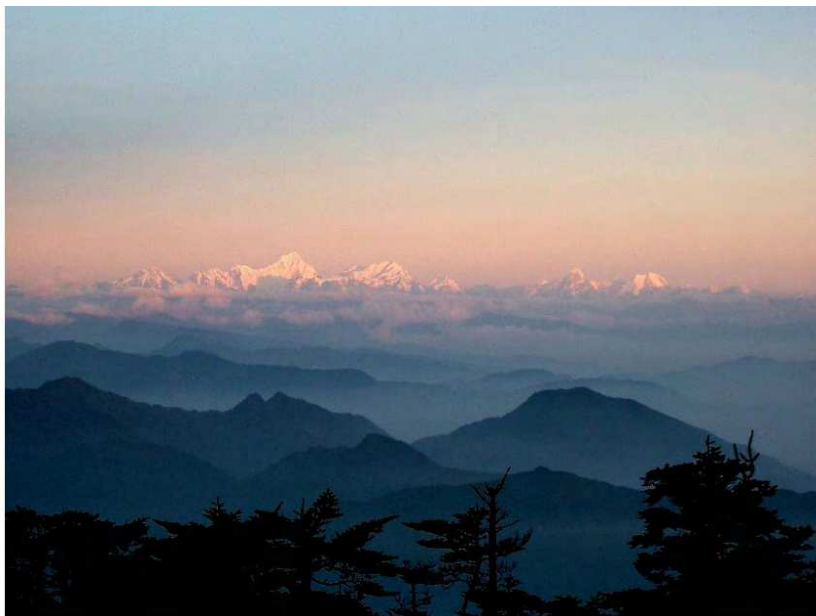


Fig. 3. A view of the Hengduan Mountains. The highest peak in the picture corresponds to Mount Gongga (7,556 m a.s.l.). Some deep gorges (of about only 1,000 a.s.l.) can be found just a few km away (photo by Jordi López-Pujol)

3. Endangered species: Current status

A second trait of Chinese flora is its high level of endangerment. Most estimates show that 3,000-5,000 species could be threatened with extinction (Fu, 1992; Wang, 1992; Gu, 1998; Zhang, 2007; CSPCEC, 2008), i.e. up to 20% of the total flora. However, and according to more recent studies (Wang & Xie, 2004; Xie & Wang, 2007), this figure could be even higher: from the ca. 4,200 angiosperm taxa (i.e. just 14% of the total number of angiosperms in China) assessed in the first phase of the elaboration of the *China Species Red List* (Wang & Xie, 2004), over 3,600 (87%) were regarded as threatened following the 2001 IUCN criteria, and up to 651 were assigned to the highest endangerment category (CR, 'Critically Endangered'), that is, facing an imminent risk of extinction. These figures, nevertheless, should be taken with caution as the red list was biased toward rare and endangered species. At the beginning of the 2000s it was estimated that, since the 1950s at least 200 plant species had become extinct (Zhang P. et al., 2000); some conspicuous examples of these are *Otophora unilocularis* (not seen since 1935) and *Rhododendron kanehirai*, whose natural populations were flooded by a dam in Taiwan although it is extensively cultivated (IUCN, 2010). However, some plant taxa have also been lost from the wild during the last decade (2000-2009), such as *Betula halophila*, *Cystoathyrium chinense* or *Plantago fengdouensis* (Zhang & Ma, 2008; López-Pujol & Zhang, 2009). In addition to these losses, many species remain on the brink of extinction. An appreciable number of taxa are in an extreme situation of risk with population sizes often consisting of fewer than 100 individuals: examples include the gymnosperms *Cupressus chengiana* var. *jiangensis* and *Abies beshanzuensis* var. *beshanzuensis*

(from which only one and three individuals are remaining, respectively), and the angiosperms *Carpinus putoensis*, *Gleditsia japonica* var. *velutina*, and *Acer yangbiense* (with just one, two and four individuals remaining in the wild, respectively); these taxa are undoubtedly among the most endangered plants on the Earth (López-Pujol & Zhang, 2009). Despite the fact that many species have been severely threatened by extinction for many decades, it was not until 1984 that the *National List of Rare and Endangered Plant Species* was issued, encompassing 388 species [8 listed as 'first grade' nationally protected (NPC-1), 159 species as 'second grade', and 221 species as 'third grade']. In 1992, the *China Plant Red Data Book* (Fu, 1992) was published, including all 388 of the endangered taxa warranting protection. Of these, 121 were listed as 'endangered', 110 as 'rare', and 157 as 'vulnerable'. Two additional volumes were originally planned (López-Pujol et al., 2006), but they are still awaiting completion. Meanwhile, based on the request of the *Regulations on Wild Plants Protection* (implemented in 1997), China promulgated the first batch of the *Catalogue of the National Protected Key Wild Plants* in 1999, which included about 300 plant taxa, distributed into two protection categories (Zhang & Ma, 2008). The second batch is currently in preparation and will mean that up to 1,900 plant taxa will be strictly protected in mainland China (CSPCEC, 2008).

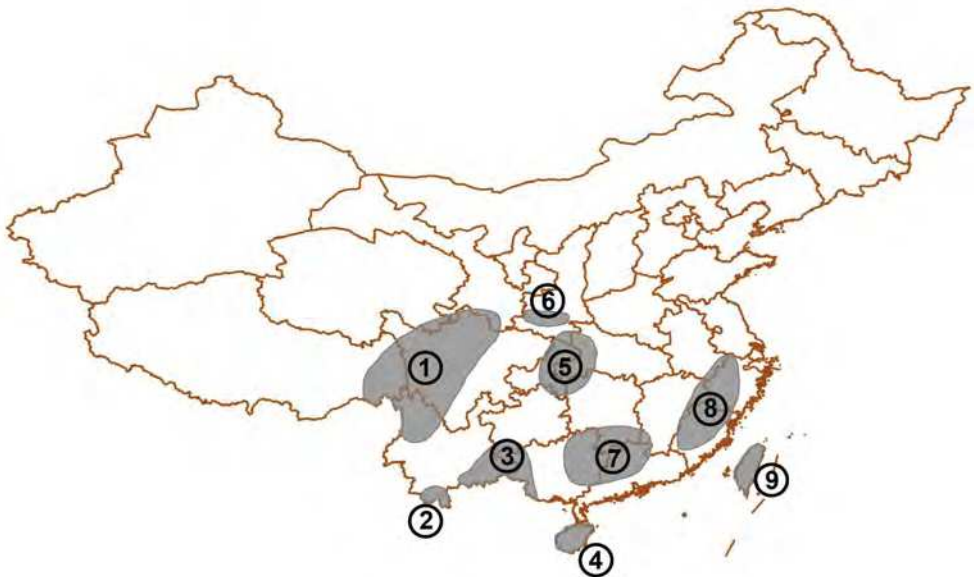


Fig. 4. Approximate location of Chinese plant diversity hotspots. The code numbers for the plant diversity hotspots correspond to those in Table 2 (see below).

In addition to these protection lists, China has also progressed in assessing plant species using the IUCN Red List Categories and Criteria. By the end of 2010 only about 740 taxa (species and subspecies) from mainland China appeared in the *IUCN Red List of Threatened Species* (<http://www.iucnredlist.org/>), however the publication of the first volume of *China Species Red List* (Wang & Xie, 2004) has given rise to the assessment of 4,408 plants using the 2001 IUCN criteria (IUCN, 2001). Moreover, several experts are working actively in

assessing new species which are added regularly in the website of the Wildlife Conservation Society (<http://www.chinabiodiversity.com/>), and an updated print version of the red list is expected to be released around 2012 (CSPCEC, 2008).

Endangered plant species tend to be concentrated in the southern part of the country (e.g. Tang et al., 2006; Zhang & Ma, 2008), thus showing high congruence with both centres of species richness and centres of endemism (Xu et al., 2008). Thus, 'biodiversity hotspots' in its broadest sense (as centres of species richness, endemism and threatened species) are entirely located in the central and southern mountainous regions of China (Table 2 and Fig. 4), which are moderately populated (much less compared to the North China Plains) and where the agricultural exploitation has been relatively limited (e.g. Huang et al., 2010) due to their low suitability (too steep). However, other practices such as extensive logging and overgrazing have significantly damaged the natural ecosystems of these areas (Cl, 2007; Morell, 2008).

Hotspot	Species richness	Endemism	Threatened plants
Hengduan Mountains (1)	+++	+++	++
Xishuangbanna (2)	+++	+	+++
SE Yunnan-SW Guizhou-SW Guangxi (3)	++	+++	++
Hainan Island (4)	++	+	++
Central China Mountains (5)	++	+++	++
Qinling Mountains (6)	+	++	+
Nanling Mountains (7)	++	++	++
East China Mountains (8)	++	++	++
Taiwan Island (9)	++	++	+

Table 2. Chinese plant diversity hotspots. Relative occurrences for each type of plant species (+++, high occurrence; ++, intermediate occurrence; +, low occurrence) and for each hotspot have been inferred after taking into account all the relevant related literature (e.g. Ying et al., 1993; Wang & Zhang, 1994; Ying, 2001; Li et al., 2003; Tang et al., 2006; Zhang & Ma, 2008; Li et al., 2009; López-Pujol et al., 2011). The code numbers for the different hotspots correspond to those in Fig. 4

4. Threats to Chinese plant biodiversity

The main threats to Chinese plant diversity, both directly and indirectly, can be described as below.

4.1 Habitat destruction

Destruction and/or fragmentation of natural habitats (Fig. 5) are the principal causes of species extinction throughout the world (Crooks & Sanjayan, 2006). China has experienced since the imperial times but particularly from 1950s onward, a tremendous loss of natural habitats, mainly due to the over-logging of forests, as well as from the conversion of forests, grasslands and wetlands into croplands (Liu & Diamond, 2005). Some areas have suffered a

dramatic reduction in forest coverage; e.g., the rainforest on Hainan Island covered 25.5% of the total area in the early 1950s; thirty years later, this coverage had decreased to below 9%, and these were mainly replaced by rubber plantations (Francisco-Ortega et al., 2010); in Xishuangbanna, the forest cover diminished, from over 60% to less than 30% during the same period (Zhang & Cao, 1995). It is thought that up to 67 million hectares of forests were logged during the 50s, 60s and 70s (Cai, 1990). Throughout China, however, forest cover has progressively increased over the most recent decades due to large afforestation and/or reforestation campaigns (Wang et al., 2008). In 1962, forest cover measured around 9.0% of the total land, while in 1981 it had increased to 12.0%, and then to over 20% by the end of 2009 (MEP, 2010). Nevertheless, nearly all of these new plantations, which replaced logged natural forests, have been mono-specific, and have sometimes consisted of exotic species, greatly diminishing the biodiversity value of the original forestlands (Xu & Wilkes, 2004; Liu & Diamond, 2005). The progressing expansion of the desert in the northwestern provinces has been caused primarily by the massive degradation of grasslands (due to overgrazing and land reclamation for agriculture); 90% of China's total grasslands are degraded to some degree (Liu & Diamond, 2005). At the end of 20th century, China exhibited an extremely fast desertification rate (with an annual increase of almost 3,500 km²), although this trend seems to have been mitigated during the last decade due to re-vegetation programs (Wang et al., 2007; but see Wang et al., 2010). Loss of wetlands (which may reach 50% of China's total; Yu, 2010), mainly through conversion to cropland, has also contributed to the rampant erosion processes experienced in China since the 1950s: severe soil erosion affects today about 20% of the country land area (Liu & Diamond, 2005; Li, 2010). Deforestation and soil erosion definitely contributed to the devastating floods in the Yangtze River basin in 1998 (Liu, 2010), and the increasing incidence of dust storms in northern China (He, 2009) is also one of the catastrophic consequences of land vegetation cover degradation (Fig. 6).

4.2 Environmental contamination and global climate change

Until recent times, the lack of any ecological consciousness by China's leaders and/or the poverty of the country (which made the government focus on development policies) have caused air, water and land to be extremely polluted. This has led not only to the degradation of natural habitats and the loss of many species but has also brought huge economic losses and numerous effects on public health (e.g. Liu & Diamond, 2005; Fu et al., 2007; Zhang J. et al., 2010). For instance, severe defoliation (up to 50%) and increased mortality rates have been detected in forests of native pines due to acid rain (Larsen et al., 2006), and it is estimated that 2.4 million premature deaths are produced in China every year due to these environmental risks (Zhang J. et al., 2010). Despite the fact that several sources are claiming a substantial amelioration in pollution levels in recent years (Xu H. et al., 2009, 2010; MFA, 2010), concerns are still unquestionable (e.g. Vennemo et al., 2009). The most problematic issue continues to be the air pollution stemming from the extensive use of coal: in 2009 it was still by far the main energy resource in China, accounting for 69.6% of the total energy consumption; renewable sources of energy continue to be very limited (below 10%; MFA, 2010). While the growth in CO₂ emissions has proven to be significantly less than was previously projected, they still remain very significant (China became the largest emitter of CO₂ since 2006, accounting for *ca.* 20% world's total; World Bank, 2010a). Other air pollutants, whose control has been increased by the authorities, have shown a declining trend, such as the SO₂ emissions and soot and industrial dust (Xu H. et al., 2009, 2010); however, China is still a very large emitter for almost all the major air pollutants

(World Bank, 2010a), and acid rain is still affecting more than half of the Chinese cities (MEP, 2010). Regarding waters, despite the verifiable improvements produced by the important governmental efforts (e.g. Xu H. et al., 2009; MEP, 2009) these remain highly polluted: a large portion of lakes and major rivers are highly contaminated (more than 40% and over 75%, respectively; MEP, 2010; Zhang J. et al., 2010).



Fig. 5. *Sinomanglietia glauca* (= *Manglietia decidua*) is one of the most endangered species of Magnoliaceae in China, with only two isolated localities (separated by ca. 450 km), and it is protected by the *Catalogue of the National Protected Key Wild Plants*. The observed habitat fragmentation (left) may account for the high differentiation and low genetic diversity detected for the species (Zhang Z.-R. et al., 2009) (photos by Zhi-Yong Zhang)

The global climate change is also producing severe impacts on plant diversity, which have already been recognized by Chinese authorities (NDRC, 2007). In addition to producing a larger incidence of natural disasters (such as floods, forest fires, landslides, storms or droughts which have direct effects on biodiversity; Liu & Diamond, 2005; Yu, 2010), the melting of glaciers is of great concern (according to the latest estimations, the total surface area of glaciers of the Tibetan Plateau has decreased by at least 17% during the last 30 years; Qiu, 2010), since its potential threat to plant diversity and distribution are very significant (e.g. Xu J. et al., 2009).



Fig. 6. Comparison between a clear (left) and a sand storm (right) day in Beijing. Pictures correspond to the campus of the Research Center for Eco-Environmental Sciences (photos by Hua-Feng Wang)

4.3 Over-exploitation of species for human use

The over-exploitation of species of economic interest may seriously threaten their survival. For example, though protected by law, many plants used in Traditional Chinese Medicine (TCM) and other traditional medicines (e.g. Tibetan) are over-collected. Among the 426 herbal, unprepared, or stir-baked drugs listed in the 1995 edition of the *Pharmacopoeia of the People's Republic of China*, 28 are included in the *China Plant Red Data Book* because of their threatened status, and 49 additional plants listed in the *Red Book* are extensively used in TCM (Peng & Xu, 1997). A clear case of over-harvesting is that of the caterpillar fungus (*Cordyceps sinensis*), profusely used both in Chinese and Tibetan traditional medicines (Winkler, 2008); other examples of valuable medicinal plants which are dwindling in wild habitats due to over-collecting include *Aquilaria sinensis*, *Eucommia ulmoides*, and *Gastrodia elata*. Moreover, despite widespread cultivation in China there are some species whose wild populations remain severely depleted, both in number and size (e.g., *Ginkgo biloba*, *Houpoëa officinalis* [= *Magnolia officinalis*] and *Juglans regia*, the latter with perhaps less than 1000 individuals; Chapman & Wang, 2002). In addition to medicinal plants, other non-timber forest products have been subjected to over-collection as well, including edible mushrooms (e.g., *Tricholoma matsutake*) and some orchids (e.g. *Paphiopedilum*) and cycads for their horticultural values (Fig. 7). Fuel wood collection and timber procurement have also severely affected many forest species; some species have been pushed to the brink of extinction such as *Acer yangbiense*, *Picea neoveitchii*, and *Pinus squamata* (López-Pujol & Zhang, 2009). Until the logging ban of 1998, the growing timber harvests since the 1950s had resulted in the loss of extensive forest areas: in Sichuan, it is thought that ca. 40% of natural forests were cut (Morell, 2008). Nevertheless, and despite the ban, illegal commercial logging still occurs on a small scale (especially in SW China) as a source for heating and construction timber (Klok & Zhang, 2008; Morell, 2008).



Fig. 7. Orchids are usually over collected for their horticultural value. In China, over 1,200 taxa of Orchidaceae are included within *China Species Red List* (Wang & Xie, 2004). *Dendrobium devonianum* (left) and *Vanda cristata* (right) are included within them (photos by Hua-Feng Wang)

4.4 Introduction of exotic species

China has a long history of introducing alien species for their potential economic values or other supposed benefits (medicinal, ornamental, soil improvement, erosion control, landscaping, etc). The first species introductions took place more than 2,000 years ago, and some examples comprise *Carthamus tinctorius*, *Medicago sativa*, *Punica granatum*, or *Vitis*

vinifera, which were brought through the Silk Road; Xie et al., 2001). However, its historical isolation has meant that large-scale introductions were relatively very few until the 1970s, despite China's particular vulnerability to invasive species as it harbors a wide range of suitable habitats and environmental conditions. Since the 1980s, when the country adopted market-oriented reforms and opened to international trade, the number of reported invasive alien species has significantly increased (Lin et al., 2007; Weber & Li, 2008). For plants, the number of invasive species has grown from 58 species listed in the 1990s (Ding & Wang, 1998) to 270 recently reported by Weber et al. (2008). This rapid increase, although it may reflect a better degree of knowledge of biological invasions (but see Lin et al., 2007), should be attributed to the unprecedented economic development of China, including an explosive growth of international trade which has enhanced the opportunities for alien species arrival (Ding et al., 2008). In addition, the domestic boom of the industrial and transportation infrastructures, coupled with an unparalleled rate of urbanization and a rampant ecological degradation, have promoted both alien species establishment and their spread within China (Lin et al., 2007; Ding et al., 2008; Weber et al., 2008).

The more than 400 invasive alien species detected in China include at least half of the 100 world's worst invasive alien species compiled by the IUCN (CSPCEC, 2008). These include plants (such as *Eichhornia crassipes*, *Lantana camara*, or *Mikania micrantha*), invertebrates (e.g. *Achatina fulica* and *Bemisia tabaci*) and vertebrates (*Bufo marinus*, *Gambusia affinis*, or *Myocastor coypus*). Some alien species have been introduced to restore vegetation without any previous assessment of their potential damage to local ecosystems, such as *Rhus typhina* (Fig. 8), massively planted throughout Beijing Municipality including the Olympic Park (Wang et al., 2011). Other invasives have been introduced unintentionally, such as the banana moth (*Opogona sacchari*), which entered inadvertently accompanying an ornamental plant (*Dracaena fragrans*) (Ding et al., 2008). A preliminary estimation of the economic losses caused by invasive alien species gave a figure of ca. 14.5 billion USD one decade ago, a considerable figure but much smaller than that for the United States (Weber & Li, 2008) because of the much larger number of invasives in the latter. Thus, more biological invasions can be anticipated in China if the current pace of urbanization and infrastructure development is maintained.

4.5 Lack of effective environmental scope of government policies and ineffective legal protection

The Chinese authorities bear a major responsibility for the tremendous losses in plant diversity in the past and even now. Development policies as well as attitudes towards wildlife in Chinese society have historically been focused on the exploitation of natural resources since imperial times, but this gained special relevance during the second half of the twentieth century (Shapiro, 2001; Liu, 2010; Yu, 2010). For example, the 'Great Leap Forward' (1958-1961), a period in which communities were encouraged to be self-sufficient in steel, involved the cutting down of at least 10% of China's forests to fuel backyard furnaces (Liu, 2010). The development policies that followed were not much better, and during the Cultural Revolution (1966-1976) numerous forests, grasslands and wetlands were transformed into farmlands (McBeath & Leng, 2006). Following the adoption of the country's 'open-door' policies in 1978, although the government began to pass numerous laws for environmental protection and biodiversity conservation, and many protected areas have been set up (see following sections), economic development has taken priority over nature conservation (Liu & Diamond, 2005; Wang et al., 2007). Laws and regulations have

shown many problems regarding their implementation and enforcement, sound monitoring and management systems are generally lacking, whereas financial resources are often insufficient (López-Pujol et al., 2006; Yu, 2010).



Fig. 8. Individuals of *Rhus typhina*, an invasive species of North American origin, grow in mountainous areas of Beijing Municipality (photos by Hua-Feng Wang)

4.6 Economic and population growth

China's economic growth over the last three decades has given rise to the fastest rate of GDP (gross domestic product) growth among the world's major economies (an average of 10% since the late 1970s; Huang & Luo, 2009), leading to strongly increased demands on their own (and their external) natural resources. China is already the world's biggest consumer of all major industrial commodities, although their per capita figures are still low compared to the developed nations (Grumbine, 2007). This means that if recent rates of economic growth are maintained, much more pressure will be put on natural resources. Economic development has involved the construction of numerous industries and power plants (the energy production increased by 105% during the period 1990-2007; World Bank, 2010a) in addition to a huge increase of the communication network (expressway length was less than 300 km in 1989 but it had reached 60,000 km in 2008, whereas more than 20,000 km of new railway were built during the same period; NBSC, 2009). All these infrastructures have contributed to the fragmentation of natural habitats; environmental impacts of some of the world's largest development projects hosted by China (e.g. Three Gorges Reservoir, South-to-North Water Transfer Project, West-East Gas Pipeline, Qinghai-Tibet Railway) cannot still be adequately quantified but would be huge (Liu & Diamond, 2005).

The unprecedented rise in the per capita income and living conditions experienced by Chinese people (the poverty incidence has declined from 84% in 1981 to 16% in 2005; World Bank 2010a) has generated a new industry, national (and international) tourism (national tourism has tripled in just 15 years; NBSC, 2009), with the subsequent construction of new hotels and holiday resorts, often located close to areas of natural and scenic interest or even inside nature reserves (López-Pujol et al., 2006; Yu, 2010). Improving accesses to these natural areas (e.g. new roads, cable cars and even airports—Jiuzhaigou-Huanglong and Shangri-La [Zhongdian] airports) has contributed to habitat degradation, since it has created an enormous influx of visitors. The number of visitors in Jiuzhaigou Nature Reserve has increased from ca. 32,000 in 1984 to over 2 million at present (Hendrickson, 2009); limitation

of mass tourism has been called for by scholars due to its pervasive effects on Jiuzhaigou's biodiversity (e.g. Zhu et al., 2006; Morell, 2008). Another illustrative collateral effect of the emergence of a prosperous middle-class in China is the number of private cars, which has skyrocketed from only 19,000 in 1985 to *ca.* 30 million in 2008 (NBSC, 2009), with the associate severe pollution.

Population growth, which has slowed down since the implementation of the one-child policy, still remains significant. At present, the estimated total Chinese population is about 1,330 million, and is predicted to rise to 1,470 million by 2035, when it would start to decrease (Chen, 2010). Moreover, given the expected economic growth (China may become the world's largest economy by 2020 in terms of GDP; Hawksworth, 2010), the pressure on natural resources will continue. However, the number of households has grown much faster than the total population because of the increase of divorces and the reduction in the number of multigenerational families sharing the same home (Liu, 2010): the average household size has decreased from 3.5 in 1990 to 2.9 persons in 2008 in urban areas; NBSC, 2009). By 2030, 250 million new homes may be needed in China (that is, more than the total number of homes in the Western Hemisphere at the beginning of the present millennium; Liu & Diamond, 2005), which will suppose an additional threat to biodiversity since smaller households are less efficient in resource use.

5. Current conservation measures

China has a long history of nature conservation; the first rules concerning wildlife protection may well predate the Zhou dynasty (1046-256 BC; Edmonds, 1994). Historical conservation of the lands surrounding Buddhist and Taoist temples, as well as their 'sacred mountains' by monks has preserved these areas intact until the present day. Moreover, there is a clear link between biodiversity conservation and ethnic minorities in China. The conservation of Holy Places, Holy Mountains, and Holy Trees by several ethnic minorities is well known (Yang et al., 2004; Xu et al., 2005), e.g. the 'spirit mountains' in Xishuangbanna (Yunnan) have historically been protected by local communities of Dai nationality (Xu et al., 2005). However, 'modern' nature conservation began late in China, and can be divided into *in situ* and *ex situ* measures, as described below.

5.1 *In situ* conservation

The development of modern protected areas (PAs) in mainland China can be divided into four main stages: 1956-1965, the initiation; 1966-1974, stagnation and devastation; 1975-1979, restoration; and 1980-present, a period of rapid growth (Li et al., 2003). China's first nature reserve was founded in 1956 at Dinghu Mountain in Guangdong Province. Nine years later, up to 19 nature reserves had been established, encompassing an area of about 0.7% of the total Chinese land surface (Table 3). By the end of 2008, there were a total of 2,538 nature reserves (Taiwan, Hong Kong and Macao not included), covering a total area of 1,489,430 km² (MEP, 2009), i.e., more than 15% of the nation's surface (Table 3), exceeding the average for developing economies (12.7%; World Bank, 2010a). Most have been established in the last 25 years, comprising the principal governmental measure for protecting China's biodiversity.

Besides nature reserves (the most comprehensive category of PAs), there are other types of protected sites which should be regarded as PAs according to the IUCN definitions (MacKinnon & Xie, 2008) and also recognized by Chinese law, such as forest parks, scenic spots, geological parks, wetland parks, and agricultural reserves, among others. Adding all

these types of protected spaces to nature reserves, the total number of PAs in China is well above 5,000, which cover over 18% of the country's land area (MacKinnon & Xie, 2008). According to the CSPCEC (2008), the present network of PAs is effectively protecting 65% of higher plant communities and 70% of national key wild flora under protection. In addition, a significant part (over 30%) of the area of the 14 Key Biodiversity Land Zones with global conservation significance (which account for nearly one quarter of China's landmass), are also protected by the PAs network (CSPCEC, 2008). Some Chinese PAs have received international recognition: 28 have been designated as Biosphere Reserves under UNESCO's Man and the Biosphere Program; 12 are Natural World Heritage Sites; and 36 have been designated globally significant wetlands under the Ramsar Convention.

Year	Number	Area (km ²)	Percentage of Chinese territory
1956	1	11	0.00
1965	19	6,488	0.07
1978	34	12,650	0.13
1982	119	40,819	0.42
1985	333	193,300	2.01
1987	481	237,495	2.47
1989	573	247,630	2.58
1991	708	560,660	5.84
1993	763	661,840	6.89
1995	799	719,070	7.49
1997	926	769,790	8.02
1999	1,146	845,090	8.80
2000	1,227	982,100	10.23
2001	1,551	1,298,900	13.53
2002	1,757	1,329,500	13.85
2003	1,999	1,397,500	14.56
2004	2,194	1,422,260	14.81
2005	2,349	1,439,500	14.99
2006	2,395	1,455,350	15.16
2007	2,531	1,458,800	15.19
2008	2,538	1,489,430	15.51

Table 3. Evolution of nature reserves in mainland China

In the last 25 years, the government of Taiwan Island has developed a complex network of PAs (up to 81) geared towards conserving its plant and animal biodiversity, which accounts for over 21% of the total land area (MacKinnon & Xie, 2008). About 44% of the land area in Hong Kong SAR is protected, the highest percentage in the Asia Pacific region (ESCAP, 2010). Nevertheless, these protected areas do not adequately cover some habitat types, such as freshwater wetlands and *feng shui* forests near urban areas (Yip et al., 2004). The most emblematic protected area in Hong Kong is the Mai Po Marshes & Inner Deep Bay, which was designated as a Wetland of International Importance (Ramsar Convention) in 1995.

5.2 *Ex situ* conservation

The most widely recognized *ex situ* conservation strategy is the preservation of living plants in botanical gardens (BGs) and arboreta. Although the first modern botanical gardens (that is, those designated for plant introduction and botanical research) were not established in China until the beginning of the 20th century, we can date their origin to *ca.* 2,800 BC (Medicinal Botanic Garden of Shennong), the earliest known botanical garden in the world (Xu, 1997). The first modern botanical garden, Hengchun Tropical Botanical Garden (Taiwan), was established in 1906, followed by Xiongyue Arboretum (Liaoning) in 1915, and Taipei Botanical Garden in 1921. Nevertheless, Hong Kong's Zoological and Botanic Garden precedes these, established in 1871. At present, over 160 botanical gardens have been set up in China (CSPCEC, 2008; Huang, 2010). The BGs belonging to the Chinese Academy of Sciences (which represents about 95% of the *ex situ* collections of all Chinese BGs) are cultivating *ca.* 25,000 species of vascular plants (Table 4), of which 20,000 are species found in China (Huang, 2010), i.e. over 60% of Chinese total flora. Living collections have increased considerably during the last decade due to a CAS innovation programme which has involved the designation of three core BGs (Xishuangbanna Tropical Botanical Garden, South China Botanical Garden and Wuhan Botanical Garden). The XTBG is currently the largest BG in China in number of plant collections (almost 15,000 taxa). The three core gardens, in addition to harboring large living collections, also maintain specialized collections: for example, SCBG holds the world's largest collection of Magnoliaceae (>130 species), Zingiberaceae (>120 species), and Palmae (>380 species) (Huang, 2010), and a collection of more than 2,000 medicinal plant species from South China (Wen, 2008).

Name	Location / Date of establishment	Area (km ²)	No. of taxa ^a / No. of species (living collections)	Red list species conserved
Xishuangbanna Tropical Botanical Garden (XTBG)	Menglun (Yunnan) / 1959	11.25	14,973 / 7,420	571
South China Botanical Garden (SCBG)	Guangzhou (Guangdong) / 1929	3.00	11,512 / 7,898	710
Wuhan Botanical Garden (WBG)	Wuhan (Hubei) / 1956	0.67	7,090 / 5,023	652
Fairy Lake Botanical Garden (FLBG)	Shenzhen (Guangdong) / 1983	8.60	6,588 / 4,956	441
Beijing Botanical Garden-CAS (BBG)	Beijing / 1956	0.72	5,001 / 3,463	177
Lushan Botanical Garden (LBG)	Lushan (Jiangxi) / 1934	3.00	4,934 / 4,378	229
Kunming Botanical Garden (KBG)	Kunming (Yunnan) / 1938	0.44	4,276 / 3,330	423
Guilin Botanical Garden (GBG)	Guilin (Guangxi) / 1958	0.67	4,056 / 3,843	445
Nanjing Botanical Garden Mem. Sun Yatsen (NMG)	Nanjing (Jiangsu) / 1929	1.86	3,790 / 2,701	263
Turpan Botanical Garden (TBG)	Turpan (Xinjiang) / 1976	1.50	506 / 490	26

^a 'Taxa' include species, subspecies, and varieties

Table 4. The 10 main BGs of Chinese Academy of Sciences. Sources: BGCI (2010) and Huang (2010)

Seed banking has also greatly progressed during recent times. The China Southwest Wildlife Germplasm Genebank project (operated by the Kunming Institute of Botany) has already stored seeds of nearly 5,000 native plant species with the next major target to store 10,000 species by 2020 (Huang, 2010), thereby aiming to secure the preservation of the germplasm resources of SW China. The KIB seedbank, which is also storing seeds for the UK Millennium Seed Bank and the World Agroforestry Center (Tsao & Zhu, 2010), is also working as a DNA bank. Regarding crop species, extant *ex situ* conservation facilities of the Ministry of Agriculture (which include long-term, medium-term and duplicate cold storage facilities) are keeping almost 400,000 accessions of seeds of *ca.* 450 crop species (Huang, 2010). In addition, perennial and vegetatively propagated crops (and their wild relatives) are preserved in 32 national field germplasm nurseries, including more than 1,300 rare and endangered species (CSPCEC, 2008; MEP, 2008a). Other *ex situ* facilities include germplasm banks specific for forest species and medicinal plants (López-Pujol et al., 2006; CSPCEC, 2008).

5.3 Environmental legislation and government planning

In addition to *in situ* and *ex situ* measures, environmental legislation and government planning (i.e. policies) are also essential to ensure adequate conservation of biodiversity. China has passed numerous laws and regulations addressing biodiversity conservation since the early 1980s (López-Pujol et al., 2006; McBeath & Leng, 2006; Yu, 2008). The most relevant laws governing plant biodiversity include the *Environmental Protection Law* (issued in 1979, revised in 1989), the *Forest Law* (issued 1984, revised 1998), the *Grassland Law* (issued 1985, revised 2002), and the *Seeds Law* (2000, revised 2004). China has also issued a significant number of regulations and rules, such as the *Regulation about Protection and Administration of Wild Medicinal Material Resources* (1987), the *Regulation about Nature Reserves* (1994), the *Regulations on Wild Plants Protection* (1996), or the recent *Regulation on the Import and Export of Endangered Wild Fauna and Flora Species* (2006) and *Regulation on Scenic Spots and Historical Sites* (2006). In addition to laws and regulations, there is some governmental supervision actively supporting biodiversity conservation in China, the most relevant being the Environmental Impact Assessment (EIA) system, which was legally implemented in 1981 and amended several times thereafter (1986, 1989, 1998, and 2002). Other mechanisms include licensing systems (such as forest logging and land use licenses), economic incentives (financial subsidies, tax-deductions, and compensation fees to enhance sustainable exploitation of natural resources, and more recently, payment for ecological and environmental services), or the quarantine system (established in the early 1980s).

Regarding government planning, China started to launch several comprehensive biodiversity-related policies from the early 1990s. Within the *Eight Five-Year Plan for Economic and Social Development* (1991-1995), China took biodiversity conservation as a key national policy. In addition to starting an inventory of biodiversity at all levels (Li, 2010), the *China Biodiversity Conservation Action Plan* (NEPA, 1994) was launched in 1994 to implement the Convention on Biological Diversity (CBD) together with *China's Agenda 21*. The Ninth Five-Year Plan (1996-2000) formally included the execution of CBD: *China's Biodiversity: A Country Study* plan was launched at the end of 1997, which analyzed the country's overall biodiversity, its economic value and benefits, the cost of implementing the CBD, and long-term objectives for biodiversity conservation and sustainable use of biological resources (SEPA, 1998). Other major plans issued before the end of the 20th century encompassed the

National Program for Nature Reserves (1996-2010) and the *National Plan for Ecological Construction* (1998-2050) (MEP, 2008a).

At the turn of the century, new environmental polices were launched to cope with the need for a more comprehensive and sustainable nature management. These new policies, commonly known as the 'Six Key Forestry Programs' (SKFP), meant an investment which exceeded the total expenditure during the period 1949-1999, and were aimed to avoid some of the pitfalls of the past in nature management (Wang et al., 2008). The SKFP, launched essentially for ecosystem rehabilitation, environmental protection and afforestation, covered more than 97% of China's counties, and included: (i) the National Forestry Protection Program (NFPP); (ii) the Shelterbelt Development Program (SDP); (iii) the Grain to Green Program (also known as the Sloping Land Conversion Program and the Cropland to Forest Program) (GTGP); (iv) the Sand Control Program for areas in the vicinity of Beijing and Tianjin (SCP), (v) the Wildlife Conservation and Nature Reserves Development Program (WCNRDP); and (vi) Fast-growing and High-yielding Timber Plantations Program (FHPPP). Most of these programs were planned to expire in 2010 except the last one, which will be alive until 2015 (Wang et al., 2007, 2008). Recent national plans include the *National Program for Conservation and Use of Biological Resources* (2007) and the *China National Environmental Protection Plan* within the Eleventh Five-Year Plan (2006-2010). At the end of 2010 the *China National Biodiversity Strategy and Action Plan* (2011-2030) was approved to replace the 1994 plan. Specific to plant biodiversity, in 2007 the *China's Strategy for Plant Conservation* (CSPC) within the *Global Strategy for Plant Conservation* (GSPC) was launched, aiming to pursue the CBD 2010 Biodiversity Target (CSPCEC, 2008).

All these plans have also been designed to fit other international treaties and conventions with implications for plant diversity signed by China, in addition to the CBD: the CITES Convention (1981), the Ramsar Convention (1992), the United Nations Convention to Combat Desertification (UNCCD) (1996), the UN Millennium Development Goals (2000), the Kyoto Protocol (2002), and the Cartagena Protocol on Biosafety (2005), among others. In addition, China also maintains international cooperation with governments and both public and private institutions, highlighting: the 'China Council for International Cooperation on Environment and Development' (with experts from several countries; <http://www.cciced.net/enciced/>), the 'China-EU Biodiversity Project' (with the European Union; <http://www.ecbp.cn/en/>), the 'Greater Mekong Subregion Core Environment Program' (with Cambodia, Laos, Burma, Thailand, and Vietnam with the support of the Asian Development Bank; <http://www.gms-eoc.org/>), and the 'Sino-American joint investigation on plant diversity in Hengduan Mountains' (with Harvard University; <http://hengduan.huh.harvard.edu/fieldnotes>).

6. Problems, prospects, and recommendations

6.1 Habitat destruction

The huge habitat destruction suffered in China, particularly since the 1950s (e.g. Shapiro, 2001), began to receive attention by the government authorities only in recent years, due at least in part, to the occurrence of natural disasters and the fall in crop productivity associated with soil erosion and land desertification (e.g. Liu & Diamond, 2005). To redress this situation, the state implemented a series of forestation and shrub or grass-planting projects (Fig. 9). Although the first plans were ratified in the late 1970s, they consisted generally of mono-culture forest plantations (often involving exotic species), lacking a

comprehensive scientific basis and failing to account for local floristic features (Zhang P. et al., 2000; Morell, 2008). However, after the disastrous floods of 1998, the National Forestry Protection Program (NFPP) was introduced, aimed at protecting the forests by logging bans and afforestation activities. In addition, the five other programs within the SKFP (Six Key Forestry Programs), launched soon after, meant that on completion, 76 million hectares should be forested (Wang et al., 2007); due to this unprecedented effort, the forest cover has increased from 16.6% in 2000 to 20.4% by the end of 2009. These new afforestation initiatives were planned to avoid past mistakes; however, several problems and limitations aroused, such as the failing of local cadres to implement the programs effectively (mostly due to corruption), the lack of clarification of land ownership (although in recent years some reforms have started to be introduced), but also a decrease in forest quality (a young plantation cannot provide the same ecological services that a mature stand can provide) and a low rate of survival of populations, sometimes due to the use of inappropriate species (Wang et al., 2008; Song & Zhang, 2010). Moreover, other shortcomings such as an overemphasis in shrub and tree planting instead of grasses (as often reported in arid zones; e.g. Cao et al., 2011) have produced undesired effects including a loss of native vegetation and the exacerbation of water shortages.



Fig. 9. Shelterbelt to protect farmland from Gobi's sand encroachment (near Jiayuguan, Gansu Province) (photo by Jordi López-Pujol)

6.2 Protected areas

The *National Program for Nature Reserves* (implemented in 1996) stipulated that the number of nature reserves must reach 1,200 by 2010 and accounting for 10% of the Chinese territory (or 12% when forest parks and scenic spots are considered) (Li et al., 2003). This goal has been widely exceeded due to the impressive rate of nature reserves establishment, particularly during the last 25 years (Table 3). Unfortunately, provisions for staffing and financing in order to manage these reserves have not increased at the same rate. For example, about one-

third of all nature reserves had neither staff nor budget (i.e., are protected only 'on paper'; Liu et al., 2003; Xu H. et al., 2009). Moreover, the staff is rarely professionally-trained (with higher education) (MacKinnon & Xie, 2008; Xu H. et al., 2009). Lack of financial investment is, however, a general problem for all the reserves including those state-funded (which are somewhat better funded but represent less than 12% of the total no. of reserves). Lack of budget compromises reserves' protection duties: they are poorly or never patrolled, species and ecosystems are not satisfactorily monitored nor inventoried, and some reserves do not even have signposts delineating their borders (Qiu et al., 2009; Xu H. et al., 2009; Quan et al., 2011). In the recent study of Quan et al. (2011), some worrying figures arose, such as a mere 2% of the nature reserves had enough financial support for their daily management activities, and that only *ca.* 11% had set up comprehensive monitoring systems.

To solve the funding shortage and to cover daily operation costs, many reserves are forced to be self-sufficient through resource exploitation (e.g. over-exploitation of plant resources including medicinal and edible plants, hunting, mining, land reclamation, hydropower development, tourism and recreation), a policy inconsistent with their intended purpose and which may cause severe harm (López-Pujol et al., 2006; MacKinnon & Wang, 2008; Yu, 2010). One illustrative example is the destruction of over 1,000 km² of natural wetlands in the Yancheng National Nature Reserve, listed both as a Ramsar site and a Biosphere Reserve (Qiu et al., 2009).

Another consequence of lack of investment is the frequent failure of compensation schemes (subsidies, compensation fees) to the local people (a problem often aggravated by the widespread corruption among local officials), who may be against the establishment of new PAs because they feel that their interests are in conflict with nature preservation. Tourism creates opportunities for local people, but this should evolve towards sustainability, and planned and managed to combine biodiversity protection while ensuring adequate economic benefits to local communities (Quan et al., 2011). Engaging local communities in conservation activities as well as in the planning and management of reserves also constitutes a useful tool for the long-term sustainability of nature reserves, since the pressures placed on reserves by local residents are largely eased (McBeath & Leng, 2006; MacKinnon & Xie, 2008). Enhancing public participation should also include the NGOs, which in other countries have demonstrated a good performance in both assisting in the PAs management as well as resolving people-park conflicts (McBeath & Leng, 2006; Qiu et al., 2009).

Nevertheless, nature reserves are afflicted with other serious problems besides insufficient budgets. Overlapping management—in some cases involving up to seven administrations—can cause confusion, inefficiency, uncertain boundaries, and multiple designations of the same reserve (López-Pujol et al., 2006; McBeath & Wang, 2009). Another common problem of Chinese PAs is that they are too small to maintain genetic diversity or to ensure species and ecosystem viability (Liu et al., 2003; Xu H. et al., 2009). This is especially true in eastern China (see Fig. 10), where nature reserves are often just occupying a very few square kilometers; for example, the 512 smallest reserves in China (which account for about 20% of their total number), accounted for *ca.* 0.13% of their total area (MacKinnon & Xie, 2008). On the contrary, very large areas can be found in western China (Fig. 10), some exceeding 10,000 km² (Qiangtang Nature Reserve, in Tibet, has almost 300,000 km²). In China, the combined area of the 20 largest nature reserves accounts for nearly 60% of the total area of all reserves (MacKinnon & Xie, 2008), which shows that the design of PAs has not been entirely rational in the past.

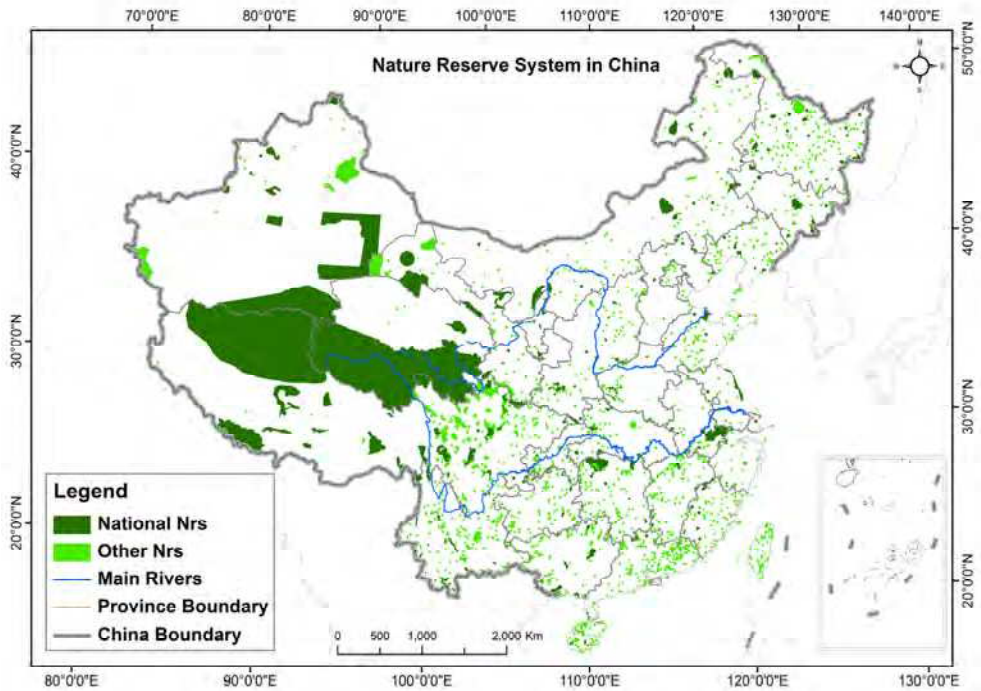


Fig. 10. Map of nature reserves in China at the end of 2008 (map elaborated by Lu Zhang, Research Center for Eco-Environmental Sciences, Beijing)

The lack of systematic planning is also evidenced when criteria of biogeographic and ecosystem representativeness are tested. Xu et al. (2008) reported a total lack of correlation between the percentage of land area occupied by nature reserves and overall species richness, endemism, or threat at provincial level; in this sense, these authors are calling for the setting up of new reserves in the provinces with less than 10% of reserves coverage (Fig. 11). Moreover, if we compare the Figs. 10 and 4 (the plant hotspots), it is clear that the areas richest in plant diversity are not protected enough. Some of the main gaps in the Chinese coverage of PAs (see Li et al., 2003; MacKinnon & Xie, 2008) are precisely those corresponding to hotspots or areas located within the hotspots (e.g. Hengduan Mountains, N Guangxi, SE Yunnan-SW Guizhou-SW Guangxi). Numerous gaps in ecosystem protection also exist, such as marine, wetland, grassland and desert vegetation (Li et al., 2003; Xu H. et al., 2009). An additional problem is their lack of connectivity through biological corridors (MacKinnon & Wang, 2008). Establishing a centralized management by a new State Agency of Nature Reserves Service at state-level, and upgrading the current *Regulation of Nature Reserves* (of 1994) to a new *Nature Reserve Law* (which is currently being drafted) seem necessary steps to achieve more comprehensive planning and management of the Chinese PAs network (Yu, 2008; McBeath & Wang, 2009).

6.3 *Ex situ* conservation measures

The increasing number of BGs during the last decades (from just 52 in 1975 to 160 at present; Fig. 12; Huang, 2010) and the launching of government programs (such as the 15-year

master plan of Chinese Academy of Sciences; Huang et al., 2002) has meant a great advancement in the *ex situ* conservation of Chinese flora. Firstly, the target to conserve 21,000 native plant species by the 15-year plan has almost been totally achieved. Secondly, the GSPC (Global Strategy for Plant Conservation; CBD, 2002) target to conserve at least 60% of threatened plants has been partially achieved: virtually all the 388 species of the *National List of Rare and Endangered Plant Species* are included in the *ex situ* living collections of Chinese BGs (although some exceptions apply; López-Pujol & Zhang, 2009), but only a small fraction (less than 40%) of the 4,408 species of *China Species Red List* (Table 4; Huang, 2010). Yet another deficiency is, despite the recent progress, that BGs are not representative of the local floras across China; some regions boasting a rich plant diversity, such as western China, have too few botanical gardens (only 10% of the total; Huang, 2010), such as the Himalayas, the Qinghai-Xizang Plateau (in the Tibet Autonomous Region there are no BGs; Cram et al., 2008), and the dry-hot valleys of southwestern China, a trend also apparent in the three primary distribution centers for endemic plants in China (He, 2002; López-Pujol et al., 2006). In addition, both the number of gardens where a given plant is cultivated and the sizes of the collections are generally insufficient: of the ca. 25,000 plant species cultivated in the CAS BGs, about two-thirds are not duplicated (that is, present in just one BG; Huang, 2010). For the threatened species, the trend is the same: only half of the Chinese threatened species are duplicated. Furthermore, despite the claim of Xu (1997), population sizes are generally not sufficient for maintaining adequate levels of genetic diversity; for example, the only *ex situ* collection of *Picea neveitchii* (a threatened species included within the *National List of Rare and Endangered Plant Species* and the *Catalogue of the National Protected Key Wild Plants*) consist of two individuals cultivated in the Xi'an Botanical Garden (Zhang, 2007). Other problems are related to lack of financial resources; the Three Gorges Botanic Garden of Rare Plants (which housed about 10,000 individuals belonging to 175 plant taxa) was closed in 2007 owing to a lack of funding (López-Pujol & Ren, 2009). Regarding seed storage facilities, there are still considerable gaps in conservation of Chinese native wild plants; for example, no seeds from any Tibetan plant species were stored in the Kunming seedbank until recently (Cram et al., 2008), although this is now being solved by the staff of Kunming Institute of Botany.

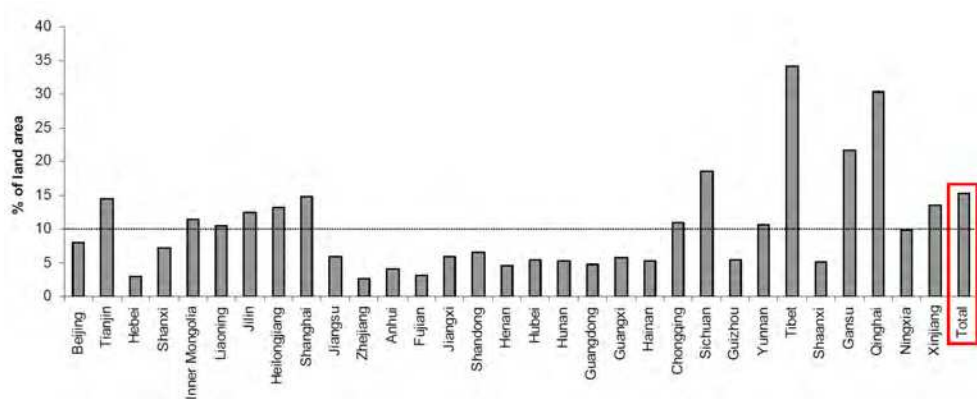


Fig. 11. Coverage of nature reserves in each province of mainland China (source: MEP, 2008b). Dotted line indicates 10% of reserves coverage.

6.4 Environmental legislation

Legislation addressing environment and biodiversity has significantly expanded in the last 25 years, and a relatively comprehensive body of laws and regulations have been enacted, some as restrictive as European laws in many aspects (Ferris, 2005). Nevertheless, two major problems remain: an insufficient and inefficient legal system and a lack of enforcement (López-Pujol et al., 2006; Johnson, 2008; Yu, 2010).

The main purpose of many biodiversity-related laws and regulations is still to manage the use of natural resources and they are poorly-focused on conservation (although this focus is increasing in recent years; McBeath & Leng, 2006) since they were mainly promulgated taking into account the natural resources' economic value and not their sustainable use (Yu, 2008). Moreover, legislation tends to focus on endangered animals and not plants (which are covered by regulations and not laws), and also do not provide explicit protection of their habitats (McBeath & Wang, 2009). In addition, specific legislation to preserve genetic resources is still very limited (Yu, 2008). There is no comprehensive law governing protected areas (although this is being drafted), and a law specifically devoted to protection of biodiversity is still pending. Another inconsistency inherent to China's legislation is the lack of a clear demarcation of responsibilities, whereas punishments only mandate damage compensation rather than ecological restoration or rehabilitation (Beyer, 2006).

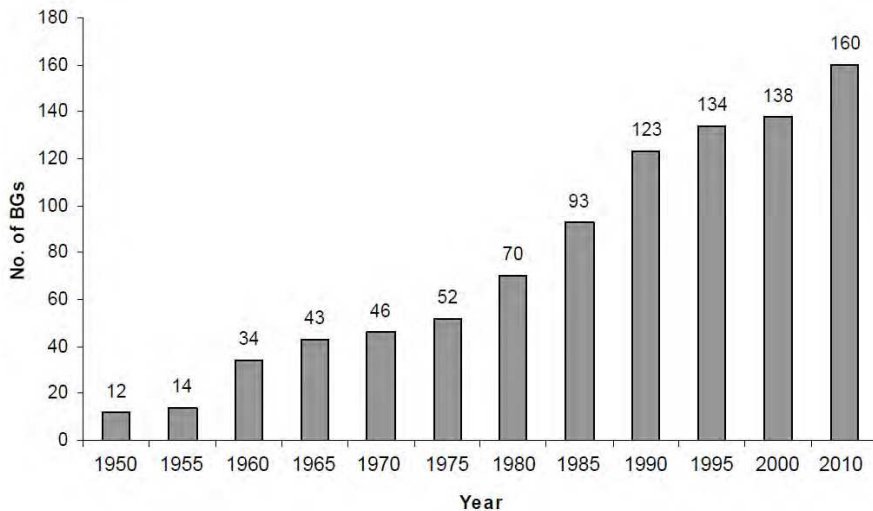


Fig. 12. Increase in the number of BGs in China. Source: Huang (2010)

Historically, law enforcement has been one of the major problems in the establishment of a sound legal system in China. As Yu (2010) states, "China is a country ruled by men rather than ruled by law", and moral precepts and customs of Confucian heritage generally outweighed formal laws (Beyer, 2006). Violations of environmental legislation are all too common and even tolerated. For example, at the beginning of the 2000s there still was a generalized lack of effective *in situ* legal protection for the nationally listed rare and endangered plant species (Xie 2003), and at present this situation is still continuing for some of them (López-Pujol & Zhang, 2009). This poor enforcement has multiple reasons apart

from historical ones, including: (i) legislation is too general and largely vague; (ii) violations of environmental and nature protection laws have, with a few exceptions, no serious penalty; this means that most companies prefer paying the fines instead of following the law; (iii) a lack of capability on the part of administration for monitoring law enforcement (staff, funding and technical expertise are insufficient); (iv) lack of coordination among the different administrative levels (several agencies are sometimes responsible for the same task); (v) conflicts of interest between national-level legislation and local interests; (vi) widespread corruption among government officials; and (vii) lack of public participation (e.g. Beyer, 2006; McBeath & McBeath, 2006; Johnson, 2008; Liu & Diamond, 2008; Nagle, 2009; Yu, 2010).

Implementation of the environmental impact assessment (EIA) has experienced enormous difficulties in the past, although the rate of enforcement has been significantly increasing since the 1990s. Prior to passage of the Law on EIA in 2003, Chinese environmental legislation only focused on individual construction projects that might pollute the environment; however, the 2003 law expanded the environmental assessment to include government-proposed plans and projects (although with some exceptions) and included public participation as part of the process (Moorman & Zhang, 2007; Zhao, 2009). However, some problems remain, as EIA is still mainly applied to assess the impact of projects that might pollute the environment rather than addressing all activities affecting biodiversity (Yu, 2008). The growing conflict between the central government and local governments is a formidable obstacle to implementing the EIA, as well as the still-limited public scrutiny and the minimal violation penalties (Zhao, 2009). In order to strengthen its enforcement, the government has implemented a moratorium on EIA approvals since 2007 (You, 2008).

6.5 Scientific research

Large-scale national surveys of vegetation and flora began in the early 1960s mainly with the aid of experts from the USSR (Li, 2010). After the difficult period of the Cultural Revolution, when all academic activities were largely stopped, scientific research received a new impulse, and some major publications started to appear, such as *Vegetation of China* (ECVC, 1980) whereas other works progressed rapidly, such as *Flora Reipublicae Popularis Sinicae*, which was started in 1958 and whose 80 volumes were finally completed in 2004 (Li, 2008). Currently, the Missouri Botanical Garden (MBG) and the Chinese Academy of Sciences (CAS) are working together on the *Flora of China* project (Fig. 13), an international effort to produce a 25-volume English-language revision of the *Flora Reipublicae Popularis Sinicae*. At present, 19 volumes have already been published and they can also be browsed online (<http://hua.huh.harvard.edu/china/>). These same institutions (MBG and CAS) have also promoted the *Moss Flora of China* project, which aims to provide an updated, English version of the bryophyte flora of China (<http://www.mobot.org/mobot/moss/china/welcome.html>). In addition, an increasing number of local and provincial floras (although rarely in English) are available today (Liu et al., 2007).

Regarding conservation biology, after the first symposium on biodiversity conservation in China took place in 1990 (Wang et al., 2000), some general surveys and studies have been published since then, including the seminal *China Plant Red Data Book* (Fu, 1992), *Chinese Biodiversity – Status and Conservation Strategy* (Chen, 1993), *A Biodiversity Review of China* (MacKinnon et al., 1996), *Conserving China's Biodiversity* (2 volumes; Wang & MacKinnon, 1997; and Schei et al., 2001), *China's Biodiversity: A country Study* (SEPA, 1998), *The Plant Life*



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cathaya argyrophylla Search
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FOC | Family List | FOC Vol. 4 | Pinaceae * | Cathaya *

1. *Cathaya argyrophylla* Chun & Kuang, Acta Bot. Sin. 10(3): 246. 1962.

银杉 yin shan

Cathaya nanchuanensis Chun & Kuang; *Pseudotsuga argyrophylla* (Chun & Kuang) Greggus; *Tsuga argyrophylla* (Chun & Kuang) de Laubenfels & Silba.

Trees to 20 m tall; trunk to 40 cm d.b.h.; bark dark gray, irregularly flaking; branchlets yellow-brown, initially densely gray-yellow pubescent, aging dark yellow and glabrous; winter buds light yellow-brown, ovoid or ovoid-conical. Leaf cushions topped with pale, orbicular or subsquare leaf scars. Leaves dark green adaxially, 4-6 cm × 2.5-3 mm on long branchlets, nearly clustered into a whorl on short branchlets where usually not longer than 3 cm, puberulent, densely pubescent along grooves, margin slightly revolute. Seed cones green, dark brown when mature, ovoid or ellipsoid, 3-5 × 1.5-3 cm. Seed scales 13-16, suborbicular or compressed orbicular-ovate, 1.5-2.5 × 1-2.5 cm, densely pubescent on exposed part. Bracts 1/4-1/3 as long as seed scales. Seeds dark green mottled with light green, slightly appressed, obliquely ovoid, 5-6 × 3-4 mm; wing yellow-brown, obliquely ovate or elliptic-ovate, 1-1.5 cm × 4-6 mm.

* Mountains, usually on open slopes and ridges; 900-1900 m. NE Guangxi (Jinxiu Yaozu Zizhixian, Longsheng Gezu Zizhixian), N Guizhou (Daozhen Xian, Tongzi), S Hunan, SE Sichuan (Nanchuan Xian, Wulong Xian).

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Fig. 13. Entry for *Cathaya argyrophylla* in the online version of *Flora of China*

of China (Chapman & Wang, 2002), or the more recent *The Green Gold of China* (MacKinnon & Wang, 2008). In parallel, publication of papers in high impact factor journals has experienced a huge rise since the 1990s: China was the seventh most productive country in biodiversity research during the period 1900-2009 (and due to obvious historical reasons, almost all of this corresponds to the last two decades (Enright & Cao, 2010), whereas the Chinese Academy of Sciences is the world's most productive research institution (Liu et al., 2011). Almost all aspects of plant biodiversity are currently being explored by Chinese researchers, including the cutting-edge ones (see Enright & Cao, 2010), and this is mostly due to the great effort of the Chinese government: spending on research and development has increased to 1.5% of GDP, well above the developing economies (0.96%; World Bank, 2010a). The total research funding of the China's National Natural Science Foundation almost quadrupled during the period 2001-2008 (He, 2009).

Other biodiversity surveys include the national survey on traditional Chinese medicinal resources, conducted between 1984 and 1994, and identifying 11,146 plant species (Xu et al. 1999). The State Forestry Administration has performed five-year surveys (including censuses) of forest resources since 1973, recently completing their 7th forest survey. Moreover, the Chinese Academy of Sciences has set up a series of ecological field stations since 1988 (about 40 at present), organized into the Chinese Ecosystem Research Network (CERN) (Fu et al., 2010), whereas the Ministry of Science and Technology has set up the Chinese National Ecological Research Network (CNERN), with over 50 field stations including those of CERN (Li, 2010). It is also noteworthy the launching of *Chinese Virtual Herbarium* (<http://www.cvh.org.cn/>), an on-line portal which allows access to the plant specimens maintained in Chinese herbaria and to related botanical databases.

Taiwan has also made considerable efforts to study the island's biodiversity, with the publication of *Flora of Taiwan* (the 6 volumes of the second edition were compiled during the period 1993-2003, and are available online at <http://tai2.ntu.edu.tw/ebook.php>) and the *Red Data Book of Taiwan Region: Criteria and Measure for Rare and Threatened Plant Species* (Lai, 1991), although a revised red book is under preparation. In the Hong Kong SAR, the first surveys on flora were conducted as early as 1861, with the *Flora Hongkongensis* of G. Bentham. Most recent works on plant biodiversity are the *Check List of Hong Kong Plants 2001* (Wu, 2002) and three out of four volumes of the *Flora of Hong Kong*, produced by HK Herbarium and the South China Botanical Garden, have been already published.

Despite the significant strides made by these various administrations, universities, and research institutes, there are still significant gaps in the knowledge of plant diversity, mainly because of the lack of intensive botanical exploration in many parts of the country, specially the western mountainous areas. For example, despite some researches that were done in the floristically wealthy Hengduan Mountains since the 1970s, comprehensive inventories were not carried out until the 2000s. In addition, Chinese botanists were not able to do sizable international collaborations until the beginning of the 1980s (e.g. Dosmann & Del Tredici, 2003). The lack of information on China's threatened plant species is still worrisome despite the recent efforts; less than one-fifth of the angiosperms of China have been assessed using the 2001 IUCN criteria, and at present no modern red book (that is, following the standards of those published in western countries, which include valuable information of each studied plant species including geographic range, habitat characterization, conservation status, threats, and present and recommended conservation measures) has been published, either at regional or national level. Finally, it should be noted that, despite the fact that a sound environmental and ecological monitoring network is already in place, there is a need for a monitoring system specific to biodiversity (e.g. Xu H. et al., 2009); that is, monitoring populations and species, with special emphasis to those threatened.

6.6 Policies, financial resources, and environmental awareness

Government policies in mainland China have promoted rapid growth since the end of the 1970s, when the 'open-door' policy was adopted in order to eradicate poverty and quickly catch up with the developed economies. These policies have often implied the seeking of short-term economic benefits by an intense and inefficient use of natural resources and high levels of pollution; such unsustainable development policies have also been applied to the conservation and management of biodiversity (Liu & Diamond, 2005; Liu, 2010; Yu, 2010). However, some optimistic signs have emerged during recent years; since the 'fourth generation' of Chinese leaders took government in mainland China, environmental protection and natural resources conservation policies (which became a national strategy since the middle of the 1990s) have been emphasized (e.g. Johnson, 2008), as a part of a broader policy aimed at achieving the *xiaokang* (a moderately prosperous society) by 2020 (MFA, 2010). In 2007, the new 'scientific development' concept was proposed during the 17th Party Congress, consisting of combining the development of the economy with the protection of natural resources and the environment, with the final goal of achieving a 'harmonious society' (that is, development based on sustainability) (Fu et al., 2007; Johnson, 2008). Consequently, the Chinese economy is progressively turning from a polluting, low-efficiency economy to an environmentally-friendly, circular economy (McBeath & Wang, 2009; Liu, 2010). For example, the Eleventh Five-Year Plan (2006-2010) includes a target to reduce energy consumption per unit of GDP by about 20% (MEP, 2008a), which was

likewise included in the 'China's National Climate Change Programme' launched in 2007 (NDRC, 2007). Furthermore, the ambitious goal of reducing CO₂ emissions per unit of GDP by 40-45% by 2020 from 2005 levels will be included within the Twelfth Five-Year Plan (2011-2015) (MFA, 2010). Other very positive steps include the upgrade of State Environmental Protection Administration (SEPA) to ministry status (Ministry of Environmental Protection) in 2008 (McBeath & Wang, 2009); however, it still remains a small body, with much less employees than the United States Environmental Protection Agency (Liu & Diamond, 2008; McBeath & Wang, 2009).

Despite these newly-oriented state-level policies, short-term economic and political gains still often outweigh the long-term benefits of preserving natural resources by the local governments; this is undoubtedly because the promotion of government officials at the local level still strongly relies on economic performance (Qiu et al., 2009; Liu, 2010; Yu, 2010). Therefore, the actual conflict between economic development and nature protection is the primary reason why many PAs are managed for purposes other than nature preservation and scientific research (e.g. McBeath & Leng, 2006; Qiu et al., 2009). Therefore, it is highly advisable, while being one of the major recommendations of the Task Force on Environmental Governance of the CCICED ('China Council for International Cooperation on Environment and Development'), that evaluating the performance of local government leaders should incorporate environmental performance (Xue et al., 2007).

One of the main reasons for the still dominant attitudes toward over-exploitation of natural resources and the lack of biodiversity protection is that the importance of biodiversity is not fully understood. The ideology of 'pollute first, clean later' is still seen as suitable for the development of the country by many Chinese, including officials (Liu, 2010), and this attitude is often fuelled by the poor example offered by the western countries in the past: although China is today the world's highest emitter of CO₂, the cumulative CO₂ emissions of China during the period 1850-2005 represent only about 8% of the world's total, much less than the 27,8% of the United States and also lower than those for Germany and Russia (World Bank, 2010b). In addition, traditional beliefs such as Confucian philosophy (which asserts control over nature; Shapiro, 2001; Yu, 2010) are still very present in contemporary Chinese society. A second reason is that the economic value of biodiversity (e.g. the ecosystem services) has also often been underestimated; ecological benefits are seldom contemplated because most of them cannot be reflected in the traditional marketplace (Zhang B. et al., 2010). However, some recent large catastrophes (e.g. the 1998 floods or the 2005 Songhua River spill) have shown the government and society that the toll (economic but also human) associated with lack of environmental protection is too high. The introduction of the calculation of Green GDP by the government showed that the costs of environmental pollution and ecological degradation accounted for about 3% of GDP in 2004 (the only year for which the figures have been released to the public; Liu, 2010), although some have estimated that these costs have ranged from 7% to 20% of GDP during the most recent decades (Liu & Diamond, 2005; Fu, 2008).

The lack of financial resources, which is a logical consequence of that explained in the previous paragraphs, is another of the root causes for the current biodiversity loss. The current investment of the Chinese government in environmental protection remains inadequate, although it has increased significantly since the 1990s; whereas the total environmental investment for the Ninth Five-Year Plan (1996-2000) was less than 1% of GDP, this rose to about 1.2% of GDP during the Tenth Five-Year Plan (2001-2005), and it has reached 1.5% in the Eleventh Five-Year Plan (2006-2010) (NBSC, 2009), but it is far from the

average of 2.5% of GDP spent by the developed countries (McBeath & Wang, 2009). Despite the commendable effort by the Chinese government in raising the funds yearly, much more investment is expected for the country with the largest monetary reserves of the world (World Bank, 2010a) and which harbors some of the most valuable flora in the world (see section 2 of this chapter); in this sense, China should play a much more leading role in plant biodiversity conservation. In addition to increasing such funding, the government should encourage the active participation of all sectors of Chinese society, the private companies in particular (e.g., through tax-exemption), as well as non-governmental organizations (NGOs) and other active groups of civil society.

The strengthening of environmental awareness can be viewed as a sign of maturation of societies. In this sense, there is room for optimism as China has witnessed during recent years a flourishing of numerous environmental groups (including over 2,000 environmental NGOs; Yu, 2010) while the control over media and scientists has been significantly reduced. Indeed, the opinion of scientists, environmental groups, and general public are now taken very seriously by the government; for example, protests forced a moratorium on the damming of the Nujiang River in 2004 (declared by the Prime Minister himself) and also halted the construction of a chemical plant in Xiamen City in 2007 (McBeath & McBeath, 2006; MacKinnon & Wang, 2008). In addition, Chinese authorities have also implicitly acknowledged the contribution of NGOs to the protection of biodiversity; at present, many international environmental NGOs are assisting the government in the design and management of conservation activities (e.g. the WWF helped in establishing the Wolong Giant Panda Reserve). The role of the NGOs is, however, weak in China compared to the western countries, and their scrutiny by the government is still significant (e.g. Xue et al., 2007; McBeath & Wang, 2009).

Despite this progress on environmental consciousness, much more should be done. Environmental education, although expanding at all levels, is still limited (CSPCEC, 2008), and specialized courses in biodiversity conservation are badly needed (McBeath & Wang, 2009). Environmental awareness should be particularly geared towards the indigenous and local communities, as they usually have more direct contact with the natural resources in both protected and unprotected areas. More importantly, most of the plant diversity (including threatened species) is located in poorer areas of the west and the southwest; therefore, it is essential to give the highest priority to those policies oriented to conciliate poverty alleviation and biodiversity conservation (CDB, 2010).

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The book covers several topics of biodiversity researches and uses, containing 17 chapters grouped into 5 sections. It begins with an interesting chapter considering the ways in which the very biodiversity could be thought about. Noteworthy is the chapter expounding pretty original "creativity theory of ecosystem". There are several chapters concerning models describing relation between ecological niches and diversity maintenance, the factors underlying avian species imperilment, and diversity turnover rate of a local beetle group. Of special importance is the chapter outlining a theoretical model for morphological disparity in its most widened treatment. Several chapters consider regional aspects of biodiversity in Europe, Asia, Central and South America, among them an approach for monitoring conservation of the regional tropical phytodiversity in India is of special importance. Of interest is also a chapter considering the history of the very idea of biodiversity emergence in ecological researches.

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