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An inventory of invasive alien species in China

Haigen Xu[1], Sheng Qiang[2], Piero Genovesi[3], Hui Ding[1], Jun Wu[1], Ling Meng[2], Zhengmin Han[4], Jinlai Miao[5], Baishi Hu[2], Jiangying Guo[6], Hongying Sun[7], Cheng Huang[8], Juncheng Lei[1], Zhifang Le[1], Xiaoping Zhang[9], Shunping He[10], Yi Wu[1], Zhou Zheng[5], Lian Chen[1], Vojtěch Jarošík[11,12], Petr Pyšek[12,11]

1 *Nanjing Institute of Environmental Sciences (NIES), Ministry of Environmental Protection, Nanjing, China* **2** *Nanjing Agricultural University, Nanjing, China* **3** *ISPRA – Institute for Environmental Protection and Research, Italy and Chair IUCN Invasive Species Specialist Group* **4** *College of Forest and Environment, Nanjing Forestry University, Nanjing, China* **5** *The First Institute of Oceanography, State Oceanic Administration, Qingdao, China* **6** *Institute of Plant Protection, Chinese Academy of Agricultural Sciences, Beijing, China* **7** *College of Life Science, Nanjing Normal University, Nanjing, China* **8** *College of Life Science, Nanjing University, Nanjing, China* **9** *College of Life Science, Anhui Normal University, Hefei, China* **10** *Institute of Hydrobiology, Chinese Academy of Sciences, Wuhan, China* **11** *Department of Ecology, Faculty of Science, Charles University in Prague, CZ 12844 Viničná 7, Czech Republic* **12** *Institute of Botany, Academy of Sciences of the Czech Republic, CZ 25243 Průhonice, Czech Republic*

Corresponding author: *Haigen Xu* (xhg@nies.org)

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Abstract
Invasive alien species (IAS) are a major global challenge requiring urgent action, and the Strategic Plan for Biodiversity (2011–2020) of the Convention on Biological Diversity (CBD) includes a target on the issue. Meeting the target requires an understanding of invasion patterns. However, national or regional analyses of invasions are limited to developed countries. We identified 488 IAS in China's terrestrial habitats, inland waters and marine ecosystems based on available literature and field work, including 171 animals, 265 plants, 26 fungi, 3 protists, 11 procaryots, and 12 viruses. Terrestrial plants account for 51.6% of the total number of IAS, and terrestrial invertebrates (104 species) for 21.3%. Of the total numbers, 67.9% of plant IAS and 34.8% of animal IAS were introduced intentionally. All other taxa were introduced unintentionally despite very few animal and plant species that invaded naturally. In terms of habitats, 64.3% of IAS occur on farmlands, 13.9% in forests, 8.4% in marine ecosystems, 7.3% in inland waters, and 6.1% in residential areas. Half of all IAS (51.1%) originate from North and South America, 18.3% from Europe, 17.3% from Asia not including China, 7.2% from Africa, 1.8% from Oceania, and the origin of the remaining 4.3% IAS is unknown. The distribution of IAS can be divided into three zones. Most IAS are distributed in coastal

provinces and the Yunnan province; provinces in Middle China have fewer IAS, and most provinces in West China have the least number of IAS. Sites where IAS were first detected are mainly distributed in the coastal region, the Yunnan Province and the Xinjiang Uyghur Autonomous Region. The number of newly emerged IAS has been increasing since 1850. The cumulative number of firstly detected IAS grew exponentially.

Keywords
Invasive plants and animals, distribution, origin, pathway, rate of introduction

Introduction

Invasive alien species (IAS) are considered one of the key pressures on world's biodiversity (Leprieur et al. 2008; Butchart et al. 2010; Rands et al. 2010), alter ecosystem services and processes (Hulme et al. 2009; Vilà et al. 2010, 2011), reduce native species abundance and richness (Cohen and Carlton 1998; Blackburn et al. 2004; Gaertner et al. 2009; Hejda et al. 2009), decrease genetic diversity of resident species (Ellstrand and Schierenbeck 2000; Daehler and Carino 2001), and cause substantial economic losses (Pimentel et al. 2005; Xu et al. 2006a; Kettunen et al. 2009). Responding to this threat is therefore particularly urgent (Lambertini et al. 2011). In October 2010, world leaders adopted the Strategic Plan for Biodiversity (2011-2020) under the Convention on Biological Diversity (CBD), including the Aichi Target 9 (Secretariat of the Convention on Biological Diversity 2010) calling to identify IAS and pathways, control and eradicate priority species, and to manage pathways in order to prevent further invasions. In order to evaluate achievements of the Aichi Targets, baseline data are needed. However, a global baseline of IAS is unavailable (Butchart et al. 2010; McGeoch et al. 2010), and national/regional data sets suitable for analysis of temporal patterns of biological invasions are rare for developing countries of the world, resulting a pattern that reflects geographical biases in information on invasion patterns (Pyšek et al. 2008; Nuñez and Pauchard 2010).

China is the world's most populous country with 1.34 billion people and one of the largest territories (Liu and Diamond 2005). China is also one of the mega-diversity countries, with half of its species found nowhere else (Liu et al. 2003; Xu et al. 2008). Its economy, ranked second, is growing at a very fast rate. The extraordinary biogeographic and economic characteristics of China make it ideal for understanding how biological invasions currently affect, and will affect in the future, the fastest growing economies in the world. Here, we present a comprehensive inventory of IAS in China, and analyze the temporal trends of biological invasions in the country in order to identify priority responses to the growing threat from biological invasions.

Methods

According to the CBD and IUCN definition, invasive alien species (IAS) are those alien species that became established in natural or semi-natural ecosystems or habitats, are an agent of change, and threaten native biological diversity (IUCN 2000; Shine et al. 2000; McNeely et al. 2001). Alien species refers to a species, subspecies or lower taxon occurring outside its natural, past or present range and dispersal potential (i.e. outside the range it occupies naturally or in a range it could not occupy without direct or indirect introduction by humans) and includes any part, gametes, or propagules (IUCN 2000). Only species with evidence of their impact on biodiversity, human activities or economy were considered in the present assessment. We included IAS that established populations in terrestrial habitats, inland waters or marine ecosystems of China.

We identified IAS and pathways of their introductions based on available literature (Ding and Wang 1998; Xie et al. 2000; Li and Xie 2002; Xiang et al. 2002; Xu and Qiang 2004; Liu et al. 2005; Wan et al. 2005, 2008, 2009; Weber et al. 2008; Xie 2008 ; Zhang et al. 2008; Wu et al. 2010; Huang et al. 2012; see Liu et al. 2012 for an overview of research in plant invasions in China), including Flora of China (126 volumes edited by the Editoral Committee of Flora Sinica, Chinese Academy of Sciences and published by Science Press in Beijing, China), and Fauna of China (100 volumes edited by the Editoral Committee of Fauna Sinica, Chinese Academy of Sciences, and published by Science Press in Beijing, China), and on field work carried out in most provinces of China. All recorded IAS with evidence of negative impacts on biodiversity, human livelihood or economy were included in the inventory, with information on their presence or absence in particular provinces or autonomous regions. A preliminary inventory of IAS was first drafted, and subsequently verified through many internal reviews and field surveys. The year or period of the first detection of a species in China was recorded, providing information on the minimum residence time (Rejmánek 2000; Pyšek and Jarošík 2005); this information was available for 396 species.

Results

The inventory included 488 IAS in China's terrestrial habitats, inland waters and marine ecosystems. Of particular taxa, there are 171 animals, 265 plants, 26 fungi, 3 protists, 11 procaryots, and 12 viruses (Appendix). Terrestrial plants account for 51.6% of the total number of IAS, and terrestrial invertebrates (104 species) for 21.3% (Table 1). Intentional introductions accounted for 67.9% of plant IAS and 34.8% of animal IAS (Table 2), such as tropic ageratum (*Ageratum conyzoides*), common pokeweed (*Phytolacca americana*), and red-eared slider (*Trachemys scripta elegans*) introduced as ornamental species. Very few animal and plant species invaded via natural spread (such as *Ageratina adenophora* and *Ondatra zibethicus*). All other taxa were introduced unintentionally (Table 2), such as the oriental wood borer (*Heterobostrychus aequalis*), and the tropical fire ant (*Solenopsis geminate*) that invaded with trade products. In terms of habitats, 64.3% of IAS occur on farmlands, 13.9% in forests, 8.4% in marine ecosystems, 7.3% in inland waters, and 6.1% in residential areas. Half of all IAS (51.1%) originate from North and South America, 18.3% from Europe, 17.3% from Asia not including China, 7.2% from Africa, 1.8% from Oceania, and the origin of the remaining 4.3% IAS is unknown.

Table 1. Invasive alien species in China classified according to the taxonomic group and environment where they invade.

Taxonomic group	Terrestrial	Freshwater	Marine	Total
Plants	252	7	6	265
Vertebrates	15	16	15	46
Invertebrates	104	4	17	125
Others				52
Total	**371**	**27**	**38**	**488**

Table 2. Pathways of introduction of IAS to China

	Plants		Animals		Others (Fungi, Protista, Procaryotae, Vira)	
Pathways	**No. of species**	**%**	**No. of species**	**%**	**No. of species**	**%**
Unintentional introduction	84	31.7	110	64.0	52	100
Intentional introduction	180	67.9	60	34.8	0	
Natural spread	1	0.4	2	1.2	0	
Total	**265**		**172**		**52**	

The distribution of IAS can be divided into three zones. Most IAS are distributed in coastal provinces and the Yunnan province; provinces in Middle China have fewer IAS, and most provinces in West China have the least number of IAS (Fig. 1). Jimsonweed (*Datura stramonium*), cotton whitefly (*Bemisia tabaci*), two-spotted spider mite (*Tetranychus urticae*), American cockroach (*Periplaneta americana*), house mouse (*Mus musculus*), and brown rat (*Rattus norvegicus norvegicus*) occur in all provinces. Seventy IAS are distributed in more than half the number of provinces, and 105 IAS in more than one third of the provinces. Sites where IAS were first detected are mainly distributed in the coastal region, the Yunnan Province and the Xinjiang Uyghur Autonomous Region (Fig. 2), but there was a shift towards northern areas that became the main points of entry of IAS into China during the last two decades (Table 3).

Only 33 IAS invaded China before 1850, including spiny amaranth (*Amaranthus spinosus*), wattle (*Acacia farnesiana*) and common lantana (*Lantana camara*). The number of newly emerged IAS has been increasing since 1850 (Fig. 3). Two hundred and twelve new IAS (53.5% of IAS with known year or period of first detection) occurred since 1950, for example pine bast scale (*Matsucoccus matsumurae*), common cordgrass (*Spartina anglica*), and erythrina gall wasp (*Quadrastichus erythrinae*). The cumulative number of IAS grew exponentially (Fig. 3). It could be partially due to increased surveillance, but our figure is based on best estimates of species arrival dates.

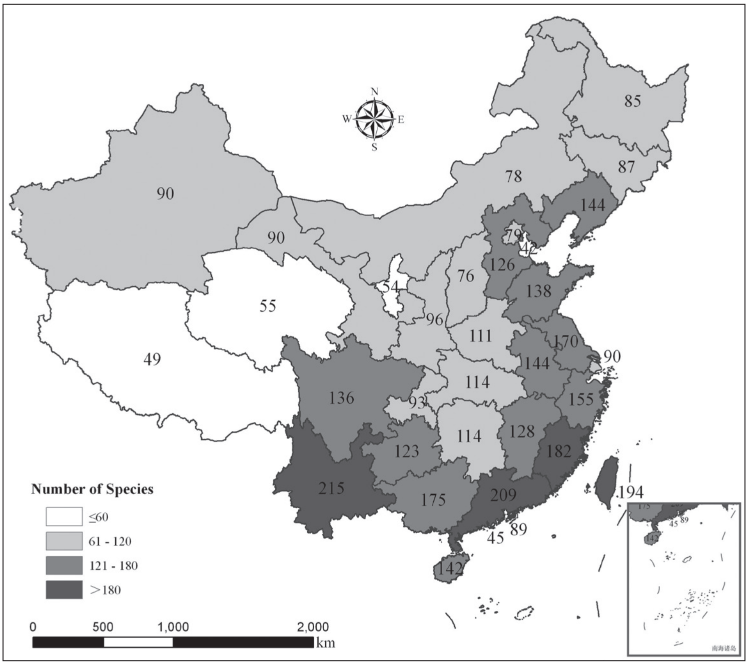

Figure 1. Regional distribution of IAS in China. Note that most IAS are distributed in coastal provinces and the Yunnan province.

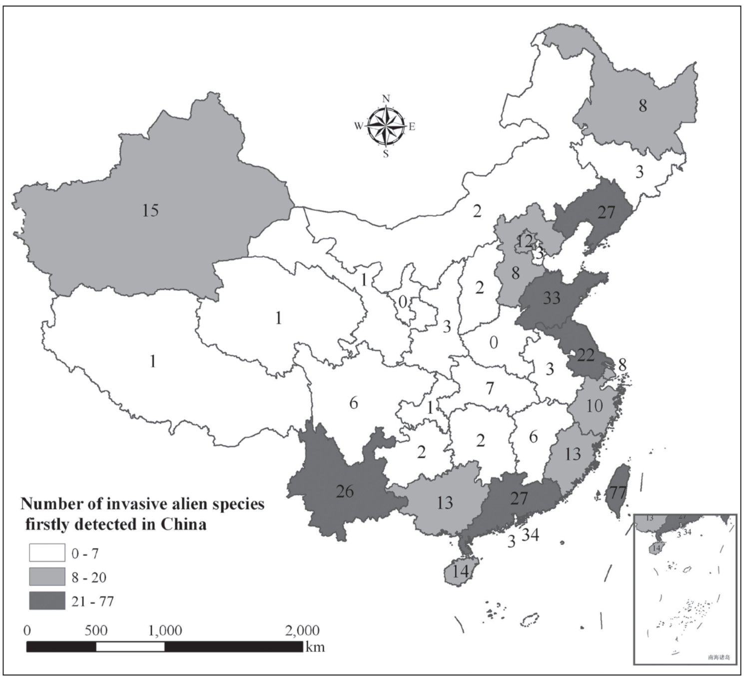

Figure 2. The distribution of first detections of IAS. First detections are concentrated in the coastal region, the Yunnan Province and the Xinjiang Uyghur Autonomous Region.

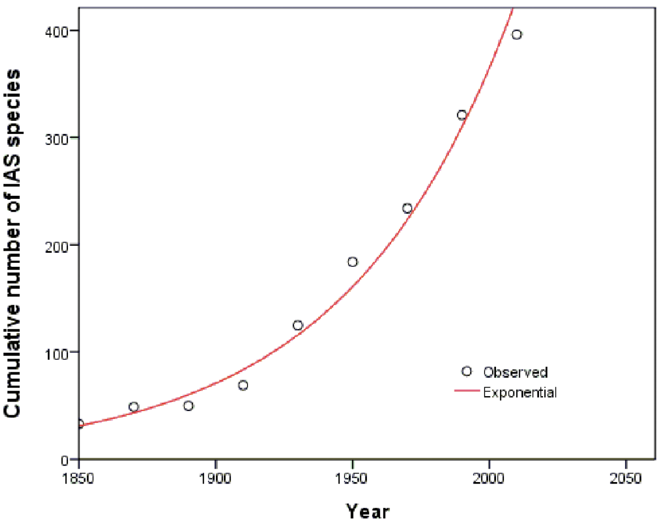

Figure 3. Temporal trends of invasions. Cumulative numbers of firstly detected IAS in China (exponential growth: R^2=0.981, P<0.001; N=396 IAS with known year or period of first detection in China) were analyzed. Only 33 IAS occurred in China before 1850, and 53.5% of the IAS were recorded after 1950.

Table 3. Temporal trends in the regions where invasive alien species were first detected in China. For each of the three periods since the 1950s, six top provinces or autonomous regions in which the most IAS were recorded are shown. The numbers are percentages of IAS that were firstly detected in the province, of the total number of species detected in China in the given period. Note that while southern areas were the most important points of entry in the first period, in the last two decades more invasions started in northern areas.

Province / region	1950–1969	Province / region	1970–1989	Province / region	1990–2009
Yunnan+	12.5	Taiwan+	21.8	Liaoning*	12.5
Taiwan+	12.5	Guangdong+	12.8	Shandong*	10.0
Guangdong+	12.5	Liaoning*	10.3	Taiwan+	10.0
Guangxi+	10.0	Shandong*	7.7	Hainan+	7.5
Xinjiang*	7.5	Beijing*	6.4	Guangdong+	7.5
Liaoning*	7.5	Yunnan+	6.4	Guangxi+	6.3

* northern provinces or autonomous regions; + southern provinces or autonomous regions

Discussion

The present study is, to our knowledge, the most up-to-date dataset of invasive species for China. However, we have to acknowledge biases that are inherent in the making of the inventory. For example, there are more plants than any other taxa, probably because plants are most numerous and easier to record. There may be biases in the timing of IAS discovery, as changes in resource allocation over time resulted in increasing opportunities for a more rigorous scientific research. It is likely that the survey pressure is not the same in all parts of China, depending on the staff numbers, among other parameters.

The cumulative number of IAS grew exponentially in China. Similar trends in historical accumulation of invasive forest insect pests and diseases have recently been reported from the United States (Aukema et al. 2010). An analysis of alien species in Europe has shown that human activity plays a key role in biological invasions (Pyšek et al. 2010, Jeschke and Genovesi 2011) and that the full effects of current socioeconomic patterns on the numbers of alien species can be delayed by several decades, resulting in what has been called an “invasion debt” (Essl et al. 2010). Our result shows that China is severely affected by invasions, with a pace of increase higher than that recorded in Europe (DAISIE 2009). Considering the fast economic growth of China, and the rapidly increasing levels of trade, tourism and transport, it is very likely that the country will face huge problems from invasive species in the future, and has already accumulated an invasion debt. For example, Europe – with a total surface similar to that of China (10 vs. 9.6 million km^2), but with about half the population of China (750 vs. 1340 million inhabitants) – hosts almost three times more IAS with ecological and/or economic impact than China (1347 species [Vilà et al. 2010] compared to the 488 reported in this paper). However, the number of IAS in China could be an underestimate due to the lower research intensity and limited monitoring activities. Nevertheless, the data from Europe and China, and taking into account China’s rapidly increasing economy suggest that the same trends will occur in other countries with fast growing economies where the levels of invasions are likely to increase as a result of economic activities. This imposes severe threats to global biodiversity and the ecosystem services of the concerned countries.

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Appendix

Brief information of IAS in China

No.	Taxon	Year when IAS was first detected	Places where IAS was first detected	Path-ways	Habitats	No. provinces / regions where IAS distributed
	Vira					
1	*Baculovirus midgut gland necrosis virus* (BMNV)	?	?	UI	OC	3
2	*Beet necrotic yellow vein virus* (BNYVV)	1978	Inner Mongolia	UI	FM	10
3	*Broad bean strain virus* (BBSV)	1998	Sichuan, Hubei, Jiangsu	UI	FM	5
4	*Cucumber green mottle mosaic virus* (CGMMV)	2005	Liaoning	UI	FM,	4
5	*Impatiens necrotic spot virus* (INSV)	2008	Yunnan	UI	FM	1
6	*Infectious hematopoietic necrosis virus* (IHNV)	1990	Liaoning	UI	OC	1
7	*Infectious pancreatic necrosis virus* (IPNV)	1980s	Liaoning, Shandong	UI	OC	5
8	*Lymphocystis disease virus* (LCDV)	1995	Shandong	UI	OC	14
9	*Poplar mosaic virus* (PMV)	1970s	Beijing	UI	FR	5
10	*Prunus necrotic ringspot ilarvirus* (PNRSV)	1999	Shaanxi	UI	FM	4
11	*Taura syndrome virus,*TSV	2000	Guangdong	UI	OC	3
12	*Tomato ringspot virus,*ToRSV	1986	Taiwan	UI	FM	1
	Procaryotae					
	Scotobacteria					
13	*Acidovorax avenae* subsp. *avenae* (Manns) Willems et al.	2003	Jiangsu	UI	FM	1
14	*Acidovorax avenae* subsp. *citrulli* (Schaad) Willems et al.	1986	?	UI	FM	8
15	*Pseudomonas savastanoi* (E.F.Smith) Stevens	1949	Guangxi	UI	FR	1
16	*Pseudomonas solanacearum* E.F.Smith	1982	Guangxi	UI	FR	3
17	*Pseudomonas syringae* pv. *actinidae* Takikawa et al.	1986	Hunan	UI	FR	11
18	*Pseudomonas syringae* pv. *tomato* (Okabe) Young, Dye & Wilkie	1998	Jilin	UI	FM	7
19	*Xanthomonas oryzae* pv. *oryzae* Swings et al.	1950	Jiangsu	UI	FM	21
20	*Xanthomonas oryzae* pv. *oryzicola* (Fang et al.) Swings et al.	1953	Guangdong	UI	FM	11
21	*Xanthomonas vesicatoria* Vauterin et al.	1991	?	UI	FM	7
	Firmibacteria					
22	*Clavibater michiganensis* (Smith) Davis et al. subsp. *michiganensis* (Smith) Davis et al.	1981	Beijing	UI	FM	5
23	*Clavibacter michiganense* subsp. *sepedonicum* Davis et al.	1996	Heilongjiang	UI	FM	15
	Protista					
	Centricae					
24	*Chaetoceros concavicornis* Mangin	1996	Hongkong	UI	OC	1

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	Pennatae					
25	*Pinnularia viridis* Nitzsch	1996	Hongkong	UI	OC	1
	Dinophyceae					
26	*Alexandrium minutum* Halim	1990s	Taiwan	UI	OC	2
	Fungi					
	Chytridiomycetes					
27	*Synchytrium endobioticum* (Schilbersky) Percival	1975	Yunnan	UI	FM	3
	Oomycetes					
28	*Albugo tragopogonis* (Pers.) S.F.Gray	2001	Xinjiang	UI	FM	1
29	*Peronosclerospora maydis* (Racib.) Shaw	1974	Shandong	UI	FM	6
30	*Peronosclerospora sorghi* (Weston & Uppal) Shaw	1974	Shandong	UI	FM	6
31	*Peronosclerospora philipinensis* (Weston) Shaw	1974	Shandong	UI	FM	6
32	*Peronosclerospora sacchari* (Miyake) Shirai & Hara	1974	Shandong	UI	FM	6
33	*Phytophthora parasitica* var. *nicotianae* (Breda de Haan) Tucker	1950	?	UI	FM	10
34	*Phytophthora sojae* Kaufm. & Gerd	1991	Heilongjiang	UI	FM	5
	Pyrenomycetes					
35	*Cryptodiaporthe populea* (Sacc.) Butinm, *Dothichiza populea* Sacc. & Br	1978	Jiangsu	UI	FR	6
	Loculoascomycetes					
36	*Botryosphaeria laricina* (Sawada) Shang	1970	Heilongjiang	UI	FR	8
37	*Mycosphaerella fijiensis* Morelet	?	?	UI	FM	1
38	*Venturia inaequalis* (Cooke)Wint, *Fusicladium dendriticum* (Wallr)	1927	Hebei	UI	FM	11
	Discomycetes					
39	*Lachnellula willkommii* (Hart.) Dennis	1975	Heilongjiang	UI	FR	5
	Teliomycetes					
40	*Cronartium ribicola* J.C.Fischer ex Rabenhorst	1958	Liaoning	UI	FR	15
	Hyphomycetes					
41	*Cephalosporium maydis* Samra, Sabet & Hingorani	1999	Taiwan	UI	FM	1
42	*Cylindrocladium scoparium* Morgan Hodges	1992	Guangxi	UI	FR	3
43	*Verticillium albo-atrum* Reinke & Berthold	1996	Xinjiang	UI	FM	1
44	*Fusarium oxysporum* Schlecht. f. sp. *asparagi* Cohen & Heald	1990	Taiwan	UI	FM	1
45	*Spilocaea oleaginea* (Cast.) Hugh	1964	Yunnan	UI	FR	7
46	*Verticillium dahliae* Kleb.	1935	?	UI	FM	20
47	*Fusarium oxysporum* f. sp. *cubense* Snyder & Hansen	1960	Guangxi	UI	FM	4
48	*Fusarium oxysporum* f. sp. *dianthi* (Prill. & Del) Snyd. & Hans	?	Shanghai	UI	FM	2

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49	*Fusarium oxysporium* Schl. f. sp. *vasinfectum* (Atk) Snyder & Hanson	1931	?	UI	FM	15
	Coelomycetes					
50	*Mycosphaerella pini* E.Rostrup	1982	Heilongjiang	UI	FR	5
51	*Phoma macdonaldii* Boerma	2008	Xinjiang	UI	FM	1
52	*Phomopsis asparagi* (Sacc.) Bubak	1993	Jiangsu	UI	FM	11
	Plantae					
	Rhodophyceae					
53	*Eucheuma striatum* Schmitz	1985	Hainan	II	OC	2
	Phaeophyta					
54	*Laminaria japonica* Aresch	1927	Liaoning	II	OC	8
55	*Macrocystis pyrifera* Agardh	1978	?	II	OC	2
56	*Undaria pinnatifida* Suringar	1984	?	II	OC	4
57	*Desmarestia ligulata* Lamouroux	2000	Liaoning	UI	OC	2
	Leptosporangiopsida					
58	*Salvinia molesta* D. S. Mitchell	?	Taiwan	UI	IW, OC	1
	Dicotyledoneae					
	Nymphaeales					
59	*Cabomba caroliniana* A. Gray	1993	Zhejiang	II	IW	3
	Ranunculales					
60	*Ranunculus arvensis* L.	Modern Times	Anhui	UI	FM	3
	Piperales					
61	*Peperomia pellucida* (L.) Kunth	Beginning of 20th century	Hongkong	UI	FR, FM	9
	Leguminosales					
62	*Acacia farnesiana* (L.) Willd.	1645	Taiwan	II	FM	9
63	*Chamaecrista minosoides* (L.) Green	Ming Dynasty	?	II	FM	8
64	*Crotalaria incana* L.	1953	Guangxi	II	FM	7
65	*Crotalaria juncea* L.	End of 19th century	Taiwan	II	FM, FR	8
66	*Crotalaria lanceolata* E. Mey.	Middle 20th century	?	II	FM	3
67	*Crotalaria mincans* L.	1910	Taiwan	II	FM	7
68	*Crotalaria ochroleuca* G. Don	1955	Guangxi	II	FM	4
69	*Crotalaria trichotoma* Bojer	1931	Taiwan	II	FM	8
70	*Desmodium tortuosum* (Sw.) DC.	1963	Hongkong	II	FM	2
71	*Indigofera suffruticosa* Mill.	1861	Hongkong	II	FM, FR	0
72	*Leucaena leucocephala* (Lam.) de Wit	1945	Taiwan	II	FM, FR	11
73	*Macroptilium atropurpureum* (Moc. & Sessé ex DC.) Urb.	1969	Guangdong	II	FM	3
74	*Medicago minima* Lam.	1910	Jiangxi	UI	FM	11
75	*Medicago polymorpha* L.	?	?	II	FM	8

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76	*Medicago sativa* L.	100 B.C	Shaanxi	II	FM	26
77	*Melilotus albus* Desr.	1918	Shandong	II	FM	17
78	*Melilotus indicus* (L.) All.	1918	Shandong	II	FM	18
79	*Mimosa bimucronata* (DC.) Kuntze	1950s	Guangdong	II	FM	3
80	*Mimosa invisa* Mart. ex Colla	1950	Guangdong	II	FM	3
81	*Mimosa pudica* L.	Ming Dynasty	?	II	FM, FR	9
82	*Neptunia plena* (L.) Benth.	1963	Guangdong	II	FM	1
83	*Robinia pseudoacacia* L.	1903	Shandong	II	FM	20
84	*Senna alata* (L.) Roxb.	1909	Taiwan	II	FM, FR	4
85	*Senna hirsuta* (L.) H. S. Irwin & Barneby	1927	Guangdong	II	FM	5
86	*Senna occidentalis* (L.) Link	16th century	?	II	FM, FR	10
87	*Senna tora* (L.) Roxb.	16th century	Shaanxi	II	FM	12
88	*Sesbania cannabina* (Retz.) Pers.	1910	Jiangsu	II	IW	10
89	*Trifolium fragiferum* L.	1931	Xinjiang	II	FM	1
90	*Trifolium hybridum* L.	1930	Shanghai	II	FM	6
91	*Trifolium incarnatum* L.	1950s	?	II	FM	15
92	*Trifolium pratense* L.	19th century		II	FM	15
93	*Trifolium repens* L.	19th century		II	FM	27
94	*Ulex europaeus* L.	1862	Sichuan	II	FM	1
95	*Vicia sativa* L.	1940s	Gansu, Jiangsu	II	FM	30
96	*Vicia villosa* Roth	1932	Shandong	II	FM	22
	Urticales					
97	*Cannabis sativa* L.	?	?	II	FM	28
98	*Pilea microphylla* (L.) Liebm.	1928	Taiwan	UI	FM	11
	Capparales					
99	*Cleome rutidosperma* DC.	1958	Yunnan	II	FM, FR	7
100	*Reseda lutea* L.	1974	Liaoning	II	FM	1
	Passiflorales					
101	*Passiflora foetida* L.	1861	Hongkong	II	FM	7
102	*Passiflora suberosa* L.	1907	Taiwan	II	FM	3
	Cucurbitales					
103	*Sicyos angulatus* L.	1999	Taiwan	II	FM, FR	4
	Cactales					
104	*Opuntia ficus-indica* (L.) Mill.	1945	Taiwan	II	FM	5
105	*Opuntia monacantha* (Willd.) Haw.	1625	Yunnan	II	FM	6
106	*Opuntia stricta* (Haw.) Haw. *var. dillenii* (Ker Gawl.) L. D. Benson	1702	Guangdong	II	FM, FR	5
	Tiliales					
107	*Waltheria indica* L.	1861	Hongkong	UI	FM	7
	Malvales					
108	*Herissantia crispa* (L.) Brizicky	1932	Hainan	UI	FM, FR	1

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109	*Hibiscus trionum* L.	?	?	UI	FM	29
110	*Malvastrum coromandelianum* (L.) Garcke	19th century	Hongkong	UI	FM	8
	Euphorbiales					
111	*Euphorbia dentata* Michx.	1976	Beijing	II	FM	6
112	*Euphorbia hirta* L.	1820	Macco	UI	FM	14
113	*Euphorbia maculata* L.	1940s	Shanghai	UI	FM	12
114	*Euphorbia marginata* Pursh	1935	Beijing	II	FM	3
115	*Euphorbia nutans* (Lag.) Small	20th century	Liaoning, Jiangsu, Anhui	UI	FM	5
116	*Jatropha curcas* L.	300 year ago	?	II	FM	8
117	*Ricinus communis* L.	?	?	II	FM	18
	Myrtales					
118	*Eucalyptus robusta* Sm.	1890	Guangdong	II	FM	5
119	*Syzygium jambos* (L.) Alston	Before 17th century	?	II	FM	6
120	*Clarkia pulchella* Pursh.	1965	Tibet	II	FM	1
121	*Gaura parviflora* Douglas ex Lehm.	1930	Shandong	II	FM, FR	7
122	*Oenothera biennis* L.	1918	Shandong	II	FM, FR	10
123	*Oenothera drummondii* Hook.	1930	Fujian	II	FM	4
124	*Oenothera glazioviana* Micheli	17th century	Yunnan, Jiangsu	II	FM	20
125	*Oenothera laciniata* Hill.	1985	Taiwan	II	FM	3
126	*Oenothera oakesiana* (A. Gray) J. W. Robbins ex S. Watson & J. M. Coult.	20th century	Fujian	II	FM	1
127	*Oenothera parviflora* L.	1951	Liaoning	II	FM	1
128	*Oenothera rosea* L'Hér. ex Ait.	1957	Jiangsu	II	FM	5
129	*Oenothera stricta* Ledeb. & Link	1917	Zhejiang	II	FM	9
130	*Oenothera tetraptera* Cav.	1935	Guizhou	II	FM	3
131	*Oenothera villosa* Thunb.	1959	Heilongjiang	II	FM	7
	Rhamnales					
132	*Parthenocissus quinquefolia* (L.) Planch.	1951	Liaoning	II	FM, FR	7
	Apocynales					
133	*Catharanthus roseus* (L.) G.Don	1661	?	II	FM	9
134	*Asclepias curassavica* L.	1928	Guangdong	II	FM	10
	Rubiales					
135	*Borreria latifolia* (Aubl.) K. Schum	1937	Guangdong	II	FM	7
	Verbenales					
136	*Duranta erecta* L.	Ming Dynasty	Taiwan	II	FM	6
137	*Lantana camara* L.	1645	Taiwan	II	FM	9
138	*Lantana montevidensis* (Spreng.) Briq.	1928	Taiwan	II	FM	5
139	*Stachytarpheta jamaicensis* (L.) Vahl	Beginning of 19th century	Hongkong	UI	FM	8

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	Cruciales					
140	*Armoracia rusticana* (Lam.) Gaertn., B. Mey. & Scherb.	Beginning of 20th century	Shanghai	II	FM	4
141	*Coronopus didymus* (L.) Sm.	1930s	Jiangsu	UI	FM	13
142	*Diplotaxis muralis* (L.) DC	1907	?	UI	FM	1
143	*Lepidium campestre* (L.) R.Br.	1925	Liaoning	UI	FM	8
144	*Lepidium densiflorum* Schrad	1931	Liaoning	UI	FM	2
145	*Lepidium virginicum* L.	1933	Hubei	UI	FM	23
146	*Raphanus raphanistrum* L.	1959	Sichuan	UI	FM	2
147	*Sinapis alba* L.	?	?	II	FM	6
148	*Sinapis arvensis* L.	?	?	II	FM	24
	Caryophyllales					
149	*Agrostemma githago* L.	19th century	?	UI	FM	6
150	*Saponaria officinalis* L.	1928	Liaoning	II	FM, FR	3
151	*Stellaria pallida* (Dumort.) Crép.	1949	Shanghai	UI	FM	5
152	*Vaccaria hispanica* (Mill.) Rauschert	?	?	UI	FM	15
153	*Portulaca pilosa* L.	1929	Taiwan	II	FM	6
154	*Talinum paniculatum* (Jacq.) Gaertn.	16th century	Jiangsu	II	FM	4
155	*Mirabilis jalapa* L.	16th century	Zhejiang	II	FM	14
156	*Anredera cordifolia* (Ten.) Steenis	1976	Taiwan	II	FM	11
	Chenopodiales					
157	*Phytolacca americana* L.	1935	Zhejiang	II	FM	21
158	*Chenopodium ambrosioides* L.	1864	Taiwan	UI	FM	11
159	*Chenopodium hybridum* L.	1864	Hebei	UI	FM	19
160	*Salicornia bigelovii* Torr.	2001	Guangxi	II	FM	5
161	*Alternanthera paronychioides* A.St.-Hil.	1969	Taiwan	II	FM	4
162	*Alternanthera philoxeroides* (Mart.) Griseb.	1930s	Shanghai	II	FM, IW	20
163	*Alternanthera pungens* Kunth	1950s	Fujian	UI	FM	5
164	*Amaranthus albus* L.	1929	Tianjin	UI	FM	5
165	*Amaranthus blitoides* S. Watson	1857	Liaoning	UI	FM	6
166	*Amaranthus caudatus* L.	Qing Dynasty	Heilongjiang	II	FM	29
167	*Amaranthus cruentus* L.	1848	?	II	FM	0
168	*Amaranthus hybridus* L.	1848	?	UI	FM	10
169	*Amaranthus palmeri* S. Watson	1985	Beijing	UI	FM, FR	4
170	*Amaranthus polygonoides* L.	1979	Shandong	UI	FM	4
171	*Amaranthus retroflexus* L.	Middle of 19th century	Hebei, Shandong	II	FM	28
172	*Amaranthus spinosus* L.	1836	Macco	UI	FM	24
173	*Amaranthus tricolor* L.	10th century	?	II	FM	29
174	*Amaranthus viridis* L.	1864	Taiwan	UI	FM	19

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175	*Gomphrena celosioides* Mart.	1968	Hongkong	II	FM	4
	Lythrales					
176	*Cuphea carthagenensis* (Jacq.) J. F. Macbr.	1960	Taiwan	II	FM	2
	Plantaginales					
177	*Plantago aristata* Michx.	1929	Shandong	UI	FM	2
178	*Plantago virginica* L.	1951	Jiangxi	UI	FM, FR	10
	Saxifragales					
179	*Bryophyllum pinnatum* (Lam.) Oken	1861	Hongkong	II	FM	7
	Umbelliflorae					
180	*Coriandrum sativum* L.	?	?	II	FM	8
181	*Cyclospermum leptophyllum* (Pers.) Sprague	Beginning of 20th century	Hongkong	UI	FM	11
182	*Daucus carota* L.	?	?	UI	FM	30
183	*Eryngium foetidum* L.	1897	Yunnan	II	FM, FR	4
	Campanulales					
184	*Triodanis biflora* (Ruiz & Pav.) Greene	1981	Anhui	UI	FM	4
185	*Triodanis perfoliata* (L.) Nieuwl.	1974	Fujian	UI	FM	3
	Asterales					
186	*Acanthospermum australe* (Loefl.) Kuntze	1936	Yunnan	UI	FM	2
187	*Achillea millefolium* L.	1918	Shandong	II	FM	7
188	*Ageratina adenophora* (Spreng.) R.M.King & H. Rob.	1940s	Yunnan	NS	FM, FR	5
189	*Ageratum conyzoides* L.	19th century	Hongkong	II	FM, FR	16
190	*Ageratum houstonianum* Mill.	1911	Taiwan	II	FM	11
191	*Ambrosia artemisiifolia* L.	1930s	Zhejiang	UI	FM	18
192	*Ambrosia trifida* L.	1930s	Liaoning	UI	FM	5
193	*Anthemis arvensis* L.	1918	Shandong	II	FM	2
194	*Aster subulatus* Michx.	1947	Hubei	UI	FM	9
195	*Bidens frondosa* L.	1926	Jiangsu	UI	FM	6
196	*Bidens pilosa* L.	1857	Hongkong	UI	FM	13
197	*Centaurea cyanus* L.	1918	Shandong	II	FM	2
198	*Chromolaena odorata* (L.) R.M.King & H. Rob.	1936	Yunnan	II	FM	4
199	*Chrysanthemum carinatum* Schousb.	1914	Hunan	II	FM	6
200	*Chrysanthemum coronarium* L.	?	?	II	FM	8
201	*Cichorium intybus* L.	1918	Shandong	II	FM	6
202	*Conyza bonariensis* (L.) Cronquist	1857	Hongkong	UI	FM	10
203	*Conyza canadensis* (L.) Cronquist	1860	Shandong	UI	FM	21
204	*Conyza sumatrensis* (Retz.) E.Walker	Middle of 19th century	?	UI	FM, FR	18
205	*Coreopsis grandiflora* Hogg ex Sweet	1932	Shandong	II	FM	2
206	*Coreopsis lanceolata* L.	1911	Taiwan	II	FM	8
207	*Coreopsis tinctoria* Nutt.	1911	Taiwan	II	FM	5

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208	*Cosmos bipinnatus* Cav.	1911	Taiwan	II	FM	8
209	*Cosmos sulphureus* Cav.	1938	Taiwan	II	FM	8
210	*Crassocephalum crepidioides* (Benth.) S. Moore	1930s	?	UI	FM	19
211	*Crassocephalum rubens* (Juss. ex Jacq.) S. Moore	2008	Yunnan	II	FM	1
212	*Erechtites hieracifolia* (L.) Raf. ex DC.	1933	Hainan	II	FM, FR	8
213	*Erechtites valerianifolia* (Wolf) DC.	1920s	Taiwan	II	FM	4
214	*Erigeron annuus* (L.) Pers.	1886	Shanghai	UI	FM	20
215	*Erigeron philadelphicus* L.	End of 19th century		UI	FM, FR	4
216	*Eupatorium catarium* Veldkamp	1980s	Hongkong	UI	FM, FR	8
217	*Flaveria bidentis* (L.) Kuntze	1980s -1990s	Tianjin	II	FM, FR	3
218	*Galinsoga parviflora* Cav.	1915	Yunan, Sichuan	UI	FM	21
219	*Galinsoga quadriradiata* Ruiz & Pav.	1943	Sichuan	II	FM, FR	10
220	*Gnaphalium pensylvanicum* (Willd.) Cabrera	1932	Hainan	UI	FM	11
221	*Gymnocoronis spilanthoides* (D. Don ex Hook. & Arn.) DC.	2006	Guangxi	II	FM	1
222	*Halianthus tuberosus* L.	1918	Shandong	II	FM	20
223	*Helenium autumnale* L.	Morden Times	?	II	FM	9
224	*Leucanthemum vulgare* Lam.	1910	Jiangxi	II	FM	5
225	*Mikania micrantha* Kunth	1919	Hongkong	II	FM	5
226	*Parthenium hysterophorus* L.	1926	Yunnan	II	FM	8
227	*Pluchea sagittalis* (Lam.) Cabrera	End of 20th century	Taiwan	UI	FM	2
228	*Pseudelephantopus spicatus* (B. Juss. ex Aubl.) C.F. Baker	1932	Taiwan	UI	FM	2
229	*Pyrethrum parthenifolium* Willd.	1933	Yunnan	II	FM	1
230	*Senecio vulgaris* L.	19th century		UI	FM	14
231	*Silybum marianum* (L.) Gaertn.	1941	Yunnan	II	FM	3
232	*Solidago canadensis* L.	1935	Shanghai	II	FM, FR	9
233	*Soliva anthemifolia* (Juss.) R.Br.	1912	Hongkong	UI	FM	5
234	*Sonchus asper* (L.) Hill.	?	?	UI	FM	30
235	*Sonchus oleraceus* L.	?	?	UI	FM	31
236	*Synedrella nodiflora* (L.) Gaertn.	1912	Hongkong	UI	FM	9
237	*Tagetes erecta* L.	?	Yunnan	II	FM	5
238	*Tagetes patula* L.	1931	Guangdong	II	FM	3
239	*Tithonia diversifolia* (Hemsl.) A. Gray	1910	Taiwan	II	FM	6
240	*Tridax procumbens* L.	1947	Hainan, Guangdong	UI	FM	8
241	*Wedelia trilobata* (L.) Hitchc.	1970s	?	II	FM	7
242	*Xanthium italicum* Moretti	1991	Beijing	UI	FM	3

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243	*Xanthium spinosum* L.	1974	Beijing	UI	FM	5
244	*Zinnia peruviana* (L.) L.	1919		II	FM	8
	Solanales					
245	*Datura innoxia* Mill.	1905	Beijing	II	FM	10
246	*Datura metel* L.	?	?	II	FM	17
247	*Datura stramonium* L.	?	?	II	FM	34
248	*Nicandra physalodes* (L.) Gaertn.	1840s	Hongkong	II	FM	14
249	*Physalis angulata* L.	Middle of 19th century	Hongkong	UI	FM	19
250	*Solanum aculeatissimum* Jacq.	end of 19th century	Guizhou	UI	FM	11
251	*Solanum capsicoides* All.	1895	Hongkong	UI	FM, FR	11
252	*Solanum erianthum* D. Don	1857	Fujian	UI	FM, FR	10
253	*Solanum rostratum* Dunal	1895	Hongkong	UI	FM	5
254	*Solanum sisymbriifolium* Lam.	1980s	Yunnan	II	FM	1
255	*Solanum torvum* Sw.	1827	Macco	II	FM, FR	10
256	*Ipomoea cairica* (L.) Sweet	1912	Hongkong	II	FM, FR	8
257	*Ipomoea indica* (Burm.) Merr.	1942	Taiwan	II	FM, FR	2
258	*Ipomoea nil* (L.) Roth	Ming Dynasty	Zhejiang	II	FM	23
259	*Ipomoea purpurea* (L.) Roth	1890	?	II	FM, FR	9
260	*Ipomoea triloba* L.	1970s	Taiwan	II	FM, FR	5
261	*Jacquemontia tamnifolia* (L.) Griseb.	End of 20th century	Guangdong	II	FM	2
	Scrophulariales					
262	*Scoparia dulcis* L.	Middle of 19th century	Hongkong	II	FM	8
263	*Veronica arvensis* L.	1910	Jiangxi	UI	FM	9
264	*Veronica hederifolia* L.	1980s	Jiangsu	UI	FM, FR	2
265	*Veronica peregrina* L.	?	?	UI	FM	15
266	*Veronica persica* Poir.	1933	Hubei	UI	FM	12
267	*Veronica polita* Fr.	?	?	UI	FM	2
268	*Justicia adhatoda* L.	1850	Hongkong	II	FM	6
269	*Orobanche brassicae* Novopokr.	1977	Fujian	UI	FM	1
270	*Martynia annua* L.	1964	Yunnan	II	FM	1
271	*Macfadyena unguis-cati* (L.) A.H.Gentry	1840	Fujian	II	FR	2
	Geraniales					
272	*Geranium carolinianum* L.	1926	Jiangsu	II	FM	15
273	*Oxalis corymbosa* DC.	Middle of 19th century	Hongkong	II	FM	31
	Boraginales					
274	*Heliotropium europaeum* L.	1934	Shanxi	UI	FM	6
	Lamiales					
275	*Hyptis brevipes* Poit.	1925	Taiwan	II	FM	3

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276	*Hyptis rhomboidea* Mart. & Galeotti	1992	Hainan	II	FM	3
277	*Hyptis suaveolens* (L.) Poit.	End of 19th century	Taiwan	II	FM, FR	7
278	*Stachys arvensis* L.	1864	Taiwan	II	FM	6
	Monocotyledoneae					
	Alismatales					
279	*Limnocharis flava* (L.) Buchenau	Modern Times	?	II	IW	3
	Liliflorae					
280	*Eichhornia crassipes* (Mart.) Solms	Beginning of 20th century	Taiwan	II	IW	16
	Arales					
281	*Pistia stratiotes* L.	Ming Dynasty	?	II	IW	18
	Graminales					
282	*Aegilops tauschii* Coss.	?	?	II	FM	6
283	*Avena fatua* L.	Middle of 19th century	Hongkong, Fujian	UI	FM	30
284	*Axonopus compressus* (Sw.) P. Beauv.	1940	Taiwan	II	FM, FR	7
285	*Brachiaria mutica* (Forsk.) Stapf	1930s	Taiwan	II	FM	2
286	*Bromus catharticus* Vahl	Middle of 20th century	Jiangsu, Yunnan	II	FM	2
287	*Buchloe dactyloides* (Nutt.) Engelm.	1950s	Beijing	II	FM	2
288	*Cenchrus echinatus* L.	1934	Taiwan	UI	FM	7
289	*Cenchrus incertus* M. A. Curtis	Beginning of 20th century	Taiwan	UI	FM	10
290	*Ehrharta erecta* Lam.	1998	Yunnan	II	FM	1
291	*Hordeum jubatum* L.	?	?	II	FM	3
292	*Lolium multiflorum* Lam.	18th century	?	II	FM	20
293	*Lolium perenne* L.	1918	Shandong	II	FM	20
294	*Lolium persicum* Boiss. & Hohen. ex Boiss.	1958	Xinjiang	II	FM	2
295	*Lolium temulentum* L.	1940s		UI	FM	17
296	*Lolium temulentum* L. var. *arvense* (With.) Lilj.	Modern Times		II	FM	6
297	*Lolium temulentum* L. var. *longiaristatum* Parnell	1940s	Qinghai	UI	FM	6
298	*Panicum dichotomiflorum* Michx.	1908	Taiwan	UI	FM	3
299	*Panicum maximum* Jacq.	1908	Taiwan	II	FM, FR	7
300	*Panicum repens* L.	1857	Hongkong	II	FM	6
301	*Paspalum conjugatum* P. J. Bergius	1912	Hongkong	II	FM	12
302	*Paspalum dilatatum* Poir.	1953		II	FM	7
303	*Paspalum fimbriatum* Kunth	1971	Taiwan	II	FM	1
304	*Pennisetum clandestinum* Hochst. ex Chiov.	1958	Taiwan	II	FM, FR	1

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305	*Pennisetum polystachyon* (L.) Schult.	1961	Taiwan	II	FM	3
306	*Pennisetum purpureum* Schumach.	1930s	Guangdong, Sichuan	II	FM	9
307	*Phalaris minor* Retz.	1958	Beijing	II	FM	1
308	*Phalaris paradoxa* L.	1958	Beijing	II	FM	1
309	*Phleum pratense* L.	1925	Henan	II	FM	8
310	*Poa compressa* L.	1914	Hebei	II	FM	5
311	*Rhynchelytrum repens* (Willd.) C.E.Hubb.	1950s	?	II	FM	5
312	*Sorghum almum* Parodi	2009	Guangxi	UI	FM	1
313	*Sorghum halepense* (L.) Pers.	Beginning of 20th century	Taiwan	UI	FM	17
314	*Sorghum sudanense* (Piper) Stapf	1922	Jiangxi	II	FM	16
315	*Spartina alterniflora* Loisel.	1979	Fujian	II	OC	7
316	*Spartina anglica* C.E. Hubb.	1964	Jiangsu	II	OC	9
317	*Vetiveria zizanioides* L.	1936	Hainan	II	FM	3
	Animalia					
	Nematoda					
	Aphelenchida					
318	*Aphelenchoides besseyi* Christie	?	?	UI	FM	13
319	*Aphelenchoides ritzema-bosi* (Schwartz) Steiner	1970s	Jiangsu	UI	FM	9
320	*Bursaphelenchus xylophilus* (Steiner & Buhrer) Nickle	1982	Jiangsu	UI	FR	9
321	*Anguina agrostis* (Steinbuch) Filipjev	1987	Inner Mongolia	UI	FM	3
322	*Ditylenchus dispaci* (Khn) Filipjev	?	?	UI	FM	4
323	*Radopholus similes* (Cobb) Thorne	?	?	UI	FM	1
324	*Heterodera glycines* Ichinohe	1899	?	UI	FM	12
325	*Meloidogyne hispanica* Hirschmann	2007	Hainan	UI	FM	1
	Gastropoda					
	Archaeogastropoda					
326	*Haliotis laevigata* Donovan	1998	Guangdong	II	OC	1
327	*Haliotis discus discus* Reeve	1986	Guangdong	II	OC	3
328	*Haliotis gigantea* Gmelin	1997	Liaoning	II	OC	2
329	*Haliotis rufescens* Swainson	1985	Liaoning	II	OC	2
330	*Haliotis fulgens* Philippi	1985	Liaoning	II	OC	3
	Mesogastropoda					
331	*Crepidula onyx* Sowerby	1979	Hongkong	UI	OC	2
332	*Pomacea canaliculata* Lamark	?	Taiwan	II	IW	13
	Stylomnatophora					
333	*Achatina fulica* Bowdich	1920s	Fujian	II	FM	6
334	*Lehmannia valentiana* (Férussac)	?	?	UI	FM	5
	Bivalvia					
	Pterioida					
335	*Argopecten irradians* Lamarck	1982	?	II	OC	3
336	*Patinopecten yessoensis* Jay	1981	Liaoning	II	OC	1

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	Ostreoida					
337	*Crassostrea gigas* Thunberg	1979	Zhejiang	II	OC	14
	Veneroida					
338	*Mercenaria mercenaria* L.	1997	Shandong	II	OC	2
339	*Mytilopsis sallei* Recluz	1977	Taiwan	UI	OC	5
	Myoida					
340	*Panopea abrupta* Conrad	1998	Shandong	II	OC	1
	Malacostraca					
341	*Litopenaeus stylirostris* Stimpson	2000	Shandong, Jiangsu, Zhejiang	II	OC	3
342	*Litopenaeus vannamei* Boone	1988	?	II	OC	9
343	*Marsupenaeus japonicus* Bate	?	?	II	OC	
344	*Procambarus clarkii* Girard	1929	Jiangsu	UI	IW	10
345	*Cherax quadricainatus* Von Martens	1980s	Jiangxi	II	IW	2
	Arachnida					
346	*Aculops lycopersici* (Maass)	1980	Guangxi	UI	FM	5
347	*Tetranychus urticae* Koch	1978	Taiwan	UI	FM	34
	Insecta					
	Blattodea					
348	*Blattella germanica* (L.)	1935	?	UI	RS	28
349	*Periplaneta americana* (L.)	?	?	UI	RS	34
350	*Periplaneta australasiae* Fabricius	?	?	UI	RS	10
	Isoptera					
351	*Cryptotermes domesticus* (Haviland)	1917	Taiwan	UI	FR, RS	5
352	*Incisitermes minor* (Hagen)	1937	HongKong	UI	RS	3
	Thysanoptera					
353	*Frankliniella occidentalis* (Pergande)	2000	Taiwan	UI	FM	6
354	*Taeniothrips simplex* (Morison)	1987	Taiwan	UI	FM	11
	Hemiptera					
355	*Eurygaster integriceps* Puton	?	?	UI	FM	4
356	*Corythucha ciliata* Say	2006	Hubei	UI	FR	7
357	*Heterosylla cubana* Crauford	1985	Taiwan	UI	FR	5
358	*Aleurodicus dispersus* Russell	1988	Taiwan	UI	FM	2
359	*Bemisia tabaci* (Gennadius)	1949	Taiwan, Yunnan	UI	FM	34
360	*Trialeurodes vaporariorum* (Westwood)	?	?	UI	FM	25
361	*Aphanostigma piri* Cholodkovsky	1979	Taiwan	UI	FM	1
362	*Moritziella castaneivora* Miyazaki	1997	Shandong	UI	FR	3
363	*Viteus vitifoliae* (Fiech)	1892	Shandong	UI	FM	5
364	*Eriosoma lanigerum* (Hausmann)	1914	Shandong	UI	FM	10
365	*Icerya purchasi* Maskell	1904	Taiwan	UI	FM	18
366	*Dysmicoccus brevipes* Cockerell	1921	Taiwan	UI	FM	7
367	*Dysmicoccus neobrevipes* (Beardsley)	1998	Hainan	UI	FM	3
368	*Oracella acuta* (Lobdell)	1990	Guangdong	UI	FR	3

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369	*Phenacoccus solenopsis* Tinsley	2008	Guangdong	UI	FM	2
370	*Parasaissetia nigra* Nietner	1989	Yunnan	UI	FM	6
371	*Matsucoccus matsumurae* (Kuwana.)	1950	Shandong	UI	FR	7
372	*Hemiberlesia pitysophila* Takagi	1982	Guangdong	UI	FR	4
	Coleoptera					
373	*Agrilus mali* Matsumura	1934	Liaoning	UI	FM	13
374	*Anthrenus verbasci* L.	?	?	UI	RS	23
375	*Trogoderma granarium* Everts	1962	?	UI	RS, FM	1
376	*Lasioderma serricorne* (Faericus)	1931	Taiwan	UI	RS	32
377	*Heterobostrychus aequalis* (Waterhouse)	1988	Guangdong	UI	RS	6
378	*Rhyzopertha dominica* (Fabricius)	?	?	UI	RS	31
379	*Necrobia ruficollis* (Fabricius)	?	?	UI	RS	9
380	*Necrobia rufipes* Degcer	?	?	UI	RS	18
381	*Cathartus advena* Walterl	?	?	UI	RS	32
382	*Tribolium confusum* Jacquelin du Val	?	?	UI	RS	19
383	*Pharaxonotha kirschii* Reitter	1987	Yunnan	UI	RS	2
384	*Xylotrechus rusticus* L.	1970s	Liaoning	UI	FR	5
385	*Acanthoscelides macrophthalmus* Schaeffer	1999	Hainan	UI	FR	7
386	*Acanthoscelides obtectus* Say	1990	Jilin	UI	RS, FM	2
387	*Acanthoscelides pallidipennis* Motschulsky	1980	Hebei	UI	FR	11
388	*Bruchidius dorsalis* Fabricius	?	?	UI	RS, FR	16
389	*Bruchus pisorum* (L.)	?	?	UI	RS, FM	32
390	*Bruchus rufimanus* Boheman	1930s – 1940s	?	UI	RS, FM	7
391	*Callosobruchus analis* (Fabricius)	?	?	UI	RS, FM	1
392	*Callosobruchus maculatus* (Fabricius)	?	Hongkong	UI	RS	12
393	*Callosobruchus phaseoli* (Chevrolate)	1998	Zhejiang	UI	RS	2
394	*Zabrotes subfasciatus* (Boheman)	1987	Chongqing	UI	RS	2
395	*Araecerus fasciculatus* (Degeer)	?	?	UI	RS	18
396	*Leptinotarsa decemlineata* (Say)	1993	Xinjiang	UI	FM	1
397	*Brontispa longissima* (Gestro)	1975	Taiwan	UI	FM	7
398	*Octodonta nipae* (Maulik)	2001	Hainan	UI	FM	1
399	*Cosmopolites sordidus* Germar	1909	Taiwan	UI	FM	6
400	*Cryptorhynchus lapathi* L.	1953	Jilin	UI	FR	9
401	*Diocalandra frumenti* (Fabricius)	1977	Taiwan	UI	FM	2
402	*Hypera postica* (Gyllenhal)	1950s	Xinjiang	UI	FM	2
403	*Lissorhoptrus oryzophilus* Kuschel	1988	Hebei	UI	FM	8
404	*Rhabdoscelus lineaticollis* (Heller)	1986	Taiwan	UI	FM	3
405	*Rhynchophorus ferrugineus* (Oliver)	?	Hainan	UI	FM	9
406	*Sitophilus granarius* (L.)	?	Xinjiang	UI	RS, FM	5
407	*Sternochetus frigidus* Fabricius	?	?	UI	FM	4
408	*Sternochetus mangiferae* Fabricius	?	?	UI	FM	2
409	*Sternochetus olivieri* (Faust)	1914	Yunnan	UI	FM	3
410	*Cylas formicarius* (Summers)	1965	Zhejiang	UI	FM	10
411	*Dendroctonus valens* Leconte	1998	Shanxi	UI	FR	4

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	Diptera					
412	*Contarinia sorghicola* (Coquillett)	?	?	UI	FM	13
413	*Mayetiola destructor* (Say)	1960 – 1970	Xinjiang	UI	FM	1
414	*Obolodiplosis robiniae* Haldemann	2005	Liaoning	UI	FR	5
415	*Liriomyza bryoniae* (Kaltenbach)	1984	Taiwan	UI	FM	10
416	*Liriomyza huidobrensis* (Blanchard)	1993	Yunnan	UI	FM	21
417	*Liriomyza sativae* Blanchard	1993	Hainan	UI	FM	33
418	*Liriomyza trifolii* (Burgess)	1988	Taiwan	UI	FM	3
419	*Bactrocera correcta* (Bezzi)	1989	Yunnan	UI	FM	2
420	*Bactrocera* (*Zeugodacus*) *cucuribitae* (Coquillett)	?	?	UI	FM	10
421	*Bactrocera dorsalis* (Hendel)	1911	Taiwan	UI	FM	15
422	*Bactrocera tsuneonis* Miy	1956	Guangxi	UI	FM	19
423	*Carpomya vesuviana* Costa	2007	Xinjiang	UI	FM	1
	Lepidoptera					
424	*Anarsia lineatella* Zeller	?	?	UI	FM	3
425	*Pectinophora gossypiella* (Saunders)	1988	Hebei	UI	FM	18
426	*Phthorimaea operculella* (Zeller)	1937	Guangxi	UI	FM, RS	14
427	*Sitotroga cerealella* Olivier	?	?	UI	FM, RS	32
428	*Opogona sacchari* (Bojer)	1987	Guangdong	UI	FR, FM	10
429	*Laspeyresia pomonella* (L.)	?	Xinjiang	UI	FM	2
430	*Corcyra cephalonica* Stainton	?	?	UI	RS	8
431	*Paralipsa gularis* (Zeller)	?	?	UI	RS	30
432	*Plodia interpunctella* (Zeller)	?	?	UI	RS, FM	33
433	*Hyphantria cunea* (Drury)	1979	Liaoning	UI	FR, FM	6
434	*Erionota torus* Evans	1940s	Fujian	UI	FM	8
	Hymenoptera					
435	*Leptocybe invasa* Fisher & LaSalle	2007	Guangxi	NS/UI	FR	3
436	*Quadrastichus erythrinae* Kim	2003	Taiwan	UI	FR, FM	4
437	*Bruchophagus gibbus* Boheman	?	Xinjiang	UI	FM	4
438	*Urocerus gigas taiganus* Benson	1984	Xinjiang	UI	FR	9
439	*Solenopsis geminate* Fabricius	1920	Guangdong	UI	FR, FM	4
440	*Solenopsis invicta* Buren	2003	Taiwan	UI	FR, FM	7
	Echinoidea					
441	*Strongylocentrotus intermedius* A. Agassiz	1989	Liaoning	II	FM	1
	Ascidiacea					
442	*Halocynthia roretzi* Drasche	2006	Liaoning, Shandong	II	OC	2
	Pisces					
	Salmoniformes					
443	*Oncorhynchus* Kisutch Walbaum	1982	Liaoning	II	IW	1
444	*Oncorhynchus mykiss* Walbaum	1959	Heilongjiang	II	IW, OC	2
445	*Salmo salar* L.	2004	Liaoning	II	IW, OC	1
	Cyprinodontiformes					
446	*Gambusia affinis* Baird & Girard	1911	Taiwan	II	IW	

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	Cypriniformes					
447	*Carassius cuvieri* Temminck & Schlegel	1959	Taiwan	II	IW	34
448	*Cirrhina mrigala* Hamilton	1982	Guangdong	II	IW	
449	*Labeo rohita* Hamilton	1971		II	IW	
450	*Tinca tinca* L.	1998	Hubei	II	IW	5
451	*Ictiobus cypinellus* Valenciennes	?	?	II	IW	
	Characiformes					
452	*Colossoma brachypomum* Cuvier	1982	Taiwan	II	IW	
453	*Serrasalmus nattereri* Kner	2002	?	II	IW	
	Siluriformes					
454	*Clarias batrachus* L.	1978	Guangdong	II	IW	
455	*Clarias lazera* Valenciennes	1981	?	II	IW	34
456	*Hypostomus plecostomus* Walbaum	?	?	II	IW	
	Pleuronectiformes					
457	*Paralichthys dentatus* L.	2002	Shandong	II	OC	1
458	*Paralichthys lethostigma* Jordan & Gilbert	2001	Shandong	II	OC	14
459	*Scophthatmus maximus* L.	1992	?	II	OC	3
460	*Verasper moseri* Jordan	2004	?	II	OC	5
461	*Solea senegalensis* Kaup	2001	Shandong	II	OC	1
462	*Solea solea* L.	2003	?	II	OC	2
	Anguilliformes					
463	*Anguilla anguilla* L.	1991	Jiangsu, Fujian	II	IW, OC	4
464	*Anguilla rostrata* Lesueur	1995	?	II	OC	4
	Perciformes					
465	*Oreochromis aureus* Steindachner		Taiwan	II	IW, OC	3
466	*Oreochromis nilotica* L.	1978	?	II	IW	
467	*Perca fluviatilis* L.	1960s	Xinjiang	II	IW	1
468	*Micropterus salmoides* Lacépède	1970s	Taiwan	II	IW	4
469	*Lepomis macrohirus* Rafinesqus	1987	Hubei	II	IW	0
470	*Morone saxatilis* Walbaum	1997	?	II	IW, OC	1
471	*Lates calcarifer* Bloch	?	?	II	OC	
472	*Sciaenops ocellatus* L.	1991	?	II	OC	1
473	*Sparus aurata* L.	2001	Tianjin	II	IW, OC	4
	Amphibia					
474	*Lithobates catesbeiana* (Shaw)	1959	Beijing	II	IW	10
475	*Lithobates grylio* (Stejneger)	1987	Guangdong	II	IW	1
476	*Lithobates heckscheri* (Wright)	1987	Guangdong	II	IW	1
	Reptilia					
477	*Trachemys scripta elegans* Wied-Neuwied	?	Hongkong	II	IW	17
478	*Trachemys scripta scripta* Wied-Neuwied	?	?	II	IW	5
479	*Chelydra serpentina* L.	1997	?	II	IW	14
480	*Macroclemys temminckii* Troost	1988	?	II	IW	11
481	*Apalone ferox* Schneider	1993	Fujian	II	IW	11
	Aves					
482	*Branta Canadensis* L.	1998	Hebei	II	IW	5

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483	*Cacatua sulphurea* Gmelin	?	?	II	FM	1
	Mammalia					
484	*Mus musculus* L.	?	?	UI	FR, FM, RS	34
485	*Rattus norvegicus norvegicus* Berkenhout	?	?	UI	FR, FM, RS	34
486	*Rattus rattus rattus* Lineaus	?	?	UI	FR, FM, RS	4
487	*Ondatra zibethicus* L.	1950	Xinjiang	NS	FM, RS	13
488	*Myocastor coypus* Molina	1953	?	II	FR, FM	6

Note: Pathways: intentional introduction (II); unintentional introduction (UI); natural spread (NS) Habitats: farmlands (FM, including fields, gardens, roadsides, grasslands, grassy slopes); inland waters (IW, including lakeshores, swamps, marshes); forests (FR, including forest margins); residences (RS); ocean (OC)

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REVIEW ARTICLE

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Advancing research on alien species and biological invasions

Plant invasions in China: an emerging hot topic in invasion science

Jian Liu[1,2], Hua Chen[2,3], Ingo Kowarik[1], Yiran Zhang[2,3], Renqing Wang[2,3]

1 *Department of Ecology, Technische Universität Berlin, Berlin, D 12165 Germany* **2** *Institute of Environmental Research, Shandong University, Jinan, 250100 P. R. China* **3** *Shandong Provincial Engineering and Technology Research Center for Vegetation Ecology, Shandong University, Jinan 250100, P. R. China*

Corresponding author: *Ingo Kowarik* (kowarik@tu-berlin.de)

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Abstract

China has shown a rapid economic development in recent decades, and several drivers of this change are known to enhance biological invasions, a major cause of biodiversity loss. Here we review the current state of research on plant invasions in China by analyzing papers referenced in the ISI Web of Knowledge. Since 2001, the number of papers has increased exponentially, indicating that plant invasions in China are an emerging hot topic in invasion science. The analyzed papers cover a broad range of methodological approaches and research topics. While more that 250 invasive plant species with negative impacts have been reported from China, only a few species have been considered in more than a handful of papers (in order of decreasing number of references: *Spartina alterniflora, Ageratina adenophora, Mikania micrantha, Alternanthera philoxeroides, Solidago canadensis, Eichhornia crassipes*). Yet this selection might rather reflect the location of research teams than the most invasive plant species in China. Considering the previous achievements in China found in our analysis research in plant invasions could be expanded by (1) compiling comprehensive lists of non-native plant species at the provincial and national scales and to include species that are native to one part of China but non-native to others in these lists; (2) strengthening pathways studies (primary introduction to the country, secondary releases within the country) to enhance prevention and management; and (3) assessing impacts of invasive species at different spatial scales (habitats, regions) and in relation to conservation resources.

Keywords

Alien invasive species, biodiversity conservation, control, flora, ecological impact, management

Introduction

Biological invasions are a major driver of biodiversity loss worldwide (Mack et al. 2000, Millennium Ecosystem Assessment 2005, Gaertner et al. 2009, Pyšek and Richardson 2010, Vilà et al. 2011), and associated costs will continue to increase with the development of international trade and global change (Pimentel et al. 2005, Dehnen-Schmutz et al. 2007, Ding et al. 2008, Perrings et al. 2010). Concepts of modern invasion science took root in the 19th and early 20th centuries (Kowarik and Pyšek 2012), but the number of invasion studies has grown enormously since Elton (1958) published the classic book of *The Ecology of Invasions by Animals and Plants* (Richardson and Pyšek 2008, Kühn et al. 2011). Yet studies on plant invasions are still geographically biased, with an overrepresentation of studies in Western countries and a low presentation of developing countries in Africa or Asia (Pyšek et al. 2008, Nuñex and Pauchard 2010, Khuroo et al. 2011). This leads to an unbalanced understanding of biological invasions, which are often context specific (Richardson and Pyšek 2006). Studies in developing countries are now trying to fill the geographical gaps in invasion science research, which will be crucial to counteract negative impacts associated with plant invasions (Khuroo et al. 2011).

China is a vast country with rich biodiversity and a long history of species introductions (Xie et al. 2001, Ju et al. 2012). For example, in 126 B.C., Zhang Qian and his assistants introduced seeds of useful plants to China from central Asia, including alfalfa (*Medicago sativa* L.), pomegranate (*Punica granatum* L.), grape (*Vitis vinifera* L.), and safflower (*Carthamus tinctorius* L.) (Xie et al. 2001). Human disturbance and diverse introduction pathways, which usually increase in the wake of economic growth, are widely recognized as important drivers of biological invasions (Hierro et al. 2006, Meyerson and Mooney 2007, Hulme 2009, Essl et al. 2011a) and this holds particularly for China (Liu et al. 2005, Lin et al. 2007, Weber and Li 2008). While plant invasions in Europe have clearly increased since the 19th century (Lambdon et al. 2008) corresponding processes might have started later in China, likely due to the longer political and economic isolation of this country (Ding et al. 2008, Wang et al. 2011). However, with the rapid economic growth of recent decades, increasing numbers of plant introductions and linking of previously isolated regions through the establishment of new transport corridors have promoted plant invasions in China (Lin et al. 2007, Weber and Li 2008, Wang et al. 2011, Ju et al. 2012).

In the face of an accelerating pace of environmental change, the awareness of environmental problems associated with plant invasions has grown significantly in China during the last ten years (Li and Xie 2002, Xu et al. 2006). Xu et al. (2012) demonstrated that the number of invasive alien plant and animal species, i.e. species with negative impacts on biodiversity, economy or human health, increased exponentially since 1850. One of these invasive plant species is *Mikania micrantha* Kuhnt, known as “plant killer” in Chinese. This species covered nearly 40–60% of the woodlands of Neilingding Island at its peak and has been found to strongly impact local ecosystems (Zan et al. 2000, Feng et al. 2002, Wang et al. 2004).

The last decade yielded an increasing number of invasion studies in China. Xie et al. (2001) and Liu et al. (2001) have reviewed early papers on non-native plants in China which were published in Chinese (e.g., Hu and But 1994, Zhang and Han 1997, Ding and Wang 1998, Peng and Xiang 1999, Zan et al. 2000, Liu et al. 2001), and Ju et al. (2012) recently provided a review on different groups of plants and animals.

Here we review recent studies and research trends related to plant invasions in China, based on an analysis of papers referenced in the ISI Web of Knowledge. In particular, we were interested in the prevailing types of research (experiments, field investigations, modeling studies, reviews, or integrative analyses), the most studied species and the research topics covered by recent studies. For the latter, we screened for papers that addressed either biological features of introduced species, mechanisms, or impacts associated with plant invasions or control options. By analyzing the research trends and gaps, we also aimed to sketch future perspectives of studies on plant invasions in China.

Methods

Scope of papers

We screened the Web of Knowledge (ISI) for all papers published between 1945 and 2010 that are related to plant invasions in China. We used "all databases" including (1) Web of Science (1945–2010), (2) Current Contents Connect (1998–2010), (3) MEDLINE (1950–2010), and (4) Journal Citation Reports and analyzed all records published through the end of 2010. We found 643 papers when searching for the terms [plant* or weed*] and [invasion* or invasive* or introduced* or non-native* or neophyte*] and [China] in topic; 329 papers when searching for [invasive species] and [China]; and 143 papers when searching for [plant invasion in China]. Combining these approaches yielded matches of a total of 1,115 papers. By reading the abstracts of these papers, 187 papers were identified as addressing plant invasions in China. In the discussion of the results we also refer to some recently published papers. We are aware of some caveats in this approach since books on plant invasions are generally not recorded in ISI nor are some papers in non ISI-listed journals (e.g., Li and Xie 2002, Wang et al. 2004, Xu 2003, Xu and Ye 2003, Zheng and Ma 2010).

Paper analysis

We classified the obtained selection of 187 relevant papers according to the publication year, research type, research topic, and studied species. We differentiated the following research types: experimental studies, field investigations, modeling studies, reviews, and integrative analyses (Table 1). Experimental studies included papers based on experiments, usually done in lab or greenhouse, or manipulative field experiments. Field investigations included studies mainly based on analyses in the field, e.g. of distribution patterns of a single species or changes in community composition due to the naturalization of introduced species. We considered studies to be integrative analyses if they were based on databases of large numbers of species at the regional or national scales (> 300 km^2), usually aiming to reveal patterns in species traits or environmental factors related to plant invasions. In addition, we differentiated papers that provided reviews or used modeling approaches.

In a second step, we analyzed whether the papers addressed one or more of the following research topics: biological features of non-native plant species (e.g., morphological and physiological characters, clonal and propagation characteristics, genetic variation); mechanism of plant invasions (competition and other biotic interactions, human interference, enemy release, ecological, economic, or health impacts of plant invasions) and management approaches (mechanical, chemical, and biological methods). Some papers have been attributed to more than one category. For example, analyses of traits belong to the topic biological features, but when the paper demonstrated plant traits to facilitate invasions it had also been assigned to the research topic of mechanisms of invasions.

Results and discussion

Number of studies

The research in ISI revealed an increasing number of studies on plant invasions in China in the last decade, with an exponentially growing number of papers since 2005 (Figure 1). The review by Xie et al. (2001) on invasive species in China is the earliest paper referenced in ISI. No publications prior to 2001 were found in ISI, illustrating a rather short period of visibility of Chinese invasion studies to the international readership.

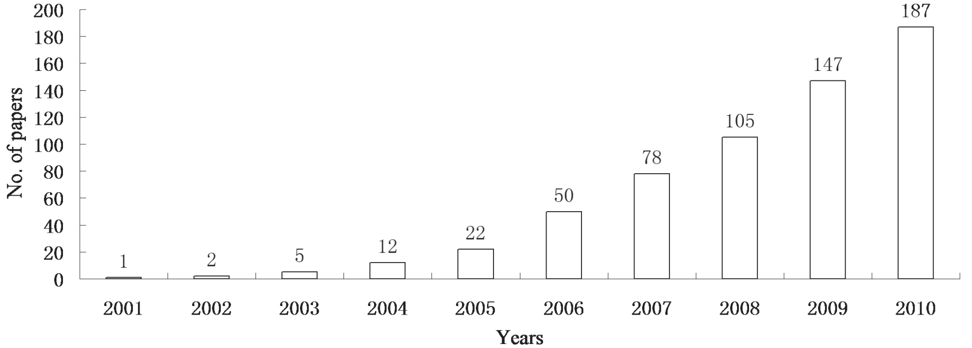

Figure 1. Cumulative numbers of papers on plant invasions in China based on a screening of the Web of Knowledge (1945–2010), see methods for details.

Main research types and topics

Most (89%) of the analyzed 187 papers addressed one or several non-native species studied by either field investigations (49%), experimental approaches (26%), modelling approaches (5%), or reviewed the existing knowledge on the species (10%; Table 1). Few papers (11%) offered integrative analyses some of which provided several lists of invasive plants in China at the national scale (Liu et al. 2005, Weber and Li 2008, Weber et al. 2008, Yang et al. 2010, He 2011). In addition, some recent papers listed naturalized plants (Wu et al. 2010, Jiang et al. 2011) and invasive plants (Liu et al. 2006, Weber and Li 2008, Huang et al. 2009, Xu et al. 2012). Yet, a comprehensive list of plant species non-native to China is missing thus far. In particular, casual non-native plant species are strongly underrepresented in existing lists. The most studied research topics (see below) included the characterization of biological features of introduced plant species and impacts associated with biological invasions, followed by studies on the control of invasive species. Compared to the latter issues, environmental factors related to plant invasions and other mechanisms that underlie plant invasions were studied to a lesser extent. Only a few papers (5%) modeled the distribution of non-native plant species in China (Table 1).

Most-studied species

The majority of the most-studied invasive plant species (Table 2) are herbaceous, and all are native to the Americas which generally comprise the most important donor regions for plants introduced to China (Feng et al. 2011, Huang et al. 2011). The perennial grass *Spartina alterniflora* Loisel. is clearly at the top of the list. An array of 47 papers cover all related research topics, with a clear focus on impacts that are complex and still in debate as reviewed by An et al. (2007) and Li et al. (2009). Consistently, most control studies

Table 1. Classifications of 187 papers on plant invasions in China, published in the period 2001–2010 and referenced in the ISI Web of Knowledge, according to research types and research topics: A – biological features of non-native plants; B – impacts of plant invasions; C – control approaches; D – invasibility or environmental factors related to plant invasions; E – mechanisms of plant invasions; F – predictions of the distribution of non-native plants. Some papers have been attributed to more than one category.

Research types	Research topics						
	All	A Traits	B Impacts	C Control	D Invasibility	E Mechanisms	F Predictions
Field investigations	91	35	33	14	8	6	1
Experimental studies	48	20	12	10	3	6	0
Integrative analyses	20	5	5	3	11	4	2
Reviews	19	11	6	9	1	1	1
Modeling studies	9	1	1	0	0	0	7
Total	187	72	57	36	23	17	11

addressed this species. The high numbers of studies on *S. alterniflora* might reflect the relevance of its impacts but might be also influenced by the fact that the invaded coastal ecosystems are easily accessible to researchers of many universities and national labs that are located in these areas. The second most studied species is *Ageratina adenophora* R.M.King & H.Rob. (=*Eupatorium adenophorum* Spreng.). *Mikania micrantha* Kuhnt, *Alternanthera philoxeroides* (Mart.) Griseb., and *Solidago canadensis* L. are the topics of between 10 and 20 papers each. Other species treated by only a few papers are *Ambrosia artemisiifolia* L., *A. trifida* L., *Chromolaena odorata* (L.) R.M.King & H.Rob., *Conyza sumatrensis* (Retz.) E.Walker, *Coreopsis grandiflora* Nutt. ex Chapm., *Flaveria bidentis* (L.) Kuntze, *Galinsoga parviflora* Cav., *Lantana camara* L., *Ipomoea cairica* (L.) Sweet, *Parthenium hysterophorus* L., *Rhus typhina* L., *Robinia pseudoacacia* L., *Solanum rostratum* Dunal, *Spartina anglica* C.E.Hubb. and *Wedelia trilobata* (L.) Hitchc.

The current focus on a rather small group of invasive plant species must not necessarily reflect the importance of a given non-native species in terms of ecological or economic impacts but might be considerably affected by the location and scientific background of research teams. Yet the much higher number of 265 invasive plant species (Xu et al. 2012) clearly indicates strong research needs to study a broader range of invasive species in China.

Usually, papers on invasive plant species address species that are non-native to the total area of China. Species that are native to a region in China but non-native to one or more others are rarely studied. One example is *Syzygium jambos* (L.) Alston, which is native to some provinces of China but has been reported as an invasive plant species in Hong Kong (Leung et al. 2009). Another example is the tree *Ailanthus altissima* (Mill.) Swingle which had been cultivated for a long time beyond its native range in China (Hu 1979). Today, it is not only invasive in many parts of the world (Kowarik and Säumel 2007) but also started to invade areas in China beyond its native range, e.g., in the Xinjiang province since the 1990s (Huang 1997, Säumel, personal communication). Analyzing the spread and impacts of species beyond their native ranges in China thus remains a challenge for future research.

Biological features

In 72 papers, the biological features of plant species non-native to China were analyzed, mostly by field studies and experimental approaches (Table 1). More than 20 papers addressed physiology, genetics, or regeneration patterns of introduced species. Fewer papers are related to morphological features and seed ecology.

Some studies illustrated physiological characters of introduced species that contribute to their invasion success. For example, Song et al. (2009) found that the increase in photosynthetic rate due to elevated CO_2 concentrations was significantly higher in non-native than in native species. In a study on the Yangtze River system, Jiang et al. (2009) revealed a competitive advantage of *Spartina alterniflora* over native species (*Phragmites australis* (Cav.) Steud. and *Scirpus mariqueter* Tang & F.T.Wang) through higher maximal net photosynthetic rate and a longer growing season.

Table 2. Most-studied alien plant species in China and related research topics (number of studies referenced in the ISI Web of Knowledge, 2001–2010). The native range and life form of the species are also shown. PG – perennial grass; BS – broadleaf shrub; PV – perennial vine; H – perennial herb; APH – aquatic perennial herb. The codes for the research topics A–F are the same as in Table 1.

Species	Life. Form	Native range	All	A Traits	B Impacts	C Control	D Mechanisms	E Invasibility	F Predictions
Spartina alterniflora	PG	America	47	9	30	8	3	3	1
Ageratina adenophora	BS	North America	27	12	5	4	3	5	1
Mikania micrantha	PV	America	19	8	4	7	1	0	0
Alternanthera philoxeroides	PH	South America	12	9	0	4	4	1	0
Solidago canadensis	PH	North America	10	4	2	1	2	1	2
Eichhornia crassipes	APH	South America	7	3	0	4	0	0	0
Total	-	-	122	45	41	28	13	10	4

Studies on the genetic variation or diversity of non-native plants revealed a very low genetic diversity in most clonal invasive plants such as *Alternanthera philoxeroides* and *Eichhornia crassipes* (Mart.) Solms (e.g., Ye et al. 2003, Ren et al. 2005, Wang et al. 2005, Li et al. 2006), while a relatively high genetic diversity has been found for a few species, such as *Coreopsis grandiflora* (Liang et al. 2008) and *Conyza sumatrensis* (Ren et al. 2010). Hao et al. (2011) illustrated a higher probability of spreading and covering a broad range in self-compatible than in self-incompatible species of Asteraceae in China.

The exploration of larger databases revealed that nearly half of 126 invasive species are clonals, and these are more frequent than other non-native plant species (Liu et al. 2006, Huang et al. 2009). Correspondingly, the top 13 plant invaders in China (based on the number of published papers) are clonally growing perennials (Huang et al. 2009). Greenhouse experiments in *Alternanthera philoxeroides* and *Spartina alterniflora* supported the idea that clonal integration and phenotypic plasticity of clonal plants have enhanced their invasion success (e.g., Geng et al. 2006, Liu et al. 2008, Yu et al. 2009, Zhao et al. 2010). By using genetic techniques, Dong et al. (2006) found that sexual reproduction facilitated the initial establishment of *Solidago canadensis* populations, while clonal growth led to a subsequent expansion of established populations.

Several studies analyzed the germination and seed banking of invasive species. Seed bank studies illustrated the role of sexual regeneration in the range expansion of clonal plants such as the highly invasive *Ageratina adenophora* and *Spartina alterniflora* (Shen et al. 2006, Xiao et al. 2009). Plasticity in seed germination and seed size was considered to be a trait that allowed for acclimation to different environmental conditions and facilitated invasions by *M. micrantha* and *Ageratina adenophora* (Yang et al. 2005, Li and Feng 2009).

Invasion impacts

There were 57 papers related to impacts of plant invasions in China (Table 1), affecting native plants, birds and other animals, soil biota, climate and economy. Several papers revealed negative effects of *Spartina alterniflora* on native plants, birds, and macrobenthic invertebrate communities (reviews by An et al. 2007, Li et al. 2009, Gan et al. 2009, Xie and Gao 2009). Yet, impacts due to *Spartina* invasions are still controversially discussed. Field research suggested a significant decrease in the abundance of native species and coverage of resident vegetation as well as negative impacts on birds feeding on native plants (Chen et al. 2004). Other studies, however, identified positive impacts of *S. alterniflora*, such as an enhanced storage of carbon dioxide (Liao et al. 2007), an increased inorganic nitrogen pool in the soil (Peng et al. 2010) and better shelter and food sources for a native crab species (Wang et al. 2008).

Ageratina adenophora was found to inhibit native species by altering soil microbial communities (Niu et al. 2007), while *Solidago canadensis* and *M. micrantha* can reduce the seed germination of native plants by allelopathic effects (Yang et al. 2007, Wu et al. 2008). A couple of papers described changes in soil conditions and soil biota due to invasions by, e.g., *M. micrantha* (Chen et al. 2009) and *A. adenophora* (Niu et al. 2007). A recent study illustrated, that allelopathic effects of the latter species can be reduced by native soil biota (Zhu et al. 2011). Chen et al. (2011) surprisingly found that *Coreopsis grandiflora* enhanced the functional diversity of soil microbial communities of invaded habitats.

Rather few papers addressed economically relevant invasion impacts. The review by Xie et al. (2001) as well as the integrative study by Xu et al. (2006) highlighted the relevance of economic impacts due to biological invasions for several sectors such as agriculture, forestry, water transportation, and human health. The authors estimated a total of USD14.45 billion in losses caused by all groups of invasive species to China in the year of 2000 but did not differentiate the contribution of plants and other organism groups to these losses. Xu et al. (2006) also stated that sound and strict case studies are lacking – and this still holds today.

To our surprise no paper particularly addressed human health, although some books recorded the harmful impacts of invasive plants on human health, in particular of allergenic plants (*Ageratina adenophora, Ambrosia* species, Li and Xie 2002, Xu 2003, Xu and Ye 2003).

Control and management

Most papers on control or management of plant invasions addressed biological control (Table 1). Several papers explored approaches of using the native parasite *Cuscuta campestris* Yunck. to restrain the non-native *M. micrantha* (e.g., Shen et al. 2005, Lian et al. 2006b, Zhao et al. 2008, Yu et al. 2008, Yu et al. 2009). The beetle *Agasicles hygrophila* Selman & Vogt was revealed to be a useful and safe biocontrol agent of *Alternanthera* *philoxeroides* as it had only limited effects on the non-target species *Alternanthera sessilis* (L.) R.Br. ex DC. (Li and Ye 2006, Lu et al. 2010a). The ant *Dorylus orientalis* Westwood was found to be a potential control agent of *Ageratina adenophora* (Niu et al. 2010).

While most studies on biological control focus on invasive plants as target species, some ecologists have begun to study the restoration of native plant communities after performing control. For example, field studies found that native *Cuscuta campestris* could not only restrain the non-native *M. micrantha* but might also contribute to the recovery of native communities by enhancing the availability of soil resources for native species (Yu et al. 2008, Yu et al. 2009).

Mechanical control approaches were studied in a few papers. The main target species were *Spartina alterniflora* (Li and Zhang 2008, Gao et al. 2009, Tang et al. 2009a) and *M. micrantha* (Lian et al. 2006a). Mostly, methods of mechanical control were tested. For example, Lian et al. (2006a) found that periodic cutting reduced the competitiveness of *M. micrantha* and fostered the growth of native species. The authors consider this approach an effective and easy method to reduce the dominance of *M. micrantha* although control of the invader was not perfect. Moreover, it also enhanced the growth of some other non-native species. Other papers covered chemical control approaches but most of them were reviews (e.g., An et al. 2007, Pan et al. 2007). The functioning of different or combined control approaches has been rarely studied thus far. As an exception, Guo et al. (2009) analyzed the effects of herbicides, uprooting, and cutting in the flowering stage on the sexual regeneration of *Solidago canadensis*. To reduce seed production the authors recommend herbicide application at the flower bud stage or uprooting at the flowering stage. Cutting flowering branches for ornamental purposes should be avoided

Policies related to the management of invasive plants were covered by a small selection of papers, and most of them were reviews. As control approaches are usually costly, an economically beneficial use of the harvested biomass could increase the efficiency and sustainability of control measures. Lu et al. (2010b) suggested an innovative approach in the field of bio-resource engineering. Their experiments showed that mixing biomass of the invasive water hyacinth (*Eichhornia crassipes*) with pig manure leads to a much higher biogas production than by using pig manure alone.

Underlying mechanisms

A couple of papers considered mechanisms underlying plant invasions in China (Table 1). Many of these were studies addressing the competition between introduced and native plant species. For example, *Spartina alterniflora* was found to have a competitive advantage over native plant species (Chen et al. 2004). Other studies illustrated significant differences in the response of native and invasive plants (*Mikania micrantha*, *Wedelia trilobata*, *Ipomoea cairica*) to elevated concentrations of CO_2 and discussed these results in terms of a future success of invasive species in the face of ongoing increases in atmospheric CO_2 concentrations (Song et al. 2009, Song et al. 2010). Wang et al. (2011)

found that increased temperatures enhance the aboveground biomass in *Ipomoea cairica* as well as the phytotoxicity of aqueous leachates from fresh leaves of this introduced liana. They concluded that global warming will foster invasions by this species.

Some studies tested the EICA hypothesis. For example, Feng et al. (2009, 2011) found that the invasive *Ageratina adenophora* have evolved an increased N allocation to growth and a reduced N allocation to defenses. Gao et al. (2011) provide evidence of the correlation between epigenetic reprogramming and the reversible phenotypic response of *Alternanthera philoxeroides* to particular environment in China. Pan et al. (2012) studied five populations of this invasive species and found that slow-growing genotypes experienced a stronger enemy release than fast-growing genotypes.

Biotic interactions other than competition have been studied to a lesser extent in China. Some papers studied the role of soil biota (e.g., Chen et al. 2011, Zhu et al. 2011). Based on studies of *Ageratina adenophora*, for example, Yu et al. (2005) suggested that non-native plants might inhibit native plants by changing soil microbe communities.

A few papers, mostly integrative data analyses, addressed ways of human interference to invasion processes. The fast growing economy of China has been often suggested to accelerate plant invasions through an enhanced international trade and associated species introductions (Lin et al. 2007, Ding et al. 2008, Wu et al. 2010), but studies on the relevance of different pathways are scarce (but see Xu et al. 2012 on the introduction pathways of invasive plant species). Data on the introduction history of non-native plants are limited, but analyzing the history of 123 plant species showed a continuous influx of non-native species to China after 1800 (Huang et al. 2011). However, much more species are known as naturalized in China (861 species according to Jiang et al. 2011), and casuals are usually underrepresented in species lists. It remains thus an open question whether the influx of non-native species since 1800 took place linearly or rather exponentially as indicated by the exponential growth of the number of invasive alien species of plants and animals since 1850 (Xu et al. 2012). Deeper insights into the history of plant invasions are an intriguing area of future research.

The example of *Parthenium hysterophorus* illustrates with evidence from nuclear and chloroplast DNA that multiple introductions were responsible for subsequent invasions in China (Tang et al. 2009b). As in other parts of the world, annuals were mostly introduced accidentally while perennials were mainly introduced intentionally (Xu et al., 2004).

Environmental factors related to invasions

Some studies related environmental factors to plant invasions. The decreasing number of invasive plant species from the south to the north of China could be related to climatic factors (Wu et al. 2006). Recent studies illustrated the relative predictive power of biogeographic and socio-economic factors in explaining current distribution patterns (Feng et al. 2011, Huang et al. 2011). Biogeographic factors mainly explained the distribution of species introduced from Central and South America, while socio-economic factors were more important for species introduced from Eurasia or North America (Huang et al. 2011). Other studies illustrated the significance of environmental factors at the habitat scale. Lu and Ma (2006) and Dong et al. (2008), for example, found that roadside habitats favor invasions by *Ageratina adenophora* and revealed a decreasing abundance of this species with increasing distance from the road. A field survey in southeast China found that *Alternanthera philoxeroides* dominates in microhabitats with high soil nutrients and water availability, whereas the cover of its native congener *A. sessilis* was relatively high in habitats with low soil nutrients and low water availability. High resource availability therefore appears to facilitate invasions by *A. philoxeroides* (Pan et al. 2006). A study on tropical East Asian islands (some belonging to China) found that closed-canopy forests appear to resist plant invasions (Corlett 2010).

Modelling the distribution of non-native plants

Only a few papers modelled the potential distribution of non-native species (Table 1), based on current environmental factors and biological features of the species, while no paper addresses the potential abundance of non-native plants in China. For example, ecological niche modelling was used to predict the invasion potential of *Ageratina adenophora* on the basis of occurrence points within colonized areas (Wang and Wang 2006). Using datasets on known localities invaded by *A. adenophora* and the environmental variables generated by the genetic algorithm for rule-set production (GARP) model, the potential future distribution of this invasive plant was modeled (Zhu et al. 2007). Using the homoclime approach, Lu et al. (2007) found that the potential range of *Solidago canadensis* in China is remarkably larger than the current range. A cellular automata model in conjunction with remote sensing and a geographical information system (GIS) was used to simulate the expansion process of *Spartina alterniflora* and support the hypothesis of space pre-emption as well as subsequent range expansion (Huang et al. 2008).

Conclusions

The exponentially increasing number of papers on plant invasions in China in the last decade (Figure 1) suggests plant invasions in China to be an emerging hot topic in invasion science. The analyzed papers cover a broad range of methodological approaches and research topics and clearly enhanced the understanding of plant invasions in China, in particular by compiling species lists, analyzing taxonomic and geographical patterns, and studying species- and environment-related mechanisms that might shape plant invasions and their associated impacts. Although plant invasions have been acknowledged as an important environmental risk to China, only six invasive species have been studied in detail thus far (Table 2). This sharply contrasts to a much higher number of invasive species (Xu et al. 2012). Further invasion research in China is thus strongly needed and would also help counteracting the continuing global imbalance in the understanding of invasion patterns (Richardson and Pyšek 2006).

We argue for an additional reason for encouraging further studies on plant invasions in China. This country has undergone far-reaching socio-economic changes in a relatively short period of about 30 years, and several drivers of this change are well known to enhance biological invasions (Lin et al. 2007, Ding et al. 2008, Weber and Li 2008, Huang et al. 2010). Experience from other regions indicates that a significant part of the "invasion echo" following economic changes might come decades or even centuries later. In different countries, recent invasion patterns could be better explained by economic parameters from the past (Sullivan et al. 2004, Essl et al. 2011a), and decades to centuries can elapse between the first introduction of a species to a region and subsequent invasions (Kowarik 1995, Aikio et al. 2010). Thus, the enormous recent increase in urbanization, transport corridors, and use of introduced plants in horticulture, landscaping, and forestry will certainly evoke a wave of future plant invasions in China (Ding et al. 2008, Ju et al. 2012). Strengthening invasion research in China, with special emphasis on the points described below, could help to mitigate foreseeable economic damage and negative impacts on the high biological diversity of this country. Some of the following points are generally relevant, others are more specific for China due to the special history of this country.

Lists of non-native species at regional and national scales

Although some lists of naturalized plants in China have recently been compiled (Wu et al. 2010, Jiang et al. 2011) and some on invasive species already exist (Liu et al. 2005, Weber and Li 2008, Weber et al. 2008, Huang et al. 2009, Xu et al. 2012), a comprehensive list of all non-native plants in China is still lacking. China is a large country in area with starkly varying environmental conditions. Thus, complete lists of non-native species compiled at the regional scale (e.g., provinces or metropolitan areas, Wang et al. 2011) and then aggregated at the national scale might be useful. As in other regions (Lambdon et al. 2008), two categories of non-native species can then be differentiated: species non-native to China and species native to one part of China but non-native to others. This differentiation is promising as species from both categories can induce severe invasion impacts. At the province scale, the number of non-native species can be underestimated if only species non-native to a larger unit, the whole country, are considered non-native (see Guo 2011, Pyšek 2011). Additional information on the taxonomy, native range, residence time, biological characters, and propagule pressure of non-native plants as well as on invaded habitats would help to reveal traits related to invasiveness and habitat invasibility and might strengthen the use of such lists in early warning approaches (Pyšek et al. 2004, Lambdon et al. 2008).

There is growing evidence that the time since the first introduction of a species to new range matters in terms of habitat occupation, impacts and response to climate (Pyšek et al. 2005, Jarošik et al. 2011). Hence, classifying non-native plants based on different lengths of their invasion history in China could help to better understand invasion processes and associated impacts in the long run. Many early introductions that have experienced a long-term selection and adaptation to their new range are known for China (Xie et al. 2001), and considering the performance of non-native species with a long invasion history might help to predict the future performance of species with a much shorter introduction history (La Sorte and Pyšek 2009). While a temporal differentiation of archaeophytes and neophytes (pre-Columbian and post-Columbian introductions, respectively) is useful for European or American studies (Pyšek et al. 2005) another definition of "early" and "late" introductions might be appropriate for China because of the different history of this country.

Invasion pathways

Human-mediated dispersal is a key process in plant invasions, and identifying and assessing the strength of dispersal vectors helps to set priorities in prevention and management (Carlton and Ruiz 2005, Kowarik and von der Lippe 2007, Hulme et al. 2008). Thus far, a few papers have studied the role of human interference on plant invasions in China. Xu et al. (2012), for example, found that about two thirds of 265 invasive plant species in China were intentionally introduced. As in other regions (Křivánek et al. 2006, Dehnen-Schmutz et al. 2007, Essl et al. 2010) horticulture and forestry are important introduction pathways. Yet as most plants are dispersal-limited even after an initial introduction to a country, subsequent secondary releases within a region often function as an important driver of plant invasions for a long period of time after the initial introduction to a country (Kowarik 2003). The case of *Spartina alterniflora* shows that repeated plantings can have highly relevant ecological consequences (An et al. 2007, Li et al. 2009). Information on negative invasion impacts of a given species can be used for regulating the relevant invasion pathways. One example is the recommendation to abandon the previously common practice of planting the North American tree *Rhus typhina* in Beijing (Wang et al. 2008).

The recent economic growth of China is associated with an increasing development of road systems, the linking of watersheds by canals, and a powerful growth of cities. The huge project of water transfer from the southern part of China to the northern part (namely the South-to-North Water Transfer Project of China), which will deliver about 45 billion m^3 of water annually from the Yangtze River to the north of China (Yang and Zehnder 2005, He et al. 2010), will also provide a new pathway for the dispersal of aquatic plants. In consequence, future invasion risks will result from interactions between the increased propagule pressure of non-native plants due to greenings of the built infrastructure and new or more effective dispersal pathways, provided either by roads (Pauchard and Alaback 2004, von der Lippe and Kowarik 2012) or waterways (Thomas et al. 2006, Säumel and Kowarik 2010, Jacquemyn et al. 2010). Consistently, road habitats appear to enhance invasions by *Ageratina adenophora* in China (Lu and Ma 2006, Dong et al. 2008). A better understanding of habitat- or dispersal-related mechanisms would help to counteract such invasions by optimizing either prevention or management efforts. As the role of socio-economic factors in driving plant invasions is often underestimated, both human-mediated and natural factors should be considered in analyzing patterns of plant invasions (Liu et al. 2005, Meyerson and Mooney 2007).

Impacts and management

Risk assessment and management of invasive plants are essential approaches to prevent potentially harmful new introductions or mitigate negative impacts of already introduced species. While classifications of species as "invasive," i.e., problematic, in other regions might be helpful in early warning systems ("invades elsewhere" criterion; Williamson 1999), setting priorities in management would certainly profit from regionally based impact assessments (Essl et al. 2011b). As the performance of non-native plant species depends on regional and local environmental conditions, impacts might differ significantly (Thiele et al. 2011, Vilà et al. 2011). The same species can affect biodiversity negatively or positively in different regions (e.g., Jäger et al. 2009, Fischer et al. 2009). As a consequence, existing (regional) lists of invasive species in China could be expanded by adding information on conservation resources that are (or might be) affected by an invasive species and on the conservation value of these resources (Bartz et al. 2010). As control actions often fail, monitoring the desired decline of non-native target species as well as effects on other species (e.g. Yu et al. 2008, 2009) would help to optimize control. Moreover, management actions can be combined with approaches of ecological restoration (Gaertner et al. 2012).

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RESEARCH ARTICLE

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Germination performance of native and non-native *Ulmus pumila* populations

Heidi Hirsch[1], Catherina Wypior[1], Henrik von Wehrden[2], Karsten Wesche[3], Daniel Renison[4], Isabell Hensen[1]

1 *Institute of Biology - Geobotany and Botanical Garden, Martin-Luther-University Halle-Wittenberg, Am Kirchtor 1, 06108 Halle/Saale, Germany* **2** *Institute of Ecology, Faculty of Sustainability & Centre of Methods, Leuphana University, Scharnhorststraße 1, 21335 Lueneburg, Germany* **3** *Senckenberg Museum of Natural History Goerlitz, Am Museum 1, 02826 Goerlitz, Germany* **4** *Centro de Ecología y Resursos Naturales Renovables – Dr. Ricardo Luti (CERNAR), FCEFyN, UNC – CONICET, Av. Vélez Sarsfield 1611, X5016GCA Córdoba, Argentina*

Corresponding author: *Heidi Hirsch* (heidi.hirsch@botanik.uni-halle.de)

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Abstract

Germination is a crucial step for invasive plants to extend their distribution under different environmental conditions in a new range. Therefore, information on germination characteristics of invasive plant species provides invaluable knowledge about the factors which might contribute to the invasion success. Moreover, intra-specific comparisons under controlled conditions will show if different responses between non-native and native populations are caused by evolutionary changes or by phenotypic plasticity towards different environmental influences.

This paper focuses on the germination of native and non-native *Ulmus pumila* populations. We expected that non-native populations would be characterized by their higher final germination percentage and enhanced germination rate, which might indicate an influence due to corresponding climatic conditions.

Germination experiments with a moderate and a warm temperature treatment did not reveal significant differences in final germination percentage. However, seeds from the North American non-native range germinated significantly faster than native seeds ($p < 0.001$). Additionally, mean time to germination in both ranges was significantly negatively correlated with annual precipitation ($p = 0.022$). At the same time, this relationship is stronger in the native range whereas mean time to germination in non-native populations seems to be less influenced by climatic conditions.

Different germination responses of the North American populations could be caused by a fast evolutionary change mediating a higher tolerance to current climatic conditions in the non-native range. However, our findings could also be caused by artificial selection during the introduction process and extensive planting of *U. pumila* in its non-native range. Nevertheless, we assume that the faster germination rate of non-native populations is one potential explanation for the invasion success of *U. pumila* in its new range since it might provide a competitive advantage during colonization of new sites.

Keywords

Climatic influence, survival analysis, biological invasions, *Ulmus pumila*

Introduction

Introduced species often face different environmental conditions in their new range compared to their range of origin. Therefore, non-native species have to overcome several factors before they can become invasive (Heger and Trepl 2003). Moreover, even after becoming established in the new range, there are consistent characteristics which can alter the ongoing invasion spread. For example, germination is crucial for dispersal and to establish populations in new sites in order to expand in range (Theoharides and Dukes 2007, Donohue et al. 2010). Therefore, data about shifts in germination characteristics could provide valuable information to predict the success of an invading species.

Differing germination characteristics can be caused by evolutionary changes mediated by corresponding environmental conditions. For example, Eckhart et al. (2011) demonstrated that germination patterns in 20 populations of *Clarkia xantiana* along a climatic gradient were linked to the corresponding temperature, mean precipitation and variation in precipitation. Additionally, several studies have shown that plant species can exhibit differing germination responses which are related to differing habitats or biotic influences (e.g. Giménez-Benavides et al. 2007, Jorritsma-Wienk et al. 2007, Grondahl and Ehlers 2008). Similarly, shifts in the germination performance towards different environmental conditions can also be an important factor during range expansion in the course of invasion. Brändle et al. (2003) showed that for 31 weedy plant species the range size is influenced by the germination niche breadth. Furthermore, enhanced germination percentages and rates of invaders compared to their native congeners or competitors have been associated with increased colonization success of the invaders (Burke and Grime 1996, Muñoz and Ackerman 2011).

Intra-specific comparisons between native and non-native populations are important to understand the mechanisms of the invasion process (Hierro et al. 2005). Furthermore, it can be useful to compare native and non-native individuals under a common environment. Such experiments will allow to distinguish if differences between ranges are caused by phenotypic responses towards different environmental conditions or by genetic changes (Leger and Rice 2003, Kawecki and Ebert 2004, Erfmeier and Bruelheide 2005, van Kleunen et al. 2010). For example, Beckmann et al. (2011) found that non-native New Zealand populations of three grassland species show increased germination compared with the native European populations, which may indicate an adaptation to new climatic conditions in the non-native range. Several other comparative studies also reported differences in germination between native and non-native populations of the same species (e.g. Kudoh et al. 2007, Hierro et al. 2009).

Although more than 300 tree species are classified as invasive, there are comparatively few studies on their invasion success (Lamarque et al. 2011, Richardson and Rejmánek 2011). Our study addresses the comparison of germination responses between native and non-native *Ulmus pumila* L. (Ulmaceae) populations. The Siberian elm is a native tree of temperate regions of East Asia, and occurs northwards up to the dry Gobi desert, where it is bound to water surplus sites and oases (Wesche et al. 2011). The flowering and fruit set production occur during the late winter to early spring (Wu et al. 2003). Each of the wind dispersed fruits (samaras) contains a single seed. The seeds lose their viability rapidly after maturity unless placed on suitable germination conditions or dried and placed at low temperatures (Baskin and Baskin 2000). *Ulmus pumila* can grow in a wide variety of habitats (e.g. slopes, valleys, plains), even with cold winters and long summer droughts (Wu et al. 2003, USDA and NRCS 2011). Since the Siberian elm performs better under harsh climatic conditions than most other trees, it has been planted in several regions outside its native range, e.g. in the semi-arid Southwestern United States as a fast growing windbreak or shade tree (Webb 1948, Leopold 1980). Furthermore, it is commonly used in elm breeding programs due to its high tolerance to the Dutch elm disease (Smalley and Guries 2000, Mittempergher and Santini 2004). Today, *U. pumila* is considered as naturalized or even invasive in 43 states of the U.S., as well as in Canada (Kartesz 2011, USDA and NRCS 2011), Mexico (Todzia and Panero 1998), Argentina (Mazia et al. 2001, Zalba and Villamil 2002), Spain (Cogolludo-Agustín et al. 2000), the European part of Russia, Estonia and Australia (NOBANIS 2012). Webb (1948) reported that different Chinese origins of the Siberian elm are characterized by differing frost hardiness. Therefore, it seems possible that specific adaptations towards local environmental conditions allow *U. pumila* to persist over such a wide distribution range. However, to our knowledge no information exists if early life cycle traits of *U. pumila* show such an adaptation and if this could contribute to the invasion success.

We focused our study on non-native populations in the Western U.S. and compared their germination performance under controlled conditions to the performance of populations from the native range in China. Thereby, we tested the following hypotheses: 1) Non-native populations will exhibit an increased percentage of germinated seeds. 2) Non-native populations are characterized by a faster germination. 3) Different germination responses might be influenced by different climatic conditions. In this context, we assume that populations located in regions with less stressful climatic conditions (e.g. higher annual precipitation) show enhanced germination characteristics.

Material and methods

Seed collection

We retrieved samaras (henceforth referred to as seeds) from seven populations from the native range (China) and seven populations from the non-native range (U.S.; Figure 1).

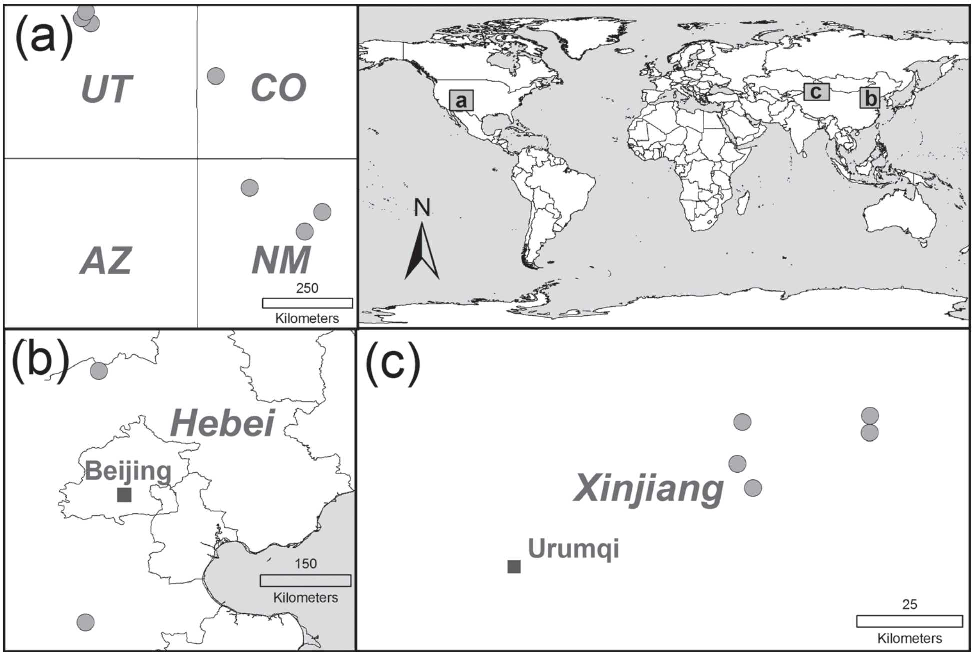

Figure 1. Sampled *Ulmus pumila* populations in the native range (**b, c**) and the non-native range (U.S.: **a**; **AZ** = Arizona, **CO** = Colorado, **NM** = New Mexico, **UT** = Utah). Populations are indicated by gray circles.

Seeds from China were collected in May and June 2009 and seeds from the U.S. in May and June 2010. We sampled at least 15 trees per population and pooled the seeds within populations. Where seeds had already been shed, they were collected from the ground. Material was stored in sealed plastic bags at 4°C following recommendations by Grover et al. (1963) to maintain seed viability.

Germination experiment

The germination experiment started in January, 2011 and was setup as a completely randomized design with eight replicates per population and treatment. Each replicate contained 20 seeds which were placed on filter paper in standard Petri dishes. In sum, we used 4480 seeds (14 populations × 2 temperature treatments × 8 replicates × 20 seeds per replicate) in our experiment. The dishes were filled with de-ionized water to keep the seeds permanently moist. Wings of the seeds were not removed due to their role in facilitating water uptake and in order to avoid seed damage (Rohmeder 1942, Namvar and Spethmann 1985). The experiment was performed in RUMED Light Thermostats germination chambers (Type 1301; Rubarth Apparate GmbH, Laatzen, Germany) under two temperature treatments (20°C/10°C and 32°C/20°C) with a photoperiod of 12 h cold white light (1200 Lux) and 12 h darkness. The two temperature treatments were used to account for the range of maximum temperatures during the main germination period of *U. pumila* (see Appendix 1: Table A1). Germinated seeds (visible radicula) were reported and removed every second or third day. After two weeks, viability of non-germinated seeds was tested with triphenyl tetrazolium chloride (ISTA Tetrazolium Commitee 2008).

Statistics

All statistics were calculated with the software R (version 2.15.0; The R Development Core Team 2012). To test if the final germination percentage (logit transformed according to Warton and Hui 2011) differs between the accessions and temperature treatments, we used a linear mixed model with populations nested in ranges as random effect (package nlme, version 3.1-103; Pinheiro et al. 2012). The Akaike Information Criterion (AIC; Akaike 1974) was used for model selection. For visualization of the germination performance and to extract the restricted mean time to germination (henceforth referred to as mean time to germination) per population we used the Kaplan-Meyer estimates of the germination functions. To test if time to germination differs between the ranges and temperature treatments, we performed a survival analysis using an Accelerated Failure Time (AFT; Bradburn et al. 2003) regression following the recommendations of Onofri et al. (2010). We used a right censoring of non-germinated, but still viable seeds. Non-viable seeds were excluded from the analysis based on the assumption that these were already non-viable at the beginning of the experiment (Onofri et al. 2010). We used the AIC values to select the most appropriate distribution (exponential, loglogistic, lognormal or Weibull), since AFT models assume parametric distributions (Kleinbaum and Klein 2005). Population was added as a random effect to test if the model is affected by variation at the population level within the ranges. The Kaplan-Meyer statistics as well as the survival analysis were calculated with the package *survival* (version 2.36-12; Therneau and Lumley 2012).

To test if the mean time to germination is adapted to different climate conditions between the native and the non-native range (see Appendix 2: Figure A1), we extracted climatic information per population (mean annual temperature and annual precipitation; see Appendix 1: Table A1) from the WORLDCLIM database (Hijmans et al. 2005). The effect of the climatic variables on germination was tested with a multiple linear regression. Additionally, we also included the effect of the population origin (native or non-native range) and the temperature treatment in our model. Selective model reduction was based on the AIC values.

Results

At the end of the germination experiment, 80.6 % of the tested seeds were germinated. From the non-germinated seeds were 2.1 % still viable (non-native origin: 1.3 %; native origin: 0.8 %).

The test for differences regarding the final germination percentage resulted in a final model containing only the temperature treatment as fixed effect. Consequently, no differences between the two ranges were detectable ($F_{1,12}$ = 0.416, p > 0.05, Figure 2). However, final germination percentages for both ranges were slightly lower under warm temperature conditions compared to a moderate temperature ($F_{1,209}$ = 6.513, p < 0.05; Figure 2).

Investigation of the time to germination revealed that the most pronounced reduction of deviance was contributed by the temperature treatment (Table 1). Lower, but still significant effects were contributed by the random effect (population), the influence of the origin of the populations (range) as well as the interaction between range and temperature treatment. These results were obtained from a final AFT model with best fit for log-normal distribution showed including range as well as temperature as predictor variables and population as random effect. The enhanced germination rates under warmer temperatures as well as the differences between the two ranges are visualized in Figure 3.

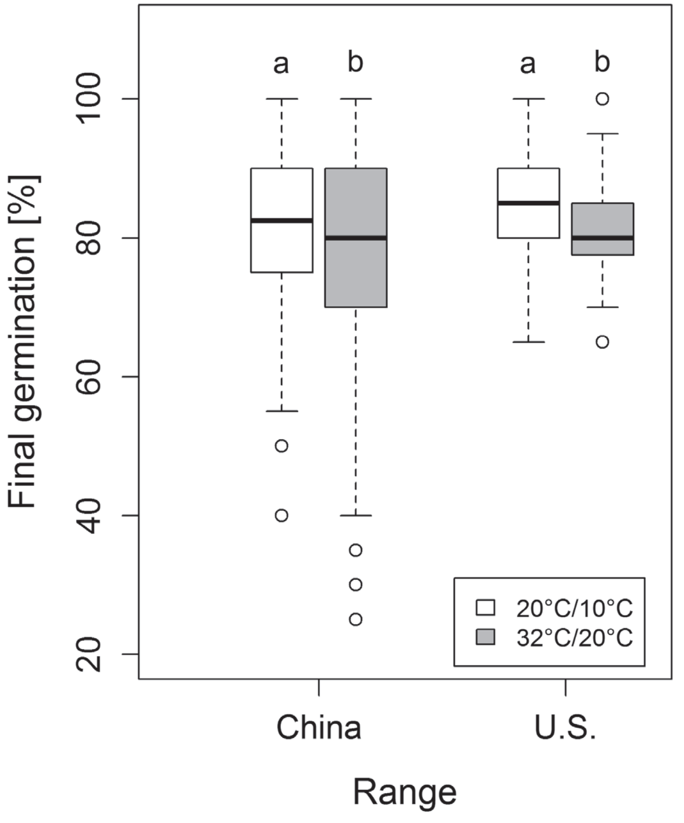

Figure 2. Final germination (%) of *Ulmus pumila* seeds from native and the non-native ranges. No differences were found between the two ranges ($F_{1,12}$ = 0.416, p > 0.05). Germination percentage was significantly decreased under warmer temperature treatment ($F_{1,209}$ = 6.513, p < 0.05; significant differences are shown by different letters above the boxes).

Table 1. Analysis of deviance results for the AFT model. The results show the differences in the time to germination of *Ulmus pumila* under consideration of range and temperature treatment (df = degrees of freedom).

Source	df	Deviance	Residual df	-2 × loglikelihood	*p*
Null model			3627	15151.90	
Range	1	165.75	3626	14986.25	<0.001
Temperature	1	1691.68	3625	13294.47	<0.001
Frailty (population in range)	12	673.16	3615	12621.30	<0.001
Range × temperature	1	6.34	3614	12614.96	0.012

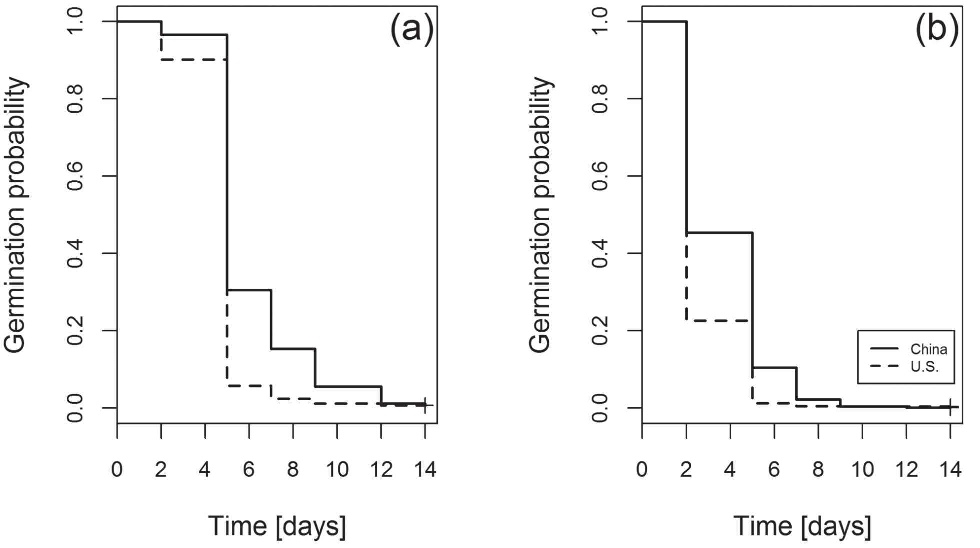

Figure 3. Kaplan-Meyer curves of the germination functions for the non-native and native origins of *Ulmus pumila*. Curves are shown for the two temperature treatments (**a**: 20°C/10°C; **b**: 32°C/20°C). Censored data is symbolized by final crosses at the curves. The curves show the probability that seeds will germinate. Therefore, the germination probability has to be 1.0 at time = 0 because all seeds are non-germinated and have consequently the chance to germinate.

Considering the mean time to germination supports our result of the AFT model regarding a faster germination at higher temperatures ($F_{1,23}$ = 88.83, p < 0.001) and in the non-native range ($F_{1,23}$ = 14.48, p = 0.001). We also found a significant negative relation between mean time to germination and annual precipitation ($F_{1,23}$ = 5.98, p = 0.022) as well as a significant interaction between range and annual precipitation ($F_{1,23}$ = 9.46, p = 0.005). This interaction shows that native populations with less annual precipitation are characterized by increased mean times to germinate. In contrast, non-native populations show only weak response in their mean time to germination towards corresponding annual precipitation conditions (Figure 4). These results were obtained from the multiple regression model retaining population origin, temperature and annual precipitation as predictor variables after stepwise model selection (multiple R^2 = 0.84, p < 0.001).

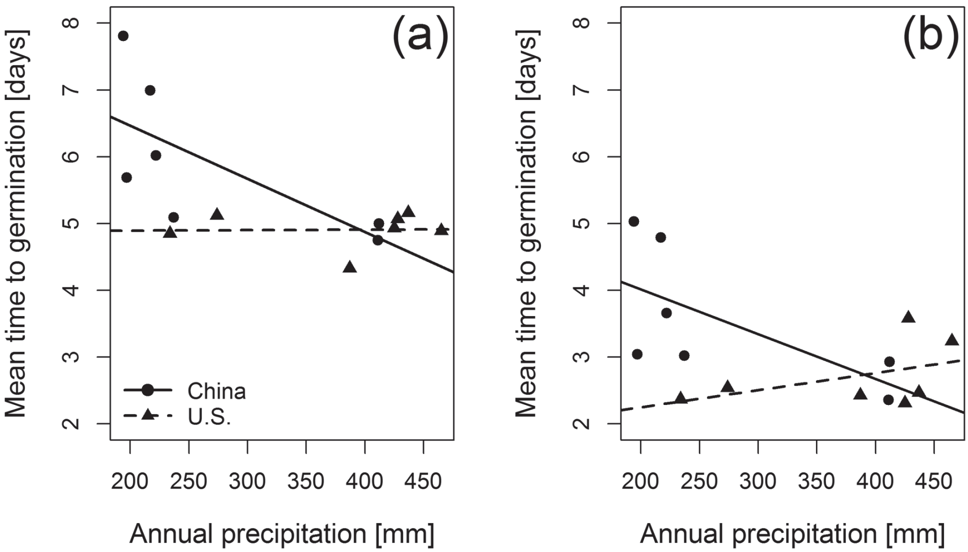

Figure 4. Relationship between mean time to germination and annual precipitation per *Ulmus pumila* population (**a**: 20°C/10°C; **b**: 32°C/20°C). To emphasize the different responses between the two ranges (non-native range: triangles; native range: circles), trend lines per range are shown (non-native range: dashed line; native range: solid line).

Discussion

Our results revealed a slightly lower final germination of *U. pumila* seeds at higher temperatures. This could be caused by an earlier and stronger infestation by mold fungi at the 30°C/20°C temperature treatment (personal observation), because these warmer temperatures provide better growing conditions of mold fungi. For example, Barnett et al. (1999) showed for *Pinus palustris* that germination can be reduced by pathogenic fungi. However, we assume that these slightly differences show no relevant effects under natural conditions, because the final germination will be still high enough to support a colonization of *U. pumila* in regions with high temperature regimes, since seeds are produced in very high numbers. Furthermore, we found only a very low amount of non-germinated but still viable seeds. Consequently, we exclude that population growth and persistence of the Siberian elm might be supported by a generated seed bank, which would be also contrary to the already mentioned short life span of *U. pumila* seeds.

Contrary to our hypothesis, the invasion success of *U. pumila* in North America does not seem to be based on an enhanced final germination percentage. However, we have evidence for enhanced times to germination in non-native populations. We propose that the fast germination is one of the contributing drivers for the invasion success of *U. pumila* because it could provide advantages during inter-specific competition in the colonization processes (Donohue et al. 2010). This hypothesis is supported by results in other studies such as a grassland experiment by Milbau et al. (2003) which revealed that regenerative traits, like germination time, are correlated with invasiveness. Furthermore, Seiwa (2000) showed that early-emerging seedlings of *Juglans ailanthifolia* are characterized by greater height than later-emerging seedlings due to a longer exposition to favorable light and temperature conditions before they are crowded by other species.

Additionally, we found that mean time to germination in both ranges seems to be influenced by climatic conditions such as annual precipitation (i.e. mean time to germination decreases with increasing annual precipitation). We assume that this relationship is based on less stressful germination conditions for the Siberian elm under climatic conditions with more rainfall since annual precipitation can be considered as a general measure of environmental quality (Philippi 1993, Hierro et al. 2009). However, the significant interaction between range and annual precipitation indicates that annual precipitation conditions show a stronger influence to the mean time to germination in native populations. In contrast, mean time to germination of non-native population seems to be less influenced by the annual precipitation conditions. Therefore, it might be possible that non-native populations are characterized by a higher tolerance towards different precipitation conditions compared to native populations. For example, evidence of germination rates related to different moisture regimes was shown for *Pinus ponderosa* in central Oregon (Weber and Sorensen 1992). Further, Maron et al. (2004) showed that such processes are also possible for introduced plants. In this context, it is often suggested that rapid evolutionary change is supported by standing genetic variance or genetic mixing (intra- or inter-specific; Lavergne and Molofsky 2007, Prentis et al. 2008, Dormontt et al. 2011). Genetic studies have also shown that non-native *U. pumila* populations in the Eastern and Central U.S. are characterized by genetic diversity levels which are comparable to native populations (Zalapa et al. 2009, 2010). Furthermore, it was demonstrated that a high proportion of these populations contain hybrids between *U. pumila* and *U. rubra* and that hybridization leads to a significant increase of genetic variability. As shown by Abbott et al. (2003), hybridization can lead to the introgression of traits which might affect the fitness of introgressants or their tolerance to novel habitats. For example, Rieseberg et al. (2007) were able to identify that introgression processes may supported range expansion of *Helianthus annuus.* However, genetic investigations are needed for our sampled populations of *U. pumila* in the Western U.S. to gain more detailed knowledge on the genetic diversity and eventually hybridization processes.

In contrast to natural evolutionary processes, the pattern of different germination reactions in our studied populations could also be caused by human-mediated selection of successful lineages during introduction (Donohue et al. 2010). For example, Chrobock et al. (2011) found evidence that cultivated non-native species germinate earlier and more successfully than related native species which indicates a human-mediated selection for these traits. Therefore, non-native species that escaped from cultivation and became invasive might be characterized by enhanced germination characteristics mediated by artificial selection. Such a type of selection could have influenced the germination performance of *U. pumila* due to selection during the introduction process and extensive planting in the U.S. afterwards (Webb 1948, Leopold 1980, Mittempergher and Santini 2004). Consequently, further research approaches should also consider seeds or seedlings obtained from commercial suppliers to test for eventually human-mediated selection. Further, we are not able to exclude that the revealed differences between non-native and native populations are influenced by maternal effects. According to Moloney et al. (2009), a bias by maternal effects could be avoided by using second-generation offspring. However, long generation times render the implementation of this approach very difficult for woody plants. Therefore, we suggest that genetic investigations are needed to proof our assumption that the different germination patterns between non-native and native populations are caused by evolutionary change rather than maternal effects.

Additionally, it should be considered that our results could be biased by two methodical factors. First, differences between both ranges might be caused by different sampling years (seeds from the native range: 2009; seeds from the non-native range: 2010). We assume that this factor induced only a negligible influence to our results, because Grover et al. (1963) observed that Siberian elm seeds did not show any different viability during the first two years of the storage conditions used for our study. Nevertheless, we strongly recommend the usage of seeds from the same sampling year for further comparative germination experiments to provide uniform test conditions. Second, it could be argued that our replicates per population and treatment are just pseudoreplicates due to their spatially non-independence. However, we exclude that the observed differences in germination resulted significantly from technically differences among the used germination chambers, because both chambers are of the same model type produced by a high quality manufacturer, both chambers had the same basic conditions (e.g. same light equipment), and both chambers are frequently cleaned and fumigated. Nonetheless, a repeated switching of the temperature treatment and the corresponding replicates in further germination experiments as applied by Zuloaga-Aguilar et al. (2011) could help to improve the experimental design of such studies, and to reduce possible different test conditions.

It should also be mentioned that several other studies have shown that changed germination characteristics are often linked to changed post-germination traits as well (Donohue et al. 2010). Erfmeier and Bruelheide (2005) studied non-native *Rhododendron ponticum* populations and showed that genetic shifts influenced the germination and the growth performance. Therefore, colonization success of non-native *U. pumila* populations could be based on both an increased germination rate and a better growth performance than native populations. In order to accept this hypothesis, research on coevolution between germination and post-germination traits is needed.

Our work suggests that changed germination characteristics could be one of the drivers for the invasion success of *U. pumila.* However, further research (i.e. genetic analyses and growth experiments) is needed to find genetic evidence for our assumption and if the assumed evolutionary change of germination responses also influenced other early life cycle traits of non-native populations of the Siberian elm.

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Appendix 1

Table A1: Location and climate information of the sampled *Ulmus pumila* populations. (doi: 10.3897/neobiota.15.4057.app1) File format: PDF.

Explanation note: Location and climate information of the sampled *Ulmus pumila* populations in China and the U.S. Maximum (max.) temperatures for the months May, June and July are provided to show the temperature range during the main germination period (lowest and highest values are italicized). Climatic information was extracted from the WORLDCLIM database (Hijmans et al. 2005).

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Appendix 2

Figure A1: Comparison of climatic conditions between the Chinese and North American locations of *Ulmus pumila*. (doi: 10.3897/neobiota.15.4057.app2) File format: PDF.

Explanation note: Comparison of climatic conditions (a: mean annual temperature; b: annual precipitation) between the Chinese and North American locations of *Ulmus pumila.* Wilcoxon rank sum tests were used to test for differences between both ranges. Mean annual temperatures are significantly higher for locations from the U.S. ($W = 7$, $p < 0.05$). Annual precipitation is marginal higher in the invasive populations compared to the native populations ($W = 9$, $p = 0.05$). Significant differences are symbolized by different lowercases above the boxes.

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REVIEW ARTICLE

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Advancing research on alien species and biological invasions

A conceptual framework for prioritization of invasive alien species for management according to their impact

Sabrina Kumschick[1], Sven Bacher[2], Wayne Dawson[3], Jaakko Heikkilä[4], Agnieszka Sendek[5], Therese Pluess[2], Tamara B. Robinson[1], Ingolf Kühn[5]

1 *Centre for Invasion Biology, Department of Botany and Zoology, Stellenbosch University, Matieland, 7602, South Africa* **2** *University of Fribourg, Department of Biology, Ecology & Evolution Unit, Ch. du Musée 10, 1700 Fribourg, Switzerland* **3** *Ecology, Department of Biology, Universitätsstrasse 10, Konstanz, D 78464, Germany* **4** *MTT Agrifood Research Finland, Economic Research, Latokartanonkaari 9, 00790 Helsinki, Finland* **5** *UFZ, Helmholtz Centre for Environmental Research – UFZ, Dept. Community Ecology, Theodor-Lieser-Str. 4, 06120 Halle, Germany*

Corresponding author: *Sabrina Kumschick* (sabrina.kumschick@alumni.unibe.ch)

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Abstract

The number of invasive alien species is increasing and so are the impacts these species cause to the environment and economies. Nevertheless, resources for management are limited, which makes prioritization unavoidable. We present a prioritization framework which can be useful for decision makers as it includes both a scientific impact assessment and the evaluation of impact importance by affected stakeholders. The framework is divided into five steps, namely 1) stakeholder selection and weighting of stakeholder importance by the decision maker, 2) factual description and scoring of changes by scientists, 3) evaluation of the importance of impact categories by stakeholders, 4) calculation of weighted impact categories and 5) calculation of final impact score and decision making. The framework could be used at different scales and by different authorities. Furthermore, it would make the decision making process transparent and retraceable for all stakeholders and the general public.

Keywords

stakeholder, decision maker, exotic, generic scoring system, impact, value

Introduction

Impacts of invasive alien species (IAS) affect different receptor environments, and are often divided into environmental and socio-economic impacts. Some of these impacts can result in substantial monetary costs and/or alterations to entire ecosystems and social systems (O'Dowd et al. 2003, Pimentel et al. 2005, Reaser et al. 2007, Vilà et al. 2010). At an international level the ecological impacts of IAS are addressed by the Convention on Biological Diversity (CBD 1992). Through this convention the need for prioritization and management of priority species has been highlighted in the Strategic Plan for Biodiversity 2011-2020 (COP 10 2010), although no guidance on how to achieve this ideal is provided.

There has been recognition that societies need to mitigate negative impacts of IAS, i.e. find appropriate means to manage IAS in a way that their impacts are at least minimized, e.g. by eradication, reduction below a specific threshold, or containment. However, resources to manage IAS are limited and with increasing globalization, the influx of potentially harmful organisms will likely continue to increase (Perrings et al. 2010, Essl et al. 2011). There are two approaches to this problem. Firstly, to limit new alien species from entering an area/country, for which purpose border control risk assessments have been developed (e.g., Pheloung et al. 1999, Bomford 2008, see Hulme 2012 and Leung et al. 2012 for reviews). Secondly, alien species that escape border controls (Bacon et al. 2012) or have never been subjected to border control risk assessment, and which cause high impact must be managed in their new ranges (see above). Hence, a system is needed that facilitates the optimal allocation of limited resources to manage those IAS that are most harmful in a given area. Such a system would ideally integrate the severity of effects on the environment, as well as on the economy and the society in question, allowing decision makers to prioritize certain high impact species for management. However, also cost-effectiveness of management needs to be taken into account.

Alien species, however, do not have only negative effects. The majority of the alien plants in Europe were deliberately introduced, e.g. as ornamental, horticultural, restoration, agricultural or forestry species (Hopper 2007, Lambdon et al. 2008, Pyšek et al. 2009) with their respective social, economic and environmental benefits. Management of such species, which are detrimental in some aspects (e.g. for biodiversity) and beneficial in others (e.g. forestry), can result in conflicts among involved stakeholders. For example, the introduction to South Africa of alien *Acacia* species which subsequently became invasive had differential effects on local communities. On the one hand, communities suffered from water scarcity due to increased evapotranspiration by the acacias. On the other hand, they benefitted from fuel wood and building timber (De Wit et al. 2001). Such situations have led to the recognition of a need for a framework to document the different consequences of an IAS for different groups of stakeholders (Stoll-Kleemann and Welp 2006, Binimelis et al. 2007, Kapler et al. 2012) and recently a few attempts have been made to develop applications which incorporate stakeholders (Cook and Proctor 2007, Hurley et al. 2010, Liu et al. 2011, De Lange et al. 2012, Forsyth et al. 2012).

Parker et al. (1999) developed a framework to assess the ecological effect of IAS, arguing that the total effect of an invader includes three fundamental dimensions: range, abundance, and the per-capita or per-biomass effect of the invader, i.e. the magnitude of ecological change it causes. Since then progress has been made in scoring the overall negative effects of alien species (e.g., Nentwig et al. 2010, Kumschick and Nentwig 2010, Pluess 2011). A very detailed assessment scheme for impacts of genetically modified organisms was presented by Kowarik et al. (2008) which can easily be adapted for invasive species. However, few of these impact scoring approaches explicitly addressed potentially competing interests of stakeholders (i.e. various ecological, economic or social interests) within the context of management of IAS. Furthermore, these studies ignore potential positive effects of IAS, which might be crucial for some species and stakeholders (see e.g., Schlaepfer et al. 2011 for a review). These points are important because biological invasions represent a complex societal issue for two reasons. First, the scientific knowledge about biological invasions and the outcome of different management options are highly uncertain and second, both conflicts of interests and values are prominent in a problem-solving context such as the management of IAS (Kueffer and Hadorn 2008). Thus, as in any environmental decision making process, IAS management has to minimize conflicts characterized by ecological, economic and social value judgments of different stakeholders (Liu et al. 2010, 2011). Furthermore, if all types of impacts of IAS are to be assessed (i.e. ecological and socio-economic impacts), then the different scientific disciplines and associated value systems require a common currency with which to measure impact. Another layer of complexity is added by the fact that different sections of administration and stakeholder groups with differing agendas need to be integrated, since plants, animals and human health are in the responsibility of different agencies and/or ministries (e.g., IPPC - International Plant Protection Convention, OIE - World Organization for Animal Health, WHO - World Health Organization).

At present, several studies have proposed prioritization methods for the management of weeds (e.g., Skinner et al. 2000, Virtue et al. 2001, Robertson et al. 2003, Tassin et al. 2006, Randall et al. 2008, Brunel et al. 2010). However, only few schemes considered potential conflicts of interest when evaluating which weed species to prioritize for control first, mainly in South Africa (Robertson et al. 2003, Forsyth et al. 2012, De Lange et al. 2012). Other studies that specifically tackled the complexities of conflicting interests and values in IAS management have usually focused on particular outbreak situations of a single species (Maguire 2004, Liu et al. 2010, 2012). Based on a participatory approach in Western Australia, the prioritization by an assessment committee differed from current resource allocations in Western Australia (Cook and Proctor 2007). A similar approach was taken by Roura-Pascual et al. (2010) for South Africa. However, such a potentially time-consuming and resource-demanding approach might not always be possible. Hence, decision makers and especially governmental bodies are still in need of a generic decision making aid that facilitates the identification of priority species whilst minimizing or reconciling potential conflicts of interest.

The basic problem with many of the systems to date is a fundamental one. Decisions made for IAS management are heavily influenced by judgements which are predominantly based on inputs from scientists (Wilhere 2008). In general, the role undertaken by scientists in decision making falls along a gradient (Lach et al. 2003). At one end of the continuum, scientists may simply report results that others use to make decisions. Alternatively, they may interpret these results and work with decision makers to integrate these results into the decision making process. At the other end of the continuum, scientists may be actively involved in decision making by advocating for a specific decision or in the extreme, making the decision themselves. While traditionally scientists largely shied away from active involvement in decision making (Lach et al. 2003), recent years have seen increasing advocacy (Marris 2006, Scott et al. 2007). This has resulted in a call for science inputs into decision making to be supportive of the process but dispassionate towards policy outcomes (Lackey 2007), recognising that science is but one vital element that needs to be considered in decision making. Against this background we argue that the process of objectively describing with scientific methods changes associated with an IAS has to be explicitly separated from the proximate subjective societal evaluation of impact, which is based on values. For the actual decision making for management, however, facts (effects on environment and socio-economy) need to be connected to values (judgement of involved stakeholders).

Here, we propose a comprehensive framework which aims to explicitly separate the scientific description of changes caused by IAS from the value systems of affected stakeholders who may have differing interests. Furthermore, it addresses both negative and positive effects of IAS, since positive effects are often neglected in purely ecological impact studies (Goodenough 2010, Davis et al. 2011, but see Pyšek et al. 2012 for a global review).

Framework for impact evaluation of IAS

The framework suggested here is divided into five steps. The different steps are shortly introduced in this section. We then present more details about the scorings and values in the following sections and in Figures 1 and 2.

Step 1: Stakeholder selection and weighting of stakeholder importance. A call for stakeholder participation is launched and they are encouraged to claim their interest. The stakeholder group is formed such that it is sufficiently representative and

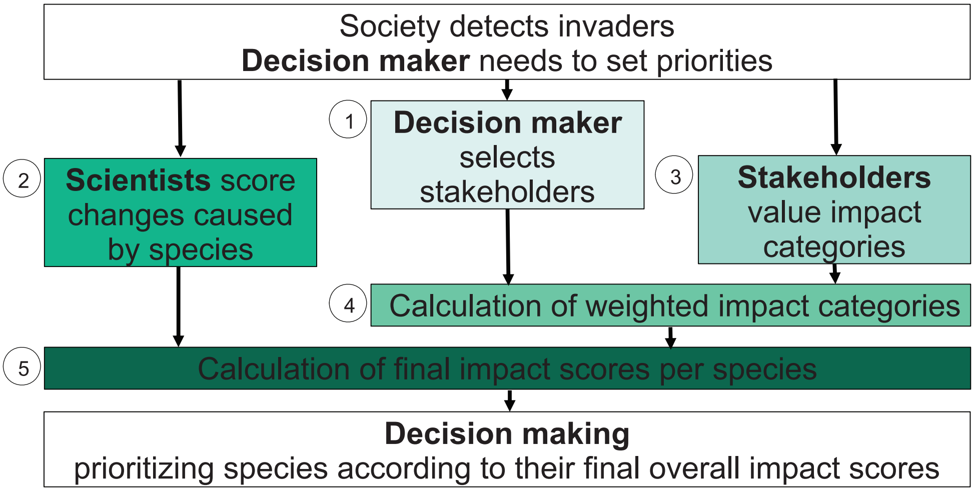

Figure 1. Schematic overview of the conceptual framework to assess change in different impact categories for each species, capture stakeholders' interests and weigh stakeholders and calculate a final impact score for each species, see chapter "framework for impact evaluation" of IAS for a brief and the following chapters for detailed explanation.

appropriate for the task at hand. The participating stakeholders are then categorized according to their importance in relation to the issue that is being evaluated. This process produces Stakeholder Weights (SW), and should be conducted in a transparent and logical way.

Step 2: Description and scoring of changes due to IAS. For this step we propose a scoring system, based on two main impact classes (ecological, socio-economic) each with several categories (e.g., agriculture, health, infrastructure, herbivory, hybridization). Negative and positive changes are separately evaluated for each IAS. The outcome of this step is hereafter referred to as Change Assessment Score (CAS).

Step 3: Valuing the relative importance of impact categories by stakeholders. After identifying affected stakeholders in step 1, each stakeholder values the relative importance of all impact categories. Negative and positive categories are valued separately. When valuing the categories, the stakeholders do not know the species assessed in step 2 and their change assessment scores. This is called Stakeholder Value Assessment (SVA).

After selection of stakeholders (step 1), the description of changes (step 2) and the assessment of stakeholder weights (step 3) can be conducted at the same time as one does not depend on the outcome of the other. The following steps in turn can only be performed if the outcomes of the former steps are known.

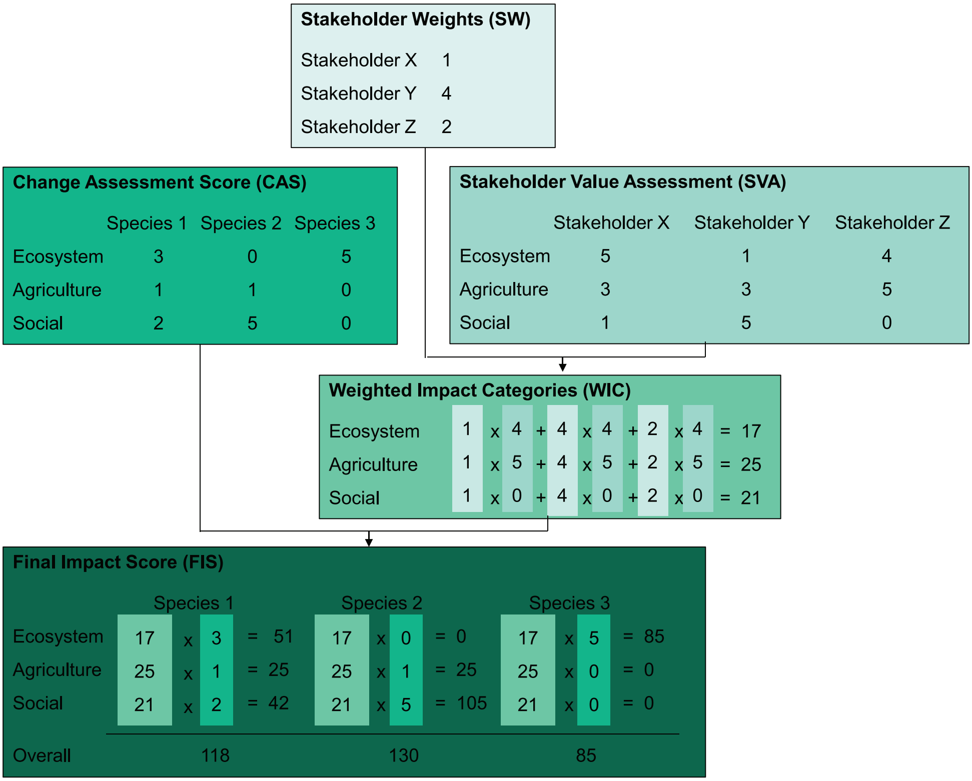

Figure 2. Fictitious example on how to calculate the different values of the prioritization framework. For brevity, we limit the number of categories from the scientific impact assessment in Step 2 of the framework to 3 (out of 24). Species 2 has the highest social Change Assessment Score (CAS), whilst species 3 has the highest CAS for ecosystem changes. The decision maker gives stakeholder Y the greatest weighting. In the Stakeholder Value Assessment (SVA), stakeholder X gives the greatest value to ecosystem, whilst Y values the social category the highest, and stakeholder Z values agriculture most. Multiplying these values by the stakeholder weights and summing the products gives Weighted Impact Categories (WIC), and multiplying these values by respective category CAS values gives the final overall impact scores per species. Note that WIC values for positive change need to be inversed to negative values upon calculation of final impact scores per category and overall.

Step 4: Calculating weighted impact categories. This is done by combining the outcomes of step 1 and step 3, which produces Weighted Impact Categories (WIC).

Step 5: Final impact scores. The Final Impact Scores (FIS) for each species are calculated by combining the CAS (step 2) with WIC (step 4).

In the following sections, the five steps are described in detail, and in the final part of the paper the usefulness of the scheme for different potential end-users and advantages and potential shortcomings are discussed.

Table 1. Prioritizing scheme in five steps, describing the action taken at each step, identifying the actor of each step and defining the output of each step. *) can be performed simultaneously.

Step	Action	Actor	Output
1)	Decision maker chooses the stakeholders and decides on the form and execution of the weighting process	Decision maker	SW = Stakeholder Weights
2*)	Assessment of change: the change a species incurs for each category from App. A is scored	Scientists	CAS = Change Assessment Score
3*)	Stakeholders value each category from App. A, regardless of the species and change a species incurred	Stakeholders	SVA = Stakeholder Value Assessment
4)	Calculation of weighted impacts by combining SVA × SW	Decision maker (or scientists or consultant)	WIC = Weighted Impact Categories
5)	Calculation of final impact score WIC × CAS	Decision maker (or scientists or consultant)	FIS = Final Impact Score

Step 1: Stakeholder selection and weighting of stakeholder importance

Stakeholders play a central role in the presented scheme, as their opinions are crucial for evaluating the subjective impact categories through formal and structured analysis. The stakeholder process needs to be carefully planned, structured and conducted, accounting for the aim and needs of the problem at hand (Renn and Schweizer 2009). There are three main things to consider: a) who should participate; b) what is the form of the participatory process; and c) whose opinion counts.

Who should participate?

A stakeholder is – put simply – someone who can affect or is affected by the issue at stake (Freeman 1984, cited in Mitchell et al. 1997). There are several methods available that one could use to come up with a list of stakeholders (see, e.g. Pretty 1995, Reed et al. 2009). For instance, an often-applied method is the so called 'snowball technique' which involves consulting every stakeholder as long as no new actors are indicated. How representative the stakeholder group should be depends on the aim of the process (e.g. Rowe and Frewer 2005, Webler and Tuler 2006, Reed 2008, Renn and Schweizer 2009, Wesselink et al. 2011). For instance, the aim of the stakeholder process may be to avoid missing information and perspectives, to try to find win-win situations or compensations from winners to losers, to represent and discuss all relevant arguments, to ensure that less-privileged groups are given the opportunity to have their voices heard, or to enlighten policy processes by illustrating the diversity of claims, opinions and values (Renn and Schweizer 2009). We suggest that at least all those who are potentially affected by the species should have a possibility to participate or have their voice heard through some other participant (e.g. an organization). In practice, stakeholders would consist of, for instance, agricultural and silvicultural producers, environmental organizations, tourist industry, city/town representatives, different outdoor associations (e.g. hunters, bird-watchers, recreationists), and so forth.

What is the form of the participatory process?

One main issue regarding the stakeholder process is the form of opinion-forming: is the process aiming at a consensus through deliberation, or are the stakeholders just asked for their individual opinion? If a consensus is sought for, the result of the process in our case would be a single set of weights (see step 3) that could directly be used to weigh the species impacts (resulting from step 2). If each stakeholder is allowed to have their own weights, then these need to be aggregated in some way unless the analysis is to be conducted separately for each stakeholder. A simple way is to average all the weights, in which case each stakeholders' opinion is valued equally. However, if the decision maker wishes to weigh the stakeholder opinions, specific weights for each stakeholder need to be produced.

Whose opinion counts?

There are both ethical and pragmatic reasons why the decision maker should attend more closely to some stakeholders than others (Colfer et al. 1999). These include fair treatment of those who are more affected by the decision, and that some stakeholders just have a greater likelihood than others of affecting (and being affected by) the issue at stake. There are theories that attempt to explain, often in business context, which stakeholders are being paid more attention to (Reed et al. 2009). One such theory is the theory of stakeholder salience (Mitchell et al. 1997), according to which stakeholders can be categorized using three attributes: power (to affect the decision maker), legitimacy (to the issue at stake), and urgency (of their claims). The more of these three properties a stakeholder has, the higher is their salience and hence their importance in the eyes of the decision maker.

Categorization provides the decision maker with the possibility to influence the prioritization procedure by weighting each of the participating stakeholders. As this may be considered as a source of bias, it should be transparent and ideally based on objective and reliable analysis of stakeholders' attributes (Mark and Shotland 1985, Mushove and Vogel 2005). For instance, Colfer et al. (1999) suggest seven criteria by which to define the most important stakeholders in the case of forest management, score them at a scale 1-3 for each criterion and obtain a weight for each stakeholder by averaging over the criteria. Alternatively there exist for instance pairwise comparison methods (Grafakos et al. 2010) that may be used to produce the required weights.

Weighting is not a straightforward process, especially when numerous attributes of the stakeholders are taken into consideration, and when stakeholders' external environment and interactions are complex (de Reynier et al. 2010, Aaltonen 2011). The decision maker undertaking the categorization ("top-down approach") is by no means the only way of attributing a weight to stakeholders. There are also methods through which the stakeholders themselves come up with a ranking order of their importance (see, for instance, Pretty 1995, Mitchell et al. 1997, Reed 2008, Reed et al. 2009).

Reed (2008) provides best practices for the stakeholder process, including for instance emphasis on empowerment and equity, representation and analysis of relevant stakeholders, need for clear objectives for the process, choosing methods depending on the context, and integration of scientific and local knowledge. Our framework requires the evaluation and categorization of stakeholder importance. However, as to what the precise format of the stakeholder process should be, we do not specify here for reasons of space. Practical examples in the context of invasive alien species can be found in Cook and Proctor (2007), Hurley et al. (2010), Skurka Darin et al. (2011) and De Lange et al. (2012). The desired outcome nonetheless is clear: the process should produce a set of weights for the different impact categories that would broadly reflect the values present in the society.

Step 2: Description and scoring of changes

The second step of the decision making process aims at recording all changes an IAS causes in the introduced range. An impact or change in this case is defined as any deviation from the state of a system before the invasion happened. In order to make comparison between species, different locations and different measurements of impact, we suggest the use of a generic scoring system (e.g., Nentwig et al. 2010, Kumschick and Nentwig 2010). Nevertheless, scoring systems and other prioritization tools suggested so far often only focus on unwanted changes and rarely take into account possible welcome changes which might result from species introductions, as e.g. increasing population densities of threatened native species (Schlaepfer et al. 2011) or economic benefits (Leung et al. 2012). For a balanced view, however, positive effects should no longer be neglected or even ignored, as many stakeholders profit from such IAS. When a decision is needed on how to deal with an IAS, all possible stakeholder interests need to be accounted for to ensure wide acceptance and support for the decision (Myers et al. 2000, Gardener et al. 2010; see step 1 for more details). However, whether changes are perceived as "positive" or "negative" depends on the value system of the stakeholders concerned (Simberloff et al. 2012). For example, the invasion of the weed Paterson's Curse (*Echium plantagineum*) in Australia was perceived as detrimental by ranchers because the plant is toxic to livestock, but beekeepers profited from its proliferous honey production (Harris 1988). In the following description of impact (changes), we continue to use the terms "positive" and "negative", but aim to define them in an objective, value-free way by describing the direction of change relative to pre-invasion state of the system.

Based on previously published scoring systems (e.g., Nentwig et al. 2010, Kumschick and Nentwig 2010, Kumschick et al. 2011, Pluess 2011, Kumschick et al. 2012), we determined a wide range of changes IAS could cause in the introduced area (Appendix A). The scoring system consists of two main classes of changes, socio-economic and environmental, and each class has 6 categories. The categories for environmental changes are hybridization, competition, transmission of diseases to wildlife, herbivory/toxicity, predation, and ecosystem effects in general. Changes to the environment can be negative or positive. Changes in the negative direction denote a decrease in an attribute of ecosystem function or native biodiversity compared to the state before the IAS was introduced and can range from no changes to the environment (score 0) to the maximum reduction possible (score 5). Positive effects can occur in systems previously altered by human-induced disturbance, e.g. alien species, land-use change, pollutants, eutrophication etc., but where an invader can fulfil some or many of the functions that previously existed or were fulfilled by species before perturbation. Thus, these scores can also range from very low changes (score +1) to the complete restoration of an expected, pre-invasion state of system functioning (score +5). Furthermore, positive effects can occur if an invasive species enhances a function still provided by other resident species. Please note that "positive" and "negative" do not denote human values, but relate to the direction of environmental change after invasion relative to the pre-invasion state of the system: "positive" indicates changes towards the pre-invasion state, "negative" changes away from the pre-invasion state. Because a species might simultaneously cause positive and negative changes within the same category, but through different mechanisms (see e.g. the *Echium* example from Australia described above), we score these positive and negative changes separately. Furthermore, it is possible that a stakeholder values positive and negative changes differently, so by keeping them separate, the categories might also be weighted differently.

The socio-economic categories are changes to human health, infrastructure, animal production, agriculture, forestry and human social life. Socio-economic changes can also be negative or positive, depending on whether they decrease or increase human well-being. Negative changes often consist of direct monetary or utility losses and can range from no changes (score 0) to the maximum negative change possible (score 5). Positive effects are also possible, for example, more possibilities for hunting an invasive species whilst alleviating hunting pressure on native mammals, or provision of a nectar source for important pollinators of agricultural crops. These effects can also range from very low (score +1) to the highest positive effects possible (score +5). Again, we have positive and negative changes within the same category, but we score these separately because they might occur through different mechanisms.

Not all changes are equally relevant for different taxa. For example, the difference in changes between alien plants and animals is likely to be quite marked in some cases. Therefore, we propose not to use the scores of change ("impact" scores) as measurement themselves, but to calculate the percentage score achieved out of the maximum possible for a given species. Hence if plants and animals are to be assessed in the same prioritization round, then for questions which are not relevant for plants or animals respectively (e.g. ecological impact through predation for plants), the overall-score should be adjusted by calculating the percentage score achieved out of the maximum attainable score for the species, and then multiplying by the maximum score attainable among all species considered. The critical point here is that in any round of prioritization, each candidate species should have an equal opportunity of attaining the same maximum score possible.

In many cases, effects of IAS on the recipient environment and economy have not been thoroughly studied. As in the whole invasion process, the uncertainty level of impacts can therefore be high and communicating these uncertainties is crucial in the decision making process (Liu et al. 2011, Leung et al. 2012). We therefore suggest including information about certainty in this step to account for the reliability of the data source used for scoring (low, medium, high). For example, this can be based on the type of data source (e.g. Low: mentioned in paper, no reference, speculation, expert judgement. Medium: evidence in literature, observational. High: demonstrated evidence in peer-reviewed literature, experimental) (Spear and van Wilgen, pers. comm.). This also deals with the fact that an impact score of 0 can be both, "no impact known" and "no impact detectable". Including a certainty level enables to distinguish these possibilities (e.g. Low: no information. Medium: unlikely based on life history, expert judgement, literature observation or speculation that there is no impact. High: demonstrated evidence in peer-reviewed literature, experimental) (Spear and van Wilgen, pers. comm.). These certainty levels are to be communicated to the decision maker and can potentially influence the final decision making. Furthermore, they can identify research needs (e.g., species with large effects with low certainty).

Step 3: Valuing the importance of impact categories by stakeholders

At this stage, we will leave the domain of objective quantification again and focus on the societal context. Scientific measurements of impact are valued differently by different stakeholders and the valuation may differ in space and time and from case to case (Sagoff 2011). Biological invasions will thus be perceived to have different impacts for different societal sectors and different groups of stakeholders. Several species may be perceived as beneficial e.g. for farming, forestry, hunting or landscape restoration, but as detrimental from a nature conservationists point of view.

The approach we suggest here (as vizualised in Figures 1 and 2) would be to let each stakeholder group give scores to each impact category according to their perceived importance for them. For example, for an impact assessment of alien species in a city, two possible stakeholder groups might be the tourism industry and environmentalists. The tourism industry is likely to assign the highest scores to the positive category on human social life, while environmentalists might give highest priority to the negative categories of change in ecosystems in general and on other species (e.g. through competition). There are a few studies on invasive species that explicitly weight the different assessment criteria (Cook and Proctor 2007, Ou et al. 2008, Hurley et al. 2010, Skurka Darin et al. 2011). The processes by which this is done in these studies are mostly rating or paired comparison (analytical hierarchy process), but also fixed point scoring or the ratio method (Grafakos et al. 2010).

We propose a fixed point scoring method to rank the importance of the categories by giving the stakeholders a fixed amount of points (e.g. 100) to freely distribute among the impact categories. This would reflect their preferences, but it is also good to note that if the stakeholders have extreme agendas, they might also end up with extreme point allocations. As mentioned, there are other mechanisms that could alternatively be used, for instance scoring each category at scale 1-5, and possibly also a combination of these. However, whatever the procedure, the stakeholders need to be given clear instructions on what is expected of them.

As indicated in the previous section, negative and positive categories are valued separately. When valuing the categories, the stakeholders do not know the Change Assessment Scores (CAS) of the alien species assigned by scientists in step 2, neither do they know which species are being assessed in order to avoid biased valuation towards certain species. In the same line of argument, stakeholders begin the valuation scoring without knowing the scores or even the participation of other stakeholder groups. If further deliberation in the stakeholder process is desired, it may follow this initial valuation.

Step 4: Calculating weighted impact categories

For each stakeholder, the values of each impact category are multiplied with the weight score given to this stakeholder by the decision maker. For example, two stakeholders, named A and B here, assign 2 and 5 points, respectively to the category "agriculture". The decision maker weights the opinion of stakeholder A as 3 times higher than the opinion of stakeholder B, thus the impact scores of A are multiplied by a factor 3 while those of B remain unchanged (see Figure 2 for another example). The overall weighted impact score for the category "agriculture" would in this example receive a score of $2 \times 3 + 5 \times 1 = 11$. This procedure yields weighted impact values for each category, incorporating the value system of all stakeholders and their importance for the decision maker. The highest weighted impact values represent the categories that are valued most across all stakeholders, i.e. those categories in which impacts would have the most serious effects for society.

Step 5: Final impact scores

The final impact scores for each species are calculated by multiplying the Change Assessment Scores (CAS) for each impact category (step 2, Figure 1) by the Weighted Impact Categories (WIC) over all stakeholders (step 4, Figure 1). This procedure essentially combines the objectively measured impact with an overall valuation of this impact by society. To calculate an overall impact score for a species, all final impact scores are summed. It should be noted here that upon calculation of the weighted impact scores, the sign of the CAS should be inverted to negative values, so that the final summed impact score reflects the net perceived impact of a given species. For example, if a high scoring negative-change species has a high positive-change score in one category, and the major stakeholders rank this positive change highly, then the finally obtained impact score will be reduced by the larger positive weighted impact score. The same procedure is applied for the associated certainty scores. We would therefore also learn about the combined certainty attached to the impact scores. Species can then be ranked according to their overall impact scores and/or by the certainty of the scores. Species responsible for large changes in an impact category that is of relatively low value to society are down-weighted, while species scoring low in impact categories that are of high societal value are given more weight (Figure 2). These final overall impact scores are a crucial element in the prioritization of management, amongst others like cost-effectiveness.

Discussion

Conservation organizations, governments and other interest groups need to prioritize which species to spend limited funds on in order to manage and achieve the best socio-economic and ecological benefit. Here, we have presented a prioritization system which combines objective ecological information on how species change the state of the invaded environment, with stakeholder assessments evaluating impact categories according to their specific interests and perception of value, to create an overall impact score. Species are thus ranked in importance by combining the overall impact score with an *a priori* stakeholder rank, according to the perceived importance of stakeholders to the decision maker, who is the ultimate funding body of management measures. The system clearly distinguishes science from values in the decision making process, which is crucial for transparent, rational and sustainable policy making (Wilhere et al. 2012).

By combining stakeholder views and scientific information on species impact, this impact prioritization system can ensure that the outcome of action to manage the most problematic species has little bias from opinions of scientists, or from unintended dominance by any one stakeholder with a loud voice. Ultimately, the decision maker can have some influence on the decision of which species to manage, by deciding which stakeholder group's opinion is the most important. This weighting of importance should be made in a transparent and repeatable way, for example by using the size of the stakeholder group (assuming that larger groups are of greater importance), but other ways of weighting stakeholder importance might be more appropriate, depending on the situation. However, any ranking of stakeholder importance should be done *a priori*.

For the system to be used in practice, it needs a few more specifications from the side of the user. For example it needs to be specified how to choose and reach the stakeholders, and according to which criteria and by whom they should be weighted. The system is very flexible and easily adaptable in this respect, as well as in relation to the impact scoring scheme that is used. For these and other possible adaptations and specifications, the system should be tested in practice and it should be documented precisely how the steps were performed and which changes were necessary. Generally, the more a system is used in practice, and the higher the awareness of its shortcomings are, the better and more broadly applicable it can become. A good example for this is the Australian weed risk assessment (WRA), which has been tested worldwide and in different ecosystems, and adapted accordingly (Pheloung et al. 1999, Weber et al. 2009). For the WRA, it has been tested whether it accurately rejects invaders and accepts harmless, non-invasive species. In the case of the framework suggested here, there is no "right" or "wrong" species to manage in any case since most (if not all) invasive species affect the newly inhabited environment in one or the other way. Naturally, cost-effectiveness of management makes managing some species a more sensible thing to do than other species. Furthermore, because of the subjective influence of the stakeholders in the decision making process, testing the system will not give a definite answer as to whether it would work in practice.

However, we believe the scheme we have presented here is general enough for different types of decision makers/funding bodies to use at different scales, with minimal modifications required. For instance, relatively local invasive species management projects by conservation organizations may only have a small number of species to assess, with few stakeholders involved. The system could just as easily be used at a regional or national level by government bodies. This process would be facilitated by the lists of problematic IAS that already exist in many countries, e.g. Australia's 'Weeds of National Significance' List (Virtue et al. 2001) and list of harmful alien mammals and birds in Europe (Nentwig et al. 2010, Kumschick and Nentwig 2010). This framework could also be of special significance in guiding actions against IAS in developing countries that may lack the policy tools to give action to their national legislation or international obligations.

Another adaptation of the system, should there be a national, multi-species management plan with sufficient funds, would be to split potential species into taxonomic groups (e.g. plants, mammals, birds, invertebrates), or according to habitat/ecosystem (e.g. freshwater or other aquatic habitats, grasslands), which would allow multiple species to be selected for management which are likely to have very different types of impacts in different areas or ecosystems. However, our proposed impact assessment in Appendix A is flexible and broadly applicable enough to allow prioritization for management across a wide range of taxa and ecosystems, if this approach is desired.

Whilst our system can be a useful tool for identifying the highest priority species for management according to society and science, it does not take into account how cost-effective management implementation might be. Ideally, we should try to target those species that are more cost-effective to control or eradicate, at least at local scales. However, the chances of successful control will also depend on other factors than the species itself. Recent studies on the feasibility of eradication found that eradication success mainly depended on the extent of the invaded area (Pluess et al. 2012a) and the habitat type (Pluess et al. 2012b). If the circumstances of the infestation by the top-priority species prohibit effective eradication, then the next species in the list could be chosen. This could also be a useful strategy for picking a single species for management, should several end up with a tied '1st place' priority score, and if funds do not allow the management of all. Alternatively the management strategy could switch from a focus on eradication to containment and damage limitation of the top species in sensitive areas. Additionally, management actions can also have potential negative effects on the environment, which can possibly be larger than the effect of the species itself. Britton et al. (2011) provide examples on how these risks can be incorporated in management decisions. Furthermore, given legislation and policies cannot be ignored; Governments have certain policy restrictions, for example regarding safety, health issues and nature conservation, which they have to comply with. Therefore, even a species that is not ranked as top priority in the proposed framework might have to be dealt with when it is required by a country's legislation.

This framework could potentially be useful for decision makers who need to set priorities for optimal resource allocation. Possible end-users of the framework besides governmental environmental agencies could also be nature conservation organizations (e.g., WWF), or any other organization interested in assessing the impact of IAS. The framework allows the end-user to set priorities for the management of problematic species across a wide range of taxa, by combining the actual change as described in step 2 (scientific input) and impact valuation in step 3 (stakeholders' valuation of impact categories).

A major strength of the approach highlighted here is the integration of scientific (i.e., objectively measurable) and social (i.e., value-based) assessments of invasive species impacts to prioritize species of concern according to impact severity. In addition, the generic nature of the impact assessment in step 2 and the category valuation by stakeholder makes the system flexible for use on different spatial scales and in different regions.

One potential weakness of the procedure proposed rests at the second step - scientific impact assessment. In practice, information on impacts generally has a high uncertainty and often is based only on expert judgements (Leung et al. 2012). Moreover, better-quantified impacts may be site-specific in their expression and magnitude, making generalization difficult (Virtue et al. 2001). Our scheme could provide an opportunity for targeting more thorough research and assessment of impacts of greatest concern to society, by communicating weighted impact values back to scientists. However, it should be borne in mind that public opinion is fluid, and may not immediately register the less tangible, but potentially detrimental impacts that invasive species can have on society.

Although the framework suggested is primarily meant to prioritize established and invasive species, it could also be used for border control of species which are invasive elsewhere and already known to cause impact, e.g. quarantine species. However, one should be aware of the problems associated with the prediction of future potentially harmful species, and also that this system does not assess entry or establishment probabilities. Particularly early during an invasion, management of species which are still harmless due to their small distributional range but may have a great potential to be detrimental in the future might be more cost-effective than trying to manage widespread species. Understanding how to predict impact is challenging but not impossible, and management decisions have to be made anyway (Leung et al. 2012).

In summary, we have presented a framework for prioritising invazive species according to impact severity, which involves the integration of scientifically assessed impacts per species, and socio-economic valuation of general impact importance across stakeholder groups. In theory, this framework could be implemented at multiple spatial scales, and for any group of species considered for management. However, the real value of the framework is revealed only once it has been thoroughly applied and tested, and we encourage the use of this framework to test whether or not it can work in practice as a useful prioritization and decision making tool in invasive species management.

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Appendix A

Definitions of change assessment (impact) scores for IAS. Effects were divided into two main classes with six categories each.

1. Environmental impact

1.1.1 Herbivory/toxicity negative

0 No impact known or detectable.
1 Very low level of herbivory (animals) or toxicity (plants or animals) on at least one native species, no major damage reported.
2 Herbivory or toxicity affecting several native species, without large impact on affected species or decline of their populations.
3 Herbivory or toxicity affecting several native species, at least one native species declining.
4 Herbivory or toxicity affecting many native species, several declining in population size, recorded community change reversible.
5 Herbivory or toxicity affecting native species listed as vulnerable, endangered or critically endangered by IUCN, decline of these species, replacement or even extinction of species, recorded community change irreversible.

1.1.2 Herbivory/toxicity positive

0 No impact known or detectable.
+1 Very low level of herbivory or toxicity affecting at least one species degrading the ecosystem, no impact on performance of affected species recorded.
+2 Herbivory or toxicity affecting one or several species degrading the ecosystem, without large impact on affected species or decline of their populations.
+3 Herbivory or toxicity affecting one or several species degrading the ecosystem, at least one species declining, some/first indications that the ecosystem changes towards its historical functional state.

+4 Herbivory or toxicity affecting one to many species degrading the ecosystem, declining in population size, strong indications that the ecosystem changes towards its historical functional state.
+5 Complete re-establishment of functional state of historical ecosystem that was degraded before invasion of alien species.

1.2.1 Competition negative

0 No impact known or detectable.
1 For animals, very low level of competition with at least one native species, exploitation competition; for plants, low abundance, native species richness not declining.
2 For animals, competition with several native species by exploitation competition, without large impact on affected species or decline of their populations; for plants, moderate abundance, decrease in native species abundance but not richness.
3 For animals, competition with several species, interference competition, at least one native species declining; for plants, high abundance, decrease in native abundance, at least one native species lost.
4 For animals, competition with many native species, several declining in population size, competition for food and/or space, behavioural changes in out-competed species; for plants, high abundance, strong decline in both abundance and richness of native species, native species still able to recruit.
5 For animals, competes with species listed as vulnerable, endangered or critically endangered by IUCN, decline of these species, replacement or even extinction of species; for plants, mono-dominant/near mono-dominant, with no or very few native species remaining; limiting native species recruitment options.

1.2.2 Competition positive

0 No impact known or detectable.
+1 For animals, very low level of competition with at least one native species degrading the ecosystem, exploitation competition, no impact on performance of affected species recorded; for plants, no loss in abundance or richness of native species.
+2 For animals, competition with one or several native species degrading the ecosystem by exploitation competition, without large impact on affected species or decline of their populations; for plants, small increase in abundance of native species, no increase in species richness.
+3 For animals, competition with one or several species degrading the ecosystem, interference competition, at least one species declining, some indications that the ecosystem changes towards its historical functional state; for plants, increase in abundance of native species, small increase in species diversity.

+4 For animals, competition with one to many species degrading the ecosystem, declining in population size, strong indications that the ecosystem changes towards its historical functional state; for plants, increase in abundance of native species and in species diversity.
+5 For animals, completely re-establish functional state of historical ecosystem that was degraded before invasion of alien species; for plants, increase in abundance of native species and in species diversity, including threatened native species.

1.3.1 Predation negative (not relevant for plants)
0 No impact known or detectable.
1 Predation known but negligible, no decline of native species.
2 Predation on several abundant species, without large impact on affected species or decline of their populations.
3 Decline of one to several native species recognized, minor change in food web structure reported.
4 Decline of many species, indirect impact by mesopredator release, clear changes in the food web.
5 Preys also on endemic or species listed as vulnerable, endangered or critically endangered by IUCN, local extinction.

1.3.2 Predation positive (not relevant for plants)
0 No impact known or detectable.
+1 Predation on species degrading the ecosystem known but negligible, no decline of species.
+2 Predation on one or several species degrading the ecosystem, without large impact on affected species or decline of their populations.
+3 Decline of one to several native species degrading the ecosystem recognized, minor change in food web structure reported, some indications that the ecosystem changes towards its historical functional state.
+4 Decline of one to many species degrading the ecosystem, clear changes in the food web, strong indications that the ecosystem changes towards its historical functional state.
+5 Completely re-establish functional state of historical ecosystem that was degraded before invasion of alien species.

1.4.1 Transmission of diseases to wildlife negative
0 No impact known or detectable.
1 Host (plant or animal) for non-specific parasites, occasional transmission of more or less harmless diseases to one native species. No population decline in native species. If a plant, species is not a breeding ground for wildlife disease vectors.

2 Occasional transmission of more or less harmless diseases, several native species affected. No or only minor population decline in native species. If a plant, species may be a breeding ground for wildlife disease vectors, but no more so than native plant species.
3 Many native species affected, frequent transmission of more or less harmless diseases or harmful diseases transmitted to one native species. Minor population decline in native species. If a plant, may be a more significant breeding ground for wildlife disease vectors than native plant species.
4 Transmits harmful diseases to several native species or more or less harmless diseases to endemic or species listed as vulnerable, endangered or critically endangered by IUCN. Moderate population decline in native species. If a plant, a major breeding ground for wildlife disease vectors, outbreaks due to species presence uncertain.
5 Transmits harmful diseases to many species and/or species listed as vulnerable, endangered or critically endangered by IUCN by direct transmission, decline of these species or extinction. If a plant, a major breeding ground for wildlife disease, outbreaks due to species presence certain.

1.4.2 Transmission of diseases to wildlife positive

0 No impact known or detectable.
+1 Occasional transmission of more or less harmless diseases to one species degrading the ecosystem; no population decline in species. Potential positive effect on health of wildlife (direct: e.g. potential medicinal species; indirect: e.g. antagonist of a health threat), but not yet reported.
+2 Occasional transmission of more or less harmless diseases, one or several species degrading the ecosystem affected. No or only minor population decline in species. Occasional, small positive effect on health of wildlife.
+3 One to many species degrading the ecosystem affected, frequent transmission of more or less harmless diseases or harmful diseases transmitted to one species degrading the ecosystem. Minor population decline in species, some indications that the ecosystem changes towards its historical functional state. Regularly small positive effect on health of wildlife, or occasional, larger positive effect on health of wildlife.
+4 Transmits harmful diseases to one to several species degrading the ecosystem. Moderate population decline in species, strong indications that the ecosystem changes towards its historical functional state. Regularly leading to larger positive effect on health of wildlife.
+5 Completely re-establish functional state of historical ecosystem that was degraded before invasion of alien species. Massive positive effect on health of wildlife caused by species.

1.5.1 Hybridization negative

0 No impact known or detectable.
1 Hybridization possible in captivity, but only rarely in the wild.
2 Hybridization is more common in the wild, no offspring, but constraints to normal mating.
3 Hybridization is more common, with offspring, but not fertile.
4 Hybridization common with fertile offspring.
5 Risk of extinction of endangered species.

1.5.2 Hybridization positive

0 No impact known or detectable.
+1 Hybrids are capable of coping with degraded ecosystem process(es), e.g. shown in laboratory experiments, but no indications are found in the field.
+2 Hybrids are able to cope with degraded ecosystem process(es) in the field.
+3 Some/first indications that hybrid changes the ecosystem towards its historical functional state.
+4 Strong indications that hybrid changes the ecosystem towards its historical functional state.
+5 Completely re-establish functional state of historical ecosystem that was degraded before invasion of the hybrid species.

1.6.1 Impact on ecosystem (other than mentioned before, i.e. chemical, physical or structural changes) negative

0 No impact known or detectable.
1 Change in chemical (e.g. eutrophication, nutrient-cycling), physical (e.g. soil compaction, structure, hydrology) and/or structural (e.g. felled trees, burrows, disturbance dynamics) characteristics detectable, but no impact on performance of natives or successional processes.
2 Moderate change in chemical, physical and/or structural characteristics, only slight impact on performance of natives or successional processes.
3 Major change in chemical, physical and/or structural characteristics, change in fauna and flora and/or successional processes, reversible.
4 Severe changes in chemical, physical and/or structural characteristics, decline of species and/or change in species composition, strong impact on successional processes, but likely to be reversible.
5 Massive changes in chemical, physical and/or structural characteristics, endemic species and/or species listed as vulnerable, endangered or critically endangered by IUCN affected, decline of species and/or change in species composition, very strong impact on successional processes, loss of habitat characteristics, damage of sites of conservation importance, irreversible.

1.6.2 Impact on ecosystem positive

0 No impact known or detectable.

+1 Change towards historical state of the ecosystem in chemical (e.g. eutrophication, nutrient-cycling), physical (e.g. soil compaction, structure, hydrology) or structural (e.g. felled trees, burrows, disturbance dynamics) characteristics detectable, but no decline of populations of species responsible for the ecosystem degradation or successional processes.

+2 Moderate change towards historical state of the ecosystem in chemical, physical or structural characteristics, only slight decline of populations of species responsible for the ecosystem degradation or successional processes.

+3 Major change towards historical state of the ecosystem in chemical, physical or structural characteristics, decline of populations of species responsible for the ecosystem degradation, major change towards historical state of the ecosystem in fauna and flora or successional processes.

+4 Severe changes in chemical, physical or structural characteristics, major decline of species responsible for the ecosystem degradation or severe change towards historical state of the ecosystem in species composition or successional processes.

+5 Complete change towards historical state of the ecosystem in chemical, physical or structural characteristics, removal of species responsible for the ecosystem degradation, re-establishment of historical habitat characteristics and successional processes.

2. Socio-economic impact

2.1.1 On agriculture negative

0 No impact known or detectable.

1 Only occasional damage or yield loss to crops or plantations (e.g. orchards), damage similar to native species; for plants, plant present, but no operational obstruction or removal/control cost.

2 Damage or yield loss to crops common, damage or yield loss similar to native species; for plants, little operational obstruction or removal/control cost. Some trade disruptions.

3 Regular damage or yield loss similar to native species through feeding on crops or through competition, occasional threat to stored food, losses exceed impact of the native fauna and flora, sometimes reaching high levels; for plants, operational obstruction and costs to remove/manage invader are still minor. Moderate trade disruptions.

4 Regular high damage or yield loss in fields or to stored food, fruit consumption; for plants, operational obstruction and costs to remove/manage invader are considerable. High trade disruptions.

5 Complete loss of yield or destruction of fields or plantations (e.g. orchards), or of stored food by consumption and contamination; for plants, operational obstruction and costs to remove/manage invader prohibit profitable agriculture on invaded land. Massive trade disruptions.

2.1.2 On agriculture positive

0 No impact known or detectable.

+1 Biological traits and life-style suggest potential positive influence on the yield or quality, but not yet reported.

+2 Occasionally leading to additional yield or increased quality, yield or quality increase small.

+3 Regularly leading to small yield or quality increase or occasionally to larger yield or quality increase.

+4 Regularly leading to larger yield or quality increase.

+5 Massive yield or quality gain caused by species.

2.2.1 On animal production negative

0 No impact known or detectable.

1 Occasional competition with, or loss of yield in livestock or animal production. Plant present but no operational obstruction or removal/control cost.

2 Competition with, or loss of yield in livestock or animal production, transmission of diseases to livestock or production animals in the native area, but not yet reported from the area of introduction. Little operational obstruction or removal/control cost. Some trade disruptions.

3 Competition more frequent with several livestock or production animal species, transmission of diseases reported, but infection rates low. Pollution by droppings on farmland which domestic stock are then reluctant to graze; for plants, loss of yield in livestock or production animals common, operational obstruction or removal/control cost minor. Plant may be toxic to livestock or production animals. Moderate trade disruptions.

4 For animals, transmission of economically important diseases or hybridization with economically important game animals; for plants, loss of yield in livestock or production animals major, operational obstruction or removal/control cost considerable. Plant toxic to livestock or production animals, fatalities uncommon. Large trade disruptions.

5 For animals, transmission of harmful diseases to or hybridization with livestock or production animals; for plants, loss of yield in livestock or production animals major, operational obstruction or removal/control costs are prohibitive. Plant highly toxic to livestock or production animals, fatalities reported. Massive trade disruptions.

2.2.2 On animal production positive

0 No impact known or detectable.

+1 Biological traits and life-style suggest potential positive influence on animal production (e.g. direct: potential livestock or game species, fur production; indirect: e.g. fodder plant, (micro-)organisms increasing yield or quality of fodder plants), but not yet reported.

+2 Occasionally leading to increased production or quality, production increase small.
+3 Regularly leading to small production or quality increase or occasionally to larger production or quality increase.
+4 Regularly leading to larger production or quality increase.
+5 Massive production or quality gain caused by species.

2.3.1 On forestry negative

0 No impact known or detectable.
1 For animals, minor impact through herbivory; for plants, little or no loss of yield or quality or operational obstruction, no change to forest structure or regeneration.
2 For animals, impact through herbivory, minor effect on forest growth, impact on seed dispersal; for plants, minor loss of yield or quality, or operational obstructions, minor changes to forest structure, minor reduction in regeneration. Some trade disruptions.
3 For animals, constrains forest regeneration through browsing on young trees, damage to plantations, gnawing of bark, damage by causing floods; for plants, moderate loss of yield or quality, changes in forest structure, impeded regeneration. Moderate trade disruptions.
4 For animals; moderate to strong damage to mature forest through seed consumption, bark stripping or antler rubbing, death of trees by felling or flooding. Killing trees by defoliating them for nesting material; for plants, strong loss of yield or quality, decline in desired canopy tree species, decline in regeneration potential, major changes to forest structure. Large trade disruptions.
5 For animals; very strong damage to mature forest through seed consumption, bark stripping or antler rubbing, death of trees by felling or flooding; for plants, very strong loss of yield or quality, complete loss or replacement of desired canopy tree species, no regeneration, complete change in forest structure. Massive trade disruptions.

2.3.2 On forestry positive

0 No impact known or detectable.
+1 Biological traits and life-style suggest potential positive influence on forest production (e.g. direct: potential forestry species; indirect: e.g. (micro-)organisms increasing yield or quality of forestry plants), but not yet reported.
+2 Occasionally leading to increased forestry production or quality, compared to native species, production or quality increase small.
+3 Regularly leading to small forestry production or quality increase, compared to native species, or occasionally to larger production or quality increase.
+4 Regularly leading to larger forestry production or quality increase, compared to native species.
+5 Massive forestry production or quality gain, compared to native species, caused by species.

2.4.1 On infrastructure negative

0 No impact known or detectable.

1 Biological traits and life-style suggest potential damage to infrastructure (e.g. potential to increase soil erosion and decrease road stability, physical damage to property and infrastructure, disruption to transport and communications) but not yet reported.

2 Occasional damage with minor economic losses, e.g. damage to fences, impact through pollution, accumulations of droppings, minor increases in soil erosion, localized damage to buildings and ground surfaces from roots and rhizomes (for plants), rare infrastructure problems (clogging up waterways, festooning power lines for plants),

3 Damage to fences and/or plantations, gnawing electricity cables etc., causing road accidents, nesting on current conductions. Moderate increase in soil erosion, moderate damage to property, buildings and infrastructure, frequent obstruction of waterways.

4 Considerable damage to property and infrastructure, with considerable economic costs, damage through burrowing or nesting in buildings, or roots and rhizomes of plants. Major obstruction of waterways.

5 Considerable damage to flood defence systems or other critical infrastructure, major soil erosion, danger to human safety, threat to transport safety.

2.4.2 On infrastructure positive

0 No impact known or detectable.

+1 Have traits or attributes likely to help preserve infrastructure, but not yet reported.

+2 Minor ability to preserve and enhance infrastructure, but performance no better than native or non-plant alternatives. No economic gain.

+3 Moderate ability to preserve and enhance infrastructure (prevent soil erosion), better than non-plant alternative. Marginal economic gain.

+4 Strong ability to preserve and enhance infrastructure better than non-plant alternative, e.g. flood defence and soil preservation, prevention of landslides. Moderate economic gain.

+5 Best option for preserving and enhancing infrastructure, better than non-plant alternative, high economic gain, preserves human safety.

2.5.1 On human health negative

0 No impact known or detectable.

1 Host of one or more harmless diseases with the possibility of infecting humans, not yet reported; for plants, known to be mildly toxic, causing mild discomfort, no cases yet reported.

2 Host of several harmless diseases, indirect transmission or possibility of direct transmission, but only a small percentage of the human population at risk, health hazard from soil and water contamination caused by drop-

pings; for plants, mildly toxic or causing mild discomfort, exposure risk low (not easily ingested, not airborne, direct contact causes no reaction), few cases reported.

3 Direct infection with one or more harmless diseases, occasional health threat through bites or other attacks; for plants, toxic, and/or causing pain, injury or discomfort, exposure risk moderate (poisoning through ingestion, airborne, direct contact causes reaction), moderate number of cases reported.

4 Direct transmission of several diseases, infection by contaminated food common, host of harmful diseases in the native range, but not yet known from the invaded range. Health threat through bites or other injuries happen more often, rarely fatal. Plants highly toxic, and/or causing strong pain/discomfort, but rarely fatal - many cases reported. Exposure risk high, through ingestion, contamination, direct contact, airborne.

5 Vector of harmful diseases to humans and/or many diseases frequently transmitted. Health threat through bites or other injuries happen frequently, more often fatal. Plants highly toxic, causing severe pain and/or discomfort, fatalities reported, or severe disruption to daily life caused through effects on human health. High risk of exposure.

2.5.2 On human health positive

0 No impact known or detectable.

+1 Biological traits and life-style suggest potential positive effect on human health (direct: e.g. potential medicinal species; indirect: e.g. antagonist of a health threat, ameliorating human living conditions), but not yet reported.

+2 Occasional, small positive effect on human health.

+3 Regularly small positive effect on human health, or occasional, larger positive effect on human health.

+4 Regularly leading to larger positive effect on human health.

+5 Massive positive effect on human health caused by species.

2.6.1 On human social life negative

0 No impact known or detectable.

1 Biological traits and life-style suggest potential for disturbance in recreational or residence areas (e.g. by noise, pollution, overgrowing), but nothing yet reported.

2 Occasional small disturbance, only small percentage of human population affected.

3 Regular small disturbance, or occasional larger disturbance.

4 Regular larger disturbance. Recreational value of a habitat or a landscape strongly affected.

5 Massive disturbance; complete loss of recreational value of a habitat or a landscape.

2.6.2 On human social life positive

0 No impact known or detectable.

+1 Biological traits and life-style suggest potential positive effect for recreational or residence areas (e.g. charismatic or decorative species, species ameliorating the environment by providing e.g. shade, or having edible parts, or species potentially used for angling or hunting), but not reported so far.

+2 Occasional small positive effect for recreational or residence areas, only small percentage of human population affected.

+3 Regular small positive effect for recreational or residence areas, or occasional larger positive effect for recreational or residence areas.

+4 Regular larger positive effect for recreational or residence areas. Recreational value of a habitat or a landscape strongly increased.

+5 Massive positive effect for recreational or residence areas. Massive gain of recreational value of a habitat or a landscape.