5 Plant Invasions in Asia

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Abstract

Asia, occupying nearly 30% of the earth's terrestrial surface, is one of the most important continent known for its highly diverse culture, economy, geography, and ecology. Three of the world's five largest economies, and nearly two-thirds of the world's population, are in Asia. The continent has a diverse range of habitats including tropical moist and boreal forests, deserts, and the Arctic tundra. Eleven out of 36 global biodiversity hotspots are in Asia, all of which are threatened due to multiple human-mediated drivers including biological invasions. The number of known invasive alien plant species (IAPS) currently present in Asia is high, and their number and distribution are expected to increase further due to a lack of effective management responses, land use and climate changes, and expanding international trade, travel, and transport. IAPS such as *Ageratina adenophora*, *Chromolaena odorata*, *Lantana camara*, *Leucaena leucocephala*, *Mikania micrantha*, *Mimosa diplotricha*, *Parthenium hysterophorus*, and *Pontederia crassipes* are widespread in the tropical and subtropical regions of Asia. Most of the known IAPS in Asia have a Central and South American origin. However, information on biological invasions, especially those of plants, is poor and fragmented, hampering efforts to develop and implement policies and management interventions. The continent is lagging behind much of the world in research effort and knowledge generation related to plant invasions. Capacity, both human and otherwise, of most countries to address biological invasions is low. Most countries (particularly in Central Asia) also lack a comprehensive database of IAPS. Ecological impact studies are also lacking in Southeast, Central, and North Asia. With a few exceptions, the eco-

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nomic cost of plant invasions is also unknown in most countries. Priority actions required for effective management of IAPS in Asia include regional collaboration for research and knowledge sharing, promotion and institutionalization of biological control, and increased focus on socioecological research related to plant invasions. Additionally, efforts are required at the continental scale to make all stakeholders aware of the problem of plant invasions for the formulation of appropriate policies and implementation of effective management strategies.

Keywords

Distribution · Diversity · Global change · Impacts · Invasive alien species · Management · Native range · Policy

5.1 Introduction

Asia is the world's largest continent and occupies nearly 30% of the terrestrial surface on earth. The continent is physically, biologically, economically, and culturally diverse, rising from below sea level (South Caspian Sea plains in northern Iran) to the highest peak in the world, Mt. Everest (8849 masl). Twelve of the 20 largest countries by population are in Asia, with China (1.4 billion) and India (1.3 billion) being the most populous (www.worldometers.info/world-population/). Other countries have very high (e.g., Singapore, Bangladesh, South Korea, Philippines) to very low population densities (e.g., Mongolia, Kazakhstan, Russia, Turkmenistan). Among the world's five largest economies, three are in Asia (China, Japan, and India) (https://www.worldometers.info/gdp/gdp-by-country/).

Ecologically diverse ecosystems including equatorial tropical rainforests, hot deserts, cold and hot arid steppe, and boreal forests occur in Asia. Eleven of the 36 Global Biodiversity Hotspots are located in Asia: two in East Asia, three in each of Southeast and South Asia, two in West Asia, and one in Central Asia (Mittermeier et al. 2011; Critical Ecosystem Partnership Fund, www.cepf.net/node/4422). The continent also has 5 (China, India, Indonesia, Malaysia, Philippines) of the 17 most mega-diverse countries in the world. Out of 238 global ecoregions of conservation priority, more than 50 terrestrial and freshwater ecoregions (out of 195 globally) are present in Asia (Olson and Dinerstein 2002).

The higher number of biodiversity hotspots in Asia (Mittermeier et al. 2011) suggests that the continent is not only rich in biodiversity, including endemic species, but is also witnessing a rapid loss of primary natural habitats. As elsewhere in the world, the rich biodiversity and natural environment of Asia have been threatened due to anthropogenic activities including biological invasions (IPBES 2018). A large number of alien plant species have already naturalized in different regions of the continent (Sect. 5.2), with many of them inflicting detrimental impacts on the environment and economy (Sect. 5.5). The national response capacities of most of the countries in Asia (except China and Japan) to address emerging risks associated with biological invasions are poor compared to some countries in North America, Western Europe, and Oceania (Early et al. 2016). This situation may lead to an increase in the number of invasive alien species (IAS) and their impacts in the future (Paini et al. 2016; Seebens et al. 2015). Many Asian countries are lagging in terms of research efforts and knowledge generation, which might contribute, along with other factors, to inadequate management and policy responses to plant invasions (Sect. 5.6).

In this chapter, we review the diversity and distribution patterns of invasive alien plant species (IAPS) across the major regions and countries in Asia, their biogeographic origin, and introduction pathways, impacts on environment and socio-economy, and management approaches including policy responses. We also highlight knowledge gaps and prospects for future research to improve the knowledge base for informed management and policy decisions. We use the terms such as "alien," "casual," "naturalized," and "invasive" species following the definition given by Pyšek et al. (2004). Considering physical and biological variation, and for ease of presentation, we divide the continental Asia into six regions: East (6 countries), Southeast (11), South

(8), West (16), Central (5), and North Asia (Russia).

5.2 Diversity

The number of alien species is continuously increasing worldwide, without any sign of abatement (Seebens et al. 2017). Increasing movement of people and goods has dramatically increased the number of organisms being moved around the world, many of which have established and proliferated outside of their native range. The key factors that determine the number of alien species at national or regional levels are per capita gross domestic product, population density, and percentage of lands used for agriculture (Essl et al. 2019). Based on available data, the numbers of naturalized plant species currently present in Asian countries are relatively low compared to countries in Western Europe and North America (van Kleunen et al. 2015), but the scenario is most likely to change in the near future because South and East Asian countries (India, South Korea, Thailand, and China) are expected to witness the highest increase in absolute number of naturalized species in future with their expanding global trade and economic growth (Seebens et al. 2015). Generally, with a very few exceptions, Asian countries lag far behind in generating biodiversity-related information (Meyer et al. 2016), which obviously includes data on the occurrence and distribution of alien species. Some countries in Asia are yet to produce national checklists of IAS. Recently, the Global Register of Introduced and Invasive Species (GRIIS), with technical help from scientists working in respective countries, compiled country-wide lists of introduced and invasive species across the world (Pagad et al. 2018). Despite the lack of capacity and resources, especially in developing countries in Asia, to develop comprehensive lists, we have used this database to reflect the state of plant invasions in countries for which information on diversity of naturalized plant species is lacking. We are also conscious of the fact that the GRIIS database may have errors because data providers often used the terms casual, naturalized, and invasive interchangeably. For example, the number of naturalized plant species reported for the small island nation Maldives (area ~300 km^2) in South Asia is 203, which is high compared to Pakistan (area ~881,912 km^2), the second largest country in the same region, which has been reported to have only 141 naturalized plant species (Table 5.1). Similarly, 2061 species are included in the GRIIS database for India, which is significantly higher than the 471 naturalized and IAPS recorded by Inderjit et al. (2018). It seems that for many countries, the GRIIS database has also incorporated those alien species, which are currently cultivated and have not escaped into the wild, or included agriculture weed species of native origin, which has led to the higher number. In essence, the major problem with documentation of IAPS is the non-uniform adoption of standard definitions of alien, casual, naturalized, and invasive species by different workers, which leads to either over- or underestimation of species numbers (Khuroo et al. 2011a, 2012a).

5.2.1 East Asia

East Asia covers about 11.9 million km^2 with a combined population of *ca.*1.6 billion people. The countries in this region include the People's Republic of China (China), Japan, Mongolia, the Democratic People's Republic of Korea (North Korea), and Republic of Korea (South Korea). Considered as regions or provinces, Hong Kong, Macau, and Taiwan were included in the data set for China. Owing to wide-ranging geographical and ecological conditions, East Asia has many naturalized plant species, especially China, and the risk is ever-increasing through cross-border trade and travel. China has 861 naturalized plant species (Jiang et al. 2011) of which 324 species are invasive (Axmacher and Sang 2013; Shen et al. 2018). Families with the most IAPS are the Asteraceae (60 species), Poaceae (42), Fabaceae (28), and Brassicaceae (22). Major IAPS in China include *Alternanthera philoxeroides*, *Ambrosia artemisiifolia*, *Ageratina adenophora*, *Pontederia crassipes*, *Mikania micrantha*, *Solidago canaden-*

Table 5.1 Number of naturalized species reported from Asian countries. The data extracted from GRIIS database (www.griis.org) on September 2019, except otherwise indicated. Data for Lebanon is not available (NA)

Region and country	Area (km^2)[a]	Number of naturalized plant species
East Asia		
China	9,708,095	861[b]
Japan	377,930	1542
Mongolia	1,564,110	35
North Korea	120,538	243
South Korea	100,210	499
Taiwan	36,193	627
Southeast Asia		
Brunei	5765	110
Cambodia	181,035	125
Indonesia	1,904,569	651[d]
Laos	236,800	250
Malaysia	330,803	287
Myanmar	676,578	117
Philippines	342,353	345
Singapore	710	532
Thailand	513,120	131
Timor-Leste	14,874	412[f]
Vietnam	331,212	243
South Asia		
Afghanistan	652,230	56
Bangladesh	147,570	107
Bhutan	38,394	244
India	3,287,590	471[c]
Maldives	300	203
Nepal	147,181	179
Pakistan	881,912	141
Sri Lanka	65,610	115
West Asia (Middle East)		
Bahrain	765	11
Cyprus	9251	341
Iran	1,648,195	118
Iraq	438,317	53
Israel	20,770	196
Jordan	89,342	69
Kuwait	17,818	12
Lebanon	10,452	NA
Oman	309,500	24
Qatar	11,586	8
Saudi Arabia	2,149,690	82
State of Palestine	6220	11
Syrian Arab Republic	185,180	30
Turkey	783,562	228[e]
United Arab Emirates	83,600	29
Yemen	527,968	208
Central Asia		
Kazakhstan	2,724,900	15
Kyrgyzstan	199,951	5
Tajikistan	143,100	7
Turkmenistan	488,100	6
Uzbekistan	447,400	7
North Asia		
Russia (including European Russia)	17,098,242	956

(continued)

[a]https://www.worldometers.info/geography/largest-countries-in-the-world/, accessed on 25 January 2020
[b]Jiang et al. (2011)
[c]Inderjit et al. (2018)
[d]Tjitrosoedirdjo (2005)
[e]Uludag et al. (2017)
[f]Westaway et al. (2018)

sis, *Flaveria bidentis*, and *Spartina alterniflora* (Wan et al. 2017). In Japan, 1552 alien species of vascular plants are naturalized (Mito and Uesugi 2004), of which 149 species are invasive (NIES 2019). The most species-rich invasive plant families are the Asteraceae (40 species), Poaceae (18), Fabaceae (9), and Scrophulariaceae (8). The most frequently reported IAPS occurring in riparian zones of Japan include *Solidago altissima*, *Robinia pseudoacacia*, *Erigeron canadensis*, *Paspalum distichum*, and *Sorghum halepense* (Miyawaki and Washitani 2004).

According to Jung et al. (2017), there are 320 alien plant species belonging to 181 genera and 46 families in South Korea with Poaceae (75 species), Asteraceae (63), Fabaceae (22), and Brassicaceae (20) being the most species-rich families. The most widely distributed species are *Phytolacca americana*, *Amorpha fruticosa*, *Robinia pseudoacacia*, *Trifolium repens*, *Ambrosia artemisiifolia*, *Bidens frondosa*, *Erigeron canadensis*, *E. annuus*, *Galinsoga quadriradiata*, and *Taraxacum officinale*. In North Korea, 226 alien plant species belonging to 162 genera and 64 families have been recorded (Son et al. 2009). Families with a high number of alien plants are Asteraceae (29 species), Fabaceae

(22), Poaceae (18), and Solanaceae (11). *Ambrosia artemisiifolia*, *Galinsoga parviflora*, and *E. canadensis* were prioritized for management due to their high invasiveness (Son et al. 2009). According to Kim and Kil (2016), South and North Korea combined (i.e., Korean Peninsula) have 504 alien plant species, of which 48 (9.5%) are invasive. In Mongolia, 51 IAPS belonging to 48 genera and 23 families are reported with the most species-rich families as Poaceae (8 species), Fabaceae (7), and Asteraceae (6) (Urgamal 2017). Based on the available data, China has the highest number of IAPS in East Asia, followed by South Korea, North Korea, Japan, and Mongolia.

5.2.2 Southeast Asia

Southeast Asia includes Brunei, Cambodia, Indonesia, Laos, Malaysia, Myanmar, Singapore, Philippines, Timor-Leste, Thailand, and Vietnam. The region is geographically south of China, east of the Indian subcontinent, and northwest of Australia. Current knowledge of invasive alien species in many countries in this region is largely based on anecdotal evidence (Peh 2010). This finding is supported by a study in the Lower Mekong Basin where it was found that there was a lack of information on the spread and impacts of invasive species in Cambodia, Lao PDR, Thailand, and Vietnam (MWBP and RSCP 2006). Available databases, mainly GRIIS (2019), and literature surveys revealed that Indonesia has the highest number of naturalized plants with 651 species, followed by Singapore (532 species) and Timor-Leste (412), with the lowest number in Brunei (110) (Table 5.1). Many of these naturalized species are invasive in the region. A review by Nghiem et al. (2013) revealed that there were 151 IAS in the region of which 75 were plant species, with the highest number of IAPS recorded from the Philippines (34), followed by Indonesia (32), Singapore (26), and the lowest number in Brunei (5). Two IAPS, *Lantana camara* and *Leucaena leucocephala*, have been reported in all 11 Southeast Asian countries while *Chromolaena odorata* and *Pontederia crassipes* from 10 countries of this region (Table 5.2).

In the Global Compendium of Weeds, Randall (2012) recorded 2150 weed species in Southeast Asia. In comparison, Waterhouse (1993) listed 232 major weed species of which 140 were highly important and 63 were believed to be alien. According to Randall (2012), only 95 species could be regarded as IAPS in Indonesia, followed by 38 in Vietnam and 32 in Cambodia. The species shared by at least ten countries in the region include *C. odorata*, *P. crassipes*, *Eleusine indica*, *L. leucocephala*, *L. camara*, *Mimosa pudica*, *Pistia stratiotes*, *Psidium guajava*, and *Scoparia dulcis*. Witt (2017) only lists 56 IAPS as posing the biggest threat to biodiversity and livelihoods in the region, which seems to be an underestimate, and lists 5 aquatic species (e.g., *P. crassipes*, *Salvinia molesta*), 3 grasses (e.g., *Brachiaria mutica*, *Cenchrus echinatus*), 9 climbers (e.g., *Mikania micrantha*, *Passiflora foetida*), 11 herb species (e.g., *Parthenium hysterophorus*, *Sphagneticola trilobata*), 13 shrub species (e.g., *C. odorata*, *L. camara*), 2 succulents (e.g., *Jatropha gossypiifolia*), and 13 tree species (e.g., *L. leucocephala*, *Mimosa pigra*).

There are a number of country reviews although many of these appear to be rather incomplete such as for Brunei and Cambodia, while other countries (e.g., Indonesia, Singapore) have more detailed information. In a review by Tamit (2003), no IAPS was reported for Brunei, with five being reported by Nghiem et al. (2013) 10 years later. Cambodia's Sixth National Report to the Convention on Biological Diversity states that "information on invasive alien species in forest ecosystems in Cambodia is very limited" and mentioned the occurrence of 13 IAPS with *M. pigra*, *Mimosa diplotricha*, *C. odorata*, and *M. micrantha* as being particularly problematic (Department of Biodiversity 2019). In Indonesia, Tjitrosoedirdjo (2005) reported the presence of 1,936 alien plant species belonging to 87 families with Asteraceae (162) and Poaceae (120) being the most speciose families. Approximately one-third (651 species) of the total alien species listed are either naturalized or agricultural weeds. The

Table 5.2 Countries of occurrence of the 21 IAPS (included in the 100 among the world's worst invasive species, Lowe et al. 2000) in different regions of Asia. North Asia has been excluded from the table because none of the listed species have been reported from that region

SN	Name of species	Regions in Asia				
		East Asia	SE Asia	South Asia	West Asia	Central Asia
1	*Acacia mearnsii*	China, Japan	Indonesia, Vietnam	India, Pakistan, Sri Lanka	–	–
2	*Cecropia peltata*	–	Malaysia	–	–	–
3	*Chromolaena odorata*	China, Taiwan	Brunei, Cambodia, Indonesia, Laos, Malaysia, Myanmar, Philippines, Singapore, Thailand, Vietnam	Bangladesh, Bhutan, India, Nepal	–	–
4	*Cinchona pubescens*	–	–	India, Sri Lanka	–	–
5	*Clidemia hirta*	Japan, Taiwan	Brunei, Malaysia, Singapore, Thailand, Vietnam	India, Sri Lanka	–	–
6	*Lantana camara*	China, Japan, Taiwan	Brunei, Cambodia, Indonesia, Laos, Malaysia, Myanmar, Philippines, Singapore, Thailand, Timor-Leste, Vietnam	Afghanistan, Bangladesh, Bhutan, India, Maldives, Nepal, Pakistan, Sri Lanka	Cyprus, Iran Israel, Palestine, Saudi Arabia, Turkey, Yemen	–
7	*Leucaena leucocephala*	China, Japan, Taiwan	Brunei, Cambodia, Indonesia, Laos, Malaysia, Myanmar, Philippines, Singapore, Thailand, Timor-Leste, Vietnam	Bangladesh, Bhutan, India, Maldives, Nepal, Pakistan, Sri Lanka	Bahrain, Cyprus, Iraq, Iran, Israel, Lebanon, Saudi Arabia, Yemen	–
8	*Melaleuca quinquenervia*	China	Malaysia, Thailand, Vietnam	India	–	–
9	*Miconia calvescens*	–	–	Sri Lanka	–	–
10	*Mikania micrantha*	China, Taiwan	Brunei, Cambodia, Indonesia, Laos, Malaysia, Myanmar, Philippines, Singapore, Thailand, Vietnam	Bangladesh, Bhutan, India, Nepal, Sri Lanka	–	–
11	*Mimosa pigra*	Taiwan	Brunei, Cambodia, Laos, Malaysia, Myanmar, Philippines, Singapore, Thailand, Vietnam	India, Sri Lanka	–	–
12	*Opuntia stricta*	China, Taiwan	Vietnam	India, Nepal, Sri Lanka	Yemen	–
13	*Pinus pinaster*	Japan	–	–	–	–
14	*Pontederia crassipes*	China, North Korea, Japan, South Korea, Taiwan	Brunei, Cambodia, Indonesia, Laos, Malaysia, Myanmar, Philippines, Singapore, Thailand, Vietnam	Bangladesh, Bhutan, India, Maldives, Nepal, Pakistan, Sri Lanka	Iran, Iraq, Israel, Jordan, Lebanon, Palestine, Syria, Turkey	–
15	*Prosopis glandulosa*	Japan	–	India, Pakistan	Iran, Israel	–

(continued)

Table 5.2 (continued)

SN	Name of species	Regions in Asia				
		East Asia	SE Asia	South Asia	West Asia	Central Asia
16	*Psidium cattleianum*	China	Malaysia	–	–	–
17	*Salvinia molesta*	China, Japan, Taiwan	Indonesia, Malaysia, Philippines, Singapore, Thailand	Bangladesh, India, Pakistan, Sri Lanka	Israel	–
18	*Spartina anglica*	China, North Korea, South Korea	–	–	–	–
19	*Spathodea campanulata*	China, Taiwan	Laos, Malaysia, Philippines, Singapore, Thailand	Maldives	–	–
20	*Sphagneticola trilobata*	China, Japan, Taiwan	Malaysia, Singapore, Thailand	India, Maldives, Nepal, Sri Lanka	Kuwait	–
21	*Ulex europaeus*	China, Japan	–	India, Sri Lanka	Turkey	Tajikistan

author also listed 5 species (e.g., *P. crassipes*, *S. molesta*) as important IAPS in aquatic habitats and 20 species (e.g., *C. odorata*, *L. camara*, *M. micrantha*) in terrestrial habitats. More recently, Setyawati et al. (2015) listed 362 plant species from 73 families as invasive in Indonesia. According to Nghiem et al. (2013), there are 20 IAPS in Malaysia, followed by Myanmar (13) and Laos (9). A recent report mentioned more than 20 IAPS (e.g., *C. odorata*, *M. micrantha*, *M. pigra*) in Myanmar (NBSAP Myanmar 2015). Bakar (2004) reported more than 100 weed species in Malaysian agro-ecosystems, many of which have been introduced including *Alternanthera philoxeroides*, *Clidemia hirta*, and *Myriophyllum aquaticum*. A floristic study of floodplain secondary forests in Peninsular Malaysia revealed that the naturalized species contributed 23% (23 of 99 species) to the total species documented (Hashim et al. 2010).

According to Sinohin and Cuaterno (2003), more than 475 plant species were intentionally introduced to the Philippines during historical times, mainly from the Malayan region. Nghiem et al. (2013) reported the presence of 34 IAPS in the Philippines with 10 terrestrial (e.g., *Gmelina arborea* and *L. camara*) and 2 wetland species (*P. crassipes* and *S. molesta*) considered to be highly problematic (Sinohin and Cuaterno 2003). In Singapore, Corlett (1988) reported the naturalization of 136 plant species, with Fabaceae (29 species) being the most speciose family followed by Asteraceae (15) and Poaceae (13). Among them, 26 species were reported as IAPS including 3 species such as *Cecropia pachystachya*, *L. leucocephala*, and *Spathodea campanulata* (Nghiem et al. 2013). There were 24 and 16 IAPS reported from Thailand and Vietnam, respectively (Nghiem et al. 2013). However, Tan et al. (2012), during a survey of 9 national parks and 1 natural conservation area in Vietnam reported 134 naturalized plant species including 25 IAPS. The National Biodiversity Strategy and Action Plan of Timor-Leste (NBSAP Timor-Leste 2015) reported the presence of at least nine IAPS including *C. odorata* and *L. leucocephala*.

5.2.3 South Asia

In South Asia, one of the most populous regions in the world, research documenting the diversity of IAPS is still insufficient (Pallewatta et al. 2003) and mostly based on reviews of the floristic literature (Khuroo et al. 2011a). In India, the largest country in the region, a number of studies have documented the diversity of alien and/or invasive flora. Khuroo et al. (2012a) compiled a comprehensive inventory of the alien flora of India, which included 225 invasive species. The families contributing the most IAPS included the Asteraceae (43 species), followed by Amaranthaceae and Euphorbiaceae (14 each) and Poaceae and Solanaceae (13 each). Inderjit et al. (2018) recently reported 471 naturalized plant species in India. Major IAPS in India included *Lantana camara*, *Mikania micrantha*, *Prosopis juliflora*, *Parthenium hysterophorus*, *Ageratina adenophora*, *Pontederia crassipes*, *Salvinia molesta*, *Nymphaea mexicana*, *Alternanthera philoxeroides*, and *Myriophyllum aquaticum*. In Pakistan, Qureshi et al. (2014) documented 73 IAPS including *P. hysterophorus*, *P. juliflora*, *L. camara*, and *Broussonetia papyrifera* which are considered to be highly problematic invasive species. Bambaradeniya (2002) listed 39 IAPS in Sri Lanka including *P. crassipes*, *P. juliflora*, *Mimosa diplotricha*, and *Leucaena leucocephala*. Wijesundara (2010) reported 28 IAPS as being common and widespread. In Nepal, there are 179 naturalized flowering plants, of which 26 are considered invasive (Shrestha 2019). Some of the highly problematic species in Nepal are *A. adenophora*, *Ageratum houstonianum*, *Chromolaena odorata*, *P. crassipes*, *L. camara*, *M. micrantha*, and *P. hysterophorus*. In Bhutan, of 964 alien plant species present, 335 species occur outside cultivated areas of which 131 are casual aliens, 103 naturalized, and 101 invasive (Dorjee et al. 2020). Among the invasive species, major ones are *M. micrantha*, *C. odorata*, *A. adenophora*, *P. hysterophorus*, and *Tithonia diversifolia* (Yangzom et al. 2018). According to the GRIIS database, the number of species naturalized in Bangladesh, Maldives, and Afghanistan are 107, 203, and 56, respectively (Table 5.1). The low number of species recorded in Afghanistan may be because of inadequate research. Bangladesh does not have a comprehensive national list of IAPS, but the National Biodiversity Strategy and Action Plan (NBSAP Bangladesh 2015) reported the occurrence of 15 IAPS including *P. crassipes*, *L. camara*, and *P. hysterophorus*. Biswas et al. (2007) reported five IAPS from Sundarbans, which is a mangrove in Bangladesh. Sujanapal and Sankaran (2016) mentioned nine IAPS (e.g., *P. crassipes*, *L. camara*, *L. leucocephala*, *Sphagneticola trilobata*) in the Maldives.

5.2.4 West Asia (Middle East)

West Asia includes Turkey, Cyprus, Syria, Lebanon, Israel, Palestine, Jordan, Iraq, Saudi Arabia, Yemen, Oman, the United Arab Emirates, Qatar, Bahrain, Kuwait, and Iran. The region serves as a bridge between the Mediterranean Sea, Black Sea, Caspian Sea, Persian Gulf, Arabian Sea, and Red Sea. According to the available literature, including the GRIIS database, countries with a high number of naturalized plants species in the region include Cyprus (341 species), Turkey (228), Yemen (208), Israel (196), and Iran (118) (Table 5.1). In the region, comprehensive documentation of the alien flora is available only for Turkey, which has 31 species in the family Asteraceae, followed by Poaceae (22), Amaranthaceae (18), and Solanaceae (15) (Uludag et al. 2017). In Cyprus, 22 naturalized plant species are invasive including *Acacia saligna*, *Robinia pseudoacacia*, and *Ailanthus altissima* (Hadjikyriakou and Hadjisterkotis 2002; Spitale and Papatheodoulou 2019). Similarly, there are 13 IAPS (e.g., *A. altissima*, *Azolla filiculoides*, *Pontederia crassipes*) in Turkey (Arslan et al. 2015) and 50 (e.g., *A. altissima*, *P. crassipes*, *Lantana camara*, *Salvinia molesta*) in Israel (Dufour-Dror 2012). In Iran, *A. filiculoides*, *Prosopis juliflora*, *P. crassipes*, *Atriplex canescens*, *Pinus eldarica*, and *R. pseudoacacia* are among the most serious IAPS (A. Naqinezhad, pers.obs.). According to Soorae et al. (2015), there are only 8 IAPS in the United Arab Emirates including *P. juliflora*, *Opuntia ficus-indica*, and *Pennisetum setaceum*. Species such as *Argemone ochroleuca*, *Nicotiana glauca*, *Opuntia stricta*, *O. ficus-indica*, *P. juliflora*, and *Trianthema portulacastrum* have been reported as invasive in Saudi Arabia (Thomas et al. 2016). Alhammadi (2010) lists 12 IAPS for Yemen, including *P. juliflora*, *O. stricta*, *O. ficus-indica*, *P. hysterophorus*, and *Verbesina encelioides*. On

Socotra Island (Yemen), 22 naturalized species have been reported, of which 4 (*Argemone mexicana*, *Calotropis procera*, *Leucaena leucocephala*, and *Parkinsonia aculeata*) are reported to be invasive (Senan et al. 2012) although the current status of some of those listed is being reviewed. *Opuntia stricta* has also been reported as being invasive on Socotra (Coles 2018), but efforts are currently underway to eradicate this species (A.B.R. Witt, pers. obs.).

5.2.5 Central Asia

Central Asia includes five nations (Kazakhstan, Kyrgyzstan, Tajikistan, Turkmenistan, and Uzbekistan) which are the Republics of the former Soviet Union. This region is located between the Caspian Sea in the west to China in the east, Russia in the north to Iran, and Afghanistan in the south. Little is known about IAPS in this region. According to the GRIIS database, the highest number of naturalized plant species (15) in Central Asia is found in Kazakhstan, which occupies nearly 68% of the land area in the region (Table 5.1). In Kyrgyzstan, there are 14 species of alien plants (Sennikov et al. 2011; Lazkov and Sennikov 2014; Lazkov et al. 2014; Lazkov and Sennikov 2017), though the GRIIS database only lists 5 species as naturalized. Similarly, Tajikistan, the smallest country in the region, has nine alien plant species (Nobis and Nowak 2011a, b; Nobis et al. 2011). We did not find any published scientific study on alien flora of the remaining two countries Turkmenistan and Uzbekistan except for the GRIIS database (Table 5.1).

5.2.6 North Asia

North Asia primarily includes the Asian part of Russia (Siberia and Far East), but for convenience, we have also included the European part of Russia. Vinogradova et al. (2018) list 354 IAPS in Russia, together with their biogeographic and ecological characteristics. Of these, 277 species are present in the European part of Russia, 70 in Siberia, and 79 in the Far East. A higher number of IAPS in the European part of Russia was mainly attributed to higher human population densities associated with high levels of urbanization and associated disturbance to natural ecosystems compared to other regions of Russia (Vinogradova et al. 2018). The most widespread IAPS in Russia include *Acer negundo*, *Echinocystis lobata*, *Erigeron canadensis*, and *Elodea canadensis* (Vinogradova et al. 2018). The GRIIS database shows that 956 alien species are naturalized in Russia (Table 5.1).

In addition to national lists, inventories of alien flora are also available for different regions within Russia. For example, in the Upper Volga region (European part), there are 770 alien plant species with 135 (17.5%) and 32 (4.2%) species naturalized and invasive, respectively (Borisova 2011). Among the IAPS, *A. negundo*, *Bidens frondosa*, and *Impatiens glandulifera* are widespread in the Upper Volga region. In the Moksha River basin within the Volga Upland, there are 314 alien vascular plants which account for 25% of the total flora of this region; among these 46 species are considered to be invasive (Silaeva and Ageeva 2016). In the Middle Volga region, there are 490 alien plant species, of which 59 are invasive or potentially invasive (Senator et al. 2017). In the Middle Urals, 328 species of alien plant species have been reported (Tretyakova 2011). Similarly, in Far East Russia, 292 alien plant species have been reported from the Magadan region (Lysenko 2011), 155 species from the Yakutia region (Nikolin 2014), and 392 species from the Khabarovsk region (Antonova 2013).

5.3 Distribution

Information on the distribution of IAPS is essential for improving our understanding of the processes which drive plant invasion and to develop effective management strategies. In this section, we review spatial distribution of selected IAPS in Asia and discuss natural (e.g., climate and elevation) and anthropogenic factors (e.g., demography and economic growth) that govern diversity and distribution of IAPS. At the end of this section, we also review the status of plant invasions in protected areas and inland aquatic and wetland ecosystems.

5.3.1 Spatial Distribution

Mapping of spatial distribution of IAPS is an important approach to rapidly assess the extent of invasions across ecosystems and track dispersal vectors and pathways. Geo-referenced distribution data have increasingly been used for the prediction of suitable habitats of IAPS as a part of risk assessment. In Asia, geographic distribution patterns of individual IAPS have been analyzed only for a few species (e.g., *Lantana camara*, *Ageratina adenophora*, *Parthenium hysterophorus*), in a limited number of countries in East, Southeast, South, and West Asia. These analyses, based on climate suitability alone, reveal that the full geographic range of these species has yet to be reached, suggesting that they are likely to increase their distribution. In this section, in addition to distribution mapping of individual species, we also review multispecies studies and highlight the distribution patterns of some of the world's worst species invading different regions of Asia.

Studies Involving Single Species

Among several IAPS in Asia, most distributional studies have been undertaken for *Lantana camara*, *Ageratina adenophora*, and *Parthenium hysterophorus*. Distribution patterns and availability of the suitable habitats for *L. camara* have mainly been undertaken in India (Kannan et al. 2013; Mungi et al. 2020) and also globally (Taylor et al. 2012; Qin et al. 2016). Kannan et al. (2013) reconstructed *L. camara* introductions in India and demonstrated that the widespread occurrence of this species in India was due to its introduction between 1800 and 1900 at different cantonments during British rule. Currently the species is found throughout India with an estimated 39% of forest area invaded (Mungi et al. 2020). The success of *L. camara* in India and elsewhere has largely been attributed to extensive deforestation leading to the creation of suitable habitats (Mungi et al. 2020). Ecoclimatic models revealed that much of Asia, which is currently uninvaded by *L. camara*, has a suitable climatic condition, and as such this species is likely to expand its distribution into tropical and subtropical regions (Taylor et al. 2012; Qin et al. 2016), in the absence of effective control measures.

Ageratina adenophora is found in several Asian countries, with most studies on its distribution being undertaken in China (Wang and Wang 2006; Zhu et al. 2007; Sang et al. 2010). In China, it was first reported from Yunnan Province in the 1940s, from where it spread north and east at rates of 7–20 km/year between the 1960s and 1990s (Wang and Wang 2006; Zhu et al. 2007). Based on ecoclimatic models, it is likely to increase its range, particularly in the southern and south-central regions including the southeastern coastlands and Taiwan, where large tracts of land are still free from invasions (Wang and Wang 2006; Zhu et al. 2007).

Similarly, *Parthenium hysterophorus* has invaded East, Southeast, and South Asia, but its distribution is only known for South Asian countries (Dhileepan and Senaratne 2009; Ahmad et al. 2019a; Shrestha et al. 2019a). It has invaded all South Asian countries except Afghanistan. In Nepal, *P. hysterophorus* is widespread in the southern part of the country (Tarai and Siwalik regions), from where it is spreading north, especially along road networks (Shrestha et al. 2019a). An ecoclimatic model revealed that parts of the western Himalaya, virtually the entire northeast, and parts of Peninsular India (particularly the coastal parts of Odisha and Andhra Pradesh, southern part of Karnataka and entire Tamil Nadu) are climatically suitable for *P. hysterophorus* (Ahmad et al. 2019a). Most of Sri Lanka and Bangladesh, southern coastal and northeastern part of India, and southern part of Nepal are also a suitable climatic match (Dhileepan and Senaratne 2009). The model also revealed that in addition to South Asia, where the occurrence of *P. hysterophorus* is currently high, there are regions of high climatic suitability in eastern China, Southeast Asia, and parts of Japan and Korean Peninsula where this species is either absent or has been recorded only at a few locations (Mainali et al. 2015).

Distribution patterns and the climatic suitability of Asia to invasions by *Mikania micrantha*, *Mesosphaerum suaveolens*, *Prosopis juliflora*, and *Ambrosia confertiflora* have also been undertaken. The Western Ghats of south India, parts of northeast India, eastern parts of Vietnam and Laos, southern China, Taiwan, northern Philippines, and parts of south and west Indonesia are a good ecoclimatic match for *M. micrantha* (Banerjee et al. 2019). Padalia et al. (2014) found that nearly 40% of India, mainly in the central part, parts of the western Himalayan foothills, and tropical areas in the northeast, were a good ecoclimatic match for *M. suaveolens*.

Distribution mapping of *P. juliflora* in West Asia revealed that invasive populations were more frequent in Jordan than in Israel, possibly due to high soil moisture and efficient dispersal by domestic herds in Jordan (Dufour-Dror and Shmida 2017). Repeated mapping of *A. confertiflora* in Israel showed that it was first recorded in 1990 at a few locations with populations exploding in the last 15 years (Yair et al. 2019). By 2015, the species was widespread, particularly in the central and northern part of Israel. Occurrence of this species declined with increasing distance from road and rivers, suggesting that they serve as dispersal corridors and provide suitable microhabitat for the establishment of *A. confertiflora*.

Studies Involving Multiple Species

Efforts have also been made to predict suitable niche areas for multiple species in Southeast (SE) and South Asian countries. In SE Asian countries, about 6 million km^2 has been predicted to be suitable for one or more of ten IAPS (Truong et al. 2017). Species which are likely to invade large areas in Asia include *Ageratum conyzoides*, *Pontederia crassipes*, *Leucaena leucocephala*, *Lantana camara*, and *Mimosa diplotricha*. Based on ecological niche modeling of 155 species currently naturalized in India, Adhikari et al. (2015) found that 49% of the geographic area of the country is susceptible to further invasions with moderate to high level of climatic suitability. Coastal regions, northeastern region, and Western Himalaya have regions with high climatic suitability. The regions with high climatic suitability that overlapped with anthropogenic drivers of invasions (e.g., dense settlements, villages, croplands) were designated as "invasion hotspots," and a large proportion of these hotspots lies in global biodiversity hotspots such as the Himalaya, Indo-Burma, Western Ghats, and Sri Lanka (Adhikari et al. 2015). In Nepal, 40% of the total area, mostly representing Tarai, Siwalik, and Middle Mountain regions, has been predicted to have a suitable climate for one to many of the 24 IAPS studied (Shrestha and Shrestha 2019). Areas predicted to be suitable for the highest number of IAPS (14–20 species), based on studies undertaken, are concentrated in central Nepal. In Sri Lanka, the southern and western parts of the country are ecoclimatically a good match for five to eight IAPS, whereas the northern and eastern parts are either unsuitable for many species included in the analysis or suitable only for 1 to 2 species (Kariyawasam et al. 2019).

Distribution of Globally Worst Species

Of the 37 species of vascular plants listed in 100 of the world's worst invasive species (Lowe et al. 2000; Luque et al. 2014), 21 are present and alien in Asian countries. Among them, the maximum number of species are present in East Asia (18 species), followed by South Asia (17), Southeast Asia (14), West Asia (8), and Central Asia (1), whereas none of these species have been reported from North Asia (Table 5.2). Global modeling also revealed that the areas at high risk to invasion by species included in the list of 100 worst species are located in East, Southeast, and South Asia (Bellard et al. 2013). Most frequently occurring plant species among them are *Pontederia crassipes* (30 countries; 64% of the total 47 countries in Asia), *Lantana camara* (30; 64%), *Leucaena leucocephala* (29; 62%), *Mikania micrantha* (17; 36%), and *Chromolaena odorata* (16; 34%) (Table 5.2, Fig. 5.1). Countries with the highest number of

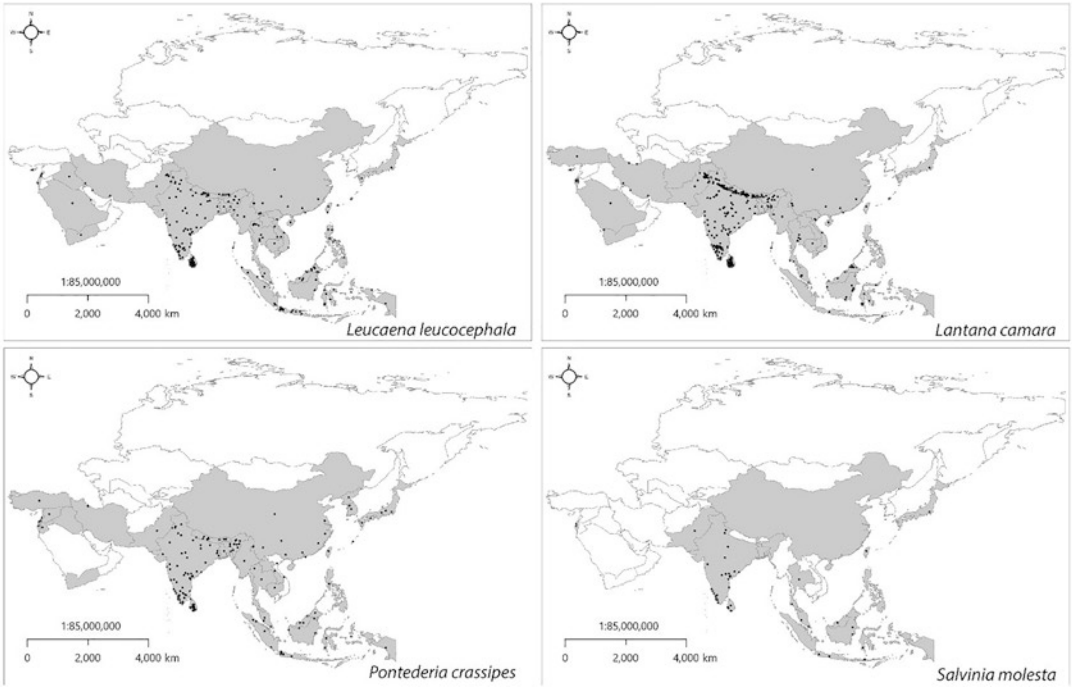

Fig. 5.1 Distribution of most frequently occurring 2 terrestrial (*Leucaena leucocephala* and *Lantana camara*) and 2 freshwater species (*Pontederia crassipes* and *Salvinia molesta*) from the list of the 100 of the world's worst invasive species that are invading Asian countries (shaded). Occurrence data was obtained from GBIF database (www.gbif.org), CABI (2020), other literatures, and individual collections (see Acknowledgment for people contributing personal collections). In few countries, presence of the species was confirmed from the literature, but there was no geographic coordinates available of their precise occurrence locations

these species are India (15 species), China (14), Malaysia (13), and Sri Lanka (12). Countries like Kazakhstan, Kyrgyzstan, Mongolia, Oman, Qatar, Russia, Turkmenistan, the United Arab Emirates, and Uzbekistan have not reported any of these species yet.

5.4 Factors Governing Plant Invasions

Occurrence of IAS is determined by complex interactions between natural (e.g., climate, native biodiversity, species traits) and anthropogenic factors (e.g., propagule pressure, disturbance). Factors that govern diversity and distribution patterns of IAPS have been analyzed in a few countries of Asia. We summarize below how climate, elevation, ecosystem attributes, infrastructure development, demography, and economic growth govern diversity and distribution of alien plants in Asia. In addition to these factors, residence time (time since introduction) also determines geographic extent of distribution of any species, but this has not been assessed in Asia except in China where it was shown that the number of provinces occupied by an invader increases with time since their introduction (Huang et al. 2010).

5.4.1 Climate and Climate Change

Understanding the role of climate in determining plant distribution is one of the classic topics in ecology (Woodward 1987). However, how climatic variables regulate distribution of alien species is a relatively understudied topic in Asia. A few studies in China suggest that the number of IAPS is high in the warm and moist regions (southeastern China) and declines in the cool and dry regions (northwest China) (Weber et al. 2008; Wu et al. 2010). In India, tropical states located southward to 20° N have a high number of naturalized plant species with the highest number (332 species) in Tamil Nadu (Inderjit et al. 2018). In the north, the states with higher precipitation during the dry season (e.g., Himachal Pradesh, 232 species) have higher numbers of naturalized species. In general, mean annual temperature and dry season precipitation are the major climate determinants of the number of naturalized plant species in India (Inderjit et al. 2018). A statistical model developed from climate anomalies also revealed a high affinity of studied IAPS to either warmer, drier, or wet places in India (Tripathi et al. 2019). In Nepal, lowland regions with tropical and subtropical climates (i.e., Tarai and Siwalik regions in the south) have higher numbers of IAPS than in the colder highlands (Shrestha 2019).

With climate change it is generally anticipated that the distribution of alien species will also change (Hulme 2017). Climate change makes ecosystems more vulnerable to invasion (Wallingford et al. 2020). It also drives the naturalization rate of introduced species and invasive potential of existing IAPS and sleeper species (Dullinger et al. 2017; Spear et al. 2021). Ecological niche modeling studies in Asia have clearly indicated that the geographic range of the majority of evaluated species will increase in future. For example, climatically suitable regions of all 11 IAPS evaluated are expected to increase, with some species establishing at higher elevations in the Western Himalaya (part of Nepal and India) (Thapa et al. 2018). In another study covering the entire Himalayan range (from Myanmar to Afghanistan), climatically suitable areas for *Ageratina adenophora*, *Chromolaena odorata*, and *Lantana camara* are likely to increase, while those of *Ageratum conyzoides* and *Parthenium hysterophorus* are likely to decrease in the future (Lamsal et al. 2018). Modeling across global ecoregions predicted an increase in plant invasion risks in ecoregions of East Asia (China) and Southeast Asia (Wang et al. 2019). Similarly, climatically suitable areas for *L. camara* may increase in China, but it may shrink in South and Southeast Asia (Taylor et al. 2012; Qin et al. 2016).

A few studies have also modeled the impacts of climate change on distribution of single or multiple IAPS in China and South Asian countries. Climatically suitable areas will increase in south and southwestern China, particularly in Guangxi, Guizhou, and Yunnan provinces, while there will be some decline in Sichuan Province in the 2080s (Wang et al. 2017). Overall, the suitable area will increase by 16%. There will be a net gain of climatically suitable areas for *L. camara* and *Senna tora* in India (Panda et al. 2018) but a net loss of suitable areas for *C. odorata* and *Tridax procumbens* (Panda and Behera 2019). In a multispecies analysis, Shrestha and Shrestha (2019) showed that the climatically suitable regions will increase for 75% of IAPS in Nepal (16 species, e.g., *L. camara*, *P. hysterophorus*, *Ageratum houstonianum*) and decline for the remaining 25% of IAPS (e.g., *Amaranthus spinosus*, *Bidens pilosa*). In Bhutan, predicted climate change (2041–2060) may increase suitable areas of four IAPS (*A. conyzoides*, *C. odorata*, *L. camara*, and *Mikania micrantha*) but reduce for two species (*A. adenophora* and *P. hysterophorus*) (Thiney et al. 2019). Areas with potential risk of invasion by a higher number of IAPS are likely to increase in Sri Lanka under future climate scenarios (2050 and 2070 for Representative Concentration Pathways, RCP 4.5 and 8.5) (Kariyawasam et al. 2019).

5.4.2 Elevation Gradient

Elevation is an important topographic feature of mountain landscapes which influences climate, such as temperature, precipitation, and solar radiation, and thus the distribution of plants and other organisms. It strongly influences the distribution of IAPS in mountain landscapes by limiting the growth of many species at higher elevations (Alexander et al. 2011). Therefore, a change in diversity of IAPS is expected along elevation gradients. A few studies in Asia have examined this pattern using interpolated, inventory, and plot-level data. While the analyses using interpolated data from species distribution range have reported unimodal relations (mid-elevation peak), other analyses using inventory and plot-level data have reported a continuous decline in the number of alien species with increasing elevation. For example, using interpolated distribution data, Bhattarai et al. (2014) reported a mid-elevation peak at *ca*. 1100 masl with lower number of naturalized species at lower and higher elevation between 60 m and 4300 masl in Nepal. Using a similar approach, Khuroo et al. (2011b) showed that the species richness of naturalized plants exhibited a unimodal relationship with elevation (500–5000 masl) in Kashmir Himalaya (India), reaching the highest species richness between 1000 and 2000 masl. A similar pattern was also observed in Himachal Pradesh (Western Himalaya, India) with the highest richness of naturalized species at 1000–1100 masl within the elevation gradient of 300–5000 masl (Ahmad et al. 2018).

Using inventory data, Akatova and Akatov (2019) reported that the number of naturalized plant species declined with increasing elevation between 100 and 2400 masl in a mountain range in the Western Caucasus, Russia. Similarly, the number of IAPS declined with increasing elevations (100–4200 masl) in the Arunachal Himalaya (India), with 13, 10, 6, and 1 species occurring in the tropical, subtropical, temperate, and subalpine zones, respectively (Kosaka et al. 2010). In Kashmir Himalaya (India), the number of naturalized species is the highest in valley plains at the lowest elevation, and it declined at higher elevation with only 14 species in the montane alpine zone (Khuroo et al. 2012b).

Using plot-level data, Leung et al. (2009) showed that the number of naturalized plant species declined linearly with increasing elevation (100–1000 masl) in the Tai Mo Shan region of Hong Kong. A similar continuous decline in richness of naturalized plant species has been reported between 1950 and 3500 masl in Eastern Himalaya, China (Yang et al. 2018), between 100 and 1000 masl in temperate mountain forests of northern China (Zhang et al. 2015), and between 1680 and 3750 masl in Kashmir Himalaya, India (Dar et al. 2018).

5.4.3 Ecosystem and Community Features

Ecosystem types and community features largely determine plant invasions at local and landscape levels. Despite the lack of consensus, ecosystems subjected to frequent disturbance that leads to the fluctuation of resources availability are, in general, vulnerable to plant invasions (Davis et al. 2000). Similarly, the diversity of native species exhibits scale-dependent responses to species invading ecosystems (Jeschke et al. 2018). However, these aspects of plant invasions have been little studied in Asia. In China, farmlands are invaded by the highest number of terrestrial IAPS species (162 species out of 170 terrestrial IAPS), followed by forests (29 species) (Xu et al. 2006). In China, regions with a high number of native plant species also tend to have a high number of naturalized species (Wu et al. 2010).

5.4.4 Infrastructure Development, Demography, and Economic Growth

Socioeconomic factors (e.g., per capita domestic growth, population density, proportion of agriculture land) are often bigger drivers of invasions than biogeographic and physical characteristics of the recipient environment (Essl et al. 2019). In regions with high population density and cross-border economic activities, propagule pressure and the proportion of disturbed habitats are high, making such regions highly vulnerable to plant invasions (Davis et al. 2000; Simberloff 2009). One of the best examples that illustrates the roles of economic growth, international trade, and population density on plant invasions is the difference between the number of alien species introduced, both intentionally and accidentally, into South Korea (256) compared to 33 into North Korea, after the division of the Korean Peninsula in 1950 (Kim and Kil 2016). According to Kim and Kil (2016), this disparity could be explained by the fact that South Korea has double the human population of North Korea and gross national per capita income which is 40 times higher and imports significantly more goods and services than its northern neighbor. In China, the number of IAPS increases with increasing road density (road length per unit area) (Weber and Li 2008). Shanghai (China) witnessed around a sixfold increase in volume of trade between 1980 and 2005, and in the same period, the number of alien species intercepted during border inspections increased more than tenfold (Ding et al. 2008). In India, demographic features such as population density and the percentage of population that live in urban areas are the major determinants of the number of naturalized plant species (Inderjit et al. 2018). In Nepal, richness of naturalized plants species is high in regions with high population density and the number of visiting tourists (Bhattarai et al. 2014).

At sub-national and local levels, transport infrastructure appears to be a major determinant for the occurrence of naturalized and invasive species. In the Kashmir valley (India), alien plant species constitute more than two-thirds of roadside flora (69%), and the richness of naturalized species declines linearly with increasing distance from the road (Dar et al. 2015). In Uttar Pradesh (India), the number of naturalized plant species increase with intensity of road use (low, medium, and high), and for all road use intensity, the species richness and relative importance of naturalized species decline as one moves away from road verges (Sharma and Raghubanshi 2009). In Manas National Park in northeast India, the occurrence of two major IAPS (*Mikania micrantha* and *Chromolaena odorata*) mainly depends on proximity to roads, among other factors (Nath et al. 2019). Similarly, distances from the nearest settlement and roads are the most important factors after tree canopy and distance from rivers in determining the occurrence of IAPS in Bardia National Park of Nepal (Bhatta et al. 2020). In general, roads facilitate plant invasions by serving as dispersal corridors for plant propagules and providing suitable microhabitats (Christen and Matlack 2006).

5.5 Plant Invasions in Special Habitats

5.5.1 Protected Areas

Plant invasions in protected areas (PAs) are increasing worldwide, and cases of successful management are very limited (Foxcroft et al. 2017; Shackleton et al. 2020), suggesting that there will be continued threats from plant invasions to global conservation goals. Despite large geographic coverage and numerous PAs in Asia, the number of studies dealing with plant invasions is very low (Hulme et al. 2014). Limited studies, however, suggest that the PAs of this region (particularly China, Southeast and South Asia, and West Asia) are invaded by a range of IAPS including species such as *Chromolaena odorata*, *Pontederia crassipes*, *Lantana camara*, and *Mikania micrantha*.

There are more than 2500 PAs in China, but studies on biological invasions have only been undertaken in 24 of these (Guo et al. 2017). The number of naturalized species reported in each PA ranged from 3 to 51, with the largest number of species in Dinghushan National Nature Reserve (51) followed by Taohongling (49) and Tianmushan (46). Some of the frequently reported species are *Alternanthera philoxeroides*, *Amaranthus spinosus*, *Euphorbia hirta*, *Erigeron annuus*, *Bidens pilosa*, *C. odorata*, and *Ipomoea purpurea* (Guo et al. 2017). In Laojun Mountain National Park (Yunnan, China), there are 61 naturalized species, of which *Galinsoga quadriradiata*, *Oxalis corniculata*, and *B. pilosa* are the most frequently occurring species (Yang et al. 2018).

In Southeast Asia, knowledge of plant invasions is limited to some PAs in Vietnam, Cambodia, and Indonesia. In a study of 10 PAs in Vietnam, Tan et al. (2012) found 8 to 15 IAPS in each PA with *C. odorata*, *P. crassipes*, *Mimosa diplotricha*, *M. pigra*, *Panicum repens*, and *M. micrantha* reported as the most problematic species. In Cambodia, Renner et al. (2011) reported seven IAPS from six PAs (Central Cardamoms

Protected Forest, Kirirom National Park, Bokor National Park, Seima Biodiversity Conservation Area, Phnom Prich Wildlife Sanctuary, and Mondulkiri Protected Forest) with one to six species in each PA. *Chromolaena odorata* was the most prevalent IAPS and found in all six PAs, even present in core areas in some cases. The Wildlife Conservation Society (2006) found 50 naturalized species, of which 15 were plants, in the Tonle Sap Biosphere Reserve (TSBR) in Cambodia. The most abundant of these were *M. pigra* and *P. crassipes*. In a study covering 8 Nationals Park's (NPs) in the Java region of Indonesia, Padmanaba et al. (2017) reported 67 IAPS (number in each NP ranging from 8 to 27 species), of which 33 occurred only in one NP and *C. odorata* and *L. camara* in all of them. *Ageratina riparia* and *L. camara* were among the most abundant species. A survey of 15 of Indonesia's NPs revealed that they were invaded by 51 plant species, of which *C. odorata* and *L. camara* were among the most problematic species (Setyawati et al. 2012).

In South Asia, information on plant invasions in PAs is available for Nepal, India, and Sri Lanka. Research on plant invasions in India and other South Asian countries is inadequate, possibly because of the traditional focus on wildlife by PA management agencies (Hiremath and Sundaram 2013). In Nepal, PAs located in the southern lowland (Tarai and Siwalik regions) have high number of IAPS (e.g., 18 and 12 species in Chitwan and Parsa National Park, respectively) compared to the PAs in mountain regions (e.g., 5 and 7 species in Langtang National Park and Manaslu Conservation Area, respectively) (Shrestha 2019). Major IAPS in lowland PAs are *L. camara*, *C. odorata*, and *M. micrantha*, while *Ageratina adenophora* is the major IAPS in mountain regions. Chitwan National Park, a major habitat of the one-horn rhino (*Rhinoceros unicornis*) in Nepal, has been severely invaded by *M. micrantha*, among others (Murphy et al. 2013), while the Bardia National Park, a major habitat of tiger, by *L. camara* (Bhatta et al. 2020). In PAs of India, 19 major IAPS have been reported, including *L. camara*, *Prosopis juliflora*, *C. odorata*, *M. micrantha*, *M. diplotricha*, and *Parthenium hysterophorus* (Hiremath and Sundaram 2013). In Manas National Park, India, *C. odorata* and *M. micrantha* are the most problematic IAPS (Nath et al. 2019). In Sri Lanka, PAs are a good climatic match for a range of species among 14 IAPS studied, including *Panicum maximum*, *L. camara*, *Leucaena leucocephala*, and *Opuntia stricta* (Kariyawasam et al. 2020). In the Himalaya (that includes parts of East, Southeast, and South Asia), 69% (338) of 493 PAs are ecoclimatically a good match for 1 or more of the 5 studied IAPS (*A. adenophora*, *Ageratum conyzoides*, *C. odorata*, *L. camara*, and *P. hysterophorus*) (Lamsal et al. 2018).

5.5.2 Inland Aquatic and Wetland Ecosystems

Inland aquatic and wetland ecosystems have disproportionately high conservation values and provide precious ecosystem services. Biological invasions are considered to be one of the main drivers of ecosystem degradation in these systems (Zedler and Kercher 2004). That said, plant invasions in aquatic and wetland ecosystems of Asia have been poorly studied. In China, Zhan et al. (2017) reported 55 naturalized plant species including algae in these ecosystems, of which 6 are invasive – *Pistia stratiotes*, *Pontederia crassipes*, *Cabomba caroliniana*, *Alternanthera philoxeroides*, *Spartina alterniflora*, and *S. anglica*. Another study, which appears to be more comprehensive, reported 152 aquatic naturalized plant species in China (Wang et al. 2016). In Japan, aquatic and wetland ecosystems are colonized by more than 40 naturalized species; many of them are highly invasive and include *P. crassipes*, *Elodea nuttallii*, *Egeria densa*, *P. stratiotes*, *Myriophyllum aquaticum*, *Gymnocoronis spilanthoides*, *A. philoxeroides*, and *Hydrocotyle ranunculoides* (Kadono 2004). In the Lower Mekong Basin (Cambodia, Lao PDR, Thailand, and Vietnam) of Southeast Asia, important wetland IAPS are *Brachiaria mutica*, *P. crassipes*, P. *stratiotes*, and *M. pigra* (Miththapala 2007).

Pontederia crassipes, *Salvinia molesta*, and *Mimosa pigra* are invasive in almost every country in Southeast Asia (Witt 2017).

In South Asia, freshwater aquatic and wetland ecosystems of India are invaded by several IAPS, of which highly invasive ones are *P. crassipes*, *S. molesta*, and *P. stratiotes* (Shah and Reshi 2012). In Kashmir Himalaya (India) alone, Shah and Reshi (2014) reported 28 species as invasive in wetlands. Wular Lake, the biggest lake in Kashmir Himalaya, is invaded by *Azolla filiculoides* and *Alternanthera philoxeroides* (Keller et al. 2018). Similarly, six IAPS in Nepal are exclusively found in wetlands including some Ramsar sites and include *P. crassipes*, *A. philoxeroides*, *P. stratiotes*, and the semi-aquatic *Ipomoea carnea*, all of which are highly problematic, while *Leersia hexandra* and *Myriophyllum aquaticum* have localized distributions (Shrestha 2019). Species such as *A. philoxeroides*, *P. crassipes*, and *P. stratiotes* are present in wetlands in the Maldives (Sujanapal and Sankaran 2016). In Israel, freshwater wetlands are heavily invaded by *P. stratiotes*, *P. crassipes*, *M. aquaticum*, *A. filiculoides*, and *S. molesta* (Dofour-Dror 2012). *Azolla filiculoides* has also invaded Ramsar sites such as Anzali wetland in northern Iran (Hashemloian and Azimi 2009). Invasions of *P. crassipes* and *P. stratiotes* have also been recently reported from Iranian wetlands (Mozaffarian and Yaghoubi 2015; Bidarlord et al. 2019).

5.6 Native Range and Introduction Pathways

Knowledge of the biogeographic origin of alien species and their introduction pathways are essential for risk assessments, screening at international ports, and Early Detection and Rapid Response (EDRR) against potential invasive species. However, these issues have been poorly studied in Asia.

5.6.1 Native Range

Biogeographic origin of species largely determines invasiveness and spatial extent of distribution in the introduced range. For instance, a species native to the tropics of South America is more likely to be invasive and widespread in India than species native to more temperate Europe (Khuroo et al. 2012a). Biogeographic origin of alien species has been a subject of analysis only in a few countries as summarized in Table 5.3. Most of the IAPS in Asia originate from tropical America, followed by Africa, Europe, and Oceania. There are obvious gaps in the data presented in Table 5.3 due to lack of adequate information from Central and North Asia. Inclusion of Russia and Central Asian countries may change the scenario. As expected, 31% of 328 alien plants found in the Middle Urals of Russia are native of Asia (outside Russia), followed by species from the Mediterranean region (22%), Europe (outside Russia, 19%), America (mostly temperate North America, 17%), Siberian region (7%), and other regions including Africa (4%) (Tretyakova 2011). Similarly, in North Korea, most (61%) of the alien species are native to other regions in Asia, followed by Europe (37%) and North America (2%) (Son et al. 2009).

Current patterns of geographic origin of alien species in Asia are most likely a result of climatic similarities and propagule pressure due to trade relations. For example, Jiang et al. (2011) attributed the highest contribution of American native plant species to alien flora of China to broad climatic similarity and high volumes of trade between China and North America. Similarly, high contributions of species from China to the alien flora of North Korea can also be attributed to high dependency of North Korea on China for the supply of essential goods (Son et al. 2009). In addition, a few species that are native of tropical America were introduced first to Europe as ornamental plants and subsequently to Asia during European colonization as exemplified by the introduction of *Lantana camara* to India (Kannan et al. 2013).

5.6.2 Introduction Pathways

Managing pathways is one of the major goals of Aichi Target 9 of the Convention on Biological Diversity aimed to combat biological invasions (IUCN-ISSG 2016). Alien species may be intro-

Table 5.3 Geographic contribution to the alien flora of selected countries in Asia

Country/region	#Species included	Native range (% of total)						References
		Americas	Asia	Africa	Europe	Oceania	Unknown/others	
China	861	52	12.7	8.6	13.9	1.6	11.2	Jiang et al. (2011)
Hong Kong (China)	144	54.2	10.4[a]	15.3	18.0[b]	–	2.1	Corlett (1992)
Indonesia	1936[d]	40	26	13	9		12	Tjitrosoedirdjo (2005)
Singapore	136	63.2	20.6	9.5	0.7	0.7	5.1	Corlett (1988)
India	1599[d]	39	21	20	11	8	1	Khuroo et al. (2012a)
Nepal	166	74.1	4.2	4.8	8.4		8.4	Bhattarai et al. (2014)
Turkey	340[d]	44.7	27.6	9.1	7.9[c]	3.8	6.9	Uludag et al. (2017)
Mean (± SD)		52.5 ± 12.8	17.5 ± 8.6	11.5 ± 5.0	9.8 ± 5.4	3.5 ± 3.3	6.7 ± 4.2	

[a]Tropical and subtropical Asia
[b]Also includes northern Asia
[c]Also include Mediterranean region
[d]Includes naturalized and casual/cultivated species

duced by one or more of the following pathways: release, escape, contaminant, stowaway, corridor, and unaided (Hulme et al. 2008). For plants, the most common dispersal pathways worldwide are "escape" (initial intentional introduction but subsequent unintentional escape) and "release" (intentional introduction for release) (Saul et al. 2017). In Asia, most of the species, for which information is available, were introduced through "escape," "release," or "contaminant." Shipping, aquaculture, and aquarium, water gardening, and ornamental trades are the major pathways of the introduction of alien species to aquatic and wetland ecosystems in China and Japan (Kadono 2004; Wang et al. 2016; Zhan et al. 2017). In Turkey, 72% of the alien flora were introduced intentionally (Uludag et al. 2017).

A large number of alien plant species introduced for ornamental purposes have escaped and naturalized in the wild, with several of them becoming serious invasive species. *Lantana camara* is probably the best and most documented example of a garden escape that has devastating environmental and socioeconomic impacts, particularly in South and Southeast Asia. The species was introduced to at least six locations in British cantonments and botanical gardens of British India, of which the first introduction occurred during the 1800s (Kannan et al. 2013). By 1874, it was reported as spreading into the wild (Kannan et al. 2013). There are several other examples of garden escapes. *Leucanthemum vulgare* was introduced as an ornamental to India during the British era and is now invasive in Kashmir and Himachal Pradesh (Khuroo et al. 2010). Mehraj et al. (2018) reported 110 cultivation escapes and 58 accidentally introduced alien plant species in Srinagar city, Kashmir (India). At least 14 IAPS, including *L. camara*, *Pontederia crassipes*, *Prosopis juliflora*, and *Clidemia hirta*, escaped from botanical gardens in Sri Lanka where they were first introduced for ornamental and educational purposes (Wijesundara 2010). Some of the species that escaped from gardens in Southeast Asia are *Caesalpinia pulcherrima*, *Thunbergia grandiflora*, *Ipomoea carnea*, *I. cairica*, *Bougainvillea spectabilis*, and *Coccinia indica* (MacKinnon 2002). Slightly more than one-third (671 species) of the total alien plant species (1936 species) present in Indonesia are ornamentals and were intentionally introduced (Tjitrosoedirdjo 2005). In Singapore, 32 naturalized plant species were initially introduced as ornamental plants and another 19 as crop species originally cultivated for food, medicine, raw materials, forage, or cover (Corlett 1988). *Salvinia molesta*, one of the worst aquatic weeds globally, was introduced to Sri Lanka for research purposes by the Department of Botany, University of Colombo (Bandara 2010). In Upper Volga region of Russia, some of the invasive woody species such as *Acer negundo*, *Fraxinus pennsylvanica*, and *Populus deltoides* were introduced as landscaping plants from 1950 to 1980 (Borisova 2016).

Several of the species introduced for habitat restoration and livestock fodder have also escaped from cultivated areas and become invasive. Several alien tree species including *Taxodium distichum*, *Cryptomeria japonica*, and *Eucalyptus camaldulensis* were introduced to West and Central Asia for the rehabilitation of degraded forests (Mozaffarian 2005; Lee and Kleine 2009); some of them are well known as invasives. *Leucaena leucocephala* was introduced as a fodder species and for nitrogen fixation to all regions except North and Central Asia (Table 5.2) where the species has invaded natural habitats in many countries (Sankaran and Suresh 2013). *Prosopis juliflora* was introduced to Western Asia for agroforestry purposes but is now invading natural habitats (Hegazy and Lovett-Doust 2016). It was also introduced to South and Southeast Asia for fuelwood where it poses a serious threat to natural ecosystems (Sankaran and Suresh 2013). *Azolla filiculoides* was introduced in ca. 1990 to Egypt (Hegazy and Lovett-Doust 2016) and almost at the same time in Iran as a green manure and fodder for livestock, but the plant soon escaped to irrigation canals and wetlands nearby (Hashemloian and Azimi 2009).

A number of species were accidentally introduced as contaminants of crop imports, especially grains and seeds for planting. For example, it is believed that *Parthenium hysterophorus* was accidentally introduced to India during the 1950s as a contaminant when wheat (*Triticum aestivum*) was imported from Mexico to Pune, Maharashtra, India (Ahmad et al. 2019a). From Maharashtra, *P. hysterophorus* has spread to all Indian states and most other countries in South Asia including Nepal as a contaminant of agricultural produce or in or on transport vehicles (Shrestha et al. 2019a, b). Similarly, *Ambrosia* spp. might have arrived in Israel through grain shipments (Yair et al. 2019). Likewise, *Ludwigia epilobioides*, *Ambrosia psilostachya*, and *Persicaria lapathifolia* are believed to have been introduced to Iranian rice fields as contaminants of rice seeds (A. Naqinezhad, pers. obs.).

5.7 Environmental and Socioeconomic Impacts

As mentioned in the previous sections, hundreds of naturalized plants have invaded a wide range of regions and ecosystems including agroecosystems and PAs. Based on studies done elsewhere, it is highly likely that the impacts of plant invasions on the environment and socio-economy of this region are significant. However, studies investigating and quantifying the impacts of biological invasions are still scarce in Asia compared to other regions (Hulme et al. 2013). This makes it difficult to assess the magnitude of the problem and hence hinders the possibility of anticipated management interventions and proactive policy responses. In this section, we have highlighted major environmental and socioeconomic impacts of plant invasions with representative examples.

5.7.1 Environmental Impacts

Biodiversity and Ecosystems

Plant invasions have caused serious negative impacts on native ecosystems, including biodiversity and ecosystem services, alteration of biogeochemical cycles, and threats to environmental safety in Asia. Change in species composition and subsequent reduction in species richness and diversity after invasion have been reported for *Ageratina adenophora*, *Carpobrotus edulis*, *Centaurea iberica*, *Chromolaena odorata*, *Pontederia crassipes*, *Lantana camara*, *Leucanthemum vulgare*, *Mesosphaerum suaveolens*, *Parthenium hysterophorus*, *Solidago canadensis*, *Spartina alterniflora*, and *Xanthium strumarium*. Fu et al. (2018) reported that *A. adenophora* reduced species richness of understory vegetation by 68% in *Pinus yunnanensis* forest in Yunnan, China, and displaced many native species, particularly those species having low leaf nitrogen content. Similarly, *C. edulis* in coastal habitats of Israel is displacing the coastal iris, *Iris atropurpurea*, a rare species endemic to Israel (Dufour-Dror 2012). In the mountain grasslands of Kashmir Himalaya, *C. iberica* has altered species assemblages, reduced the number and abundance of palatable native species, and reduced species diversity (Reshi et al. 2008). Native plant species richness was 1.25 times higher in non-invaded plots (1 m^2) than in plots invaded by *C. odorata* in Nepal (Thapa et al. 2016). Vigorous growth of *P. crassipes* outcompeted native hydrophytes, reducing species richness from 16 to 3 in parts of Dianchi Lake of Yunnan Province in China (Wu 1993). In West Asia, *P. crassipes* has replaced many native aquatic plants in wetlands and aquatic channels (Mozaffarian and Yaghoubi 2015; Hegazy and Lovett-Doust 2016).

Invasion by *L. camara* reduced species richness and diversity by 41% and 16%, respectively, in Siwalik Hills of Himachal Pradesh, India (Singh et al. 2014). In Nepal's Bardia National Park, *L. camara* reduced native plant species richness by more than 50% (Bhatta et al. 2020). Plots invaded by *L. vulgare* had, on average, 4.3–6.7 fewer species than non-invaded plots in Kashmir, India (Ahmad et al. 2019b). Species diversity of non-invaded plots was 3.4 times higher compared to plots invaded by *L. vulgare* (Khuroo et al. 2010). The number of species declined by 46–52% in areas heavily invaded by *M. suaveolens* in Chandigarh, India (Sharma et al. 2017). Locally useful species such as *Justicia adhatoda*, *Dioscorea deltoidea*, and *Murraya koenigii* were completed displaced by *M. suaveolens*. There was a 60–70% reduction in abundance and 35–60% reduction in the number of native species due to invasion by *P. hysteroph-*

orus in Chandigarh, India (Kaur et al. 2019). *Solidago canadensis* has partially displaced more than 30 native species which accounted for 10% of total local native species in Shanghai alone (Lei et al. 2010). Non-invaded plots had 1.3 and 1.7 times higher species richness and diversity, respectively, than plots invaded by *X. strumarium* in the Pothwar region of Pakistan (Qureshi et al. 2019).

In forests, plant invasions inhibit tree regeneration. For example, *Leucaena leucocephala* had detrimental impacts on seed germination and seedling establishment of native tree species on the subtropical oceanic island of Chichijima, Japan (Hata et al. 2007). Similarly, seedling density of *Shorea robusta*, the most important timber species in Nepal, was 2.6 times higher in non-invaded plots than in plots invaded by *C. odorata* (Thapa et al. 2016).

In addition to changes in species composition and diversity, plant invasions also have impacts on a range of other ecological processes. For example, *S. alterniflora* has converted mudflats to meadows and degraded native wetland ecosystems in the Yangtze River estuary (Li et al. 2009; Liu et al. 2012). Though there is no empirical evidence, it is believed that invasive species like *L. camara* alter fire regimes, particularly in regions with a dry climate, contributing to the loss of forests (Hiremath and Sundaram 2005).

Impacts on Animals

Only a few studies have examined the impacts of plant invasion on animals in Asia. *Spartina alterniflora* has resulted in loss of shorebirds' foraging habitats and change in community structure and diets of native arthropods in the Yangtze River estuary, China (Li et al. 2009; Liu et al. 2012). During extensive field studies, one of the authors (A.B.R. Witt) observed some impacts of IAPS on flagship wildlife species in Southeast Asia: the Sumatran rhino, Sumatran elephant, and Sumatran tiger in the Bukit Barisan Selatan National Park (and other protected areas in Sumatra island) that are greatly affected by the dense smothering habit of *Merremia peltata* and the near extinction of the rare banteng (*Bos javanicus*) in Baluran National Park due to over 70% loss of its primary habitat of grass savanna by *Acacia nilotica* (ABR Witt, pers. obs.). Invasion by *Prosopis juliflora* in Vettangudi Bird Sanctuary of south India has degraded nesting habitat of breeding birds due to the high probability of eggs and chicks falling to the ground from the nests in this plant (Chandrasekaran et al. 2014). In Nepal's Chitwan National Park, a World Natural Heritage Site, *Mikania micrantha* has invaded 44% of the habitat of endangered one-horn rhino with potential negative impacts on forage supply due to smothering of many native species by the weed (Murphy et al. 2013).

Impacts on Soil

Changes in soil chemistry, nutrient content, and availability have been reported due to invasions by *Ageratina adenophora*, *Chromolaena odorata*, *Parthenium hysterophorus*, *Mikania micrantha*, *Mesosphaerum suaveolens*, *Leucanthemum vulgare*, and *Spartina alterniflora*. Soil in *A. adenophora*-invaded sites of southwestern Yunnan Province, China, had 4.32 mg/kg more nitrogen than non-invaded soil (Zhao et al. 2019). The invaded soil also had higher rates of microbial-mediated functional processes such as nitrogen fixation, nitrification, and ammonification than in the non-invaded soil. Invasion by *C. odorata* also significantly increases labile and total carbon and nitrogen fractions in tropical savanna soils (Wei et al. 2017). Organic carbon, nitrogen, phosphorus, and potassium were higher in *P. hysterophorus*-invaded grassland soils than in non-invaded ones in Nepal (Timsina et al. 2011). However, in Chandigarh, India, the concentrations of organic matter, nitrogen, phosphorus, and potassium were lower in the *P. hysterophorus*-invaded soil than in non-invaded sites (Kaur et al. 2019). Invasion by *M. micrantha* increases soil enzyme activities and abundance of aerobic bacteria but reduces the abundance of anaerobic bacteria in comparison to non-invaded sites (Li et al. 2006). *Mikania micrantha* also enhances nutrient cycling during early stages of secondary succession following slash-and-burn agriculture (Swamy and Ramakrishnan 1987). *Mesosphaerum suaveolens* invasions increase soil organic matter, organic carbon, and electrical conductivity (Sharma et al. 2017). Ahmad et al. (2019c) reported that invasion by *L. vulgare* in Kashmir Himalaya, India, had a significant impact on key soil properties with soil pH, water content, organic carbon, and total nitrogen significantly higher in the invaded plots as compared with the uninvaded plots. In contrast, the electrical conductivity, phosphorous, and micronutrients, viz., iron, copper, manganese, and zinc, were significantly lower in the invaded plots as compared with the uninvaded plots. The results indicated that *L. vulgare*, by altering key properties of the soil system, influences nutrient cycling processes and facilitates positive feedback for itself. In wetland ecosystems of Yangtze River estuary, China, *S. alterniflora* has enhanced storage of carbon dioxide and increased the inorganic nitrogen pool (Li et al. 2009).

5.7.2 Socioeconomic Impacts

Agriculture and Aquaculture

IAPS are reported to have negative impacts on agricultural production. In terms of the threats of biological invasions to the agricultural sector, four of the five countries most threatened by IAPS are located in Asia; they are Mongolia, Nepal, Bangladesh, and Cambodia (Paini et al. 2016). In Nepal, reduced agriculture production, forage supply, and livestock poisoning are the major impacts of IAPS among farming communities (Shrestha et al. 2019b). Local communities ranked *Ageratum houstonianum* as the most problematic weed in their agriculture production system, mainly due to its toxicity to livestock and high labor cost of weeding. *Mikania micrantha* invasion reduces fodder supply and subsequently increases time to collect fodder by local communities from forests in Nepal (Rai and Scarborough 2015). *Pontederia crassipes* blocks waterways, affects water transport for agriculture and tourism, covers lakes and rivers, causes algal blooms, and reduces aquatic production in China (Ding et al. 2001). Invasions by *Azolla filiculoides* and *Alternanthera philoxeroides* in Wular Lake, Kashmir (India), impact negatively on fishing and the availability of wild edible plants (Keller et al. 2018). In Turkey, 40 of 51 alien plant species have socioeconomic impacts, mainly on agricultural production and human health (Yazlik et al. 2018a). The highest ranking species in terms of socioeconomic impacts are *P. crassipes* and *Lantana camara*. Similarly, *Ipomoea triloba* has substantially increased weeding cost in cotton farms of Turkey (Yazlik et al. 2018b).

Human and Animal Health

Invasive alien plant species also threaten public health and social well-being. In China, *Pontederia crassipes* is reported to provide habitats for mosquitoes and flies, thereby affecting public health (Ding et al. 2001). *Ambrosia artemisiifolia* and *A. trifida* produce copious amount of pollen, compounding health problems like rhinitis, oculorhinitis, asthma, and skin irritations (Li et al. 2015). *Ageratina adenophora* pollen contains aromatic and pungent chemicals causing allergenic reactions in people (Zhu et al. 2007). In Japan, the recurrent bouts of sneezing, nasal congestion, and tearing and itching of the eyes are caused by seasonal allergies to the pollen of certain plants including alien *Ambrosia* species and alien meadow grasses such as *Lolium multiflorum*, *L. perenne*, *L. x hybridum*, and *Dactylis glomerata* (Saito and Ide 1994). In Israel, the allergenic effect of *Ambrosia confertiflora* pollens to humans has been reported (Yair et al. 2019). Respiratory allergy and dermatitis caused by *Parthenium hysterophorus* are the most common type of plant dermatitis in India, which may be life threatening to sensitive individuals (Sharma and Verma 2012). It mainly affects exposed body parts such as the face, neck, hands, and legs. Similar negative health impacts of *P. hysterophorus* to human have been also reported in Nepal (Shrestha et al. 2015).

There are few studies reporting impacts of IAPS to livestock health in Asia. For example, consumption of *A. adenophora* has been reported to cause acute asthma, diarrhea, depilation, and even death of livestock in China (Zhu et al. 2007). *Ageratum houstonianum* is reported to have poisoning effects on livestock in Nepal (Shrestha et al. 2019b). Several cases of livestock death due to consumption of *Mimosa diplotricha* have been also observed in southeastern districts of Nepal (BB Shrestha, pers. obs.). Impact of IAPS to wildlife health has not been reported yet.

Economic Costs

Few studies have evaluated the economic costs of invasive alien species in Asian countries. In China, economic losses due to invasive alien species (plants and other organisms) were estimated to be 14.45 billion USD per year in 2000 (which was 1.36% of GDP) (Xu et al. 2006). Of the total losses, the direct losses associated with damage and control costs in agriculture, forestry, aquaculture, transportation, and health accounted for 16.59% and the indirect losses associated with loss of ecosystem services 83.41%. Nghiem et al. (2013) estimated that the total annual cost of all invasive alien species associated with agriculture, human health, and environment in Southeast Asia amounted to 33.2 billion USD but clearly stated that this was likely to be a conservative estimate. Most of these impacts (90%) were associated with the agricultural sector (29.3 billion USD) where information is more readily available. Economic losses in India due to IAPS on crop production and pasture were estimated to be 38.7 USD billion per year (Pimentel et al. 2001).

A few studies have estimated economic cost of individual species. In China, the annual losses in livestock production due to the effect of *Ageratina adenophora* were estimated to be 162 million USD, and the losses in services of grassland ecosystems were 0.4 billion USD (Xu et al. 2006; Ding et al. 2007). On Nei Lingding Island (Guangdong Province, China), the economic loss caused by *Mikania micrantha* was reported to range from 0.56 to 1.6 million USD per year (Zhong et al. 2004). Over 12 million USD per year was spent in China on the manual removal of *P. crassipes* between 1991 and 2001, and 128 million USD was spent in 1996 for manual removal of several weeds in Wenzhou City of China's Zhejiang Province (Ding and Xie 1996; Ding et al. 2001). In India, total cost associated with damage and control of *Parthenium hysterophorus* in agroecosystems between 1955 and 2009 was estimated to be 2.067 trillion INR (equivalent to 26.8 USD billion as per the exchange rate of 15 April 2020) (Sushilkumar and Varshney 2010). Reduced profitability of teak (*Tectona grandis*) plantations due to invasion by *M. micrantha* has been also reported from Kerala, India (Muraleedharan and Anitha 2000). In Punjab Province of Pakistan, the annual cost of *P. hysterophorus* invasion associated with crop and livestock production, health, and social well-being was estimated to be 913 USD per household (Bajwa et al. 2019).

5.8 Management

A variety of management interventions have been developed and implemented in Asia. The management options for IAPS may vary according to the species in question, stage of invasions, the habitat invaded, land use, farming system, size of invasion, time, socioeconomic condition, and available resources. According to Padmanaba et al. (2017), current management efforts are reactive, localized, and intermittent, with currently available resources being insufficient for early detection and prompt responses in PAs in Java, Indonesia. Unfortunately, a similar scenario is prevalent in most parts of Asia. To improve control measures against IAPS, many comprehensive management approaches are widely adopted and used in China and some other Asian countries, combining different physical, chemical, ecological, and biological control methods (Yang et al. 2017; Clements et al. 2019). Generally, these integrated control methods for IAPS are usually designed to make up for shortcomings of individual control applications and can achieve better environmental protection, economic returns, and control (Clements et al. 2019). Integrated pest management interventions that incorporate ecosystem-based and environment-friendly approaches have been initiated in Central Asia (Maredia and Baributsa 2007). In the following sections, we discuss various management approaches being developed and implemented in Asia, including community participation and policy responses.

5.8.1 Physical Methods

Physical control techniques for IAPS include hand pulling or uprooting, slashing, ringbarking, ploughing, and similar interventions, most of which are widely practiced in Asia by farmers and local communities. However, these approaches are seldom documented in the scientific literature, and their effectiveness has been rarely investigated. Physical control techniques could be effective for small, localized invasions but are largely ineffective for widespread and abundant invasions across the landscape. For example, cutting *Lantana camara* during the wet season for biomass by local communities can significantly reduce its abundance at local level, allowing recolonization by native species (Kannan et al. 2016). Similarly, frequent manual removal of *Mikania micrantha* biomass at a local scale while retaining native vegetation may reduce its competitiveness (Rai et al. 2012). Extra precautions are needed to prevent regeneration and dispersal from plant parts which can easily regenerate from stem fragments (Huang et al. 2015). Physical control is also labor intensive and difficult when the IAPS is thorny (e.g., *Mimosa diplotricha*). Despite some limitations, physical methods can be important components of an integrated management strategy.

In Nepal, local communities remove *Chromolaena odorata*, *Ageratina adenophora*, and *Lantana camara* from forests and use their biomass to produce compost and bio-briquettes (Shrestha 2019; Shrestha et al. 2019a, b). Wetland IAPS such as *Pontederia crassipes*, *Pistia stratiotes*, and *Alternanthera philoxeroides* are being removed manually or by using weed harvesters (Shrestha 2019). Site restoration and follow-up control activities after physical removal of IAPS are essential to sustain efficacy. For example, in Kashmir Himalaya, India, control of aquatic IAPS (e.g., *Azolla filiculoides*, *Nymphaea mexicana*) through manual and mechanical measures in Dal Lake has failed due to lack of follow-up action. However, these programs have benefits beyond biodiversity because of local community support. The mechanical removal of aquatic invasive plants provides livestock fodder to local population, and the manual control programs provide daily wage-based employment opportunities (Khuroo et al. 2009; McDougall et al. 2011).

In Israel, physical control is being practiced in a few nature reserves under the supervision of the Israel Nature and Parks Authority (INPA) (Dufour-Dror 2012). In most cases, it is carried out either by uprooting individual plants, by cutting them down, or, in the case of wetland species, by simply collecting the plants from the water bodies and disposing them. Another method attempted to control *Acacia saligna* in Israel is solarization, which uses transparent plastic sheets to cover the soil surface in order to induce seed germination (Cohen et al. 2008). High temperature maintained beneath the plastic eventually kills the seedlings and reduces the persistent soil seed bank.

5.8.2 Chemical Methods

Herbicides are generally an effective control method for IAPS, especially in regions where herbicides are affordable, due to their relatively high efficacy and better returns on application costs (Clements et al. 2019). A broad selection of herbicides has been evaluated for use on IAPS. These herbicides containing the active ingredients 2,4-dichlorophenoxyacetic acid, glyphosate, sulfometuron methyl, paraquat, glufosinate, and picloram are mostly used for control of IAPS, especially *Ageratina adenophora* and *Mikania micrantha* (Yang et al. 2017; Clements et al. 2019). Various combinations of triclopyr, picloram, glyphosate, and diuron have been found effective in controlling *M. micrantha* in teak plantations in Kerala, India (Sankaran et al. 2017).

In Israel and Cyprus, improved methods of chemical applications such as drill-fill (drilling holes on the lower part of trunks and injecting herbicides), cut-stump (felling trees by chainsaw and application of herbicides on outer rim of the stump), and frilling (removal of bark by knife and application of herbicides) techniques have been successfully used for control of invasive tree species such as *Acacia saligna*, *Ailanthus altissima*, *Robinia pseudoacacia*, and *Dodonaea viscosa* (Dufour-Dror 2013). While these methods have minimum undesirable chemical impacts to the environment, they are labor intensive and require access to every individual tree to be treated.

5.8.3 Biological Control

Biological control of IAPS is environmentally friendly and sustainable (Seastedt 2015). Despite a large number of IAPS, biological control has only been practiced in a few countries in Asia. The first biological control agent, *Dactylopius ceylonicus* (Hemiptera: Dactylopidae), was accidentally introduced from Brazil to India in 1795 where it successfully controlled *Opuntia monacantha* (Cactaceae) in 5–6 years (Rabindra and Bhumannavar 2009). The intention was to introduce *D. coccus* for dye production, but the wrong cochineal was inadvertently introduced. *Dactylopius ceylonicus* was then introduced to Sri Lanka in 1865 to control *O. monacantha*, the first deliberate international transfer of a biological control agent (Rabindra and Bhumannavar 2009). In 1933, China initiated a biological control program with the introduction of two agents, *Ophiomyia lantanae* and *Lantanophaga pusillidactyla*, into Hong Kong for the control of *Lantana camara* (Shen et al. 2018). Over the period of more than 100 years, several biological control agents have been released in Asia with variable success. A literature review revealed that 36 biological control agents (31 arthropods and 5 fungi) targeted for 17 species (1 pteridophyte and 17 angiosperms) are established in different Asian countries (Table 5.4). According to Day and Witt (2019), 15 countries in Asia have intentionally released 42 biological control agents against 19 weed species. The highest number of biological control agents are present in China (18 species) followed by India (16), Thailand (11), Vietnam (6), Timor-Leste (4), Sri Lanka (4), Myanmar (3), Nepal (3), Indonesia (2), Malaysia (2), the Philippines (2), Pakistan (1), Laos (1), and Israel (1) (Table 5.4).

The largest number of biological control agents (10 species) targeted *Lantana camara;* 3 for each of *Chromolaena odorata*, *Pontederia crassipes*, and *Mimosa pigra*; 2 for each of *Ageratina adenophora*, *Ambrosia artemisiifolia*, and *Parthenium hysterophorus*; and 1 agent each for the remaining 11 species (Table 5.4). Only a subset of these species was deliberately introduced while others have spread from neighboring countries. For example, ten biological control agents targeting seven IAPS from neighboring countries have spread naturally and established in China (Shen et al. 2018). Similarly, three agents have established in Nepal after spreading from other Asian countries (Shrestha 2019). Relatively high damage has been observed on *Salvinia molesta* (in India), *A. artemisiifolia* (China), *Opuntia* spp. (India, Sri Lanka, and Israel), and *Mimosa diplotricha* (Timor-Leste) by their respective biological control agents. Impact of many other agents on the target species is either low or moderate in Asia. While it is essential to understand the factors that determine the effectiveness of established biological control agents through regular monitoring, search for new and effective biological control agents targeting highly problematic IAPS should also be continued. Further promotion of biological control programs by countries in Asia as a major component of the integrated management is imperative for long-term and sustainable control of IAPS.

5.8.4 Ecosystem-Based Approaches

Invasibility of any ecosystem depends on its attributes such as successional stage, disturbance regime, and species composition, among others. Minimization of disturbance and manipulation of species composition in semi-natural ecosystems (e.g. agroforestry system, managed pasture) can improve performance of native communities and competitively suppress IAPS, thereby complementing the traditional methods of physical, chemical, and biological controls. One emerging field of research on ecosystem-based management is the use of native or useful and noninvasive alien species to suppress growth and reproduction of IAPS in managed forests, graz-

Table 5.4 Established biological control agents with their targeted invasive alien plant species in different Asian countries

Targeted species [family]	Biocontrol agents [family]	Countries with established population	General impacts	References
Salvinia molesta [Salviniaceae]	*Cyrtobagous salviniae* [Curculionidae]	India	High	Rabindra and Bhumannavar (2009)
Alternanthera philoxeroides [Amaranthaceae]	*Agasicles hygrophila* [Chrysomelidae]	China, Thailand	Moderate	Shen et al. (2018), Day et al. (2018)
Ageratina adenophora [Asteraceae]	*Passalora ageratinae* [Mycosphaerellaceae]	China	Low	Shen et al. (2018)
	Procecidochares utilis [Tephritidae]	China, Nepal, India, Thailand	Low	Day et al. (2018), Shen et al. (2018), Shrestha (2019)
Ambrosia artemisiifolia [Asteraceae]	*Ophraella communa* [Chrysomelidae]	China	High	Shen et al. (2018)
	Epiblema strenuana [Tortricidae]	China	Moderate	Shen et al. (2018)
Ambrosia trifida [Asteraceae]	*Puccinia xanthii* ssp. *ambrosiae-trifidae* [Pucciniaceae]	China	Variable	Shen et al. (2018)
Chromolaena odorata [Asteraceae]	*Acalitus adoratus* [Eriophyidae]	China, Lao PDR, Myanmar, Thailand, Vietnam	Slight	Day et al. (2018), Shen et al. (2018)
	Cecidochares connexa [Tephritidae]	India, Indonesia, Philippines, Timor-Leste	Moderate	Rabindra and Bhumannavar (2009), Shen et al. (2018)
	Pareuchaetes pseudoinsulata [Arctiidae]	India	??	Rabindra and Bhumannavar (2009)
Mikania micrantha [Asteraceae]	*Puccinia spegazzinii* [Pucciniaceae]	Taiwan (China), India	Moderate	Rabindra and Bhumannavar (2009), Shen et al. (2018)
Parthenium hysterophorus [Asteraceae]	*Puccinia abrupta* var. *partheniicola* [Pucciniaceae]	China, Nepal	Low	Shen et al. (2018), Shrestha (2019)
	Zygogramma bicolorata [Chrysomelidae]	Nepal, India, Pakistan	Moderate	Shen et al. (2018), Shrestha et al. (2019a)
Xanthium strumarium [Asteraceae]	*Puccinia xanthii* [Pucciniaceae]	Sri Lanka, Timor-Leste	Moderate	Shen et al. (2018)
Opuntia ficus-indica [Cactaceae]	*Dactylopius opuntiae* [Dactylopiidae]	Israel	High	Shen et al. (2018)
Opuntia stricta [Cactaceae]	*Dactylopius opuntiae* [Dactylopiidae]	India, Sri Lanka	High	Shen et al. (2018)
Opuntia elatior [Cactaceae]	*Dactylopius opuntiae* [Dactylopiidae]	India	High	Rabindra and Bhumannavar (2009)
Opuntia monacantha [Cactaceae]	*Dactylopius ceylonicus* [Dactylopiidae]	India, Sri Lanka	High	Rabindra and Bhumannavar (2009)
Leucaena leucocephala [Fabaceae]	*Acanthoscelides macrophthalmus* [Chrysomelidae]	China	Low	Shen et al. (2018)
Mimosa diplotricha [Fabaceae]	*Heteropsylla spinulosa* [Psyllidae]	Timor-Leste	High	Shen et al. (2018)

(continued)

Table 5.4 (continued)

Targeted species [family]	Biocontrol agents [family]	Countries with established population	General impacts	References
Mimosa pigra [Fabaceae]	*Carmenta mimosa* [Sesiidae]	Malaysia, Vietnam	Moderate	Shen et al. (2018)
	Acanthoscelides puniceus [Chrysomelidae]	Thailand, Vietnam	Low	Day et al. (2018)
	Acanthoscelides quadridentatus [Chrysomelidae]	Thailand, Vietnam	Low	Day et al. (2018)
Pontederia crassipes [Pontederiaceae]	*Neochetina bruchi* [Erirhinidae]	China, Thailand, India	Moderate	Shen et al. (2018), Day et al. (2018);
	Neochetina eichhorniae [Erirhinidae]	China, Thailand, India	Moderate	Shen et al. (2018), Day et al. (2018)
	Orthogalumna terebrantis [Galumnidae]	India	Low	Rabindra and Bhumannavar (2009)
Lantana camara [Verbenaceae]	*Calycomyza lantanae* [Agromyzidae]	China, Thailand, Vietnam	Low	Day et al. (2018), Shen et al. (2018)
	Hypenalaceratalis [Erebidae]	China	Low	Shen et al. (2018)
	Lantanophaga pusillidactyla [Pterophoridae]	China	Low	Shen et al. (2018)
	Crocidosema lantana [Tortricidae]	China	Low	Shen et al. (2018)
	Ophiomyia lantanae [Agromyzidae]	China, India, Myanmar, Thailand, Vietnam	Low	Rabindra and Bhumannavar (2009), Day et al. (2018), Shen et al. (2018)
	Lantanophaga pusillidactyla [Pterophoridae]	China, Myanmar, Thailand	Low	Shen et al. (2018)
	Octotoma scabripennis [Chrysomelidae]	India	Moderate	Rabindra and Bhumannavar (2009), Shen et al. (2018)
	Teleonemia scrupulosa [Tingidae]	India, Indonesia, Malaysia, Philippines, Sri Lanka, Thailand, Timor-Leste	Moderate	Shen et al. (2018)
	Uroplata girardi [Chrysomelidae]	India, Philippines	Moderate	Rabindra and Bhumannavar (2009), Shen et al. (2018)
	Epinotia lantana [Tortricidae]	India	Low	Rabindra and Bhumannavar (2009)

ing grasslands, and agroecosystems. These ecological methods are widely used through plant-plant competition, utilizing parasitic plants, soil fungi competition, and allelopathy (Clements et al. 2019). A number of greenhouse or field experiments were conducted in China to evaluate the competitive capacity of replacement plants against the invasive *Ageratina adenophora*. Many local plants have been demonstrated as ideal candidates of replacement plants, such as *Trifolium repens*, *T. pratense*, *Pennisetum hydridum*, *Setaria yunnanensis*, *Eupatorium fortunei*, *Chenopodium serotinum*, *Setaria sphacelata*, and *Pennisetum clandestinum* (Yang et al. 2017). Similarly, some plant species such as *Cuscuta campestris*, *Macaranga tanarius*, and *Heteropanax fragrans* can suppress *Mikania micrantha* in China (Clements et al. 2019). Sweet potato (*Ipomoea batatas*), an important cash and food crop widely grown in the world, is reported to suppress four IAPS, *M. micrantha*, *Ageratum conyzoides*, *Bidens pilosa*, and *Galinsoga parviflora* (Shen et al. 2015, 2019). In a field experiment conducted in northern Pakistan, growth of *Parthenium hysterophorus* was suppressed by >70% when grown together with fodder species such as *Sorghum almum*, *Cenchrus ciliaris*, and *Chloris gayana* (Khan et al. 2014).

Habitat restoration by introducing native species has also been suggested for the control of *Acacia saligna* in sand dunes (El-Bana 2008). Some of the ecosystems inherently resist plant invasions. For example, soil and vegetation of undisturbed, late-successional forests may confer resistance to the establishment of *M. micrantha* (Hou et al. 2011). When density of native species is maintained at a high level, the negative impacts of invasive species such as *A. adenophora* may be weakened (Thapa et al. 2017).

5.8.5 Community Awareness and Public Participation

Community participation is important for the successful implementation of IAS management strategies. It is also essential from the ethical point of view and to meet legal compliance requiring public participation in decision-making, including access of communities to information related to environmental matters (Boudjelas 2009). Efforts have been made to produce community awareness and education materials (e.g., identification kit, booklets) for wider dissemination. For example, the International Centre for Integrated Mountain Development (ICIMOD), an organization working in Hindu Kush Himalaya, produced a community training manual for the management of IAPS in this region (Joshi et al. 2016). Publication of a bilingual (English and Nepali) field guide with descriptions of 27 IAPS found in Nepal has been planned in 2021 (Adhikari et al. 2021). A similar field guide is available for IAPS of Bhutan (Yangzom et al. 2018), Indonesia (Setyawati et al. 2015), Israel (Dufour-Dror 2012), Southeast Asia (Witt 2017), forests of Asia, and the Pacific Region (Sankaran and Suresh 2013). Attempts in creating awareness of IAPS at subnational level include publication of the *Handbook on Invasive Plants of Kerala*, India, by the Kerala State Biodiversity Board (Sankaran et al. 2013). A number of countries are also implementing participatory IAPS control programs by involving local communities. For example, the people of Ranupani Village of Indonesia, with support from the Bromo Tengger Semeru National Park management, have managed to clear about 65% of *Salvinia molesta* from the surface of the lake (UN Environment 2019). Community-based organizations are involved in the removal of IAPS from wetlands (including Ramsar sites – Pokhara lake cluster and Beeshajari lake system) and community managed forests in Nepal (Shrestha 2019). Parthenium awareness week is an annual event which has been regularly observed in India to motivate communities for the management of *Parthenium hysterophorus*. For example, tens of thousands of people, from school children to politicians, in 19 states of India actively participated during Parthenium Awareness Week-2009 (Varshney and Sushilkumar 2009). We envisage that millions of people and thousands of community-based organizations are involved in the management of IAPS in Asia, but these efforts and activities are yet to be documented and recognized.

Local communities, as "citizen scientists," are important stakeholders in generating knowledge that can support scientific publications and implement invasive species policy decisions (Groom et al. 2019). However, the citizen science approach is relatively rare in Asia compared to other regions of the world. For example, Johnson et al. (2020) reported 26 citizen science initiatives reporting invasive alien species that had led to publication of 31 scientific papers; these initiatives were mostly from Western Europe (11) and North America (10) and surprisingly none from Asia.

5.8.6 Policy Responses

Asian countries formulate policies, devise programs, form institutions, and invest in research and community awareness to tackle the challenges posed by the IAS. We reviewed national reports of Asian countries submitted to the Convention on Biological Diversity (https://www.cbd.int, accessed 15 April 2020) and found a wide range of variations in the policy response to manage IAS (unless stated, please refer country national reports for details). Some countries, like Japan, have separate legislations that is solely focused on IAS, the Invasive Alien Species Act, promulgated in 2005. Similarly, different laws and regulations regarding IAS, such as the Domestic Animals Epidemic Prevention Regulation and Plant Quarantine Regulations, the Quarantine Law on Import and Export of Animals and Plants, the Protection Law for Wildlife, the Law on Hygienic Quarantine, the Living Modified Organisms Act, and so forth, have been issued in East Asia (Xie et al. 2001; Washitani 2004; Son et al. 2009; Yan et al. 2012). South Korea and India have other legislations that deal with IAS. In South Korea, Conservation and Use of Biodiversity Act has a provision which designates potentially high-risk species that may harm the ecosystem if introduced to the country. Under this provision, species are subject to evaluations of their risk to the ecosystem and require approval from the Ministry of Environment when imported or introduced to South Korea.

Asian countries have also formed formal and informal institutions from central to local level dedicated to IAS management. In Malaysia, a high-level National Committee on Invasive Alien Species was established for the management of IAS to implement the National Plan of Action for Prevention, Eradication, Containment, and Control of Invasive Alien Species 2014–2018. Russia also created a National Center for Foreign Species to oversee programs and activities related to IAS management in the Russian Federation territory. Similarly, national plans were also prepared in the Philippines (National Invasive Species Strategy and Action Plan 2016–2026), Indonesia (National Strategy and Directive Action Plan for Management of Invasive Alien Species), and Malaysia (National Action Plan for the Prevention, Eradication, Containment, and Control of Invasive Alien Species) to serve as a roadmap in preventing the introduction and spread of IAS. In Nepal, a national strategy for the management of IAS is in the process of approval from the Ministry of Forest and Environment (Shrestha 2019). China has formally set up a dedicated institution under the Ministry of Agriculture and henceforth set up an emergency response office to address the invasion of alien species and organized on-site elimination of IAS and emergency responses. In Thailand, a Working Group on Alien Species has been formed under the National Sub-committee on Convention on Biological Diversity. This Working Group provides the operational guidelines that were endorsed by the Cabinet on February 2, 2018, to control and prevent the loss of biodiversity due to IAS. A country scale risk assessment framework for IAS was developed in Malaysia. Plans were not only seen at the central level but also at the local level in some Asian countries. For example, local-level plans were prepared by 17 municipal governments across South Korea to develop and implement their own annual plans in addition to the Ministry of Environment's plans to manage alien species. Local-level plans for controlling specified IAS have also been formulated in Japan.

Despite some exceptions, many Asian countries have prepared databases of IAS. Countries like Qatar, Saudi Arabia, and Maldives have little or almost no information on IAPS, whereas countries like Kazakhstan, Uzbekistan, and Bhutan have realized the threats that IAS pose to their biodiversity, agriculture, and economy but still lack formal policy, plans, and programs. Nevertheless, countries including South Korea, Japan, Thailand, the Philippines, Malaysia, India, China, Russia, and Nepal have maintained database of IAS in their countries.

Other than controlling IAPS, countries are undertaking activities to prevent IAS entering their countries by developing and implementing quarantine regulations (e.g., Mito and Uesgi 2004; Son et al. 2009; Ju et al. 2012). Inspections have been strengthened at all borders and ports in many countries. For example, in Kazakhstan, there are some measures to control pests and diseases in agriculture under plant protection and plant quarantine programs. Border control measures have also been strengthened in North Korea and Japan. In China, many professional research teams, offices, and centers on IAS have also established in universities, research academies, and government agents (Ju et al. 2012; Yan et al. 2012).

Most of the policies and programs to control and manage IAS set by the Asian countries were either guided by or aligned with Aichi Biodiversity Target 9. The Target stated that by 2020, IAS and pathways were to be identified and prioritized, priority species were to be controlled or eradicated, and measures were to be in place to manage pathways to prevent their introduction and establishment (www.cbd.int/sp/targets/). Despite some successes to manage IAS in some Asian countries and progress made to formulate policies and implement programs and form institutions, a collective initiative at the continental and/or regional scale in Asia is urgently required because the Aichi goals have yet to be met. Given the interconnectedness among Asian countries through trade and travel, global as well as regional cooperation is essential to control and manage IAS. Therefore, it is high time for Asian countries to make a common regional strategy and take action against the threat posed by IAS on their environment and economy including human health. However, before the development of such a strategy, each country needs to identify management of IAS as a priority conservation issue, develop exclusive policies to deal with biological invasions, and designate offices and staff to implement policy decisions involving all stakeholders. In addition, continued funding to support such activities needs to be sought.

5.9 Conclusions and Way Forward

Hundreds of alien plant species are naturalized in Asia, with many of them being notorious invasive species of global significance. Yet, the knowledge base generated in the continent that is essential for IAPS management is insufficient and fragmented. For instance, most countries in this region do not have prioritized lists of IAPS endorsed by government authorities for management, though researchers have attempted to do so in a few countries like India (Mungi et al. 2019), Nepal (Tiwari et al. 2005; Shrestha et al. 2019a, b; Adhikari et al. 2021), and Turkey (Yazlik et al. 2018a). Biodiversity hotspots are shared between all countries in Southeast Asia with thousands of endemic plant species, but studies examining ecological impacts of IAPS are surprisingly lacking in this region. Similarly, ecological impact studies are also lacking in Central and North Asia. With the exception of a few estimates available for Southeast Asia, China, India, and Pakistan, economic cost valuation is not available for most of the species and countries in Asia. On top of the poor knowledge base, the national response (reactive) as well as the capacity (proactive) of most of the Asian countries to manage IAPS is low to medium (Early et al. 2016). Early Detection and Rapid Response (EDRR) is the most effective approach, after "prevention," to manage biological invasions, yet this is the most neglected measure of IAPS management in Asia. In a nutshell, the number of IAPS is already high in Asia, and their number as well as spatial extent of their invasion is very likely to increase further in the near future due to lack of effective management responses (Early et al. 2016) and expanding international trade and economy of many countries in the continent (Seebens et al. 2015).

Increasing number of IAPS and their geographic extent of invasions not only threatens biodiversity and ecosystem services but also directly affects the livelihoods and well-being of millions, if not billions, of people in Asia. This necessitates some transformative approaches, as mentioned below, which would prevent the introduction of new IAPS and mitigate the impacts of established IAPS. Since IAS do not recognize political borders, their management needs to extend beyond international borders. This could be possible through regional collaboration for research and information exchange among countries that share a common pool of IAS. While this kind of cooperation has already been successful to some extent in generating scientific knowledge, and subsequently managing IAS in Europe (DAISIE 2009), it is glaringly absent in Asia. International collaboration beyond Asia, such as the one that China and the USA have for the exchange of biological control agents against IAPS (Ding et al. 2006), needs to be promoted for effective management of plant invasions. Another important approach that needs promotion in Asia are biological control programs which are currently absent in many countries. Uncertain national funding and poor infrastructure including human resources together with low awareness among stakeholders have prevented many countries to initiate biological control programs (Day and Witt 2019). Research on biological invasions has traditionally focused on ecology, with socioeconomic dimensions poorly represented not only in Asia but also throughout the world (Vaz et al. 2017). Expanding biological invasions research to include socioeconomic dimensions of IAPS will help to generate socially relevant additional data and knowledge (Abrahams et al. 2019) that not only better inform the current management and policy decisions but also may better predict future invasions in an era of global environmental change (Kueffer 2010). The citizen science approach has emerged as an important tool for generating knowledge relevant to addressing the problems of biological invasions by tapping the potential of emerging information and communication technologies (August et al. 2015). This approach may help to narrow the geographic gaps in data availability though community engagement while disseminating useful information to communities themselves. Furthermore, lack of adequate awareness of the damage caused by IAS is a serious issue among most stakeholders, especially policy makers, foresters, agriculturists, and the general public. Major efforts are required to make all stakeholders adequately aware of the problem for the formulation of appropriate policies and implementation of effective management approaches.

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