Chapter 12
Invasive Plant Species in Indian Protected Areas: Conserving Biodiversity in Cultural Landscapes

Ankila J. Hiremath and Bharath Sundaram

Abstract Invasive plant species in Indian protected areas have received relatively little attention until recently. This may partly be due to a historical emphasis on wildlife protection, rather than on a broader science-based approach to conservation of biodiversity and ecosystem functioning. A literature review of invasive plant species in India showed that nearly 60 % of all studies have been done since 2000, and only about 20 % of all studies are from protected areas. Studies from protected areas have largely focused on a small subset of invasive alien plants, and almost half these studies are on a single species, *Lantana camara*, probably reflecting the species' ubiquitous distribution. The spread of alien plants in India has been both ecologically and human mediated. Efforts to manage plant invasions have, in the past, been diluted by the ambivalence of managers attempting to find beneficial uses for these species. Despite growing knowledge about the harmful impacts of certain invasive plants on native species and ecosystems, their deliberate spread has continued, even till quite recently. And, despite the successful implementation of management initiatives in some protected areas, these efforts have not expanded to other areas. The lack of a national coordinated effort for invasive species monitoring, research, and management largely underlies this.

Keywords Invasive alien species • *Lantana camara* • Management • *Prosopis juliflora*

A.J. Hiremath (✉)
Ashoka Trust for Research in Ecology and the Environment 1, K-Block Commercial Complex (2nd floor), Jangpura Extension, New Delhi 110014, India
e-mail: hiremath@atree.org

B. Sundaram
Azim Premji University, PES Institute of Technology Campus, Electronics City, Hosur Road, Bangalore 560100, Karnataka, India
e-mail: bharath.sundaram@azimpremjifoundation.org

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12.1 Introduction

Two decades ago, Usher (1991) ventured tentatively that it was unlikely tropical nature reserves (as with reserves elsewhere) were free from alien invasive species. With the tremendous growth in global trade and travel, and with increasing landscape fragmentation, this can now be categorically stated (Mack and Lonsdale 2001; Denslow and deWalt 2008; Weber and Li 2008). This is likely to be especially true in a country like India, with its network of relatively small protected areas (PAs) set in a matrix of altered, human-dominated landscapes.

Worldwide there is a growing catalogue of the potential impacts of invasive species on native species, wildlife habitats, disturbance regimes, and ecosystem services (e.g. Pyšek et al. 2011; Foxcroft et al. 2014; Simberloff et al. 2013). Yet in Indian PAs invasive plant species have received relatively little attention until recently, whether from researchers, managers, or the general public. This neglect may, at least in part, lie in the history of forest management and conservation, and in the genesis of PAs in India.

In this chapter we review the available scientific literature on invasive alien plant species (IAPs) in Indian PAs. We then trace the history of introductions of the better-known invasive species that have been reported from Indian PAs. Using two examples of widespread invasive plant species in India, we assess their impacts. Finally, we look at patterns regarding the drivers of invasion that are starting to emerge from these studies. These findings provide valuable insights for future management of invasive species in these PAs, with their long and continuing history of human habitation and use.

12.2 India's Protected Areas

India's 668 PAs account for about 4.9 % of the country's geographic area (Krishnan et al. 2012). The categories of PAs under the Indian Wildlife Protection Act (WLPA) include national parks, wildlife sanctuaries, conservation reserves, and community conserved areas, varying in the degree of human use permitted within them. Indian parks are generally small relative to PAs in some other parts of the world, being, on average, on the order of a few 100 km^2. Many PAs have had a long history of forest management and use by communities that lived in these forests prior to their notification. Historical management and use by these communities included shifting cultivation, burning, hunting, grazing, and fuel-wood and non-timber-forest-product (NTFP) collection. Other PAs had historically been managed for the harvest of timber. Thus, these PAs are cultural landscapes as much as they are natural landscapes.

Even today a large proportion of PAs have resident forest-dependent communities. Other than in national parks, forest-dwelling communities have rights to NTFP and fuel-wood collection and grazing in protected areas, though shifting cultivation, hunting, and burning have been curtailed (Krishnan et al. 2012). An amendment to the WLPA in 2002 (Government of India 2002) banned NTFP collection for commercial use, permitting only subsistence collection. The exception to this general pattern is the case of tiger reserves, which are a subset of PAs especially earmarked for tiger (*Panthera tigris*) protection. In tiger reserves the most recent amendment to the WLPA (Government of India 2006) mandates the setting aside of a core inviolate zone, the critical tiger habitat, which is to be free from all human habitation and use.

Initially, the modern era of forest management in India (1864 onwards) was dominated by production forestry. The first PAs were established only at the turn of the twentieth century. They owed their origins to diminishing populations of valuable game animals as a result of overhunting and on-going habitat transformation. As animal numbers declined, the numbers of hunters-turned-conservationists, and their influence, increased (Burton 1953; Rangarajan 2001). Prominent amongst them was Colonel James Corbett, of the eponymous national park, and the first such park in India (Rangarajan 2001). Other protected areas owed their origins to erstwhile princely rulers who were prescient in setting aside portions of their hunting preserves – *shikargahs* – for the protection of valuable endangered species (Rangarajan 2001; Krishnan et al. 2012). One example is that of Gir in the western Indian state of Gujarat, and the last remaining home of the Asiatic lion (*Panthera leo*), which was protected by the rulers of Junagadh. Another example is Bandipur in the southern Indian state of Karnataka, where the ruler of Mysore had set aside tiger protection blocks in which hunting of wildlife was strictly prohibited (Rangarajan 2001).

Having begun in this manner, conservation in India had a single-minded focus, the protection of charismatic mammals, as is evident from the earliest reserves (e.g. Kaziranga established for the one-horned rhinoceros, *Rhinoceros unicornis*, and Kanha for the tiger; Krishnan et al. 2012). Many protected areas continue to be synonymous with individual species, for example, Gir (the Asiatic lion) and Corbett (tiger). Management in this context was largely focused on inventorying and maintaining stocks, and preventing illegal activities such as hunting and poaching (Burton 1953; Stracey 1960). This preoccupation with numbers only increased in the 1970s; a nationwide tiger census in 1972 showed that its numbers had declined markedly, leading to the initiation of Project Tiger, under which a series of tiger reserves were established (Rangarajan 2001). Even when the emphasis of conservation broadened to include other species in the landscape, management remained largely unchanged. The argument was that conservation of big mammals, so-called umbrella species, automatically guaranteed conservation of all other plants and animals, though this is not always the case (e.g. Das et al. 2006).

It is only in the last three or four decades that the focus of conservation in India has broadened to include not only species, but unique habitats and ecosystems. A biogeographic assessment of the country in 1992, under the National Wildlife Action Plan of 1983, led to the identification of gaps in the PA network, and the setting up of new reserves (Krishnan et al. 2012). Other initiatives during this time have included the establishment of biosphere reserves and world heritage sites under the aegis of UNESCO, as part of a global programme to conserve unique landscapes. In the last couple of decades the growing global focus on biodiversity hotspots (biodiversity rich areas with high degrees of endemism and threats; Myers et al. 2000) has led to recognition of the conservation value of large regions like the Eastern Himalayas and the Western Ghats (Critical Ecosystems Partnership Fund 2013). Even so, the management of PAs has remained largely unchanged, with its emphasis on protection from illegal activities, and on inventorying and maintaining wildlife numbers, with the role of science continuing to be largely insignificant (Madhusudan et al. 2006).

Given this history of forest management and conservation, it is not surprising that the insidious spread of invasive plant species in Indian PAs went largely unnoticed until quite recently. Although IAPs have occasionally been included as part of habitat management plans in PAs (e.g. *Lantana camara* in the Melghat Tiger Reserve; Sawarkar 1984), these have tended to be isolated instances. It is only in the past decade or so that both managers and researchers have become increasingly interested in the issue of IAPs in India's PAs.

12.3 Invasive Plants in Indian Protected Areas: An Overview

From the list of 100 of the world's worst invasive species (see Lowe et al. 2000, for criteria they use), 11 plant species occur in India and several of these occur in PAs. These 11 species comprise *Acacia mearnsii* (black wattle), *Arundo donax* (giant cane), *Chromolaena odorata* (Siam weed), *Clidemia hirta* (Koster's curse), *Imperata cylindrica* (cogon grass), *Lantana camara* (lantana), *Leucaena latisiliqua* (= *Leucaena leucocephala*, false koa), *Mikania micrantha* (mile-a-minute weed), *Opuntia stricta* (prickly pear), *Ulex europaeus* (gorse), and *Sphagneticola trilobata* (= *Thelechitonia trilobata, Wedelia trilobata*, Singapore daisy). These are a subset of the 225 alien plant species in India that Khuroo et al. (2012) have classified as invasive, using a modification of the classification proposed by Pyšek et al. (2004). They also recognise an additional 134 species as naturalised, but with potential to become worst i81.9(in)-283.1(the)-285.7(near)-i81.7(fut)-7(ure.)TJ1.2007-1.2007TD(Our)-372.4 string (exotic OR worst 321.8(OR)-328.8(.9(wd*)-327.8(OR)-323.20alien)-331.6(OR)-323.20nc (India). The same search string was used to search the Agricola database. In addition to these two sources, we also 2earched the available archives for the journal Tropical Ecology, which is not indexed in either of the databases searched, and relied on our knowledge of the Indian literature (especially articles in other journals not indeorst in either of the two databases, e.g. the Journal of the Bombay Natural History Society, Conservation and Society, and Indian Forester). We supplemented our findings with other relevant articles and reports cited in records retrieved from these searches. Overall, this constitutes a reasonably complete review of the

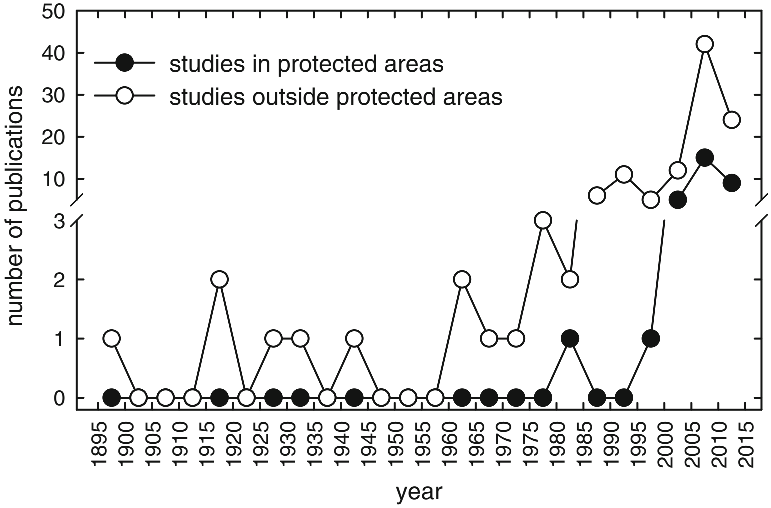

Fig. 12.1 Temporal trends in the published information on invasive alien plants in India, showing the relative numbers of studies conducted in protected areas compared to studies outside protected areas (Publications for the period 2010–2015 are still accruing)

plant species in India, but by no means is it a complete review of the grey literature. Most reports of IAP occurrences and accounts of management undertaken by the Forest Department are almost certain to remain in departmental reports and forest management plans that are difficult to access; very few such accounts make their way into the public domain (in this case, the Indian Forester, which is the journal of the Indian Forest Department). Abstracts of the initial list of articles retrieved were screened to eliminate studies that exclusively pertained to weeds of agricultural systems, though articles pertaining to IAPs in shifting cultivation systems were retained. Only articles pertaining to alien plant species that are recognised to be invasive (i.e. those that are widespread and dominant, Colautti and MacIsaac 2004) were retained. Those that dealt with other introduced or naturalised species, for example alien species in plantation forests or agro-ecosystems, were excluded. Finally, studies that were not typically ecological in their emphasis (e.g. those looking at chemical or molecular characteristics of particular species) were also excluded.

The most noteworthy finding to emerge from this review was how few studies exist on IAPs in India. We did not restrict the search to any particular time period, so the earliest report dates back to the end of the nineteenth century (Anon 1895), long before there was widespread interest in the biology of species invasions (generally attributed to Elton 1958, but see Chew 2011). For a period spanning more than a century, the search yielded less than 150 studies, with more than 60 % of these studies after the year 2000.

A second noteworthy finding from this review is how few studies are from PAs. The research on invasive plant species in PAs accounts for barely a fifth of all IAP research in India (Fig. 12.1). Considering that Indian PAs constitute only a small fraction of the country's area, making them valuable repositories of the country's biological diversity, this result is striking.

Third, the work on invasive plant species in Indian PAs has focused only on a handful of species, with *L. camara* being the subject of almost half of these studies (Table 12.1). *Lantana camara* is also the most studied invasive plant species in India, and is the focus of over a third of all studies generally. This may be a reflection of its status globally. Cronk and Fuller (1995) classify it as one of the most ubiquitous invasive plant species worldwide, ranging from tropical to subtropical and warm temperate regions of the world. Based on the current state of knowledge in India, an account of IAPs in Indian PAs is largely, though not exclusively, an account of *L. camara*.

12.4 A History of Introductions

Studies of IAPs in Indian PAs repeatedly mention only a small subset of species. One reason for this could be that most of the species in this subset have been in India for at least 100 years, presumably long enough to have become invasive (Wilson et al. 2007; Table 12.2). Second, at least two of these species (*L. camara* and *Prosopis juliflora*, mesquite) are very widespread. *Lantana camara*, particularly, occurs in a wide variety of different ecosystems (Table 12.1). *Prosopis juliflora*, on the other hand, forms extensive and conspicuous stands, even though it is restricted to the arid and semi-arid regions of the country (Saxena 1998). Third, most of the species in this subset tend to represent a new life form in the systems they have invaded. Whether it is the shrubby or clambering *L. camara* invading relatively open deciduous forests or woodland savannas, or *P. juliflora* (a tree), *Cytisus scoparius* (Scotch broom, a shrub) or *Mimosa diplotricha* (giant sensitive plant, a clambering vine) invading grasslands, the impacts of these species are more readily visible than they would be if the alien invader merely resulted in more of the same, for example, an alien grass invading a grassland.

In India, as with other regions of the world, invasive species have arrived in a variety of ways. Most alien plant species that are known to be invasive in PAs in India were first introduced into the country as garden ornamentals. Other reasons why alien species were introduced intentionally was to meet fuel-wood requirements, to prevent desert spread, and for commercial cultivation (Table 12.2). The most unusual, and perhaps apocryphal, example of an intentional introduction is that of *M. micrantha,* a climber known for its rapid growth in humid tropical environments. *Mikania micrantha* is thought to have been introduced by the allied forces during World War II to camouflage airfields built along the Indo-Burmese border as a defence against the advancing Japanese forces (Randerson 2003).

An example of an accidentally introduced species that has become invasive is *Parthenium hysterophorus* (congress grass). Reports suggest that it arrived in India as a contaminant of imported wheat in the mid-1950s, though there is evidence that it may already have been in India as early as 1810 (Paul 2010). Attempting to

Table 12.1 Protected areas in India for which published information on invasive alien plants is available. Note the disproportionate number of studies on *Lantana camara* compared with other invasive plant species, and the range of ecosystem types in which *L. camara* occurs

Protected area	Ecosystem type	Invasive species reported	Source
Kalakad Mundanturai Tiger Reserve	Tropical evergreen forest	*Lantana camara*, *Chromolaena odorata*	Chandrasekaran and Swamy (2002, 2010)
Protected forest, Anamalais	Tropical evergreen forest	*Coffea arabica*, *Coffea canephora*	Joshi et al. (2009)
Greater Nicobar Biosphere Reserve	Tropical evergreen forest	*Mikania micrantha*, *Chromolaena odorata*, *Lantana camara*, *Ageratina* spp., *Merremia peltata*	Babu and Leighton (2004)
Southern Western Ghats (no specific protected area mentioned)	Tropical evergreen forest	*Mikania micrantha*	Sankaran and Srinivasan (2001)
North-eastern India (no specific protected area mentioned)	Tropical evergreen forest	*Mikania micrantha*	Gogoi (2001)
Achanakmar-Amarkantak Biosphere Reserve	Tropical moist-deciduous forest	*Lantana camara*	Sahu and Singh (2008), Shukla et al. (2009)
Mudumalai National Park	Tropical moist-deciduous, dry deciduous forest, scrub forest	*Lantana camara*, *Chromolaena odorata*, *Opuntia stricta* var. *dillenii* (=*O. dillenii*)	Mahajan and Azeez (2001), Ramaswami and Sukumar (2011)
Biligiri Rangaswamy Temple Tiger Reserve	Tropical moist-deciduous, dry deciduous forest	*Lantana camara*, *Chromolaena odorata*	Murali and Setty (2001), Sundaram and Hiremath (2012)
Bandipur National Park	Tropical dry deciduous forest	*Lantana camara*, *Chromolaena odorata*	Puyravaud et al. (1995), Prasad (2009, 2010, 2012)
Chinnar Wildlife Sanctuary	Tropical dry deciduous forest, scrub forest	*Lantana camara*, *Ageratum houstonianum*	Chandrashekara (2001)
Melghat Tiger Reserve	Tropical dry deciduous forest	*Lantana camara*	Sawarkar (1984)
Tadoba-Andhari Tiger Reserve	Tropical dry deciduous forest	*Lantana camara*, *Hyptis suaveolens*, *Parthenium hysterophorus*	Giradkar and Yeragi (2008)
Kumbalgarh Wildlife Sanctuary	Tropical dry deciduous forest	*Prosopis juliflora*	Waite et al. (2009)
Ranthambore National Park	Tropical dry deciduous forest	*Prosopis juliflora*	Dayal (2007)
Corbett Tiger Reserve	Subtropical moist-deciduous, dry deciduous forest	*Lantana camara*	Babu et al. 2009; Love et al. (2009)

(continued)

Table 12.1 (continued)

Protected area	Ecosystem type	Invasive species reported	Source
Rajaji National Park	Subtropical moist deciduous, dry deciduous forest	*Lantana camara*	Rishi (2009), Kimothi and Dasari (2010), Kimothi et al. (2010)
Valley of Flowers National Park	Alpine meadow	*Polygonum polystachyum*	Saberwal et al. (2000), Kala and Shrivastava (2004)
Mukurti National Park	Montane forest and grassland (grassland-shola mosaic)	*Cytisus scoparius, Chromolaena odorata, Ulex europaeus, Acacia mearnsii*	Zarri et al. (2006), Srinivasan et al. (2007), Srinivasan (2011)
Kaziranga National Park	Floodplain grassland	*Mimosa invisa* (= *Mimosa diplotricha*), *Mikania micrantha*	Vattakkavan et al. (2005), Lahkar et al. (2011)
Orang National Park	Floodplain grassland	*Mimosa diplotricha, Mikania micrantha, Chromolaena odorata*	Lahkar et al. (2011)
Pabitora Wildlife Sanctuary	Floodplain grassland	*Mikania micrantha, Ipomoea carnea*	Lahkar et al. (2011)
Manas National Park	Floodplain grassland	*Mikania micrantha, Chromolaena odorata*	Lahkar et al. (2011)
Jaldapara Wildlife Sanctuary	Floodplain grassland	*Mikania micrantha*	Lahkar et al. (2011)
Garumara Wildlife Sanctuary	Floodplain grassland	*Mikania micrantha, Chromolaena odorata*	Lahkar et al. (2011)
Gulf of Mannar Marine Biosphere Reserve	Marine	*Kappaphycus alvarezii*	Bagla (2008), Chandrasekaran et al. (2008), Namboothri and Shankar (2010)

reconcile these disparate reports, Kohli et al. (2006) suggest that it may have arrived in the nineteenth century, but only became widespread in the mid-twentieth century.

The introduction of the seaweed *Kappaphycus alvarezii* for the commercial production of carrageenan deserves special mention. It was first introduced in 1993 to the Central Salt and Marine Chemicals Research Institute in western India. From there it was introduced into the Palk Bay at the southern tip of India in 2001, even though it was known to be invasive in other analogous environments (in Hawaii and the Caribbean; Namboothri and Shankar 2010). It has since spread to the Gulf of Mannar Marine Biosphere Reserve, where it is now rapidly growing over coral colonies, forming dense mats and smothering the corals below (Chandrasekaran et al. 2008).

Table 12.2 The subset of invasive alien plants in India that have been reported from protected areas, with the motives for their introduction and their source regions

Invasive species	Year of introduction	Source region	Reason for introduction	Source
Acacia mearnsii	1840s	Australia	Intentional (fuelwood)	Nair (2010)
Ageratina spp.	1800s	Mexico	Intentional (ornamental)	Muniappan et al. (2009)
Ageratum conyzoides	prior to 1882	South America	Possibly intentional (ornamental)	Kohli et al. (2006)
Ageratum houstonianum	[a]	Mexico	Possibly intentional (ornamental)	Khuroo et al. (2012)
Chromolaena odorata	1800s	Central, South America	Intentional (ornamental)	Bingelli et al. (1998)
Coffea arabica, C. canephora	1500s	Yemen	Intentional (cultivation)	Coffee Board of India (www.indiacoffee.org)
Cytisus scoparius	Prior to 1930	United Kingdom/ Europe	Intentional (ornamental)	Zarri et al. (2006), Srinivasan et al. (2007)
Hyptis suaveolens	[a]	Central, South America	[a]	Raizada (2006)
Ipomoea carnea	Late 1800s	South America	Intentional (ornamental)	Chaudhuri et al. (1994)
Kappaphycus alvarezii	1993	Philippines	Intentional (commercial)	Namboothri and Shankar (2010)
Lantana camara	1809, introduced several times during nineteenth century	South America (via Europe)	Intentional (ornamental)	Anon (1895), Iyengar (1933), Thakur et al. (1992), Day et al. (2003), Kannan et al. (2013)
Merremia peltata	[a]	Indo-Pacific region	[a]	Paynter et al. (2006)
Mikania micrantha	1940s	Central, South America	Intentional (camouflage)	Randerson (2003)
Mimosa invisa (syn. *Mimosa diplotricha*)	1960s	South America via Southeast Asia	Intentional (soil improvement)	Vattakkavan et al. (2005)
Opuntia stricta var. dillenii (= O. dillenii)	[a]	Mexico	[a]	Khuroo et al. (2012)
Parthenium hysterophorus	1950s (or prior to 1810; see text)	Latin America	Accidental	Kohli et al. (2006)

(continued)

Table 12.2 (continued)

Invasive species	Year of introduction	Source region	Reason for introduction	Source
Polygonum polystachyum	–	Indigenous weed	–	Kala and Shrivastava (2004)
Prosopis juliflora	1857, 1878	Central & South America (Mexico, Jamaica, Peru, Argentina, Uruguay)	Intentional (to halt desertification, for fuelwood)	Pasiecznik et al. (2001)
Ulex europaeus	Prior to 1910	Europe	Intentional (ornamental)	Bingelli et al. (1998)

[a]Denotes lack of information

12.5 Introduction, Invasiveness and Impacts: The Example of Two Widespread Invasive Species

In this section, the invasion history, invasiveness and impacts of IAPs in India's PAs is illustrated by using examples of two widespread invasive plant species, *L. camara* and *P. juliflora*.

12.5.1 Case Study 1: Lantana camara

12.5.1.1 Introduction and Spread

The European 'plant hunters' of the seventeenth and eighteenth centuries brought back a number of botanically and horticulturally interesting plants from their voyages and introduced them to botanical gardens across Europe, from where they travelled to other parts of the world. One such species was *L. camara*, whose earliest recorded introduction to India was in 1809 as an ornamental plant brought to the Calcutta Botanical Gardens (Thakur et al. 1992; but see also Kannan et al. 2013). There are also other later accounts of *L. camara* arriving in India, for example, in Coorg around 1865 (Anon 1895), and in peninsular India via Sri Lanka (Iyengar 1933). By the time *L. camara* was introduced into the old world tropics it had already been in cultivation as a garden ornamental in Europe since the mid- to late-seventeenth century (Day et al. 2003; Kannan et al. 2013). Plants that were introduced from Europe were thus likely to have been a complex of hybrids, which then hybridised further in their introduced environments (Day et al. 2003). This may be what underlies *L. camara's* wide ecological amplitude both in India and elsewhere (e.g. Vardien et al. 2012). Today it occurs in a variety of habitats across India, from tropical forests in the south all the way up to the subtropical and warm temperate lower reaches of the Himalayas in the north (e.g. Table 12.1;

Fig. 12.2 *Lantana camara* in the Biligiri Rangaswamy Temple Tiger Reserve, India. *Lantana camara* exhibits substantial morphological variation, (**a**) forming dense thickets 3–4 m tall, or (**b**) clambering up into tree crowns (Photo: (**a**) AJ Hiremath, (**b**) B Sundaram)

Kannan et al. 2013). It also manifests tremendous morphological variability (see Fig. 12.2). The extent to which these differences are genotypic or phenotypic is unknown, though the tools to investigate these differences now exist (Ray et al. 2013).

The earliest reports of *L. camara* spread date back to the late nineteenth century (Anon 1895; Kannan et al. 2013). Between 1917 and 1931 it was recorded to have spread at the rate of 600–1,280 ha per year across four forest ranges in North Salem, southern India, going from 3 % to 42 % of all forests in the district during this period (Iyengar 1933). The report does not, however, mention the abundance of *L. camara*, or how exactly this spread was measured. Another account indicates that *L. camara* spread at the rate of more than 2 km/year between 1911 and 1930, from a location where it was introduced in the Himalayan foothills (Hakimuddin 1930).

A recent account of *L. camara* from the Biligiri Rangaswamy Temple Tiger Reserve (BRT) constitutes perhaps the first systematic, long-term monitoring record of an invasive species' spread in a PA in India (Sundaram and Hiremath 2012). Over an 11-year period, the frequency of *L. camara* occurrence doubled across the 540 km^2 of this reserve. In 1997 *L. camara* was encountered in about 40 % of all plots surveyed, while by 2008 it was encountered in over 80 % of these plots (Fig. 12.3). The increase in spatial extent was accompanied by a commensurate, and disproportionate, increase in density. *Lantana camara* increased from one in every 20 stems in 1997, to one in every three stems by 2008. This increase in *L. camara* density was accompanied by a reduction in stems of native species, because there was no overall increase in the total numbers of stems recorded.

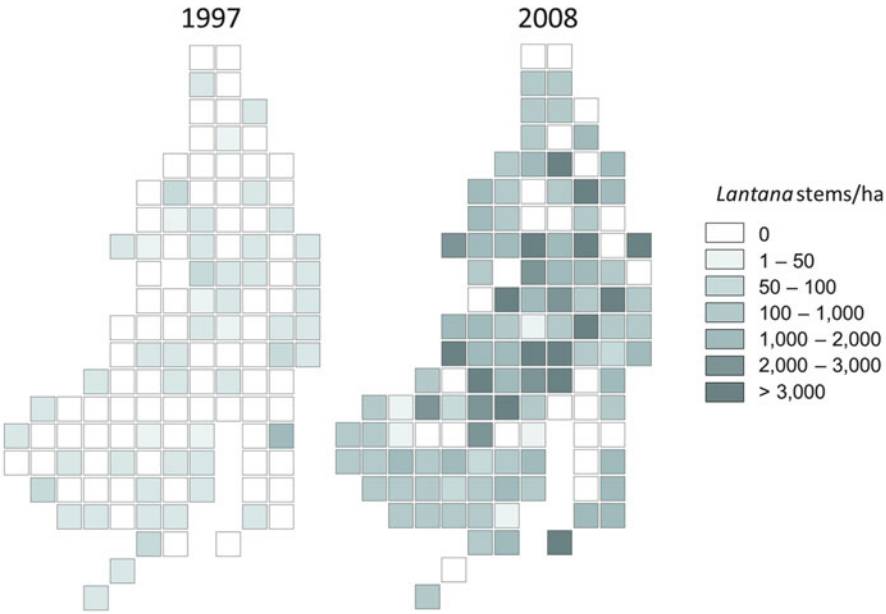

Fig. 12.3 *Lantana camara's* spread in the Biligiri Rangaswamy Temple Tiger Reserve, India, between 1997 and 2008. The squares represent cells of a 2 × 2 km grid across the 540 km^2 park. *Lantana camara* density is depicted based on vegetation surveys in a 400 m^2 plot at the centre of each cell (Sundaram 2011, with permission)

12.5.1.2 Invasiveness and Impacts

There have been several recent reviews of hypotheses pertaining to species traits that make them invasive, or community characteristics that make them invasible (Catford et al. 2009; Gurevitch et al. 2011; Jeschke et al. 2012). Several of these alternative mechanisms appear to play a role in the successful invasion of *L. camara*. One explanation is that of enemy release, which argues that species tend to grow uncontrolled in introduced environments in the absence of herbivores or pathogens that would keep them in check in their native environments (Keane and Crawley 2002). *Lantana camara* is not palatable to herbivores and thus does not appear to be preferentially browsed in Indian forests. Another explanation is that invasive species are able to take more rapid advantage of available resources and at the same time use nutrients more efficiently in low resource environments, when compared with native species (Funk and Vitousek 2007), a combination of characteristics typically thought to trade off against one another (Berendse and Aerts 1987). *Lantana camara* has been shown to be efficient at nutrient uptake and use (Bhatt et al. 1994), which would potentially give it a competitive advantage over other species, especially on nutrient poor soils.

It has also been suggested that invasive species typically produce large numbers of fruits that are widely dispersed, as do pioneer species, thus enabling them to exert propagule pressure (Lockwood et al. 2005). *Lantana camara* flowers and fruits year round, and in southern Indian deciduous forests it has been estimated to produce on the order of 10^4 fruits per individual during a single fruiting season (M. Kaushik, unpublished data). A combination of prolific fruiting and dispersal aided by avian frugivores can result in *L. camara* dominating the soil seed bank. In BRT, Sundaram (2011) found over 600 seeds/m^2 in the top 10 cm of soil, which is more than twice the number of seeds of all other native woody species (i.e. trees, shrubs, lianas) combined. These characteristics could enable *L. camara* to pre-emptively take advantage of opportunities to germinate and establish. Indeed, it has been shown to effectively colonise edges of fragmented forests (Sharma and Raghubanshi 2010) and colonise rapidly after disturbances (Duggin and Gentle 1998).

Apart from the ecological reasons, there are also human-mediated reasons underlying the successful establishment and spread of invasive species in Indian PAs. In the case of *L. camara*, despite extensive documentation of its spread and harmful impacts on agriculture and forestry, early work was focused on its potential uses (Hakimuddin 1930; Iyengar 1933), diluting attempts at control (e.g. Tireman 1916). While this was in keeping with the production-oriented approach to forest management of the time, surprisingly, this ambivalence continues. Soni et al. (2006), for example, list potential economically beneficial uses of *L. camara*, despite the accumulating ecological literature on its harmful impacts.

Invasive alien plants can have impacts at multiple scales (Parker et al. 1999). They may not be a significant cause of species extinction, other than in very specific environments like oceanic islands (Davis et al. 2011). However, there are other well-documented types of impacts on, for example, community structure and composition (e.g. Hejda et al. 2009), plant-animal interactions (e.g. Ghazoul 2004), disturbance regimes (e.g. D'Antonio and Vitousek 1992), and ecosystem processes (e.g. Vitousek and Walker 1989; Le Maitre et al. 2001). In Indian PAs these potential effects of IAPs take on added significance, especially considering the small area of remaining natural ecosystems that these PAs represent, relative to the country as a whole.

Studies indicate that *L. camara* invasion is correlated with changes in nitrogen cycling (Sharma and Raghubanshi 2009). This has been attributed to changes in litter quality and turnover under *L. camara* compared to background levels. There are also indications that *L. camara* is correlated with changes in community structure and composition (Sharma and Raghubanshi 2010; Sundaram and Hiremath 2012; Prasad 2012). The mechanism by which this happens may be the suppression of native regeneration. Although seedlings of native trees are found beneath *L. camara*, very few appear to recruit into the sapling stage (R. Ganesan, unpublished data). *Lantana camara* presence also appears to be associated with adult tree mortality (Prasad 2009; Sundaram and Hiremath 2012). A plausible explanation for this, based on observation, is that *L. camara* alters fuel characteristics (see also Berry et al. 2011), leading to fires that are more intense and severe than they would be in its absence (Tireman 1916; Hiremath and Sundaram 2005). In the long-term this could drastically alter the physiognomy of *L. camara*-invaded forests, with dire consequences for the wildlife they are meant to conserve.

In addition to changes in plant community structure and composition, there is growing evidence to suggest that *L. camara* may also have cascading trophic impacts. Increased abundance of unpalatable *L. camara* has been correlated with reduced abundances of native species. This means increased susceptibility of native vegetation to browsing, forage scarcity for herbivores, and, in turn, implications for predators like the tiger (Prasad 2010). *Lantana camara's* prolific fruiting attracts large numbers of frugivores, especially birds, potentially disrupting native plant-frugivore interactions (M. Kaushik, unpublished), and altering bird community composition, especially the abundance of certain feeding guilds (Aravind et al. 2010). Finally, the suppression of natural regeneration by *L. camara* can also have detrimental demographic consequences for important non-timber-forest-product species and for forest-dependent communities that harvest these fruits (e.g. *Phyllanthus emblica* and *P. indoficheri*, collectively known as the Indian gooseberry; Ticktin et al. 2012).

12.5.2 Case Study 2: Prosopis juliflora

12.5.2.1 Introduction and Spread

Prosopis juliflora was first brought to India in the latter half of the nineteenth century. There are at least two different accounts of its introduction, in 1857 and 1878, to halt the spread of the Thar desert in Northwest India, and for use as a fuel-wood species in peninsular India. There are indications that the sources of these two introductions differed, with seeds of the former coming from Mexico, and the latter from Jamaica. There are also records of subsequent introductions of seeds from Peru, Argentina and Uruguay (Pasiecznik et al. 2001). Following the initial success of *P. juliflora*, it was planted on a large scale in the western Indian states of Gujarat (in 1894), Rajasthan (in 1913) and Maharashtra (in 1934) (Tiwari 1999). In 1940 *P. juliflora* was even declared a “Royal Plant”, and given special protection in the erstwhile princely kingdom of Jodhpur in Rajasthan (Pasiecznik et al. 2001).

Prosopis juliflora was also introduced to several PAs as a way to alleviate pressure for fuel wood from local forest-dependent communities (Robbins 2001), from where it has spread rapidly. Two prominent examples of this are Keoladeo Ghana (Anoop 2010), and Ranthambore (Dayal 2007), both in the desert state of Rajasthan in Northwest India. Yet, while it is widely recognised as a problem in PAs in Rajasthan – Keoladeo Ghana, Ranthambore, and also Kumbalgarh (Robbins 2001) – in other parts of the country *P. juliflora* is still cited as a successful and desirable example of afforestation. In the neighbouring state of Gujarat, for instance, Saxena (1998) talks of how the entire region of Kutch was successfully converted to *P. juliflora* woodland in just a 30-year period. In the Banni grasslands, which form part of Kutch, *P. juliflora*'s rate of spread between 1980 and 1992 was estimated (using remote sensing) to be as much as 25.5 km^2 per year (Jadhav et al. 1993; Tewari et al. 2000).

12.5.2.2 Invasiveness and Impacts

Unlike *L. camara*, much less is known about what contributes to *P. juliflora's* success as an invasive species. In a study comparing it with its only native congener, *P. cineraria* (khejri), Sharma and Dakshini (1996) suggest that its seed characteristics enable it to establish and grow faster than the native species. *Prosopis juliflora* has also been shown to be tolerant of drought and salinity (Pasiecznik et al. 2001). These characteristics, in combination with its low palatability, probably give it an advantage over native species.

The impacts of *P. juliflora* in PAs have not been as well documented as those of *L. camara*. The invasion of *P. juliflora* has, however, been shown to be replacing natural habitat in India's premier bird reserve, the Keoladeo Ghana (Anoop 2010), and to have allelopathic impacts on native vegetation (Kaur et al. 2012). Others have documented its impacts on native vegetation and forest-dependent communities in and around Kumbalgarh wildlife sanctuary, where *P. juliflora* invasion, accompanied by other un-palatable shrubby species (including *L. camara*), has led to the exclusion of important fodder grasses (Robbins 2001). *Prosopis juliflora* has also been documented to be encroaching on unique habitats for grassland birds such as the rare Houbara bustard (*Chlamydotis undulata*; Tiwari 1999).

In a bio-economic analysis of Ranthambore National Park, Dayal (2007) examined the impacts of *P. juliflora* on different stakeholders, namely, wildlife managers and local villagers (a composite of fuelwood collectors, cattle grazers, and goat owners). Findings suggest that the spread of *P. juliflora* is potentially reducing forage availability for wild herbivores as well as for cattle, though not for goats, which browse on the fruits and help to disperse seeds. The detrimental impacts of the tree on wild herbivores could, in turn, have bottom-up consequences for their predators, the tiger. Different scenarios for managing *P. juliflora* may potentially lead to very different outcomes for each of the stakeholders. In the context of this biological and socio-economic complexity that characterises many of India's PAs, a key question may be whether certain types of *P. juliflora* utilization (e.g. fuel wood collection, but not browsing by goats) could also further conservation goals by benefiting both managers and villagers (Dayal 2007).

With the exception of *L. camara* and *P. juliflora,* other invasive species in Indian PAs have barely been studied. Evidence is gradually accruing to suggest that *M. diplotricha* is encroaching floodplain habitat to which rhinoceros are restricted (Lahkar et al. 2011). Also, that the spread of *C. scoparius* in the montane grasslands of the Nilgiris (southern India), is altering community composition in these unique ecosystems (Srinivasan et al. 2007). However, these examples are probably just the tip of the iceberg (see Table 12.3). For most PAs in India even basic information regarding invasive species presence or absence is lacking, while information about their impacts is virtually non-existent.

Table 12.3 The subset of invasive alien plant species in Indian protected areas for which there is documented information on impacts

Invasive species	Impact at the population or community level	Impact at the ecosystem level	Source
Coffea canephora	Correlated with altered plant community composition	[a]	Joshi et al. (2009)
Chromolaena odorata	Invading floodplain grasslands, reducing habitat for the rhinoceros	[a]	Talukdar et al. (2008), Lahkar et al. (2011)
Cytisus scoparius	Correlated with altered plant community composition	[a]	Srinivasan et al. (2007)
Ipomoea carnea	Invading floodplain grasslands, reducing habitat for the rhinoceros	[a]	Lahkar et al. (2011)
Kappaphycus alvarezii	Forms dense mats over corals, eventually killing them	[a]	Chandrasekaran et al. (2008)
Lantana camara	Correlated with altered plant community composition, altered bird community composition; potential impacts on higher trophic levels (due to impacts on herbivores)	Increased soil nitrogen cycling, change in fire regime	Tireman (1916), Prasad (2009, 2010, 2012), Sharma and Raghubanshi (2009, 2010), Aravind et al. (2010), Sundaram and Hiremath (2012)
Mikania micrantha	Smothers other vegetation, eventually killing it; invasion of floodplain grasslands, reducing habitat for the rhinoceros	[a]	Gogoi (2001), Sankaran and Srinivasan (2001), Talukdar et al. (2008), Lahkar et al. (2011)
Mimosa diplotricha	Invading floodplain grasslands, reducing habitat for the rhinoceros, toxic to herbivores	[a]	Talukdar et al. (2008), Lahkar et al. (2011)
Prosopis juliflora	Replacing grasslands, reducing habitat for grassland birds; correlated with altered plant community composition; allelopathic	Increased soil organic nitrogen, organic carbon, exchangeable phosphorus, accumulation of phenolics	Tiwari (1999), Robbins (2001), Kaur et al. (2012)

[a]Denotes lack of information

12.6 Drivers of Invasion

12.6.1 Fire

Findings are starting to emerge from Indian PAs that seem to counter prevailing theoretical (Davis et al. 2000) and empirical (e.g. Myers 1983; Hobbs 1989; Larson et al. 2001) evidence for disturbed systems being more vulnerable to invasion than undisturbed systems. In BRT, for example, anecdotal evidence suggests that initial *L. camara* colonization and establishment had been preceded by widespread fires following bamboo flowering and dieback. More recent observations suggest that *L. camara* is able to recover from fire faster than the surrounding native vegetation, leading to a self-perpetuating fire-*Lantana* cycle (proposed by Hiremath and Sundaram 2005), analogous to the invasive grass-fire cycle reported from the Americas (D'Antonio and Vitousek 1992).

In a study aimed at testing this *L. camara*-fire cycle hypothesis, Sundaram (2011) found, contrary to expectation, that areas that had burned more frequently over an 11 year period (1997–2008) showed less abundant *L. camara* than areas that burned less frequently over the same period. It is possible that this may just reflect the time since fires, with areas that burned more frequently still recovering. However, a related ethnographic study suggests otherwise. Sundaram et al. (2012) found that the local inhabitants of the BRT, the *Soliga*, date the beginning of *L. camara's* spread in these forests to about 40 years ago. This roughly coincides with the notification of BRT as a PA and the cessation of the local community's traditional forest management practices (including cool early season ground fires, termed '*taragu benki*'). The *Soliga* maintain that the annual occurrence of *taragu benki* helped to suppress *L. camara*.

Needless to say, *L. camara* is now so abundant that the occurrence of cool ground fires or *taragu benki* would be impossible; fires would today rapidly become crown fires, causing widespread mortality of native vegetation, as witnessed annually during the dry season. Thus, with the spread of *L. camara* in BRT, these forests appear to have changed from a state where fire possibly halted the spread of *L. camara*, to a state where fire promotes the spread of *L. camara* (or at least causes damage to native vegetation, which could benefit *L. camara*), all in the space of about 40 years.

12.6.2 Cessation of Disturbance: A Paradoxical Driver of Invasions

Lantana camara in BRT is not an isolated example of a possible link between cessation of a particular disturbance regime and the spread of an IAP. Studies on *C. scoparius* in the montane grasslands of the Mukurti National Park in the Nilgiris

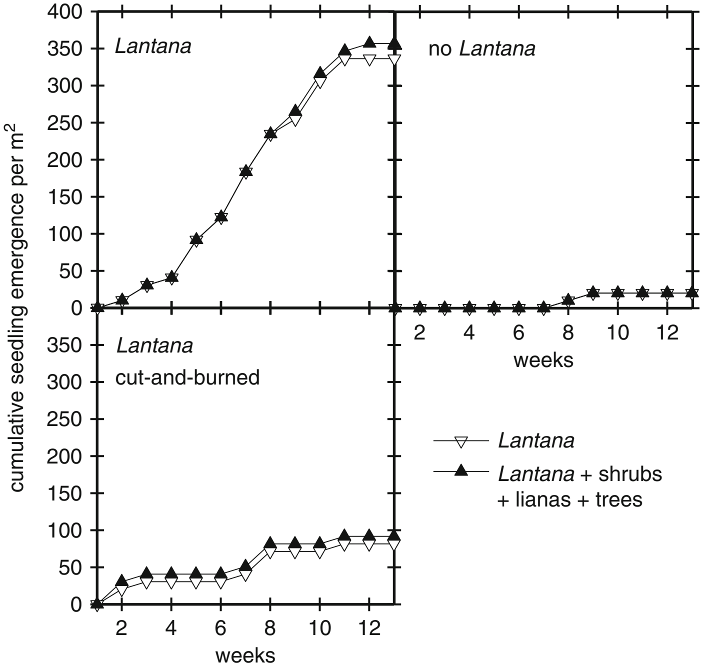

Fig. 12.4 Cumulative emergence of seeds of *Lantana camara* and other woody species from the soil seedbank over a 13-week period. Samples are from the dry season, when there are fewer viable seeds in the soil than at other times. Note the relatively large number of *L. camara* seedlings emerging in invaded sites compared with uninvaded sites. Also note the reduction in numbers of *L. camara* seedlings emerging in invaded sites that were burned

point to a similar situation. Srinivasan et al. (2012) suggest that suppression of fires may be the proximate cause of the spread of *C. scoparius* populations in Mukurti over the past few decades. This region has been home to Toda pastoralists, a community that traditionally practised fire management. The earliest records of the Todas in Mukurti go back to around 1117 A.D., suggesting that until burning ceased there had been an almost 900-year history of anthropogenic fires (Noble 1967). Similarly, in the Valley of Flowers National Park in the Himalayas, the spread of *Polygonum polystachyum* (Himalayan knotweed, a 'native invader') has been related to the cessation of grazing by nomadic pastoralists when the area became a national park (Naithani et al. 1992; Saberwal et al. 2000; but see Kala and Shrivastava 2004).

For *L. camara* in BRT, experiments suggest that fires kill seeds in the soil seed bank (Sundaram and Hiremath, unpublished; Fig. 12.4). This may be one mechanism by which *taragu benki* suppressed *L. camara* when it was not yet as widespread as it is today. Another mechanism may have been that fire helped to maintain the largely grassy understory of these deciduous woodland-savanna forests and that fire suppression enabled *L. camara* to out-compete grasses.

The cessation of a historical management regime (whether fire or grazing) could be considered equivalent to the removal of top-down control, thus exposing weaker competitors to competitive exclusion by stronger competitors (Paine 1966; Miller et al. 2001). This may be especially important where native vegetation has historically evolved with a prolonged anthropogenic disturbance regime. Anthropogenic fires may thus have played a role in preventing *C. scoparius* from becoming dominant in the montane grasslands of the Nilgiris, or in preventing *L. camara* from saturating the soil seedbank in BRT. Likewise, livestock grazing may have played a role in keeping *P. polystachyum* in check in the Valley of Flowers. Unfortunately, we only have observational evidence for these patterns. Additionally, two or three examples are insufficient to extract widespread trends. But these observations suggest that understanding the dynamics of IAPs in Indian PAs is unlikely to be complete without factoring in the ubiquitous human influence, both historical and on-going.

12.7 Conclusions and Implications for Management

Of the more than 200 IAPs and almost 700 PAs in India, information has been published on only a few invasive species and from only about 20 PAs. Despite the growing worldwide awareness of alien species invasions, India still lacks specific legislation to screen and regulate the introductions of potentially invasive species into the country. There is also no action plan for IAP management, no coordinated national research programme, and not even a repository for information on the distribution, extent and impact of even the better known invasive species (Khuroo et al. 2011). This has resulted in, for example, the seaweed *Kappaphycus alvarezii* being introduced into the vicinity of a marine biosphere reserve as recently as 2001, in spite of knowledge about its invasiveness in other similar environments.

Lantana camara and *P. juliflora* are excellent examples of the shortcomings that need to be addressed in managing invasive species in India's PAs, as well as of the opportunities that exist to do so. To develop creative solutions for long-term management and monitoring, PAs in India need to develop an adaptive management plan that promotes collaboration between researchers and managers. The lack of shared information on the distribution and impact of IAPs underlies the ambivalence with which forest managers have treated invasive species, and continue to do so. Thus, for instance, *P. juliflora* continues to be planted extensively (e.g. Saxena 1998), even as intensive efforts are in place to remove it in other areas (Anoop 2010).

In a recent review of *L. camara*, Bhagwat et al. (2012) described its management as a lost battle, suggesting that attempts to eradicate it have failed and ways of adaptively managing it need to be developed. Given how ubiquitous *L. camara* is today, it would be hard to disagree; any attempt to completely eradicate it is bound to fail. In the context of human-dominated landscapes, utilizing *L. camara* to enhance livelihoods and offset some of its costs may be one of the few viable options available (e.g. Uma Shaanker et al. 2009). However, in the context of high conservation value landscapes, there is a strong case to be made for *L. camara* control. Indeed, there are examples to demonstrate that this can be a realistic goal if researchers and managers collaborate to integrate the growing ecological understanding of invasive species with attempts to manage them, as in the case of the Corbett Tiger Reserve (Babu et al. 2009; Love et al. 2009). A similar successful collaborative programme between forest managers, researchers, and non-governmental conservation organizations is the monitoring and removal of *M. diplotricha* (= *M. invisa*) in Kaziranga National Park (Vattakkavan et al. 2005).

Another example of a successful attempt to control an invasive species in a PA comes from Keoladeo Ghana (Anoop 2010). Here park management has taken advantage of an existing poverty alleviation programme, the Mahatma Gandhi National Rural Employment Guarantee Scheme, to employ local villagers for *P. juliflora* removal (cf. the Working for Water programme in South Africa). The villagers use the *P. juliflora* for fuel wood and are employed to continue monitoring. However, despite the success of this initiative, it has not yet been expanded to other PAs across the country.

Invasive species management in PAs in India needs to move beyond just invasive plant removal. It needs to include an ecosystem approach that also considers drivers of invasion. Understanding the interaction between IAPs and the long-term anthropogenic disturbance regimes that these landscapes have evolved with may be as important to their management as understanding the biology and impacts of IAPs (Hobbs et al. 2011). Moles et al. (2012) have suggested that it is not disturbance, per se, but rather a change in disturbance regime, including the cessation of past disturbance, which may better explain ecosystem invasibility. Sharp changes in disturbance or management regimes have historically accompanied PA creation, with strict protection replacing past management (e.g. grazing, burning, etc.). Such practice is rooted in what Hobbs et al. (2006) argue is a "one-dimensional dichotomy between natural and human-dominated". They go on to suggest that we need to move away from these simplistic depictions to a more realistic understanding of how human beings interact with nature. Though they were referring to contemporary landscapes that are increasingly human-modified, it would be equally relevant in the context of PAs in India. Neither scientists nor managers can neglect the historical and on-going role of people in shaping Indian PAs. Engaging with this management history, rather than its abrupt cessation, may be a critical element in the management of IAPs in these landscapes.

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