Possible control of *Senna spectabilis* (Caesalpiniaceae), an invasive tree in Mahale Mountains National Park, Tanzania

James V. Wakibara and Bakari J. Mnaya

Abstract *Senna spectabilis* is a tree native to South and Central America. Thirty-five years ago it invaded the Mahale Mountains National Park in western Tanzania where it presently covers *c.* 225 ha. We quantified its occurrence relative to that of sympatric species of native trees, and compared girdling and felling as methods for its control in three 0.25 ha plots. Within invaded areas of forest this exotic species was both the most abundant and dominant of the 26 species of tree recorded. During 4 years of monitoring the experimental plots the abundance of *S. spectabilis* declined markedly in the plots where control methods were practised, but increased slightly in the unmanipulated plot. In contrast, the abundance of native tree species increased markedly in the plots where *S. spectabilis* had been removed or killed, with higher densities in the girdled rather than the felled plot. *S. spectabilis* appears to suppress the recruitment of native trees in the Park, and its removal can encourage regeneration of the degraded forest without the need for artificial seeding.

Keywords Caesalpiniaceae, control, exotic species, invasive tree, *Senna spectabilis*, Tanzania.

Introduction

Invasive plants are undesirable, especially in nature reserves, because their disruptive effects on the biota of invaded habitats are hard to predict (Lodge, 1993). They may, for instance, reduce the diversity of native species or alter soil chemistry, sedimentation levels, hydrological processes, fire regimes and even animal food sources (Cronk & Fuller, 1995; Callaway & Aschehoug, 2000; Christian, 2001). The best way to avoid these problems is to prevent the introduction of invasive plants into non-native regions, but increasing global trade and human mobility has facilitated mixing of fauna and flora across biogeographical boundaries (Sykora, 1990). Where plant invasion succeeds, the cost of restoration can be high (David *et al.*, 2001). Thus, exotic plants should always be viewed as potentially harmful, and be carefully monitored.

Senna spectabilis H.S. Irwin & R.C. Barneby (Caesalpiniaceae) (syn. *Cassia spectabilis* DC) is a tree native to Central and South America (Irwin & Barneby, 1982). In 1967 it was transferred from the village of Katumbi, western Tanzania, into what is now Mahale Mountains National Park, located *c.* 17 km south of the village (Fig. 1). It was initially planted to create shade but later the farmers grew it as living fences to prevent crop damage by animals (M.K. Seif, pers. comm.). At the time, slash-and-burn cultivation was common at the site and probably prevented the spread of this fire-intolerant exotic (S. Uehara, pers. comm.). In 1975 the inhabitants of Mahale were relocated following government plans to establish a National Park, following which *S. spectabilis* has flourished.

S. spectabilis grows extremely fast (Garrity & Mercado, 1994) and flourishes on acidic and infertile soils (Maclean *et al.*, 1992). It flowers and sets seed precociously, (Mbuya *et al.*, 1994), and the seed remains viable for up to 3 years (Watkins, 1960). It resprouts quickly, profusely, and persistently when cut. Garrity & Mercado (1994) were able to harvest a high biomass of cuttings in each of the four consecutive years that they pruned *S. spectabilis*. The species is non-nodulating, but accumulates nitrogen efficiently, at times even exhausting soil nitrogen reserves (Maclean *et al.*, 1992; Hulugalle & Ndi, 1993; Ladha *et al.*, 1993).

S. spectabilis is not recorded in the Global Invasive Species Database (2002), even though it has also escaped from Trinidad and Tobago and invaded the northern parts of Orinoco in Venezuela (Irwin & Barneby, 1982). It is a useful tree in Tanzania (Mbuya *et al.*, 1994), but is also recognized as invasive in parts of both Tanzania and Uganda (Anon., 2000; C.A. Chapman, pers. comm.). At Mahale it is found in *c.* 225 ha of once native forest (Ruffo, 1995). Such coverage may seem relatively minor,

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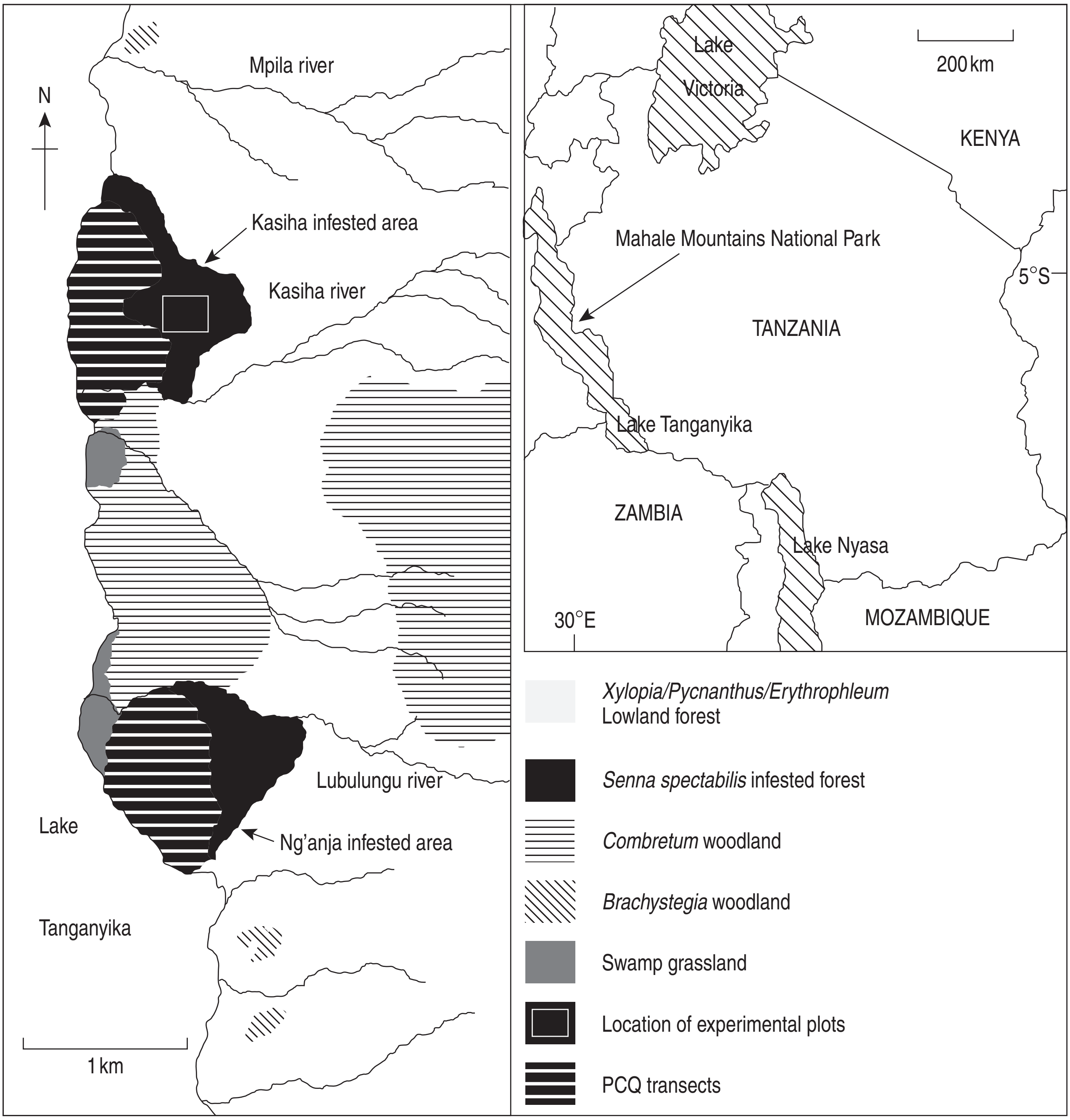

Fig. 1 Map of the study site showing the location of the three experimental plots, the survey transects, major vegetation types (after Turner, 2000), and the areas heavily infested with *Senna spectabilis*. The location of Mahale Mountains National Park is indicated on the inset.

given the Park's total area of *c.* 161,300 ha, but it amounts to 10% of the core feeding range of one group of chimpanzees *Pan troglodytes schweinfurthii*, the species that the park was primarily established to conserve (Fig. 1). Garrity & Mercado (1994) noted that *S. spectabilis* is unpalatable to ruminants, and of the 13 forest mammals at Mahale only two have been confirmed to consume parts of this exotic (Ihobe, unpub. data).

Mahale Mountains National Park (1,613 km^2) is located in western Tanzania (Fig. 1). The park's name describes its massif of several mountains, the highest of which is Nkungwe at 2,460 m. Annual rainfall averages 1,500–2,300 mm, and falls mainly during November-May. June to October is usually dry (Takasaki *et al.*, 1990). The vegetation of the area is a mosaic of forests, woodlands and swamp grassland (Fig. 1) (Turner, 2000).

Ruffo (1995) quantified *S. spectabilis* invasion at Mahale, and Wakibara (1998) initiated experiments for its control. However, both of these studies were preliminary. Here, we further quantify its occurrence relative to that of native tree species in the section of forest that has been invaded, and quantify recruitment of the native tree species in experimental plots four years after girdling or felling of *S. spectabilis*.

Methods

From 27 December 1999 to 5 January 2000 we used the Point-Centred Quarter technique (Cottam & Curtis, 1956) to assess the intensity of *S. spectabilis* invasion. A survey baseline was set approximately parallel to the shore of Lake Tanganyika, and 11 transects of 500–700 m length (a total length of 5.9 km), separated by 100 m or more, were run eastwards, perpendicular to the baseline; the length of individual transects depended on the nature of the terrain (Fig. 1). We chose at random 82 sampling points, at least 50 m apart along the transects. There were 3–10 sampling points per transect depending on its length. At each sampling point we identified the nearest tree in each quadrant of the compass with a diameter at breast height (DBH, 1.3 m above the ground) $\geqslant$10 cm and recorded its DBH and its distance from the sampling point. We used the average distance from the sampling points to the trees to calculate the number of trees per ha (Cottam & Curtis, 1956). We also determined the relative dominance of each tree species recorded.

In March 1996 we demarcated three 0.25 ha plots in an area of forest severely infested with *S. spectabilis* (Fig. 1). Delimiting growth stages as trees (DBH $>$10 cm), poles (DBH 2–10 cm), saplings (DBH $<$2 cm, height $>$1.5 m) and seedlings (height $<$1.5 m) (Okimori & Matius, 2000), we counted all individuals in the three plots. In the first plot we girdled all *S. spectabilis* trees *c.* 60 cm above ground level, in the second we felled all *S. spectabilis* trees, and we left the third untouched as a control. In the two experimental plots we cut all *S. spectabilis* poles and saplings and uprooted germinating seedlings 2–3 times per week. It took 14 days for a team of 10 to fell and girdle the two plots. Intervention was terminated after 90 days, following which relatively few new *S. spectabilis* seedlings germinated. We neither treated the plots with herbicides nor artificially enriched them with native tree propagules.

Results

We sampled a total of 328 'closest individual' trees of 26 species in 15 plant families along the 11 transects. The density of *S. spectabilis* was 586 trees ha^{-1}, whereas that of the native species was 1–43 trees ha^{-1} (Table 1). *S. spectabilis* was both the most abundant and most dominant tree species (Table 1).

Before intervention there were no significant differences between the three plots in the combined number of seedlings and saplings, or poles and trees, for either *S. spectabilis* or native species (Table 2). Four years after intervention the numbers of *S. spectabilis* were significantly lower in the felled and girdled plots compared to the control plot, whereas in the latter the number of *S. spectabilis* had increased slightly (Table 2). Intervention enhanced the regeneration of native tree species, with significant increases compared to the control, and with the greatest increases in the girdled plot (Table 2).

Twenty-seven native tree species were recorded in the experimental plots after 4 years (Table 3). Nineteen (70%) of these were also recorded in the transect survey of the infested forest (Table 1). After 4 years, 11, 6 and 3 tree species, respectively, were recorded in the girdled, felled and control plots that had not been present at the time of the initial survey.

Discussion

Our results indicate that *S. spectabilis* suppresses the growth of native tree species at Mahale, and that its removal allows their natural regeneration. Similarly Turner (2000) recently found that tree diversity at Mahale was lower in *S. spectabilis* dominated areas than in intact forests. A caveat to our results is that we only monitored the plots for 4 years, and used only three plots, in an area where *S. spectabilis* invasion was most severe. In areas where *S. spectabilis* is less dense its response to intervention could differ, and therefore caution is necessary in extrapolating our findings to all of the invaded area.

How this exotic tree suppresses the growth of native trees at Mahale is unknown. Hulugalle & Ndi (1993) suggested that *S. spectabilis* is allelopathic, but it is the legume of choice in hedge cropping systems (Mathews *et al.*, 1992a, b), and Maclean *et al.* (1992) showed that it is not allelopathic to maize or rice.

A previous attempt to control *S. spectabilis* at Mahale by felling alone was frustrated by profuse sprouting of the stumps (Turner, 1996; A.H. Seki, pers. comm.). Successful control appears to require either girdling or felling, in combination with removal of seedlings and sprouts. In particular, we found that the removal of *S. spectabilis* seedlings from the cleared plots is crucial for the regeneration of native tree seedlings, but it takes more time than tree felling or girdling alone. The girdled plot recruited both the highest number of species and

Table 1 Density and dominance of *Senna spectabilis* relative to sympatric tree species in the infested forest.

Species	Family	Number of trees ha^{-1}	Relative density[a] (%)	Relative dominance[b] (%)
Senna spectabilis	Caesalpiniaceae	586.10	72.87	47.76
Trema orientalis	Ulmaceae	42.62	3.66	2.94
Annona senegalensis	Annonaceae	34.34	3.05	2.63
Harungana madagascariensis	Guttiferae	27.90	2.74	0.24
Sterculia tragacantha	Sterculiaceae	20.70	3.05	6.76
Bridelia micrantha	Euphorbiaceae	12.72	1.22	2.13
Antidesma membranaceum	Euphorbiaceae	12.57	1.83	0.68
Unidentified 1		12.49	0.61	0.73
Erythrina abyssinica	Papilionaceae	11.50	0.61	5.36
Acacia sieberiana	Mimosaceae	11.27	0.91	9.03
Dracaena reflexa	Agavaceae	9.97	0.91	0.38
Azanza garckeana	Malvaceae	9.58	0.91	0.09
Brideria atroviridis	Euphorbiaceae	9.43	0.61	0.38
Xylopia parviflora	Annonaceae	8.51	0.61	0.62
Tabernaemontana holstii	Apocynaceae	7.27	1.22	0.49
Ficus exasperata	Moraceae	6.82	0.91	0.49
Ziziphys mucronata	Rhamnaceae	6.29	0.61	0.43
Stereospermum kunthianum	Bignoniaceae	5.52	0.61	0.14
Unidentified 2		5.21	0.61	0.08
Spathoidea nilotica	Bignoniaceae	3.90	0.61	2.75
Ficus vallis-choudae	Moraceae	3.83	0.34	3.83
Albizia gummifera	Mimosaceae	2.75	0.30	0.72
Albizia glaberrima	Mimosaceae	2.29	0.30	8.52
Celtis africana	Mimosaceae	2.06	0.30	2.67
Erythrophleum suaveolens	Caesalpinaceae	1.76	0.30	0.12
Pseudopondias microcarpa	Anacardiaceae	0.92	0.30	0.03
TOTAL		858.32	100	100

(a) Number of stems of a given species/number of stems of all tree species $\times 100$.
(b) Total basal area of a given species/total basal area of all species $\times 100$.

Table 2 Density (stems ha^{-1}) of seedlings and saplings combined, and poles and trees combined (see text for details) of *Senna spectabilis* and native tree species in two 0.25 ha experimental plots and a control plot at Mahale National Park before (March 1996) and after (Feb. 2001) *S. spectabilis* trees were felled or girdled. Figures in parentheses are the percentage change in numbers following intervention.

	Plot type				
	Felled	Girdled	Control	χ^2	P
S. spectabilis (seedlings & saplings)					
Before	11,592	11,004	11,120	2.8	0.3020
After	496 (−95.7)	1,368 (−87.6)	11,620 (+4.3)	2005.7	<0.0001
S. spectabilis (poles & trees)					
Before	5,684	4,800	5,248	9.3	0.0094
After	144 (−97.5)	792 (−83.5)	7,132 (+35.9)	1311.4	<0.0001
Native species (seedlings & saplings)					
Before	312	364	404	1.5	0.4792
After	600 (+92.3)	2,812 (+672.5)	496 (+22.8)	293.9	<0.0001
Native species (poles & trees)					
Before	448	348	288	4.5	0.1070
After	1,592 (+255.4)	3,992 (+1047.1)	388 (+34.7)	448.5	<0.0001

individuals of native trees, perhaps because the retention of some forest structure is favourable for recolonization by native species. As girdling is also less time-consuming than felling, it is probably a better method for the control of *S. spectabilis*. In the experimental plots *Harungana madagascariensis* and *Trema orientalis* are the two pioneer native tree species most likely to regenerate in areas cleared of *S. spectabilis*.

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Table 3 Number of trees per ha in the experimental plots before and after girdling or felling of *Senna spectabilis*. Numbers in parentheses are those before intervention.

		Plot type		
Species	Family	Felled	Girdled	Control
Senna spectabilis	Caesalpinaceae	0 (640)	31 (604)	856 (528)
Trema orientalis	Ulmaceae	600 (32)	564 (28)	20 (16)
Harungana madagascariensis	Guttiferae	44 (8)	276 (28)	20 (12)
Ficus exasperata	Moraceae	80 (8)	44 (4)	32 (4)
Grewia platyclada	Tiliaceae	0 (0)	92 (0)	0 (0)
Cordia millenii	Boraginaceae	32 (0)	56 (0)	0 (0)
Ficus vallis-choudae	Moraceae	32 (12)	48 (4)	4 (4)
Antidesma membranaceum	Euphorbiaceae	0 (0)	48 (4)	32 (8)
Canthium venosum	Rubiaceae	0 (0)	40 (8)	4 (4)
Pseudopondias microcarpa	Anacardiaceae	4 (0)	36 (4)	4 (0)
Xylopia parviflora	Annonaceae	0 (0)	16 (0)	4 (4)
Tabernaemontana holstii	Apocynaceae	12 (4)	8 (8)	0 (0)
Stereospermum kunthianum	Bignoniaceae	4 (4)	4 (4)	12 (0)
Sterculia tragacantha	Sterculiaceae	0 (0)	16(16)	0 (0)
Erythrophleum suaveolens	Caesalpinaceae	0 (0)	12 (4)	4 (0)
Dracaena reflexa	Agavaceae	4 (4)	12 (4)	0 (0)
Cordia africana	Boraginaceae	4 (0)	12 (4)	0 (0)
Brideria atroviridis	Euphorbiaceae	4 (4)	0 (0)	18 (16)
Annona senegalensis	Annonaceae	4 (0)	0 (0)	12 (12)
Azanza garckeana	Malvaceae	0 (0)	8 (0)	0 (0)
Albizia glaberrima	Mimosaceae	4 (4)	8 (0)	8 (8)
Ziziphys mucronata	Rhamnaceae	0 (0)	4 (0)	0 (0)
Margaritaria discoidea	Euphorbiaceae	4 (0)	4 (0)	0 (0)
Blighia unijugata	Sapindaceae	0 (0)	4 (0)	4 (4)
Grewia mollis	Tiliaceae	4 (0)	0 (0)	0 (0)
Croton sylvaticus	Euphorbiaceae	0 (0)	4 (0)	0 (0)
Celtis africana	Ulmaceae	0 (0)	4 (0)	0 (0)
Acacia sieberiana	Mimosaceae	0 (0)	4 (0)	0 (0)
TOTAL		836 (720)	1355 (724)	1034 (620)

Seedling enrichment, as suggested by Ruffo (1995), was not necessary. In a separate trial (unpub. data) we found that native seedlings transplanted into a patch cleared of *S. spectabilis* were soon uprooted by forest mammals. The seeds of the native tree species that germinated in the experimental plots could have been part of the soil seed-bank and/or deposited by frugivores. Fruit-eating animals, mostly primates, frequently traversed the experimental plots. In the forest of Kibale National Park, Uganda, Chapman & Chapman (1996) observed that chimpanzee dung contained viable seeds which assisted in the regeneration of forest patches degraded by logging.

Although *S. spectabilis* was introduced into Mahale 35 years ago, it has spread slowly and currently infests only *c.* 225 ha of the *c.* 3,000 ha of forest immediately vulnerable to its invasion frontier. *S. spectabilis* competes aggressively in disturbed forests and forest gaps but not in closed canopies (Irwin & Barneby, 1982), which is typical of most invasive plant species (Kornas, 1990; Duggin & Gentle, 1998). Duggin & Gentle (1998) considered an intact canopy to be the most effective barrier against invasion of forest by *Lantana camara* in Australia. Similarly, Mugasha *et al.* (2000) concluded that the spread of *Maesopsis eminii* on Tanzania's East Usambara Mountains declined following reduced forest disturbance by humans. The restricted distribution and slow rate of spread of *S. spectabilis* at Mahale offers an opportunity for its containment and possible control. It would probably be best to destroy the outlying pockets of invasion before tackling the main infestation (Moody & Richard, 1988).

In conclusion, *Senna spectabilis* appears to suppress the regeneration and growth of native tree species at Mahale Mountains National Park, and we have shown that there are potential options for its control. The Tanzania National Parks authority forbids the introduction of exotic species into the parks and supports initiatives for their removal. Following on from our experimental results, Tanzania National Parks is presently soliciting funds for a control programme for this damaging exotic tree species.

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