Published by SET Publisher

Journal of Basic & Applied Sciences

ISSN (online): 1927-5129

Invasion, Impact and Control Techniques for Invasive *Ipomoea hildebrandtii* on Maasai Steppe Rangelands

Ndaki Marco Manyanza[1] and Fredrick Ojija[2,*]

[1]*Department of Natural Sciences, College of Science and Technical Education, Mbeya University of Science and Technology, P.O. Box 131, Mbeya, Tanzania*

[2]*Department of Applied Sciences, College of Science and Technical Education Mbeya University of Science and Technology, P.O. Box 131, Mbeya, Tanzania*

Article Info:

Keywords:
Forages,
invasive plant,
weed survey,
Simanjiro,
Tanzania.

Received: March 17, 2021
Accepted: April 23, 2021
Published: April 26, 2021

Citation: Manyanza NM, Ojija F. Invasion, Impact and Control Techniques for Invasive Ipomoea hildebrandtii on Maasai Steppe Rangelands. J Basic Appl Sci 2021; 17: 25-36.

DOI: https://doi.org/10.29169/1927-5129.2021.17.03

Abstract:

The ecosystem integrity of the Simanjiro Maasai steppe rangeland in Tanzania is threatened by the invasive plant *Ipomoea hildebrandtii* Vatke. However, its invasion status, impact and control techniques are unclear in the country. We conducted a study in Terrat and Sukuro villages in Simanjro District, Tanzania, to assess its invasion status and impact across grassland–woodland habitats using point sampling techniques. Key informant interviews and questionnaires were used to assess techniques used by the Maasai pastoralists to control *I. hildebrandtii*. A total of 10 plots (70 m^2 each) with 9 quadrats (1 m^2 each) in the invaded and non–invaded sites were established to study *I. hildebrandtii* invasions. The impact of *I. hildebrandtii* on rangelands was investigated by comparing herbage (herbaceous vegetation) species composition, richness, basal cover and biomass productivity between invaded and non–invaded plots. Results revealed that *I. hildebrandtii* invasion was higher in grass woodland habitats (90%) than in non-invaded plots. Non–invaded plots exhibited higher biomass productivity (0.289 ± 0.03 t DM/ha) than invaded plots (0.202 ± 0.02 t DM/ha). Furthermore, non–invaded plots had a higher basal cover (grasses: 54.71 ± 1.95%, forbs: 45.29 ± 1.95%) compared with invaded plots. We also recorded high native plants abundance in quadrats with low *I. Hildebrandtii* density (22.00 ± 1.36). Additionally, 81% of Maasai pastoralists reported to manually (uproot) control *I. hildebrandtii*. Based on the results of our study, we recommend further research and novel control techniques coupled with education to be implemented in the Simanjiro.

*Corresponding Author
E-mail: fredrick.ojija@yahoo.com

INTRODUCTION

Rangelands provide vital resources for wildlife and livestock, and support human livelihoods and recreational activities [1,2]. However, rangelands integrity and sustainability are threatened by weeds and/or invasive plant species in addition to anthropogenic changes [1,3–5]. Earlier studies have described factors which facilitate the spread of invasive plants on rangelands [2]. These include, but not limited to overgrazing by livestock, habitat degradation, change in climatic conditions, frequency and timing of fires that may cause shifts in native biodiversity composition, deliberate or accidental introduction of alien plants or their seeds for agricultural purposes or food [2,6]. But, many invasive plants have been introduced to rangelands deliberately to satisfy human needs i.e. for shade provision, erosion control, and improving forage productivity [1,7]. Following their establishment, invasive plants have the potential to reduce rangelands' ability to provide ecosystem services i.e. provision of fodder quality and quantity for livestock and wildlife [1,8–11]. Correspondingly, their negative impact on rangelands have been reported to disrupt human wellbeing [12,13].

The spread of some invasive plants on rangelands are frequently enhanced by wind, flood water, and contaminated crops seeds or grains, and agricultural implements [6,8,14]. Also, the establishment of tropical botanical gardens have been reported to play a significant role in the spread of invasive plants on rangelands [15]. As invasive plants dominate on rangelands they modify vegetation structure by suppressing native species through allelopathy or competition for resources i.e. water, nutrients, space, pollinators and light [16–19]. Allelopathy has been defined as effect of a plant species on seeds germination, growth and development of nearby plants by the release of allelochemicals into the habitat [20]. Due to allelopathy and competition, invasive plants transform rangelands into novel ecosystems which are unable to carry out their ecosystem function [1].

Moreover, previous studies show that invasive plants exert negative effect on crop and livestock production [21], pollination [17,22,23] as well as human and animal health [10,24]. The magnitude of the problem is supposed to be augmented by the increase of invasions and spreading nature of invasive plant species [21]. Furthermore, the cost related to the control of invasive plants in rangelands have been claimed to be substantial [2,8]. For instance, in the United States, it was estimated that the annual cost of controlling invasive plants in rangelands is US$5 billion [2].

There are over 100 rangeland invasive plant species in Tanzania with negative impact on biodiversity [7,25]. Of these, some of the problematic invasives with deleteriously effects on native biodiversity include *Parthenium hysterophorus* L. (Asteraceae), *Chromolaena odorata* (L.) R.M.King & H.Rob. (Asteraceae), *Ipomoea hildebrandtii* Vatke (Convolvulaceae), *Leucaena leucocephala* (Lam.) de Wit (Fabaceae), *Acacia mearnsii*De Wild (Fabaceae), *Acacia melanoxylon* R. Br (Fabaceae), *Agave angustifolia* Haw. (Agavaceae), *Caesalpinia decapetala* (Roth) Alston (Fabaceae), *Calotropis procera* W.T. Aiton (Apocynaceae), *Clidemia hirta* (L.) D. Don (Melastomataceae), *Datura stramonium* L. (Solanaceae), *Lantana camara* L. (Verbenaceae), *Opuntia stricta* (Haw.) Haw. (Cactaceae), and *Xanthium strumarium* L. (Asteraceae) [7,19,26,27]. These and other invasive plant species have harmful effects on rangeland and pastures through altering vegetation dynamics [25,27]. They suppress rangelands' native vegetation through allelopathy or resource competition, modify plant community structure, reduce rangeland productivity and crop yields, and cause health problems to human and animals [25]. In general, their invasions on rangelands are associated with economic losses [12,28,29].

In Tanzania, many rangelands are vulnerable to invasive weeds [7,18,19,26,30]. This may be due to climatic conditions coupled with anthropogenic habitat change [15]. Maasai steppe rangelands outside the Tarangire national park in Tanzania are examples of rangelands prone to invasive plants i.e. *I. hildebrandtii*. The Maasai steppe rangelands are important recruitment sites for wildlife from the park and for Maasai pastoralists [31–33]. Although the Maasai steppe rangelands are vital source of nutritive forage value [32], they are endangered by *I. hildebrandtii* and land–use change [31]. *Ipomoea hildebrandtii* has also become very abundant in other rangelands in Tanzania [30]. Similar to the impact of an invasive *Ipomoea kituensis* in Kenya [34], also, *I. hildebrandtii* causes environmental and socio–economic damage, and serious threats to native biodiversity. For instance, a study in Kenya reported that invasion of *I. hildebrandtii* on rangelands caused a decline or loss of nutritious fodder species and livestock loss, and altered ecological systems and edaphic characteristics [21,35].

Despite the threats posed by *I.* hildebrandtii on biodiversity and livelihoods [21,30,35], the invasive is still under–reported and understudied in Tanzania. *Ipomoea hildebrandtii* is a sub–woody shrub with hairy stems [35]. It is perennial herb species which grows up to 4 m tall [35]. It is native to east Africa, Kenya, Tanzania, Uganda, Ethiopia, Democratic Republic of the Congo and Rwanda [21,30,35]. However, it has become invasive in Ethiopia, Kenya and Tanzania [21,25,36]. It grows and invades wastelands and degraded and disturbed habitats i.e. roadsides, overgrazed rangelands and savannah [35]. In east Africa, the spread of *I. hildebrandtii* is increasing in the region because it is often grown as an ornamental [35]. Apart from frequent drought and overgrazing which decrease grass forage in Maasai steppe rangelands [33], *I. hildebrandtii* invasions seems to escalate the problem of inadequate forage. Similarly, in Kenya, particularly in Mashuru and Namanga divisions the invasion of *I. hildebrandtii* has been considered as a major problem to livestock production as it reduces grass forage [35].

Furthermore, *I. hildebrandtii* is inedible to livestock and wildlife and therefore competes for resources with other plants, and thus, causes a decline in production of forage biomass. As a result, it threatens wildlife sustainability and their distribution in Tarangire–Maasai steppe rangelands. This is because the Maasai steppe rangelands which are the pastoral areas are also important seasonal dispersal zones of wildlife i.e. zebra, wildebeest and buffaloes. Despite growing knowledge about biological invasions in Tanzania, there is still a wide gap in understanding of *I. hildebrandtii* in Maasai steppe rangelands. This is because the invasion status of *I. hildebrandtii* is under–reported, and its impacts and control techniques are unclear in the country, particularly in Simanjiro district. Thus, this study was carried out to assess the invasion status, impact and control techniques of *I. hildebrandtii* across different habitats i.e. grassland, bushland, and woodland in Maasai steppe rangelands. We hypothesized that; (i) *I. hildebrandtii* negatively impact biomass production, species richness and basal cover of herbage plants in the invaded habitats, (ii) Maasai pastoralists practice cultural ways to control the *I. hildebrandtii* and (iii) Grasslands are highly invaded by *I. hildebrandtii* compared with woodland habitat.

MATERIALS AND METHODS

The study was conducted in Terrat and Sukuro villages in Simanjiro district, Tanzania (Figure **2**). Simanjiro district's plains are located between 3°52'S, 36°05' E and 4°24' S, 36°39' E. The temperature in Simanjiro ranges between 19 ^{o}C and 26 ^{o}C, and the average annual rainfall is 450 – 600 mm. The Maasai ethnic communities in Simanjiro rangeland plains are pastoralists. These rangelands are also used by wildlife. Agro–pastoralists and crop farmers in Simanjiro mainly cultivate maize and beans. Common vegetation in Simanjiro rangeland plains include *Digitaria macroblephara* (Hack.) Stapf, *Panicum coloratum* L., *Acacia tortilis* (Forssk) Hayne, *Commiphora schimperi* (O. Berg) Engl., *Acacia stuhlmanni* Taub. and *Pennisetum mezianum* Leeke.

Assessing the Invasion and Control Techniques of *I. hildebrandtii*

We used point sampling technique to assess the invasion status of *I. hildebrandtii* across grassland and woodland habitats in Terrat and Sukuro villages [21]. The surveys of *I. hildebrandtii* in Maasai steppe rangelands involved habitat categories; grassland, woodland, grass bushland, wood grassland, bush

Figure 1: *Ipomoea hildebrandtii* spreading on Maasai steppe rangelands in the Simanjiro, Tanzania.

woodland, and grass woodland. In each habitat, we estimated the density of *I. hildebrandtii* based on percentage acre estimation [35,37]. Furthermore, we assessed techniques used by the Maasai pastoralists to control the impact and spread of *I. hildebrandtii* on rangelands. We used key informant interviews (18 individuals) and questionnaire surveys (60 individuals) in each study villages [21,27]. Participants for questionnaire surveys in the study villages were selected based on probability sampling technique through simple random sampling [21]. The percentage of *I. hildebrandtii* invasion status was compared across grassland, woodland, grass bushland, wood grassland, bush woodland and grass woodland.

Assessing the Impact of *I. hildebrandtii* on Herbage Plant Species

A total of 10 plots of 70 m^2 each with 9 quadrats of 1 m^2 each in the invaded and non–invaded sites were established. The plots were at an interval of 300 m apart. The impact of *I. hildebrandtii* on rangelands were investigated by measuring and comparing the composition of herbage species, richness, basal cover and biomass productivity between invaded and non–invaded areas [35,37,38]. Herbage species composition, species richness, basal cover and biomass productivity were determined using a metal frame quadrat (1 m x 1 m) thrown at 30 m paces in each of the 10 plots both in the invaded and non–

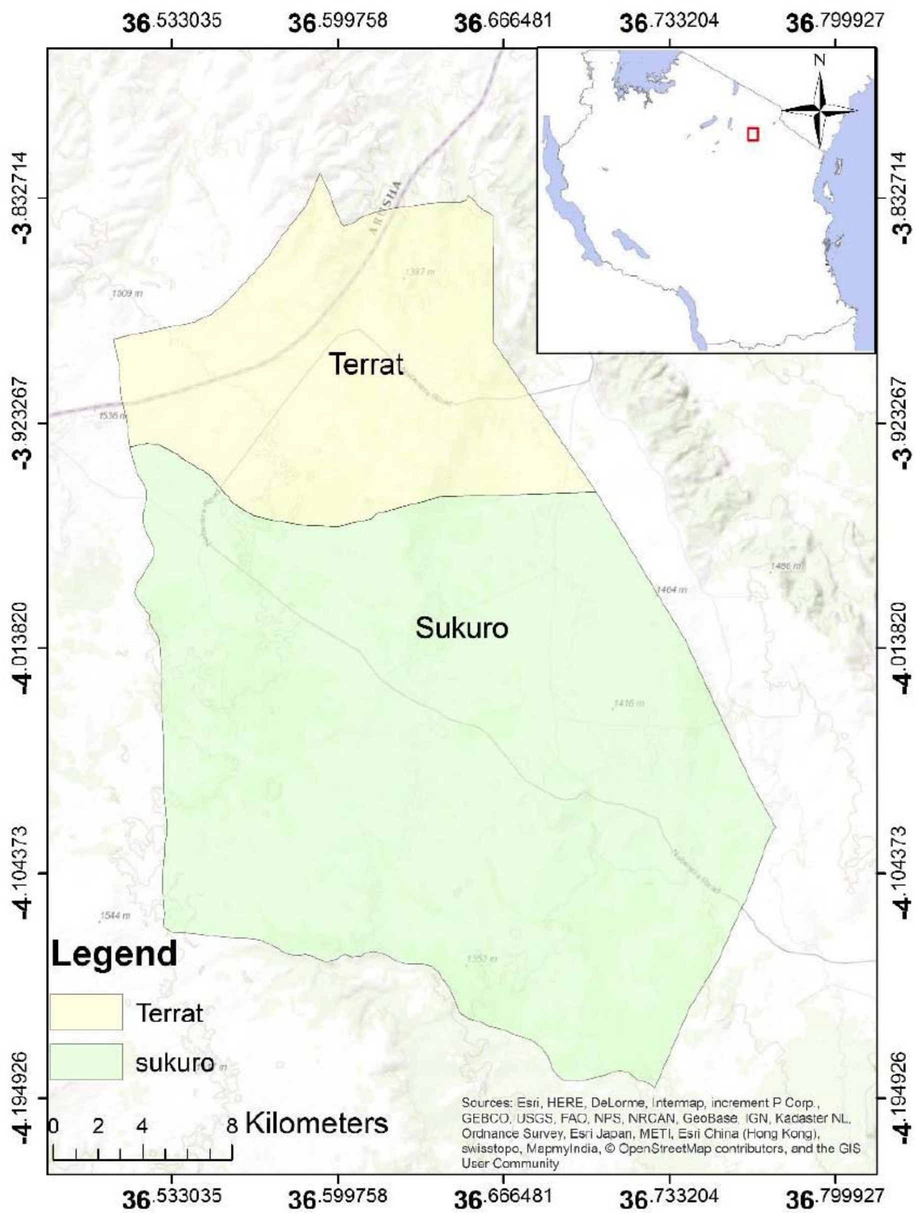

Figure 2: Map showing study villages (Terrat and Sukuro) in the Simanjiro plains.

invaded areas. We counted the number plant species and estimated the herbage species composition and basal cover (%) visually in invaded and non–invaded plots [37,38]. Herbage foliage was clipped by hand sickle at 2.5 cm above the ground to determine biomass productivity. Fresh weight of the sample (W_1) was determined in the field using a 0.01 precision digital scale. The fresh samples were then transferred in the laboratory and dried in an oven at 60 °C for 48 h [18,39]. Dried samples were measured to determine a dry weight (W_2). The W_1 and W_2 were used to calculate dry matter (DM) (equation 1) and herbaceous biomass or yield productivity (tonne, DM/ha) (equation 2).

$$DM\ (\%) = \frac{W_2}{W_1} x\, 100 \qquad \text{(eqn 1)}$$

$$t\,DM\ /\ ha = \frac{Average\ DM\ \ yield\ x\, 10000\ m^2}{1 m^2} \qquad \text{(eqn 2)}$$

where t and ha are tonne and hectare, respectively.

Furthermore, we estimated visually the density of *I. hildebrandtii* and native plant species in each nine quadrats (1 m^2) as high, medium, and low when the invasive individuals were > 4, 3–4, and 1–2 in 1 m^2 quadrat respectively [40]. We compared the density of *I. hildebrandtii* and the abundance of native plant species.

Statistical Methods

The effect of *I. hildebrandtii* on herbage species composition, species richness and basal cover were compared between invaded and non–invaded sites using a Mann–Whitney test. The impact of *I. hildebrandtii* on abundance of native plant species was analysed using one–way ANOVA (general linear model procedure) with the number of quadrats as the unit of replication and density categories as categorical predictor. Respectively, homogeneity of variance and normality were tested using Levene's and Shapiro–Wilk's test. Mean comparisons were performed using the post hoc Tukey–Kramer Honest Significant Difference (HSD) test. For all the tests, a 0.05 significance level was used. Statistical tests were performed with Origin version 9.0 SR1 (2013).

RESULTS

Higher invasions of *I. hildebrandtii* on Maasai steppe rangelands was found in grass woodland and wood grassland represented by 90% and 80%, respectively (Table **1**). It was found that the local people especially the Maasai pastoralists control the spread of *I. hildebrandtii* on rangelands through mechanical means which involves manual removal or uprooting and burning of the invasive seedlings (Table **2**). This control technique was reported as the most preferable approach for suppressing *I. hildebrandtii* in the study villages (Table **2**). Uprooting and burning of *I. hildebrandtii* were claimed to be cheap and simple techniques to use by any person in his or her area (s). Moreover, we learnt that these techniques were practiced at family level as there was no prevention or control measures for *I. hildebrandtii* at the community level. Additionally, awareness about biological invasions, management and impact of invasives on biodiversity and livestock were very low to Maasai communities in the study villages. Additionally, we did not find any information from local people about the use of herbicides and/ or biological control as an approach to suppress *I. hildebrandtii* during our study.

Table 1: Percentage (%) of *Ipomoea hildebrandtii* Invasion in Different Habitats

Habitat	Invasion status (%)
Grass bushland	30
Grassland	20
Wood grassland	80
Woodland	70
Bush woodland	0
Grass woodland	90

In Terrat village, invaded site had a higher *I. hildebrandtii* basal cover of 51.86 ± 1.84% compared with Sukuro village which had 40.15 ± 2.44% (Z = 1.14, $p < 0.001$; Table **3**). The invasive *I. hildebrandtii* reduced basal cover of both grass and forb species by 46 % in the invaded site (p< 0.001). The mean basal cover of *I. hildebrandtii* in invaded site was 46.01 ± 2.14% compared with grasses and forb (Table **3**). Invaded site had lower basal cover for grasses (29.38 ± 1.34%) and forb species (24.70 ± 1.20%) than non–invaded site (Table **3**). It was observed that presence of *I. hildebrandtii* cover in the invaded plots was associated with 46% decrease in basal cover for both grass and forb species. Furthermore, invaded plots had lower herbage biomass productivity compared with non–invaded plots (0.202 ± 0.02 and 0.289 ± 0.03 t DM/ha, respectively, Z = 1.14, p< 0.002, Figure **3**).

Table 2: Control Techniques for *Ipomoea hildebrandtii* in Maasai Steppe Rangelands

Control techniques	Respondents (n)	
	n = 78	Per cent (%)
Preventing importation of alive plant via immigrants	12	15
Manual removal/ uprooting of *I. hildebrandtii* seedlings	63	81
Burning of uprooted *I. hildebrandtii* seedlings	36	46
Chemical control of *I. hildebrandtii*	0	0
Biological control of *I. hildebrandtii*	0	0

Table 3: Herbage Basal Cover between Invaded and Non–Invaded Site

Herbage type	Mean basal cover (%)		
	Invaded sites	Non–invaded sites	
Ipomoea hildebrandtii	46.01 ± 2.14	n.a	n.a
Grasses	29.38 ± 1.34^{b}	54.71 ± 1.95^{a}	*
Forb species	24.70 ± 1.20^{c}	45.29 ± 1.95^{b}	*

Values with different letter (s) in a row differ significantly at p < 0.05, *indicates significant difference, n.a = not applicable.

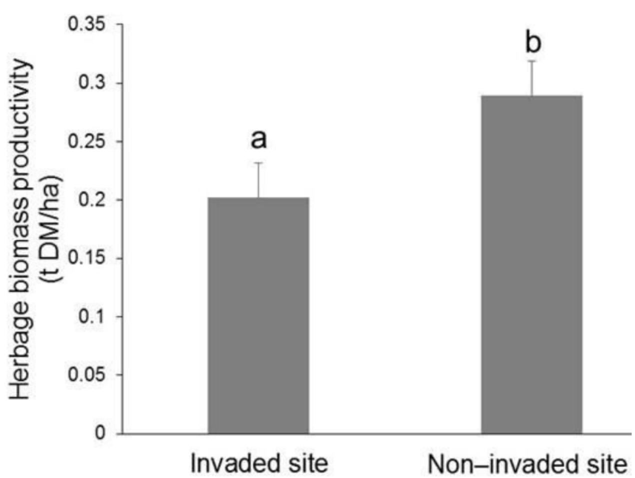

Figure 3: Herbage biomass productivity in site with (invaded site) and without (non-invaded) *Ipomoea hildegrandtii.* Different letters indicate significant difference at p < 0.05.

The abundance (mean ± S.E) of plant species, both grasses and forbs in high (13.33 ± 1.09), medium (16.56 ± 1.19) and low (22.00 ± 1.36) density of *I. hildebrandtii* differed significantly ($F_{(2,\ 24)} = 12.86$, $p = 0.0001$, Figure **4**).

Results of the study revealed that quadrats with *I. hildebrandtii* low density had higher native plant species abundance (Figure **5**). Grass species composition between invaded and non–invaded sites did not differ significantly ($p = 0.589$). However, composition of some plant species were less abundant in the invaded plots, for instance, *Cenchrus ciliaris* L. (2.3%), *Brachiaria spp*. (1.7%), *Hyperrhenia* spp. (1.7%), *Eragrostis* spp. (1.3%) and *Dactyloctenium aegyptium* (L.) Willd (0.4%) (Table **4**). *Digitaria* spp. (27.1%), *Cynodon dactylon* (L.) Pers. (21.8%) and *Aristida stipoides* Lam. (11.4%) were abundant in the invaded plots (Table **4**). Moreover, forb species composition between invaded and non–invaded plots

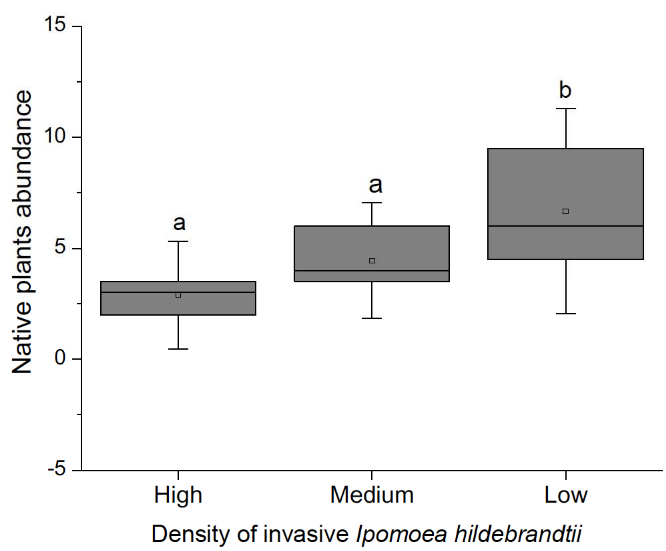

Figure 4: Impact of *Ipomoea hildebrandtii* density on the abundance of native plant species. High, medium and low density is represented by > 4, 3–4, and 1–2 individuals in 1 m^2 quadrat, respectively. Box plots show the mean (a square within boxes) and ranges from 25 to 75% quartile, and the tips of the whiskers indicate the 5th and 95th percentiles. Different letters on bars indicate significant difference based on Tukey's HSD test at p = 0.05.

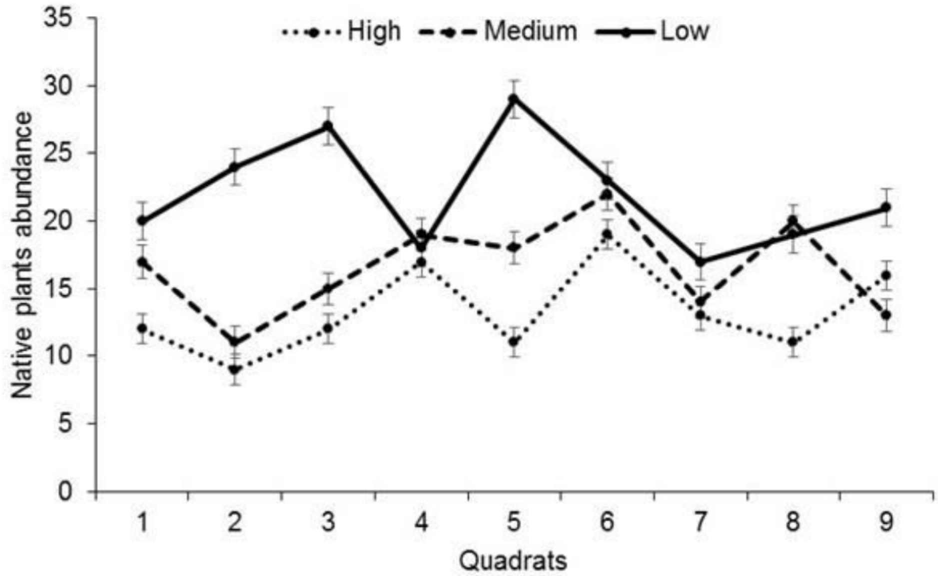

Figure 5: Variation in the number of native plant species in quadrats with respect to *Ipomoea hildegrandtii* density. High, medium, and low represent *I. hildebrandtii* density when the individuals were > 4, 3–4, and 1–2 in 1 m^2 quadrat, respectively.

Table 4: Grass Species Composition (%) in Invaded and Non–Invaded Sites

Grass species	Species composition (%)	
	Invaded sites	Non–invaded sites
Digitaria spp.	27.1	20.5
Cynodon dactylon (L.) Pers.	21.8	14.4
Aristida stipoides Lam.	11.4	1.4
Urochloa spp.	8.3	4.7
Nandi setaria Stapf. Ex. Hubb	7.0	3.7
Cyperus rotundus L.	6.6	6.5
Melinis minutiflora P. Beauv.	5.2	0.0
Sporobolus spicatus (Vahl) Kunth	2.6	1.9
Cenchrus ciliaris L.	2.3	7.8
Themeda spp.	2.2	1.9
Brachiaria spp.	1.7	21.4
Hyperrhenia spp.	1.7	2.8
Eragrostis spp.	1.3	6.5
Bothriochloa insculpta (A.Rich) A. Camus.	0.4	0.9
Dactyloctenium aegyptium (L.) Willd	0.4	5.6

did not differ significantly (p = 0.611, Table **5**). *Gutenbergia cordifolia* Benth. (14.3%), *Cyathula cylindrical* Moq. (2.2%), *Barleria ramulosa* C. B. Clarke (1.3%) were lower in the invaded plots (Table **5**). We recorded 15 grass and 14 forb species in the invaded and non–invaded plots, respectively (Table **4**). Further, many forb species were recorded in the invaded compared with non-invaded plots i.e. 21 and 16 respectively (Table **5**).

DISCUSSION

Ipomoea hildebrandtii which is rapidly spreading in eastern African rangelands is considered an invasive plant with deleterious effects on the wider ecosystem [21,35]. In our study we found that *I. hildebrandtii* invasion is strongly associated with disturbed habitats as it has been reported in southern region of Kenya. The invasive plant is invading disturbed or degraded

Table 5: Forb Species Composition (%) in Invaded and Non–Invaded Sites

Forb species	Species composition (%)	
	Invaded sites	Non–invaded sites
Gutenbergia cordifolia Benth.	14.3	23.4
Oxygonum sinuatum (Hochst. & Steud. ex. Meisn.) Dammer.	14.3	14.7
Cyathula cylindrica Moq.	2.2	11.9
Tephrosia ehrenbergiana (Schweinf.) Brummitt	10.2	11.2
Solanum campylacanthum L.	13.1	10.4
Convolvulus sagittatus Thunb.	10.8	8.6
Commelina benghalensis L.	5.6	3.6
Crotalaria sp.	0.0	3.2
Macrotyloma maranguense (Taub.) Verdc.	6.7	2.9
Barleria ramulosa C. B. Clarke	1.3	2.5
Emilia javanica (N. L. Burm.) C. B. Robinson	0.0	2.5
Justicia exigua S. Moore	11.8	2.2
Crotalaria polysperma Kotschy	0.0	1.4
Leucas glabrata (Vahl) Sm.	1.9	0.7
Pentas lanceolata (Forssk.) Deflers	3.2	0.4
Centroscema spp.	0.0	0.4
Stylosanthes spp.	1.6	0.0
Asytasia schimperi T. Anderson	0.6	0.0
Leonotis nepetifolia (L.) R. Br.	0.6	0.0
Crotalaria spinosa Benth.	0.3	0.0
Euphorbia crotonoides Boiss.	0.3	0.0
Justicia nyassana Lindau.	0.3	0.0
Sida ovata Forssk.	0.3	0.0
Zehneria scabra (Linn. f.) Sond.	0.3	0.0
Senecio ruwenzoriensis S. Moore	0.3	0.0

habitats affecting mostly overgrazed rangelands [21]. Furthermore, similar to studies on other species [19,35,36,38,41], we observed *I. hildebrandtii* invasion being common in overgrazed areas in the Maasai steppe rangelands. In this study however, we found that *I. hildebrandtii* has the potential to invade diverse habitat types including grass bushland, grassland, wood grassland, woodland, bush woodland and grass woodland. However, *I. hildebrandtii* was found to be abundant on wood grassland and grass woodland habitats in Maasai steppe rangeland. The invasions of *I. hildebrandtii* across different habitats on Maasai steppe plains perhaps is facilitated by climatic condition and the variations in terrain, edaphic factors and vegetation cover on rangelands. Moreover, differences in *I. hildebrandtii* invasion in our study villages may also be due to land use practices which influence habitats degradation. Therefore, education about biological invasions and environmental management is key and should be implemented in the villages exploiting the Maasai steppe rangelands in Simanjiro area, Tanzania.

Since *I. hildebrandtii* pose negative impact on ecosystem health of Maasai steppe rangelands through reducing forage biomass, the local people particularly the Maasai pastoralists use physical means to suppress the invasive. The control techniques (uprooting and burning) are mainly carried out by family members to prevent *I. hildebrandtii* growth and its spread in their areas. This is done so to allow rejuvenation of more native plants in the invaded areas and increase forage biomass productivity for feeding livestock (pers. comm. with villagers, 2019). These techniques have also been used to control *I. hildebrandtii* in semi–arid ecosystems in southern region of Kenya [21]. Furthermore, overgrazing and lack of awareness about management of invasive plants seem to facilitate the spread of *I. hildebrandtii* on the rangeland in Simanjiro. This may have a significant impact to wildlife ungulates particularly zebra, wildebeest, and antelopes which sometimes forage on Maasai steppe rangelands.

It was further observed that local people in the study villages do not use herbicides to control *I. hildebrandtii* and other invasive on Maasai steppe rangelands. This is a vital step towards protecting and ensuring the ecosystem health and integrity of rangelands because the use of herbicides may have far–reaching impacts on abundance and diversity of native plants on rangelands compared with the effect caused by *I. hildebrandtii* per se [18]. Several studies have asserted that herbicides can cause damage to the environment, human health and other organisms [4,18,42,43]. Also, synthetic herbicides are often broad–spectrum i.e. they would have lethal non–target impacts on native plants, endangered, threatened or ecologically important species e.g. insect natural enemies, decomposers, pollinators and soil macrobes and other microbes responsible for nutrient cycling [44]. Overall, the use of chemical herbicides to control invasives in natural and semi–natural habitats such as protected areas and rangelands is not recommended.

Furthermore, our study reveals that *I. hildebrandtii* negatively affects forage and grass biomass productivity on Maasai steppe rangelands. This means that *I. hildebrandtii* has the potential to affect livestock productivity and wildlife distribution on Tarangire–Maasai steppe ecosystem. This is because the Maasai steppe rangelands are important recruitment site for wildlife from Tarangire national park and livestock from Maasai communities as the rangelands offer nutritive forages [32,33]. It is assumed that rangelands in low or poor condition (e.g. degraded or invaded rangelands) are often biologically less productive compared with those in good conditions e.g. without invasives [37]. Also, in some cases, production of forage or fodder for livestock is greater in rangelands of good condition than on poor condition. Similar to other studies by Belnap *et al*. [1], DiTomaso *et al*. [2] and Duncan *et al*. [3], our study also report same effect of invasive *I. hildebrandtii* on rangelands which is associated with alteration of vegetation structure and reduced pasture quality and quantity. Hence, we advise that *I. hildebrandtii* should be controlled to ameliorate its negative impacts on native plant communities and ecosystem health of the rangelands that support wildlife and livestock.

Additionally, our results showed that *I. hildebrandtii* negatively impact the abundance and species composition of herbage plants. Some plant species had relatively low abundance of *I. hildebrandtii* in invaded plots. Moreover, plant species such as *Cenchrus ciliaris, Brachiaria* spp., *Hyperrhenia* spp., *Eragrostis* spp. and *Dactyloctenium aegyptium* had lower species composition in invaded sites. This indicates that these herbage species may be suppressed by *I. hildebrandtii* and thus, its control is important. Despite the number of herbage species being high in the invaded plots compared with uninvaded plots, herbage biomass productivity was low in the invaded plots. This is because, like other invasives, the large cover of *I. hildebrandtii* can reduce growth vigour and health of herbage plants. The high abundance of plants such as *Digitaria* spp., *C. dactylon*, and *A. stipoides* in the invaded plots indicate that these species perhaps are not affected by *I. hildebrandtii*. Because of this, these plant species could be used as competitor or suppressor species to control the growth and development of *I. hildebrandtii* [39,46]. Nevertheless, they can be used following detail field surveys and competition experiments to test their suppressive ability against *I. hildebrandtii* [39,41,46].

Additionally, we found that broad basal cover of *I. hildebrandtii* in the invaded plots was associated with the decline of grass and forb species basal cover. Overall, *I. hildebrandtii* invaded plots reduced the basal cover of grasses and forb species. For instance, the larger basal cover of *I. hildebrandtii* in the plots significantly reduced basal cover of herbage species by 46 % in the invaded plots. This effect depicts that *I. hildebrandtii* could be a driver of ecosystem health of the Tarangire–Maasai steppe rangeland. Thus, *I. hildebrandtii* and other invasive plants can have harmful effects on Maasai steppe rangelands and pastures by altering vegetation dynamics and limiting rangeland capacity to provide ecosystem services. With sufficiently high abundance to influence change in native biodiversity, *I. hildebrandtii* can further affect the economy of local people particularly the Maasai pastoralists who use the rangelands to graze their livestock, and earn income from tourists who visit Tarangire national park. Thus, if *I. hildebrandtii* is not controlled, it may eventually invade Tarangire national park and decrease wildlife forage, plant and animal diversity, and deplete water resources. Therefore, we strongly recommend that it may be worthwhile to manage the native plants abundance instead of maximizing diversity on altered habitats in Tarangire–Maasai steppe rangeland where *I. hildebrandtii* has already established.

CONCLUSIONS

This study has established that the invasion of *I. hildebrandtii* on Maasai steppe rangelands is associated with the decrease of herbaceous basal cover and biomass productivity of some plant species. The spread of *I. hildebrandtii* in Terrat and Sukuro villages is facilitated by low knowledge about invasive plants, poor land use and management of invasive plants. Our findings, therefore, should be a catalyst to encourage the government authorities and other stakeholders to take necessary measures to prevent and control the spread and impact of *I. hildebrandtii* in Simanjiro district. Nonetheless, local communities should be involved in any initiative to control invasive species because they are the main users of Maasai steppe rangelands. In doing so, this would enable to control and reduce the spread of *I. hildebrandtii* on rangelands. Further, we recommend that successful management of invasive plants on rangeland requires the development of a long–term strategic plan which may encompass integrating prevention programs, education, and sustainable multi–year integrated methods that may improve degraded rangeland habitats, and prevent reinvasion and/ or encroachment by other invasives.

ACKNOWLEDGEMENTS

We thank the Maasai pastoralists, the district officers and local people who provided useful information and facilitating the access to Simanjiro Conservation Easement Area. Thanks are also due to field assistants who assisted in data collection and Kafula Chisanga of the Nelson Mandela African Institution of Science and Technology, Department of Sustainable Agriculture, Arusha, Tanzania who provided input to the manuscript.

DECLARATION OF CONFLICTING INTERESTS

The author(s) declared no potential conflicts of interest with respect to the research, authorship and/or publication of this article.

FUNDING

The study was funded by the Higher Education Student Loan Board (HESLB) of Tanzania.

REFERENCES

[1] Belnap J, Ludwig JA, Wilcox BP, Betancourt JL, Dean WRJ, Hoffmann BD, *et al*. Introduced and invasive species in novel rangeland ecosystems: Friends or Foes? Rangeland Ecology Management 2012; 65: 569-78. https://doi.org/10.2111/REM-D-11-00157.1

[2] DiTomaso JM, Masters RA, Peterson VF. Rangeland Invasive Plant Management. Rangelands 2010; 32(1): 43-7. https://doi.org/10.2111/RANGELANDS-D-09-00007.1

[3] Adkins S, Shabbir A. Biology, ecology and management of the invasive parthenium weed (*Parthenium hysterophorus* L.): Management of Parthenium weed. Pest Management Science 2014; 70(7): 1023-9. https://doi.org/10.1002/ps.3708

[4] Dhileepan K. Biological control of Parthenium (*Parthenium hysterophorus*) in Australian rangeland translates to improved grass production. Weed science 2007; 55(5): 497-501. https://doi.org/10.1614/WS-07-045.1

[5] Ojija F, Arnold SEJ, Treydte AC. Plant competition as an ecosystem-based management tool for suppressing *Parthenium hysterophorus* in rangelands. Rangelands 2021; 41(6): 239-243. https://doi.org/10.1016/j.rala.2020.12.004

[6] Axmacher JC, Sang W. Plant invasions in China - challenges and chances. van Kleunen M, editor. PLoS ONE 2013; 8(5): e64173. https://doi.org/10.1371/journal.pone.0064173

[7] Sheil L. Naturalized and invasive plant species in the evergreen forests of the East Usambara Mountains, Tanzania. African Journal of Ecology 2008; 32(1): 66-71. https://doi.org/10.1111/j.1365-2028.1994.tb00556.x

[8] Chance DP, McCollum JR, Street GM, Strickland BK, Lashley MA. Native species abundance buffers non-native plant invasibility following Intermediate forest management disturbances. Forest Science 2019; 65(3): 336-43. https://doi.org/10.1093/forsci/fxy059

[9] Leweri C, Ojija F. Impact of anthropogenic habitat changes on insects: A case study of mount Loleza forest reserve. International Journal of Entomology Research 2018; 3(4): 36-43.

[10] Masters RA, Sheley RL. Invited synthesis paper: Principles and practices for managing rangeland invasive plants. Journal of Range Management 2001; 54: 502-17. https://doi.org/10.2307/4003579

[11] Ojija F, Manyanza NM. Distribution and Impact of Invasive *Parthenium hysterophorus* on Soil Around Arusha National Park, Tanzania. Ecology and Evolutionary Biology 2021; 6(1): 21-27: 7.

[12] Bajwa AA, Farooq M, Nawaz A, Yadav L, Chauhan BS, Adkins S. Impact of invasive plant species on the livelihoods of farming households: evidence from *Parthenium hysterophorus* invasion in rural Punjab, Pakistan. Biol Invasions 2019; 21(11): 3285-304. https://doi.org/10.1007/s10530-019-02047-0Ojija F, Ngimba C. Suppressive abilities of legume fodder plants against the invasive weed *Parthenium hysterophorus* (Asteraceae). Environmental and Sustainability Indictors 2021; 1-22.

https://doi.org/10.1016/j.indic.2021.100111

[13] Vilà M, Espinar JL, Hejda M, Hulme PE, Jarošík V, Maron JL, *et al*. Ecological impacts of invasive alien plants: a meta-

analysis of their effects on species, communities and ecosystems. Ecology Letters 2011; 14(7): 702-8. https://doi.org/10.1111/j.1461-0248.2011.01628.x

[14] Dawson W, Mndolwa AS, Burslem DFRP, Hulme PE. Assessing the risks of plant invasions arising from collections in tropical botanical gardens. Biodiversity and Conservation 2008; 17(8): 1979-95. https://doi.org/10.1007/s10531-008-9345-0

[15] Didham RK, Tylianakis JM, Hutchison MA, Ewers RM, Gemmell NJ. Are invasive species the drivers of ecological change? Trends in Ecology & Evolution 2005; 20(9): 470-4. https://doi.org/10.1016/j.tree.2005.07.006

[16] Lopezaraiza-Mikel ME, Hayes RB, Whalley MR, Memmott J. The impact of an alien plant on a native plant-pollinator network: An experimental approach. Ecology Letters 2007; 10(7): 539-50. https://doi.org/10.1111/j.1461-0248.2007.01055.x

[17] Ojija F, Arnold SEJ, Treydte AC. Bio-herbicide potential of naturalised *Desmodium uncinatum* crude leaf extract against the invasive plant species *Parthenium hysterophorus*. Biological Invasions 2019; 21(12): 3641-53. https://doi.org/10.1007/s10530-019-02075-w

[18] Ojija F, Arnold SEJ, Treydte AC. Impacts of alien invasive *Parthenium hysterophorus* on flower visitation by insects to co-flowering plants. Arthropod-Plant Interactions 2019; 13(5): 719-34. https://doi.org/10.1007/s11829-019-09701-3

[19] Callaway RM, Ridenour WM. Novel weapons: Invasive success and the evolution of increased competitive ability. Frontiers in Ecology and the Environment 2004; 2(8): 436-43. https://doi.org/10.1890/1540-9295(2004)002[0436:NWISAT]2.0.CO;2

[20] Bosco K, John MK, Everlyne KC, Robert N, Halima N, William MN. Key informant perceptions on the invasive *Ipomoea* plant species in Kajiado County, South Eastern Kenya. AFF 2015; 4(4): 195-9. https://doi.org/10.11648/j.aff.20150404.17

[21] Aizen MA, Morales CL, Morales JM. Invasive mutualists erode native pollination webs. Simberloff D, editor. PLoS Biology 2008 Feb 12; 6(2): e31. https://doi.org/10.1371/journal.pbio.0060031

[22] Albrecht M, Ramis MR, Traveset A. Pollinator-mediated impacts of alien invasive plants on the pollination of native plants: the role of spatial scale and distinct behaviour among pollinator guilds. Biological Invasions 2016 Jul; 18(7): 1801-12. https://doi.org/10.1007/s10530-016-1121-6

[23] Vilà M, Pujadas J. Land-use and socio-economic correlates of plant invasions in European and North African countries. Biological Conservation 2001 Aug; 100(3): 397-401. https://doi.org/10.1016/S0006-3207(01)00047-7

[24] Witt ABR, Luke Q. Guide to the naturalized and invasive plants of Eastern Africa. Wallingford, CAB International; 2017. https://doi.org/10.1079/9781786392145.0000

[25] CABI. Invasive species compendium: *Parthenium hysterophorus* (parthenium weed). https://www.cabi.org/isc/datasheet/45573. Accessed on 30-10-2019 2019.

[26] Shackleton RT, Witt ABR, Piroris FM, van Wilgen BW. Distribution and socio-ecological impacts of the invasive alien cactus *Opuntia stricta* in eastern Africa. Biol Invasions 2017; 19(8): 2427-41. https://doi.org/10.1007/s10530-017-1453-x

[27] Duncan CA, Jachetta JJ, Brown ML, Carrithers VF, Clark JK, DiTOMASO JM, *et al*. Assessing the economic, environmental, and societal losses from invasive plants on rangeland and wildlands. Weed Technolog 2004; 18: 1411-6. https://doi.org/10.1614/0890-037X(2004)018[1411:ATEEAS]2.0.CO;2

[28] Pratt CF, Constantine KL, Murphy ST. Economic impacts of invasive alien species on African smallholder livelihoods. Global Food Security 2017; 14: 31-7. https://doi.org/10.1016/j.gfs.2017.01.011

[29] CABI. Invasive species compendium: *Ipomoea hildebrandti*i. https: //www.cabi.org/ISC/datasheet/51512130. Accessed 25-09-2020 2020.

[30] Msoffe FU, Kifugo SC, Said MY, Neselle MO, Van Gardingen P, Reid RS, *et al*. Drivers and impacts of land-use change in the Maasai Steppe of northern Tanzania: an ecological, social and political analysis. Journal of Land Use Science 2011; 6(4): 261-81. https://doi.org/10.1080/1747423X.2010.511682

[31] Nelson F. Natural conservationists? Evaluating the impact of pastoralist land use practices on Tanzania's wildlife economy. Pastor Res Policy Pract 2012; 2(1): 15. https://doi.org/10.1186/2041-7136-2-15

[32] Pius ZY, a, Christopher W. Livelihoods diversifications and implications on food security and poverty levels in the Maasai plains: The case of Simanjiro district, Northern Tanzania. Afr J Environ Sci Technol 2010; 4(3): 154-66. https://doi.org/10.5897/AJEST09.177

[33] Mganga KZ, Musimba NKR, Nyariki DM, Nyangito MM, Mwang'ombe W, Ekaya WN, *et al*. The challenges posed by *Ipomoea kituensis* and the grass-weed interaction in a reseeded semi-arid environment in Kenya. International Journal of Current Research 2010; 10(12): 1-5.

[34] Mworia JK, Kinyamario JI, John EA. Impact of the invader *Ipomoea hildebrandtii* on grass biomass, nitrogen mineralisation and determinants of its seedling establishment in Kajiado, Kenya. African Journal of Range and Forage Science 2009; 25(1): 11-6. https://doi.org/10.2989/AJRFS.2008.25.1.2.380

[35] Witt A, Beale T, van Wilgen BW. An assessment of the distribution and potential ecological impacts of invasive alien plant species in eastern Africa. Transactions of the Royal Society of South Africa 2018; 73(3): 217-36. https://doi.org/10.1080/0035919X.2018.1529003

[36] Frost WE, Smith EL. Biomass productivity and range condition on range sites in Southern Arizona. Journal of Range Management. 1991; 44(1): 64. https://doi.org/10.2307/4002641

[37] Fan L, Chen Y, Yuan J, Yang Z. The effect of *Lantana camara* L. invasion on soil chemical and microbiological properties and plant biomass accumulation in southern China. Geoderma 2010; 154(3-4): 370-8. https://doi.org/10.1016/j.geoderma.2009.11.010

[38] Ammondt SA, Litton CM. Competition between native Hawaiian plants and the invasive grass *Megathyrsus maximus:* Implications of functional diversity for ecological restoration. Restoration Ecology 2012; 20(5): 638-46. https://doi.org/10.1111/j.1526-100X.2011.00806.x

[39] Wabuyele E, Lusweti A, Bisikwa J, Kyenune G, Clark K, Lotter WD, *et al*. A roadside survey of the invasive weed *Parthenium hysterophorus* (Asteraceae) in East Africa. Journal of East African Natural History 2015; 103(1): 49-57. https://doi.org/10.2982/028.103.0105

[40] Li W, Luo J, Tian X, Soon Chow W, Sun Z, Zhang T, *et al*. A new strategy for controlling invasive weeds: Selecting valuable native plants to defeat them. Scientific Reports 2015; 5(1): 1-11. https://doi.org/10.1038/srep11004

[41] Ellison CA, Cock MJW. 10 Classical biological control of *Mikania micrantha*: The sustainable solution. CAB International 2017; 162-90. https://doi.org/10.1079/9781780646275.0162

[42] Khan Z, Pickett J, Hassanali A, Hooper A, Midega C. *Desmodium* species and associated biochemical traits for

controlling Striga species: present and future prospects. Weed Research 2008; 48(4): 302-6.
https://doi.org/10.1111/j.1365-3180.2008.00641.x

[43] Frimpong JO, Ofori ESK, Yeboah S, Marri D, Offei BK, Apaatah F, *et al*. Evaluating the impact of synthetic herbicides on soil dwelling macrobes and the physical state of soil in an agro-ecosystem. Ecotoxicology and Environmental Safety 2018; 156: 205-15.
https://doi.org/10.1016/j.ecoenv.2018.03.034

[44] DiTomaso JM. Invasive weeds in rangelands: Species, impacts, and management. Weed Science 2000; 48(2): 255-65.
https://doi.org/10.1614/0043-1745(2000)048[0255:IWIRSI]2.0.CO;2

[45] Čuda J, Skálová H, Janovský Z, Pyšek P. Competition among native and invasive *Impatiens* species: the roles of environmental factors, population density and life stage. AoB PLANTS 2015; 7.
https://doi.org/10.1093/aobpla/plv033