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Integrated Plant Invasion and Bush Encroachment Management on Southern African Rangelands

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1. Introduction

1.1. Background

Rangeland could be defined as the land on which indigenous vegetation (climax or natural potential) is predominantly grass, grass-like plants, forbs, or shrubs that are grazed or have the potential to be grazed, and which is managed as a natural ecosystem for grazing livestock and wildlife habitat [1]. Rangeland productivity is threatened by land degradation mostly characterised by soil erosion and invasion by alien plant species. Plant invasion is considered a threat to rangelands because of the suppression of productivity of herbaceous plant species due to the increase of bush cover [2]. In an endeavour to understand the concepts of plant invasion in rangelands, it is important to acknowledge that the terms invasion and encroachment are normally used loosely and commonly interchangeably. However, it is crucial to understand their distinction so that the approaches in addressing their different characteristics and effects on rangelands are informed by clear comprehension. Bush encroachment refers to the spread of plant species into an area where previously it did not occur [18]. Invasion on the other hand, refers to the introduction and spread of an exotic plant species into an area where previously did not occur. Thus, bush encroachment could occur even with indigenous species and it is more defined by plant density than species themselves. Whilst invasion on the other hand, although it includes plant density, focuses on the exoticism of species in question and it is, therefore, more species specific. Furthermore, while encroachment focuses on the woodiness of the species, invasion is not limited to woody species but includes the alien herbaceous species; thus, there are grasses that are classified as invaders. However, in this chapter bush encroachment and invasion are used interchangeably and treated as synonyms.

Other than the suppression of herbaceous by encroaching species, the higher bush density in rangelands reduces land accessibility by livestock, and that subsequently negatively affects the utilisation of rangelands. Furthermore, due to competition for light, water, and nutrients between native and invading species, the grazing capacity of rangelands declines [2, 4] and plant biodiversity becomes compromised [3]. Therefore, invasions are considered one of the largest threats to the ecosystems of the earth [5- 6], and the services that they provide to humanity [5]. These species are characterised by rapid spread and they displace native vegetation and disrupt important ecosystem processes, and that leads to serious environmental impacts [5- 7]. There are a number of sources for invading species, however, in natural ecosystems such as rangelands some alien tree species used in commercial forestry and agroforestry cause major problems as invaders [8]. The effects of bush encroachment, such as an increase in woody vegetation density and cover, and reduction of biomass production in rangelands [9], have been widely reported in Southern Africa [10 – 11]. Invader species can be found in different ecosystems, however, in South Africa, they are a significant environmental problem in terrestrial and freshwater ecosystems [12]. Bush encroachment and invasion on rangelands, therefore, have negative effects on rangeland biological and economic value. Thus, bush encroachment and invasion results in rangeland degradation, which leads to declination of rangeland functional capacity and subsequently on the increased food insecurity and poverty. Hence, introduction of woody plant cover in grasslands and their increase in savanna ecosystems is an indication of rangeland degradation [13]. The foregoing assertion is aligned with the definition of rangeland degradation, which states the reduction or loss of biological and economic productivity arising from inappropriate land use practices [13]. Therefore, if bush encroachment in rangeland is left unchecked, it progresses within grassland ecosystems until a closed canopy woodland thicket occurs [15], which influences vegetation species composition and in turn threatens the sustainability of livestock production as well as wildlife habitat [16] and grassland birds [17]. Thus, the increase in vegetation cover of encroaching species can significantly reduce grass productivity through competition, shading and allelopathic effects.

Invasion phenomenon is becoming an increasing concern to land managers who are seeking cost-effective ways of combating the spread of invasive species [6]. It is important to acknowledge that factors causing invasion are complex [10, 19]. This is because of a large number of predisposing factors and that species behave differently at various environments. Therefore, any ecological and/or economic intervention in managing bush encroachment in rangelands should be anteceded by the comprehensive understanding of the drivers for this phenomenon. Nevertheless, bush encroachment is often associated with overgrazing [20]. This is because of a positive relationship between grazing pressure and woody vegetation cover [13]. There are other reported drivers of bush encroachment such as increased rainfall [21], fire suppression [22], and soil characteristics [23]. It is acknowledged, therefore, that bush encroachment threatens livestock production particularly, grazers [24] and in turn livelihoods of pastoral communities hence researchers, policy makers and practitioners need to understand bush encroachment dynamics and characteristics in order to adapt to live with or control it. Invasive plants in rangelands in the long-term affect livestock industry by lowering forage yield and quality, interfering with grazing accessibility and poisoning animals and subsequently

increasing costs of management and production of livestock, and eventually reducing land value. In the wildlife ecosystems, these species affect the wildlife habitat and forage production, deplete soil and water resources, and reduce plant and animal diversity [25]. In general, woody and succulent species invasion in rangelands result in a decline in biodiversity [26], reduction in ecosystem resilience [27] and a greater likelihood of irreversible changes in plant species composition [28].

Grazing is one of the economic ways of utilising rangelands especially in communal and/or pastoral areas. The provisions of biodiversity conservation and ecosystem stability within rangelands maintain the ecological value of these ecosystems. Therefore, maintaining or restoring rangeland ecosystem health and resilience is a critical social imperative to ensure the future supply of the ecosystem services, which are vital for the future well-being of human societies [29]. Such services include provision of stable soils, reliable and clean supply of water, and the natural occurrence of plants, animals and other organisms to meet the aesthetic and cultural values, and to enhance the livelihoods of people living around rangelands [30]. This review chapter explores the phenomenon of plant invasion and bush encroachment in the southern African region; however, reference is made to invasion and encroachment reported beyond the southern African boundaries. Furthermore, although this chapter emphasises bush encroachment and invasion in rangelands or natural ecosystems, the reference is further made from other ecosystems such as cultivated, riparian, and marine areas. This chapter explores plant invasion and encroachment phenomenon in terms of its identified causes, its ecological and economic impact. Furthermore, bush encroachment control practices in rangeland ecosystems and their significance in restoring invaded ecosystems were evaluated. Finally, different methods and approaches used in management of invasion in rangeland are synthesised into an integrated rangeland management approach.

2. Bush encroachment and invasion in rangelands

2.1. The concept of bush encroachment and invasion

Bush encroachment could be defined as an increase in woody plant abundance in grassland and savanna regions accompanied by changes in the herbaceous cover and composition of the natural vegetation [31 - 33]. This section addresses the question of whether bush encroachment and/or invasion are the problem in rangelands and if the phenomenon poses a challenge to natural ecosystems and human livelihoods. South Africa's natural ecosystems such as rangelands are under threat from invasive alien plants [12, 34], the scale of the problem facing managers of invasive alien plants in South Africa is huge, and thus, about 10 million ha has been invaded [35]. There is some sort of cosmopolitan concern about the effects of bush encroachment and invasion on rangeland ecosystem productivity and sustainability. Thus, human communities and natural ecosystems worldwide are under siege from a growing number of destructive invasive alien species [36]. These species erode natural capital, compromise ecosystem stability, and threaten economic productivity of rangeland ecosystems. Besides the effects of invasion in agriculture, forestry, and human health, biological invasions

are also widely recognised as the second-largest global threat to biodiversity. The problem of invasion in rangelands is growing in severity and geographic extent as global trade and travel accelerate, and as human mediated disturbance and increased dissemination of propagules makes ecosystems more susceptible to invasion by alien species [36]. One of the remarkable characters of invasive alien plants is that few, if any, of them are invasive in their countries of origin. Thus, their ability to grow vigorously and produce copious amounts of seeds is kept in check by a host of co-evolved invertebrates and pathogens [6]. Some of these plant species, when transported to a new continent without the attendant enemies, they exhibit “ecological release.” This phenomenon allows the introduced species to multiply rapidly in the absence of a host of attendant invertebrates and diseases, with associated tendencies to spread rapidly and to out-compete native species [6].

Mostly livestock and wildlife production depend on rangelands for sustenance as a source of feed and habitat. Rangelands are represented by a variety of ecosystems including desert and rich alluvial valleys, coastal and inland foothills, high mountain meadows and arid inland plains [25]. In the southern African context, the larger space of rangelands is represented by savanna and grassland ecosystems. Savannas are extensive, socioeconomically important ecosystems with a mixture of two life forms, thus, trees and grasses [37, 38, 39]. Whilst in Africa, savannas are the most important ecosystems for raising livestock [40]. Thus, domestic livestock, particularly *Bos* (cattle), *Ovis* (Sheep) and *Equus* (Horses) have grazed many of these areas for many years. As a result, the plant composition has changed greatly from the original ecosystems [41].

Factors and mechanisms regulating bush encroachment by invasive woody plants in rangeland ecosystems are not fully apprehended [2, 42]. However, the dynamics and modalities of bush encroachment are mostly widespread in African [13, 20], Australian [43], and North American and Latin American rangelands [39]. The increase in the tree-grass ration in the savannas has been attributed to the replacement of indigenous herbivores by domestic grazing animals and the intense utilisation of the natural vegetation by domestic livestock [33, 44]. Furthermore, heavy grazing results in reduced fuel loads leading to less frequent and low intensity fire, which reduces the effectiveness of fire in the control of woody vegetation. This heavy grazing further leads to altered competitive interactions between the woody and herbaceous layers due to the removal of grasses [32]. However, a number of times, these phenomenon have been linked to climate change [45] or land use patterns [24] or combination of number of factors [13], both biotic and abiotic in nature. Thus, local climate and long-term climate change in conjunction with grazing effects and fire limitation have been identified as possible causes of bush encroachment [46, 47, 48]. Long-term prohibition of range fire, cultivation of bottomlands and continuous grazing on the remaining portion of the communal rangelands have been reported to have induced the invasion of bush encroachment to a level of more than 60%. This has resulted in reduced grass cover, poor range condition, and subsequently poor livestock productivity [13, 49, 50].

Although there are a myriad of explanations about bush encroachment and invasion in rangelands, the first attempt at a general explanation for bush encroachment was a two-layer hypothesis for tree-grass coexistence [2, 51, 52, 53]. In this model, water is assumed be the major

limiting factor for both grassy and woody plants growth. Based on this analogy, it is hypothesized that grasses use only topsoil moisture, and woody plants mostly use subsoil moisture [54]. Therefore, reduction of grass plant density and vigour through practices such as severe grazing, allows more water to percolate into the subsoil, where it is made available for woody plant growth. Subsequently, reduction of grassy vegetation has demonstrated an increase in shrub and tree abundance under heavy grazing [55, 56]. The two-layer model is still widely accepted to explain bush encroachment phenomenon, however, field data and other theoretical models have indicated the contravening evidence [20]. Thus, the release of trees from competition with grass is not required for mass tree recruitment to occur; for example, encroachment of certain species such as *Prosopis glandulosa* is unrelated to herbaceous biomass or density [57]. Furthermore, a spatially explicit simulation model indicates that rooting niche separation might not be sufficient to warrant coexistence under a range of climatic situations [58].

This indicates that the concepts of bush encroachment and invasion in rangelands are by far still complex in terms of causation and/or predisposition factors. There were great differences reported in a number of studies in the degree of niche separation. These variations depend on various abiotic factors, and plant species involved [59, 60, 61]. Therefore, the two mechanisms, heavy grazing and rooting niche separation, do not suffice to serve as the one-dimensionally exclusive explanations for bush encroachment. This is justified by the fact that at initiation of bush encroachment young trees use the same subsurface soil layer as grasses in the sensitive early stages of growth. In addressing the relationship between bush encroachment and grazing, bush encroachment has been reported in areas where grazing was not severe. Therefore, overgrazing in combination with rooting niche separation are not the solitary predisposing factors for bush encroachment; bush encroachment sometimes also occurs on soils too shallow to allow for root separation [62]. This further shows the complexity of comprehending the causes of bush encroachment in grasslands and savannas and that further translates to the complexity of controlling the problem. This, therefore, suggests that there is no panacea in addressing the bush encroachment; therefore, integration of bush encroachment control measures and practices could lead to a sustainable solution than accrediting one method over others.

There are a number of disturbances that have been mooted to be the major determinants of savanna vegetation structure, and savannas have been portrayed as inherently unstable ecosystems. Thus, they are considered to be oscillating in an intermediate state between those of stable grasslands and forests. This is because they are pushed back into the savanna state by frequent disturbances related to human impact, herbivory, fire [61], or drought, and spatial heterogeneities in water, nutrient, and seed distribution [58]. The disturbance hypotheses suggest that bush encroachment occurs as disturbances shift savannas from the open grassland towards the forest extreme of the environmental spectrum. Although disturbance theories may be valid for specific situations, however, they may lack generality [2].

Bush encroachment and invasion by alien plant species may further be, to a certain degree, attributed to climate change. Climate change causes a number of variations in the atmosphere, and such changes could positively or negatively affect vegetation growth performance. One of the effects of climate change is an accumulation of carbon dioxide (CO₂) concentrations in

the atmosphere. These increased CO₂ concentrations are likely to have an effect on tree-grass dynamics in savannas. This is because savanna trees and grasses have different photosynthetic pathways, which will respond differently to changes in atmospheric CO₂ accumulations. It is predicted that atmospheric CO₂ is exponentially increasing and will likely double to 700 parts per million (ppm) within the next century [62]. This has a further potential beneficial effect on plant life; the benefit is attributed to the fact that plants take up CO₂ via photosynthesis and use it in photosynthesis to produce carbohydrates. Thus, the higher CO₂ concentration could significantly increase the capacity of plants to absorb and temporarily store excess carbon. The efficiency of plants in the savanna to utilise the high CO₂ concentrations will be influenced to a larger extent by the photosynthetic pathways of different plant species and, therefore, that will influence plant species composition and ecosystem structure. For example, *Acacia* trees have the C3 photosynthetic pathway, which is less efficient, hence, they have a lower net photosynthetic rate at current atmospheric CO₂ levels than the C4 pathway used by most of savanna grasses [62]. However, at the higher atmospheric CO₂ levels than currently experienced, C3 plants will have a higher net photosynthetic rate than C4 plants. Thus, C3 plants should show increases in yield of 20–35% with a doubling of atmospheric CO₂, while C4 plants such as grasses should only experience a 10% increase in yield. Furthermore, the increased CO₂ concentrations will improve the competitive ability of trees against grasses. Thus, *Acacia* trees will have more carbon to invest in carbon-based defences against herbivory such as condensed tannins [63, 64].

In an attempt to further explain bush encroachment phenomenon in semi arid and arid environments, it is hypothesised that it is a natural phenomenon occurring in ecological systems governed by patch-dynamic processes [65]. This hypothesis has been based on field observations gained on the spatial distribution of *Acacia reficiens* trees in arid central Namibia. It is argued that encroachment of *A. reficiens* along rainfall gradient increases with increasing rainfall in spite of a relatively constant level of grazing [65]. However, any form of vegetation disturbance in rangelands (grazing, fire, etc.) can create space, and thus, making water and nutrients available for tree establishment due to reduced competition. However, under low soil nitrogen conditions, the nitrogen-fixing trees have a competitive advantage over other plants and, given enough rainfall, may germinate as a group in the bare patches created by the disturbances. The mechanism underlying this hypothesis, which demonstrates how it may be used to explain this phenomenon are such that both tree-grass coexistence and bush encroachment occur in a patch-dynamic system with stochastic rainfall patterns [2]. Nevertheless, it was suggested that in arid and semi-arid savanna ecosystems, woody vegetation needs above-average precipitation for germination and subsequent establishment [66]. To keep the soil moist for a period sufficient for germination and survival through the sensitive early stages of seedling development, several rain events close in succession are necessary [67]. However, in a savanna ecosystem, rainfall is often patchily distributed, in terms of both time and space [46, 68, 69]. Therefore, the spatial overlap of several rainfall events of high frequency in a single year is a rare occurrence in semi-arid and arid ecosystems. In addition to local seed availability, this rainfall frequency is a necessary condition for the creation of a bush encroachment patch.

2.2. Spatial distribution of encroaching and invasive plant species in rangelands

Several estimates have been made of the spatial extent of alien plant invasions in South Africa [36]. The rapid reconnaissance in 1996/97 [35] suggested that about 10 million hectares of South Africa has been invaded by the approximately 180 species that were mapped. In South Africa, there are a number of invading species; however, the principal invaders are trees and shrubs in the genera *Acacia*, *Hakea* and *Pinus*. However, the majority of invasive and/or encroaching species in rangelands are in the Fabaceae family, which are normally nitrogen-fixing legumes [70]. Localization of invading species distribution is influenced by the landscape formation gradient, thus, there are dense invasions in the mountains and lowlands and along the major river systems [12]. The susceptibility of rangelands to bush encroachment and/or invasion varies between the vegetation types. Thus, vegetation types such as grassland and savanna biomes are extensively invaded mostly by species such as Australian wattles (*Acacia* species), other tree species, and a variety of woody scramblers (notably, trifid weed, *Chromolaena odorata*, and brambles, *Rubus* species). Invading trees such as jacaranda (*Jacaranda mimosifolia*) and syringe (*Melia azedarach*) have spread into semi-arid savanna by spreading along perennial rivers. In the Nama Karoo, woody invaders, notably mesquite (*Prosopis* species), have invaded large areas of alluvial plains and seasonal and ephemeral watercourses. Several cacti (*Opuntia* species) and saltbushes (*Atriplex* species) have invaded large areas of the Nama Karoo and Succulent Karoo [71] and the thicket biome in the Eastern Cape [12].

There are a number of species introduced from other continents and can cause significant problems on rangelands. The temporal and spatial spread of an invading organism including plants generally follows a sigmoid curve [72, 73]. Thus, the initial expansion is slow as the founder colony expands and starts new colonies, decreasing again as the potential habitat (invadable area) becomes fully occupied. The increase of invasive species on the given space and time leads to significant changes on the ecosystem integrity. Thus, invasive plants in the new region lead to profound changes in ecosystem processes, community structure, and displacing native species [74]. Therefore, it is fundamental to determine the spread of invading species in terms of time and space prior to development of a plan to control them. Several attempts have been made to prioritize alien species according to their invasive potential in different parts of the world. However, most attention has been given to screening species for their invasive potential prior to their introduction to a region [75, 76].

The ranking of Weeds of National Significance was developed for Australia based on expert scoring of four criteria [77]. These are grounded on their invasiveness, impacts, potential for spread, and socio-economic and environmental values. In South Africa, invasive species were prioritized based on their potential invasiveness, spatial characteristics, potential impacts, and conflicts of interest [78]. The Southern African Plant Invaders Atlas (SAPIA) database contains records for over 500 species of invasive alien plants in South Africa, Lesotho, and Swaziland, with information on their distribution, abundance, and habitat types [79]. There are two lists of invasive alien plants, classified into group species based on similarities in their distribution, abundance, and/or biological traits [80]. The first list contains those species that have already had a substantial impact on natural and semi-natural ecosystems such as rangeland in South Africa. Species demonstrating high value for any of the three components was considered to

have high impact and species with high values for all three components have the highest impact. These species are perceived to constitute the prime concern for managers and, therefore, are referred to as the major invaders. Therefore, the presence and abundance of this species could be regarded to be above the economic threshold and warrant economic and ecological attention. Thus, the projects aimed at the prevention and/or control of these species should receive the largest proportion of available funding over the next few decades.

The second list contains those species that currently have a lower impact on natural or semi-natural ecosystems in South Africa. Thus, these species exhibit a lower product of range, abundance, and effect, but appear to have the capacity to exercise greater influence in the future. They are, therefore, termed “emerging invaders,” and are currently afforded lower priority in management. However, some of these species are likely to become more important in the future, and could become targets for pre-emptive action such as biocontrol. These species should be carefully monitored to ensure that they do not become major problems. There are 117 major invaders identified in South Africa, and black wattle (*Acacia mearnsii*), white and grey poplars (*Populus alba/canescens*) and mesquite (*Prosopis glandulosa* var. *Torreyana/velutina*) are the three species/species-groups falling within the ‘very wide spread-abundant’ category [80]. The distribution pattern of these ‘very widespread/widespread-abundant’ species corresponds to the areas where high overall numbers of invasive alien plants were recorded. Most of the major invaders are found within the ‘widespread common’ and localised abundant categories. The highest numbers of species in the ‘localized-abundant’ category are restricted to Western Cape and Natal coasts, and northeastern Mpumalanga and Gauteng (Table 1). A list of 84 emerging invaders identified in South Africa was also presented; a majority (60%) of these species have been listed by the regulations under the Conservation of Agricultural Resources Act (CARA). Emerging invaders account for approximately 2500 records, or 5%, of the SAPIA database, and those species added from other sources [81, 82] and expert knowledge. Almost 20% of the emerging species are classified as riparian species according to expert opinion. A further 17% of these species are estimated to have the potential of expanding over a large part of the country if unmanaged (categories ‘large habitat–large propagule pool’, ‘large habitat–moderate propagule pool’ and ‘large habitat–small propagule pool’), and almost 80% of species falling in these categories have been afforded legal status [80]. These species are distributed along the eastern coast and northeastern interior, but have not yet been recorded in the Northern Cape and Western Cape.

Most of the emerging invaders (61%) are estimated to have a moderate amount of invadable habitat available within South Africa (categories ‘moderate habitat–large propagule pool’ and ‘moderate habitat– moderate propagule pool’). These categories show a slight difference in species distribution; distribution patterns of the ‘moderate habitat–large propagule pool’ category are similar to the ‘localized–abundant’ category of major weeds, whilst distribution patterns for the ‘moderate habitat-moderate propagule pool’ category show a lower incidence of fynbos invaders. The emerging invaders that are estimated to have a small amount of invadable habitat available but a large current propagule pool size (Table 2) show a very similar distribution pattern to the species which fall into the ‘moderate habitat–large propagule pool’ category.

Range-abundance	Scientific name	Common name	No of grids-cells	%Grid-cells abundant	Riparian or landscape	CARA category
Very widespread-abundant	<i>Acacia mearnsii</i>	Black wattle	432	28	Both	2
	<i>Poplars alba/canescens</i>	White and grey poplars	557	20	Riparian	2
	<i>Prosopis glandulosa</i> var. <i>Torreyana/velutina</i>	Honey mesquite/ prosopis	453	15	Both	2
Very widespread-common	<i>Agave americana</i>	American agave	433	1	Landscape	Proposed
	<i>Arundo donax</i>	Giant reed	377	14	Riparian	proposed
	<i>Eucalyptus spp.</i>	Gum trees	506	4	Both	1
	<i>Melia azedarach</i>	Seringa	558	7	Both	
	<i>Nicotiana glauca</i>	Wild tobacco	396	3	Both	3
	<i>Opuntia ficus-indica</i>	Sweet prickly pear	863	4	Landscape	1
	<i>Ricinus communis</i>	Castor-oil plant	471	7	Riparian	2
	<i>Salix babylonica</i>	Weeping willow	475	12	Riparian	2
Widespread-abundant	<i>Acacia cyclops</i>	Red eye	167	29	Both	2
	<i>Acacia dealbata</i>	Silver wattle	256	24	Riparian	1/2
	<i>Acacia longifolia</i>	Long-leaved wattle	95	24	Both	1
	<i>Acacia saligna</i>	Port Jackson willow	160	28	Both	2
	<i>Ageratina adenophora</i>	Crofton weed	11	19	Riparian	1
	<i>Ageratum colyzoides/houstonianum</i>	Invading ageratum	74	26	Riparian	1
	<i>Argemone mexicana</i>	Yello-flowered Mexican poppy	29	18	Riparian	1
	<i>Atriplex lindleyi spp. inflata</i>	Sponge-fruit saltbush	164	43	Landscape	3
	<i>Azolla filiculoides</i>	Red water fern	206	36	Riparian	1
	<i>Caesalpina decapetala</i>	Mauritius thorn	128	19	Both	1
	<i>Campuloclinium macrocephalum</i>	Pompom weed	17	25	Both	1
	<i>Cardiospermum grandiflorum/halicacabum</i>	Balloon vines	63	22	Both	1
	<i>Cestrum aurantiacum/laevigatum</i>	Inkberry	80	24	Both	1
	<i>Chromolaena odorata</i>	Triffid weed	96	36	Both	1
	<i>Eichlomia crassipes</i>	Water hyacinth	95	22	Riparian	1
	<i>Lantana camara</i>	Lantana	261	27	Both	1
<i>Pinus pinaster</i>	Cluster pine	86	26	Landscape	2	

	<i>Psidium guajava</i>	Guava	167	17	Both	2
	<i>Rubus cuneifolius</i>	American bramble	75	34	Both	1
	<i>Rubus fruticosus</i>	European blackberry	89	20	Both	2
	<i>Salix fragilis</i>	Crack willow	75	22	Riparian	2
	<i>Solanum mauritianum</i>	Bugweed	268	21	Both	1
Widespread- common	<i>Acacia decurrens</i>	Green wattle	101	21	Both	2
	<i>Acacia melanoxylon</i>	Australian blackwood	138	15	Both	2
	<i>Achyranthes aspera</i>	Burweed	77	4	Both	1
	<i>Ailanthus altissima</i>	Tree-of-heaven	32	5	Both	3
	<i>Anredera cordifolia</i>	Bridal wreath	24	8	Both	1
	<i>Araujia sericifera</i>	Moth catcher	36	2	Both	1
	<i>Atriplex nummularia</i> spp. <i>nummularia</i>	Old-man saltbush	173	7	Both	2
	<i>Bidens formosa</i>	Cosmos	48	11	Riparian	
	<i>Cardiospermum halicacaburn</i>	Heart pea	30	0	Riparian	
	<i>Casuarina equisetifolia</i>	Horsetail tree	24	3	Both	2
	<i>Cereus jamacaru</i>	Queen of the night	127	9	Landscape	1
	<i>Conyza bonariensis</i>	Flax-leaf fleabane	5	0	Riparian	
	<i>Crotalaria agatiflora</i> subsp. <i>imperialis</i>	Bird flower	18	0	Both	Proposed
	<i>Cuscuta campestris</i>	Common dodder	82	1	Both	1
	<i>Datura</i> spp (<i>D. Ferox</i> / <i>D. Innoxia</i> / <i>D. Stramonium</i>)	Thorn apples	84	1	Riparian	1
	<i>Echium plantagineum/vulgare</i>	Patterson's curse/blue echium	44	14	Both	1
	<i>Eucalytus camaldulensis</i>	Red river gum	123	15	Riparian	2
	<i>Hakea sericea</i>	Silky hakea	78	12	Landscape	1
	<i>Ipomoea alba</i>	Moonflower	23	3	Riparian	1
	<i>Ipomoea indica/purpurea</i>	Morning glories	98	8	Both	1
<i>Jacaranda mimosifolia</i>	Jacaranda	201	6	Both	3	
<i>Mirabilis jalapa</i>	Four-o'clock	7	0	Landscape	Proposed	
Widespread- common	<i>Morus alba</i>	White or common mulberry	130	4	Riparian	3
	<i>Opuntia aurantiaca</i>	Jointed cactus	61	5	Landscape	1
	<i>Opuntia imbricata</i>	Imbricate cactus	131	10	Landscape	1
	<i>Opuntia monacantha</i>	Cochineal prickly pear	48	1	Both	1
	<i>Opuntia robusta</i>	Blue-leaf cactus	225	1	Landscape	

	<i>Opuntia stricta</i>	Australian pest pear	108	10	Landscape	1
	<i>Pinus halepensis</i>	Aleppo pine	85	3	Landscape	2
	<i>Pinus patula</i>	Patula pine	90	12	Both	2
	<i>Pinus radiata</i>	Radiata pine	71	12	Landscape	2
	<i>Pinus spp.</i>	Pine trees	126	9	Landscape	
	<i>Pyracantha angustifolia</i>	Yellow fire thorn	143	1	Both	3
	<i>Robinia pseudoacacia</i>	Black locus	110	9	Both	2
	<i>Schinus molle</i>	Pepper tree	232	1	Both	Proposed
	<i>Senna didymobotrya</i>	Peanut butter cassia	142	13	Both	3
	<i>Senna occidentalis</i>	Wild coffee	56	8	Both	
	<i>Sesbania punicea</i>	Red sesbania	325	13	Riparian	1
	<i>Solanum seaforthianum</i>	Potato creeper	33	7	Both	1
	<i>Solanum sisymbriifolium</i>	Dense-thorned bitter apple	40	6	Both	1
	<i>Sorghum halepense</i>	Johnson grass	44	4	Riparian	2
	<i>Tamarix spp. (T. chinensis/T. ramosissima)</i>	Tamarisk	92	4	Riparian	1/3
	<i>Verbena bonariensis</i>	Purple top	58	5	Riparian	
	<i>Verbena tenuisecta</i>	Fine-leaved verbena	14	4	riparian	
	<i>Xanthium strumarium</i>	Large cocklebur	151	12	Both	1
	<i>Zinnia peruviana</i>	Redstar Zinnia	4	0	Both	
Widespread-scarce	<i>Acacia baileyana</i>	Bailey's wattle	87	0	Both	3
	<i>Populus nigra var. italica</i>	Lombardy poplar	90	0	Riparian	Proposed
Localized-abundant	<i>Acacia pycnantha</i>	Golden wattle	35	25	Landscape	1
	<i>Albizia lebbeck</i>	Lebbeck tree	5	33	No data	1
	<i>Azolla pinnata var. imbricata</i>	Mosquito fern	3	25	Riparian	
	<i>Colocasia esculenta</i>	Elephant's ear	10	21	Riparian	
	<i>Echinopsis spachiana</i>	Torch cactus	57	3	Landscape	1
	<i>Eucalyptus lehmannii</i>	Spider gum	41	13	Landscape	1/2
	<i>Flaveria bidentis</i>	Smelter's bush	19	26	Riparian	
	<i>Hakea drupacea</i>	Sweet hakea	28	7	Landscape	1
	<i>Hakea gibbosa</i>	Rock hakea	18	11	Landscape	1
	<i>Harrisia martinii</i>	Moon cactus	21	43	Both	1
	<i>Hedychium coccineum</i>	Red ginger lily	3	20	Riparian	1
	<i>Hedychium flavescens</i>	Yellow ginger lily	5	40	Both	1
	<i>Hedychium spp.</i>	Ginger lilies	7	25	Riparian	1

<i>Helianthus annuus</i>	Sunflower	5	17	No data	
<i>Leptospermum laevigatum</i>	Australian myrtle	38	30	Landscape	1
<i>Ligustrum vulgare</i>	Common privet	3	20	Riparian	3
<i>Lilium formosanum</i>	Formosa lily	16	21	Landscape	3
<i>Litsea glutinosa</i>	Indian laurel	8	44	Both	1
<i>Macfadyena unguis-cati</i>	Cat's claw creeper	27	27	Both	1
<i>Melilotus alba</i>	White sweet clover	15	40	Riparian	
<i>Metrosideros excelsa</i>	New Zealand bottlebrush	2	25	Riparian	3
<i>Myriophyllum aquaticum</i>	Parrot's feather	48	19	Riparian	1
<i>Nassella trichotoma</i>	Nassella tussock	12	21	Landscape	1
<i>Nerium oleander</i>	Oleander	24	6	Riparian	1
<i>Opuntia fulgida</i>	Chainfruit-cholla/rosea cactus	11	17	Landscape	1
<i>Opuntia lindheimeri/Opuntia engelmannii var. linderheimeri</i>	Small round-leaved prickly pear	11	21	Landscape	1
<i>Paraserianthes lophantha</i>	Stinkbean	54	10	Both	1
<i>Parthenium hysterophorus</i>	Parthenium weed	24	37	Riparian	1
<i>Paspalum dilatatum</i>	Common Paspalum	6	33	Riparian	
<i>Pennisetum villosum</i>	Feathertop	22	21	Landscape	1
<i>Pinus elliottii</i>	Slash pine	34	15	Landscape	2
<i>Pistia stratiotes</i>	Water lettuce	27	17	Riparian	1
<i>Pittosporum undulatum</i>	Australian cheesewood	3	0	Both	1
<i>Rumex usambarensis</i>	Rumex	4	20	Landscape	
<i>Salvinia molesta</i>	Salvinia	33	20	Riparian	1
<i>Schinus terebinthifolius</i>	Brazilian pepper tree	32	16	Both	1

N.B: Major invaders grouped according to categories. 'No. grid-cells' is the number of grid-cells where the species has been recorded in the Southern African Plant Invaders Atlas (SAPIA) database; '% grid-cells abundant' is the percentage of grid-cells in South Africa where the species is recorded as very abundant or abundant in the SAPIA database (note: where more than one record with the same species and abundance code occurred within a grid-cell, it was counted as one record); 'Riparian or landscape' is the classification given to a species if more than 75% of its records in the SAPIA database fell into the respective category (if neither the landscape nor riparian records exceeded 75% then the species was classified as 'both'); and 'CARA category' lists the species regulated by the Conservation of Agricultural Resources Act (Act 43 of 1983), where 1 refers to Category 1 prohibited weeds that must be controlled in all situations; 2 includes Category 2 plants with commercial value that may be planted in demarcated areas subject to a permit, provided that steps are taken to control spread; 3 includes Category 3 ornamental plants that may no longer be planted or traded, but may remain in place provided a permit is obtained and steps taken to control their spread; and 'proposed' includes those species that were proposed for listing under the Conservation of Agricultural Resources Act, but require further investigation before they can be included.

Table 1. Major invaders plants species in South Africa according to their categories (Source: [80])

3. Effects of bush encroachment and invasion on rangelands

3.1. Ecological impact

It is important to establish an understanding of ecological effects of bush encroachment on rangeland ecosystems prior to embarking on any bush encroachment intervention. Thus, the degree of invasion should be quantified to help justify the need for, and determine the type of intervention. It is fundamental to characterise invasion and these could be in terms of identification of invading species (morphology, phenology, anatomy, physiology, mode of spread), plant population density, spatial localization (along the landscape, vegetation types, soil type, water distribution), seasonal distribution, their impact on the ecosystem stability (soil cover and biodiversity) and productivity (primary and secondary). The global reviews of plant invasions suggest that the most damaging species transform ecosystems by using excessive amounts of resources, notably, water, light, and oxygen. Invading species achieve these by adding resources such as nitrogen, promoting or suppressing fire, stabilising sand movement, and/or promoting erosion, accumulating litter and accumulating or redistributing salt [82]. Such changes potentially alter the flow, availability, or quality of nutrient resources in biogeochemical cycles. They further modify trophic resources within the food web and alter physical resources such as living space or habitat, sediment, light and water. In addition, invaders are most likely to have substantial effects on ecosystems by rapidly changing the disturbance regime [36]. Thus, dense stands of alien trees and shrubs in rangelands can rapidly reduce abundance and diversity of native plants [83].

Different invading species have similar or specific effects on rangeland ecosystem dynamics. Thus, invasion of black wattle (*Acacia mearnsii*) in South African rangeland ecosystems has negative ecological impacts [8]. These impacts include reduction of surface stream flow, loss of biodiversity, increase in fire hazard, and increases in soil erosion, destabilisation of riverbanks, and loss of recreational opportunities, aesthetic costs, and nitrogen pollution and subsequently loss of grazing potential. An increase in the height and biomass of vegetation increase rainfall interception and transpiration, and decreases stream flow [8]. Alien trees and shrubs increase above ground biomass and evapotranspiration and thereby decrease both surface water runoff and ground water recharge [84]. The reduction of surface water runoff as a result of current invasions was estimated to be 3 300 mm³, which is about 7% of the national total [35], most of which is coming from the fynbos and grassland biomes [85]. The increased biomass and evapotranspiration rates associated with invasive alien plants arise because of their greater height, root depth, and senescence, compared to the native species that they replace [86]. Invasive plants may influence native ecosystems by exerting resource competition on native plants to altering fire dynamics [87]. Thus, the increased biomass that accompanies plant invasions also result in more intense fires [8, 36, 70] due to an accumulation of fuel loads. On the other hand, the dense stands of invasive trees hamper access for fire management

purposes [36], which makes it difficult for fire control in rangelands. The increase in fire intensity due to accumulation of sufficient fuel load subsequently damages vegetation and soil [70], which in turn leads to excessive soil erosion due to soil water repellency caused by fire [36].

Therefore, it suffices to indicate that the alien invasive plants reduce the functional capacity of rangeland ecosystems such as support for livestock and wildlife [36, 70]. This is among others due to competition between invasive plants and grasses that are important for grazing. This competition leads to reduction on performance of a number of ecosystem functions such as grass cover, which subsequently contributes to loss of grazing potential [36]. There is also a significant loss of biodiversity due to competition [70], resulting from the displacement of species-rich indigenous plant communities by single-species stands, and disruption of important ecosystem processes [8]. On the other hand, invasion of riverbanks causes deep channelling followed by slumping during floods and that result in destabilized riverbanks. Subsequently, the invasion along the riverbanks leads to loss of recreational opportunities due to reduction of access for anglers, canoeists, white-water rafters, and swimmers. Invasive plants further detract from the wilderness character of many rural landscapes and conservation areas and that imposes reduction of the aesthetic value of ecosystems. An increase in soil nitrogen levels in nutrient-poor environments can make habitats unsuitable for indigenous plants and more susceptible to invasion by other species, and, in turn, reducing biodiversity.

In order to develop the effective invasion control in rangelands, it is significant to understand the mechanisms that are employed by the invader species to survive and colonise the new ecosystems. There are a number of ways through which invasive plants survive and outcompete the indigenous species in rangelands; one of the mechanisms is their ability to grow rapidly compared to indigenous plants. Thus, invasive alien plants typically grow more rapidly, often increasing the proportion of biomass contributed by alien plants. The large biomass contributed by invasive plants is composed of leaves, bark, seed, flowers, and twigs that become 'terrestrial litter' after abscission [88]. Such litter enters and is retained in water bodies where its rate of breakdown by invertebrate feeding as well as decomposition through fungal and bacterial activity differs from that of inputs from indigenous plants [89]. The often large differences in litter inputs from invasive alien plants relative to indigenous species leads to reduced decomposition rate and dramatically alters the nutrient cycle in rangeland ecosystem [90]. Additions in the biomass contributed by alien plants can increase the amount of metabolised nutrients, which in turn escalates natural eutrophication processes [91] as well as free-floating and rooted aquatic macrophyte invasions [92]. Thus, eutrophication leads to gradual changes in the plant and animal populations and the development of potentially toxic algal blooms and, therefore, a slow decline in water and habitat quality [91]. The level of impact that litter from invasive alien plants has on nutrient cycles is determined by vegetative spread, plant structure, phenology, plant water and nutrient uptake efficiency, photosynthesis type, presence of symbionts and nitrogen fixation, phosphorus content and tissue chemistry such as allelopathy [93].

Habitat-propagule pool size	Scientific name	Common name	Impact	Weediness	Biocontrol	% Weedy relatives	Combined Score	CARA category
Large-large	<i>Bromus diandrus</i>	Ripgut brome	0	2	10	5	53	
	<i>Pinus taeda</i>	Loblolly pine	10	1	10	4	87	2
	<i>Tecoma stans</i>	Yellow bells	5	1	10	3	69	1
	<i>Tipuana tipu</i>	Tipu tree	5	1	10	10	73	3
Large-moderate	<i>Celtis sinensis</i> /	Chinese nettle tree/						
	<i>Celtis occidentalis</i> /	Common hackberry/						
	<i>Celtis australis</i>	European hackberry	0	1	10	1	45	Proposed
	<i>Cytisus scoparius</i>	Scotch broom	5	5	10	4	86	1
	<i>Pennisetum purpureum</i>	Elephant grass	10	3	10	2	95	Proposed
	<i>Pereskia aculeata</i>	Pereskia	10	1	10	2	87	1
	<i>Rosa rubiginosa</i>	Eglantine	10	3	10	3	96	1
	<i>Toona ciliata</i>	Toon tree	5	1	10	2	64	3
	<i>Ulex europaeus</i>	European gorse	5	5	10	1	80	1
	Large-small	<i>Acacia paradoxa</i>	Kangaroo thorn	5	2	10	3	69
<i>Pueraria lobata</i>		Kudzu vine	5	3	10	5	76	1
<i>Triplaris americana</i>		Triplaris	5	0	10	1	62	1
Moderate-large	<i>Acacia elata</i>	Peppertree wattle	5	2	10	3	69	3
	<i>Acacia podalyriifolia</i>	Pearl acacia	5	1	10	3	67	3
	<i>Ardisia crenata</i>	Coralberry tree	5	1	10	0	66	1
	<i>Cinnamomum camphora</i>	Camphor tree	10	2	10	0	90	1/3
	<i>Cotoneaster franchetii</i>	Orange cotoneaster	5	2	10	1	69	3
	<i>Cotoneaster pannosus</i>	Silver-leaf cotoneaster	5	2	10	1	69	3
	<i>Eucalyptus cladocalyx</i>	Sugar gum	5	1	10	2	68	2
	<i>Eucalyptus saligna</i>	Saligna gum	5	1	10	2	66	
	<i>Eugenia uniflora</i>	Surinam cherry	5	2	10	0	68	1

Habitat-propagule pool size	Scientific name	Common name	Impact	Weediness	Biocontrol	% Weedy relatives	Combined Score	CARA category
	<i>Hedychium coronarium</i>	White ginger lily	10	2	10	1	87	1
	<i>Hedychium gardnerianum</i>	Kahili ginger lily	10	3	10	1	92	1
	<i>Ligustrum japonicum</i>	Japanese wax-leaved privet	5	1	10	3	66	3
	<i>Ligustrum lucidum</i>	Chinese wax-leaved privet	5	4	10	3	78	3
	<i>Ligustrum ovalifolium</i>	Californian privet	5	1	10	3	68	3
	<i>Ligustrum sinense</i>	Chinese privet	5	4	10	3	80	3
	<i>Lonicera japonica</i>	Japanese honeysuckle	5	6	10	1	83	Proposed
	<i>Myoporum serratum</i>	Manatoka	5	0	10	2	84	3
	<i>Myoporum tenuifolium</i> ssp. <i>montanum</i>	Manatoka	5	0	10	2	69	
	<i>Nephrolepis exaltata</i>	Sword fern	10	0	10	3	82	1
	<i>Pyracantha coccinea</i>	Red firethorn	5	0	10	8	61	
	<i>Spartium junceum</i>	Spanish broom	5	3	10	10	82	1
	<i>Syzygium paniculatum</i>	Australian water pear	5	0	10	0	61	
Moderate-moderate	<i>Albizia procera</i>	False lebeck	5	1	10	2	64	1
	<i>Alhagi maurorum</i>	Camelthorn bush	5	2	10	10	79	11
	<i>Anacardium occidentale</i>	Cashew nut	5	1	10	1	63	
	<i>Callistemon rigidus</i>	Sitt-leaved bottlebrush	0	1	10	1	45	Proposed
	<i>Catharanthus roseus</i>	Madagascar periwinkle	0	2	10	3	51	
	<i>Cestrum parqui</i>	Chilean cestrum	10	3	10	1	91	1
	<i>Cynodon nlemfuensis</i>	East African couch	5	2	10	10	76	

Habitat-propagule pool size	Scientific name	Common name	Impact	Weediness	Biocontrol	% Weedy relatives	Combined Score	CARA category
	<i>Cytisus monspessulanus</i>	Montpellier broom	5	0	10	4	66	1
	<i>Duranta erecta</i>	Forget-me-not	0	1	10	1	44	Proposed
	<i>Eriobotrya japonica</i>	Loquat	0	2	10	0	50	3
	<i>Ficus carica</i>	Fig	0	2	10	0	50	
	<i>Gleditsia triacanthos</i>	Honey locust	5	2	10	1	68	2
	<i>Leucaena leucocephala</i>	Leucaena	5	3	4	3	52	1
	<i>Mangifera indica</i>	Mango	0	1	10	0	46	1
	<i>Montanoa hibiscifolia</i>	Tree daisy	0	1	10	1	44	
	<i>Passiflora edulis</i>	Passion fruit	0	2	10	1	50	1
	<i>Passiflora subpeltata</i>	Granadina	0	1	10	1	46	
	<i>Physalis peruviana</i>	Cape gooseberry	0	2	10	5	54	
	<i>Phytolacca octandra</i>	Forest inkberry	0	2	10	6	55	
	<i>Pyracantha crenulata</i>	Himalayan firethorn	5	1	10	8	73	3
	<i>Senna bicapsularis</i>	Rambling cassia	5	0	10	1	62	3
	<i>Senna pendula</i> var. <i>glabrata</i>	Rambling cassia	5	2	10	1	68	3
	<i>Sesbania bispinosa</i> var. <i>bispinosa</i>	Spiny sesbania	0	0	10	4	45	
	<i>Sophora japonica</i>	Japanese pagoda tree	0	0	10	2	42	
	<i>Syzygium cumini</i>	Jambolan	5	1	10	0	66	3
	<i>Syzygium jambos</i>	Rose apple	5	1	10	0	66	3
	<i>Tithonia diversifolia</i>	Mexican sunflower	0	1	10	3	48	1
	<i>Ulmus parvifolia</i>	Chinese elm	0	0	10	5	46	
	<i>Verbena brasiliensis</i>	Slender wild verbena	0	1	10	2	45	
Riparian-large	<i>Canna indica</i>	Indian shot	5	2	10	10	79	1
	<i>Canna x generalis</i>	Garden canna	5	1	10	10	72	

Habitat-propagule pool size	Scientific name	Common name	Impact	Weediness	Biocontrol	% Weedy relatives	Combined Score	CARA category
	<i>Casuarina cunninghamiana</i>	Beefwood	5	1	10	4	69	2
	<i>Cortaderia jubata</i>	Purple Pampas	5	3	10	2	75	1
	<i>Cortaderia selloana</i>	Pampas grass	5	5	10	2	81	1
	<i>Oenothera biennis</i>	Evening primrose	5	1	10	4	67	
	<i>Populus deltoides</i>	Match poplar						Proposed
	<i>Eucalyptus microtheca</i>	Coolabah	0	0	10	2	42	
	<i>Mimosa pigra</i>	Giant sensitive plant	5	4	10	1	76	3
	<i>Myriophyllum spicatum</i>	Spiked water-milfoil	5	4	10	3	80	1
	<i>Oenothera glazioviana</i>	Evening primrose	5	2	10	4	72	
	<i>Oenothera indecora</i>	Evening primrose	5	1	10	4	68	
	<i>Oenothera jamesii</i>	Giant evening primrose	5	0	10	4	64	
	<i>Oenothera laciniata</i>	Cutleaf evening primrose	5	1	10	4	67	
	<i>Oenothera tetraptera</i>	White evening primrose	5	0	10	4	66	
	<i>Parkinsonia aculeata</i>	Jerusalem thorn	5	1	10	0	66	
Small-large	<i>Alpinia zerumbet</i>	Shell ginger	5	0	10	0	62	
	<i>Grevillea robusta</i>	Australian silky oak	5	2	10	0	67	3
	<i>Quercus robur</i>	English oak	5	1	10	1	67	

N. B: Scores for 'Impact', 'Weediness', 'Biocontrol' and 'Weedy relatives' are standardized by dividing the maximum score for that criterion and multiplying by 10. Scores for these four criteria were weighted, with 'Impact', 'Weediness' and 'Biocontrol' receiving an equal weighting of four, and 'Weedy relatives' receiving a lower weighting of one. The weighted criteria were summed to obtain the 'Combined score' for each species. 'CARA category' lists the species regulated by the Conservation of Agricultural Resources Act (Act 43 of 1983), where 1 refers to Category 1 prohibited weeds that must be controlled in all situations; 2 includes Category 2 plants with commercial value that may be planted in demarcated areas subject to a permit, provided that steps are taken to control spread; 3 includes Category 3 ornamental plants that may no longer be planted or traded, but may remain in place provided a permit is obtained and steps taken to control their spread; and 'proposed' includes those species that were proposed for listing under the Conservation of Agricultural Resources Act, but require further investigation before they can be included.

Table 2. Emerging invaders grouped according to categories (Source: [80])

The majority of invasive and/or encroaching species in rangelands is dominated by the genus *Acacia*, which is the second largest with over 900 species [70]. Australian acacias are important invaders of South African rangeland areas [94]. In the fynbos ecosystems where soil nutrients are generally poor, the invasion by nitrogen-fixing acacias increases nitrogen inputs, and subsequently leads to an increase in soil fertility. Therefore, the massive increase in soil fertility permits acacia species to propagate and outcompete indigenous species [90]. There are a number of *acacia* species found in rangelands and their ability to fix nitrogen has been widely reported; these include *Acacia cyclops*, *A. dealbata*, *A. mearnsii* and *A. saligna* [90, 95]. The groundwater on places that were invaded by *A. saligna* has shown elevated NO_3^- and NO_2^- concentrations compared to groundwater in natural ecosystems [94]. The presence of *A. saligna*, as well as the nutrient leaching that occurred after its removal, result in seasonal nitrogen concentrations that are higher than the water quality targets for domestic use ($\text{NO}_x < 6 \text{ mg/l}$) [94, 96]. Therefore, the removal of alien plants would be beneficial from both a water quantity as well as water quality perspective [94].

In natural communities, plants compete in different ways; one of these ways is chemical interactions in the form of allelopathy [87, 97]. Invasive plants interfere with other plants by releasing allelochemicals into the environment and that negatively affects surrounding plants, thus giving the producer a competitive advantage. Invasive plants possess physiological traits that enable them to exploit ecological opportunities. The word allelopathy comes from the Latin words *allelon*, which means of each other and *pathos*, which means to suffer, which is commonly associated with the chemical inhibition of one species of plants by another [98]. Allelopathy is the process through which invasive plants such as *eucalyptus*, *Pinus*, *Chromolaena* and *Lantana* produce biochemicals that influence the growth, survival, and reproduction of indigenous species. However, it is important to note that most of the plant species naturally produce number of allelopathic substances such as monoterpenes and phenols [97]. Phenolics and volatile compounds can be released from eucalyptus foliage. These biochemicals can act as antibiotics in certain soils, possibly affecting nitrogen cycles.

Although it has not been evaluated, the impacts of allelochemicals may subsequently influence water quality through soil erosion or surface runoff processes [70]. Allelochemicals are believed to be present in almost all plant tissues such as leaves, flowers, fruits, stems, roots, rhizomes, seeds, and pollen where they may be released from plants into the environment by means of volatilization, leaching, root exudation, and decomposition of plant residues [99, 100]. Invasive plants use the mechanism of allelopathy to outcompete other plants [87]. Allelochemicals can be found present in litter and on the soil surface where plants grow. Rain assists with the leaching of allelopathic substances into the soil, where they may affect the germination and growth of other plants [97, 101]. Allelopathic substances might play a role in shaping plant community structure in semi-arid and arid environments [97]. Thus, allelopathic substances inhibit plant growth depending on the concentration, leachability, season, and age of the plants [101]. Phytotoxins can persist in the soil and litter layer for long after allelopathic plants senesce, thereby reducing the establishment potential of an area. Allelopathic substances can be present in the soil and often determined by a number of important factors [97]. These factors include the density at which the leaves fall, the rate at which this material decomposes,

the distance from other plants and, finally, rainfall [101, 102, 103]. Phenolics signify the main allelopathic compounds that inhibit seed germination, plant growth and other physiological processes that result in changes of floristic composition within a plant community.

Competition between plants can lead to the allelopathic inhibition of germination or growth via phytotoxic chemical releases, which are caused by competing species. However, allelopathy can be extremely difficult to demonstrate in the field due to difficulties in differentiating allelopathic effects from resource competition [87, 99]. Allelochemical compounds are in fact released into the soil and accumulate to levels of toxicity, which leads to inhibition of germination [100]. Allelochemicals released by invasive plants may affect native plant survival and production in a number of ways. These include the modification of the soil microbiota [74, 104], and enhancement of growth of beneficial microbes in their rhizosphere leading to an establishment of positive feedbacks that can contribute to the decrease of native biodiversity [74]. Allelochemicals are further known to inhibit absorption of ions [105]. Other than allelopathic effects, invasive plants exert competition of resource especially through light [87]. Therefore, allelopathy and resource competition operate simultaneously influencing each other and, in the meantime, they are influencing plant community structure [106].

Allelochemicals, as soon as released into the soil, may inhibit germination, shoot, and root growth of other plants, which will affect nutrient uptake thereby destroying the plant's usable source of nutrients [107]. Allelopathy of invasive plants delays the germination and growth of seedlings of other species and eventually hinders their growth completely. Therefore, degree of inhibition due to allelopathy is largely dependent on the concentration of the extracts and, to a lesser extent, on the species from which they were derived [101, 108]. The effects of allelopathy on germination and growth of plants occur through a variety of mechanisms including reduced mitotic activity in roots and hypocotyls, suppressed hormone activity, reduced rate of ion uptake, inhibited photosynthesis, and respiration, inhibit protein formation, decreased permeability of cell membranes and/or inhibition of enzyme action [97]. Plants that germinate at slower rates are often smaller; thereby, this may seriously influence their chances of competing with neighbouring plants for resources such as water [109]. Indirectly, allelopathic effects of invasive species on germination and growth of native species determine their competitive ability against them [97]. The roots of *Aloe ferox* have allelopathic inhibition on tomato seed germination [97]. Accumulation of allelochemicals in the rhizosphere because of root and microbial exudates and/or metabolism may affect the germination. However, under arid conditions germination will be less affected since microbial activities are very low due to low availability of soil moisture [101]. The effects of allelochemicals on the root growth are due to cell division destruction [105]. *L. maackii* also exudes allelopathic compounds from its leaves or roots that inhibit germination and growth of species that grow on the same site [87]. Allelochemicals could be found on any part of the plant; however, the concentration varies with plant parts. The leaf extracts of *L. maackii* appeared to have a more negative effect on seed germination than root extracts [87]. Generally, leaf extract concentrations have a stronger effect on germination of seeds of other plants [87]. However, it is important to note that allelopathic chemicals from one plant can hinder germination of seeds of the same plant. For example, chenopod seed germination can also be inhibited by extracts generated from its leaves [97].

However, all extracts, except the one obtained from the leaves of *E. tomentosa* significantly inhibited the germination of lettuce seed and appeared to stunt the growth of roots and shoots of germinants [97].

There are different allelochemicals exuded by invasive plants; these may have direct and indirect effects on germination and establishment of native species. However, phenolics are widely recognized for their allelopathic potential in plants, and can be found in a variety of tissues. Phytotoxic activity of allelochemicals in soil has been considered as plant-to-plant interaction, which is mediated by chemicals released from the plants [99]. Indirect effects of allelochemicals include its influence on the availability of nutrients in the soil, which may cause changes in soil chemical characteristics [110]. Allelochemicals might inhibit the growth of nitrifying bacteria, which would decrease N-availability at the plant level [111]. Additionally, chemical compounds produced in the process of litter decomposition are inhibitory for both heterotrophic and autotrophic bacteria and fungi [110,111] and, thus, rates of mineralization may be reduced. Allelochemicals such as phenolic acids are considered to have an important influence on nutrient cycling in terrestrial ecosystems [110]. The allelochemicals can produce some changes in the resource exploitation competition in such way that allelochemicals affect the mycorrhizae that allow the plant to absorb the nutrients, which leads to decrease in the soil productivity [106, 112]. Soil microorganisms are affected by root exudates that eventually affect other plant roots. Some chaparral species produce substances, which accumulate on the soil surface and make the soil less wettable [111]. The allelochemicals affect availability and accumulation of inorganic ions, although their activities are influenced by ecological factors such as nutrient limitation, light regime and soil moisture deficiency [106].

Allelochemicals, such as phenolics and terpenoids, play an important role in the inhibition of nitrification and, thus, influence soil productivity of a plant community [113]. Thus, any influence on nutrient dynamics may ultimately affect the growth of plants in the community, which will lead to the increase of invasive plants. Reduced soil fertility may enhance the production of allelochemicals from invasive plants [106]. The addition of plant litter to soil may influence nutrient mobilization and soil pH, which can further influence nutrient immobilization and microbial activity [114]. Therefore, litter can alter the chemistry of the soil in such a way that it inhibits germination of other plants [106]. Chemicals released into the environment by a plant may not necessarily have direct effects on community structure but abiotic soil factors can influence these chemicals. Many phenolic acids have potential to influence microbial population, cause a shift in the microbial community, and eventually affect soil productivity of the area [106]. The soil microflora is directly responsible for decomposition and mineralisation processes and soil fauna is of considerable importance in regulating these processes through influencing the growth and activity of soil microbes [115]. Allelochemicals exuded from roots of invasive plants and residue decomposition play an important role in inhibiting plant pathogens particularly those borne in soil [116]. However, amended soils with allelopathic residues tend to be rich in organic matter [117]. Electrical conductivity (EC) of the amended soils increased as compared to the control and all nutrients were significantly more [117]. Although, earlier reports show that inclusion of plant litter, in addition to releasing putative phytotoxins into the soil medium, alters the soil nutrient dynamics and, thus, affects

the plant growth [106, 112, 116]. A similar increase in electrical conductivity of the soil incorporated with residues of allelopathic plants was reported [118]. In fact, the behaviour of the allelopathic compounds present in soil remains unclear [119].

The modes of release of the allelopathic compounds are not specific because they vary from plant to plant [120]. Thus, allelochemicals are released into the environment by root exudation, leaching from aboveground parts, volatilisation, and decomposition of plant material and ultimately enter into the soil [99, 110, 121]. Therefore, allelochemicals may reach other plants through transport such as root exudates into the soil and may induce the inhibitory activity on the other plants. The behaviour of allelochemicals in soil is run by the physicochemical properties including soil organic matter and organisms [99]. The model that has assumptions such as "allelochemicals are released into the soil from living plants and degraded into non-allelopathic substances was developed. Therefore, rate of the release is proportional to the amount of allelochemicals in living plants and rate of allelochemicals degradation is proportional to the amount of allelochemicals released [121]. However, the soil microorganisms were also reported to produce and release allelochemicals [112]. The release of allelochemicals by mature shrubs may inhibit plant germination, survival or growth [111]. Allelopathic content of a plant varies according to its maturity [122]. Allelopathic compounds released from different plant parts can be either released continuously within specific periods such as specific developmental stages or influenced by external factors such as precipitation [123]. The synthesis and exudation of allelochemicals via roots is usually enhanced by stress conditions that the plant encounters such as extreme temperature, drought, and ultraviolet exposure [124].

The visible effects of allelopathy frequently observed are inhibited or delayed seed germination or reduced seedling growth. The diversity of structure among allelochemicals suggests that they have no common mode of action [110]. Plant exudates can also have an indirect effect on the surrounding environment and reduce neighbouring plant germination or growth, independent of toxicity [111]. Allelopathic activities are more pronounced when allelopathic potential species grow under water stress [125]. Phenolic acids that were tested had a similar mode of action such as inhibition of nutrient uptake by roots of plants [126]. In most cases, various allelochemicals take action as growth regulators by inhibiting growth and changing development [112]. The common mode of action of allelochemicals is quite related to the membrane destruction [126]. It was discovered that allelochemicals affect plants on cell division, cell elongation, cell structure, cell wall, ultrastructure of the cell [112, 127]. Phenolic allelochemicals can also lead to increased cell membrane permeability; cell contents spill which lead to the increase of lipid peroxidation, and eventually, slow growth or death of plant tissue occurs [112, 126, 127]. Furthermore, nutrient uptake can be affected negatively by allelochemicals. This occurs when these allelochemicals inhibit nutrient absorption of the plant [127]. The mode of action of benzoic acid involved the inhibition of nutrient uptake by plant roots, which resulted in growth inhibition [126]. The radicle elongation was significantly reduced by the extract of leaves, and leaves and stem at the three concentrations of *Acacia mearnsii*, which signifies that *A. mearnsii* has allelopathic potential [128]. The impact of allelochemicals also have

been observed on the respiration of the plants which affect oxygen absorption capacity [127], eventually inhibit photosynthesis by reducing the chlorophyll content which affect photosynthesis rate [98, 112, 126]. There is an inhibition of the activity of hydroxyphenylpyruvate dioxygenase (HPPD) enzyme due to isoxaflutole, which results in the inhibition of meristmatic tissue, which leads to inhibition of shoot growth [126]. Therefore, the modes of action of most allelochemicals and phytotoxins are complex and are not clearly understood [126].

The active compound or compounds must be isolated in an amount adequate for identification and for further characterisation in bioassays [110]. Screening of fractions of plant extracts or leachates for their effects on seed germination of various plant species are frequently used to identify phytotoxic compounds [110]. The identification of an active phytotoxic compound from a suspected allelopathic plant does not establish that this is the only compound involved in allelopathy. The release of allelochemicals of different chemical classes from allelopathic plant species has been documented including tannins, cyanogenic glycosides, several flavonoids and phenolic acids [129]. The most clearly identified compounds can be divided into four groups: phenolic acids, hydroxamic acids, alkaloids, and quinones. In the study of allelopathy, plants are identified based on the allelochemical release [120]. Most studies utilized some parts of the plants such as roots, leaves and leaves plus stem to establish the existence of allelochemicals on the identified plants [107, 128].

3.2. Economic impacts

Rangelands contribute to the economy of Southern Africa in a number of ways. They provide agricultural commodities that can be valued in the market such as wool, meat, milk etc. These are the major source of forage for grazing animal which in turn influence animal production. Rangelands further provide benefits that, are not directly related to the agricultural sector, such as wildlife habitat, however, have an impact on the economy through activities that make use of them [130]. Increases in the density of woody plants worldwide are a major threat to livestock production [13, 131], and rangeland biodiversity. Invasive species pose problems for managers of rangelands because they reduce the land's usefulness for grazing activities. In addition, they interfere with other non-agricultural functions that rangelands provide, such as acreage of wildlife habitat and watershed quality. Therefore, in order to realise the impact of invasion on rangelands, it is important to understand the total economic loss that invasive plant infestations create on the economy in relation to both its agricultural and non-agricultural products of the ecosystems [130].

Economic impact of invasive species could be defined as the product of a species' range, abundance and per capita [36, 80, 132]. Although the invasive plants have an ecological implication they also have some economic implications; these could be either positive or negative. Species such as *Acacia mearnsii* (Black wattle) are highly invasive and have spread over an area of almost 2.5 million ha in South Africa [133]. It has significant negative impacts on water resources, biodiversity, and the stability and integrity of riparian ecosystems [8]. These two features, a commercial value on the one hand, and an invasive, damaging ability on the other, give rise to a classic conflict of interests, where the benefits accrue to a number

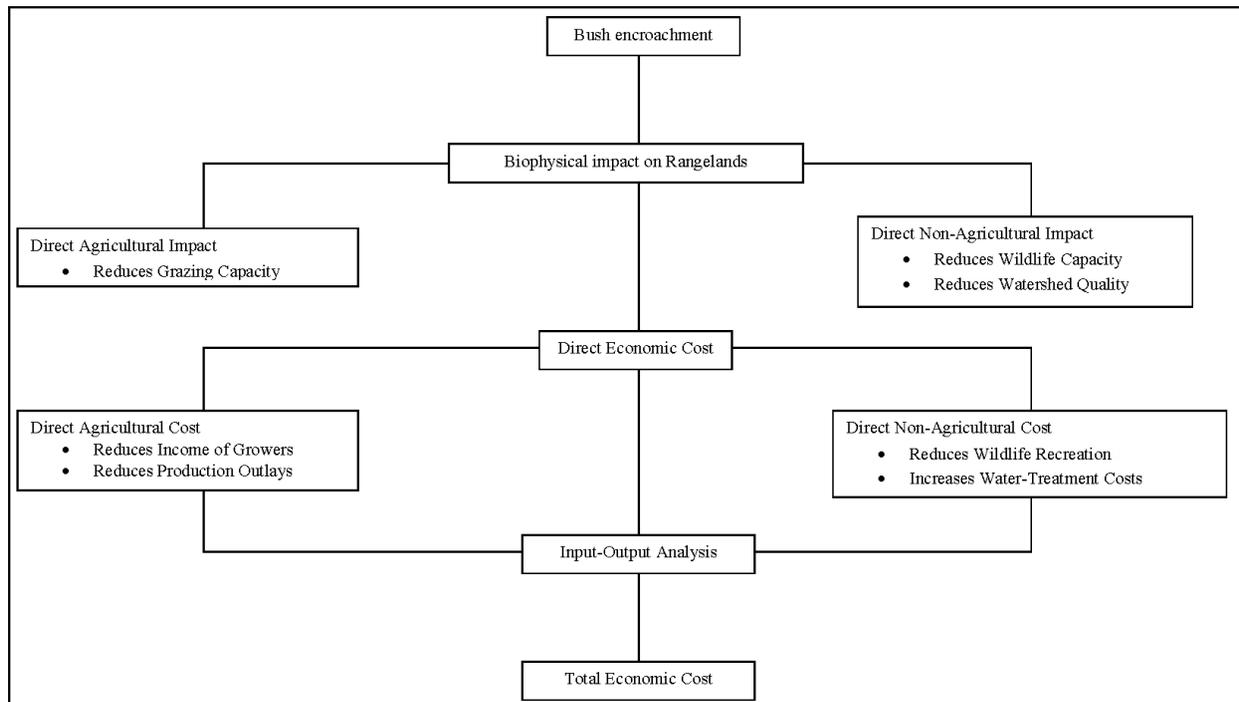


Figure 1. Hypothetical flow chart indicating economic impact of bush encroachment in rangelands (Source [80]).

of people, while the society at large bears the external costs. Furthermore, there are larger reductions of water resulting from the presence and densities of invasive plants. Thus, the potential water reductions in South Africa would be more than 8 times greater if invasive alien plants were to occupy the full extent of their potential range [85]. These invasions come at a significant cost to the economy, estimated at about R6.5 billion per annum, which is about 0.3% of South Africa's GDP of around R 2 000 billion, and with potential to rise to > 5% of GDP if invasive plants were to be allowed to invade all of the suitable habitat [134]. Economic of bush encroachment in rangelands can be divided into agricultural and non-agricultural, direct and indirect impacts, and, further, into primary and secondary impacts (Figure 1). Economic impacts of plant invasions may be related to a decline in cattle carrying capacity (agricultural impact), wildlife carrying capacity, and watershed quality (non-agricultural impacts). Reductions in cattle grazing outlays may account for the direct agricultural costs. In addition, economic impacts may be estimated as reductions in wildland-associated recreation expenditures and increases in expenditures to mitigate damages from runoff and soil erosion to account for the non-agricultural losses. These estimated losses are incorporated into an input-output model of economy to compute total (direct plus secondary) economic costs incurred due to the invasion of noxious weeds [130]. Secondary economic effects of bush encroachment include indirect and induced losses on the economy. Indirect losses are linked to economic sectors not necessarily directly affected by the infestations, but these sectors supply inputs needed by directly affected industries. Induced effects represent changes in household spending patterns, caused by changes in employment that the direct and indirect effects generate.

4. Management of rangelands for bush encroachment and invasion

4.1. Bush encroachment control

Bush encroachment forms dense infestations that rapidly deplete soil moisture, preventing the establishment of other species. As it displaces native vegetation, it reduces wildlife habitat and ecosystem diversity, and suppresses production of nutritious, palatable forage for wildlife and livestock, which leads to a reduction in grazing and wildlife carrying capacity. Soil and water conservation benefits of the regions rangelands also decline; watershed quality declines in areas where the weeds have advanced [135].

Bush encroachment is considered a threat to forage production, which is the feed for the grazing livestock [42]. The threat to the pastoral economy by bush encroachment and invasion is often the main reason for the control of bush encroachment [136]. Bush encroachment control is a disturbance that reduces the threat of bush encroachment by disrupting the invasive woody plant community structure through transformations of biotic environments and habitat conditions in which colonization of the disturbed microhabitat takes place. Bush control methods shift the rangeland vegetation from dominance by woody vegetation to dominance by herbaceous vegetation. This control of the bush is aimed at creating suitable habitat for grazers [137, 138]. Thus, forage production of herbaceous vegetation increases with reduction of woody species. The principle of bush encroachment control is based on the ability of the control method to shift the competition between desired and undesired species. Encroaching species have the higher competitive ability over the native species, which is why they colonise. They build up this competitive advantage by modifying the environment in such a way that growing conditions will suit their needs through a number of ways. These include release of chemical substances that suppresses germination and growth of their competitors (Allelopathy) and modification of soil fertility in the case of acacias through higher nitrogen inputs, which in turn favours their growth. Encroaching species also impose competition for light and through shading and subsequently growth for native species becomes negatively affected. There is also a competition for soil moisture and soil nutrient; in this manner, most of the invasive plants win because of their deeper root systems. Other invasive species produce large numbers of seeds, which normally are dispersed faster, have a shorter dormant time before germination, and colonise. Invasive plants use one or a combination of these mechanisms for survival. Therefore, bush encroachment control reduces the ability of invasive plants to exhibit these survival mechanisms. The use of selective herbicides is aimed at reducing the competitive ability of invasive species through killing them and, in that, species that are not affected by this herbicide gain an advantage. Mechanical methods such as hand clearing targets unwanted plants and create a competitive space for desired plants, thus, without this clearing the invasive species are more competitive. Use of fire to control invasive woody plants is justified by the fact that when woody plants are burned they do not recover or they take a longer time to recover which gives the herbaceous species time to grow with minimal or no competition. In the biological control method, use of herbivores such as goats to selectively-browse on the encroaching species or use of invertebrates that feed on the seed of invading species also reduces the competition against native plants.

It is important to mention that the shift towards herbaceous species dominance, in turn, may induce shifts in herbaceous species that tolerate bush cover and such species might decline in numbers [139]. The changes could cause partial or total reduction of plant biomass [140] by shifting vegetation structure and composition [141]. Furthermore, disturbance can produce changes in the life history strategies of individual species in response to intensities of disturbance forces [140] and the created micro-environmental conditions [142]. Although livestock-forage production of rangelands may support removal of encroaching species to enhance forage production, it is important to note that bush encroachment control methods are management systems [137] that might have varied policy implications for bush control [143]. Therefore, understanding the potential role of different bush encroachment control methods for promoting herbaceous species composition requires recognition of the objectives of resource users and policymakers [144]. Thus, the intended ecosystem status is dependant of the functional characteristics of such an ecosystem.

4.2. Bush encroachment management methods

4.2.1. Rangeland management practices

Grazing management entails management of livestock and vegetation resources. The main livestock decisions made by farmers both in the commercial and communal areas are concerned with livestock type, number and seasonal pattern of movement [145]. Commercial and communal livestock farming are generally regarded as the rangeland management systems and they are distinct in grazing management practices. Thus, communal grazing areas are generally characterised by continuous grazing, which is perceived by most of the scientists to be the root cause of the often-reported land degradation in this system. On the other hand, commercial livestock farming is characterised by structured and objective grazing management practices such as assigning the correct livestock units in proportion to the carrying capacity of the land. These would be done in rotation to give vegetation in grazed areas time to recover such that the rested areas can be grazed again. Understanding the dynamics of bush encroachment in relation to rangeland management systems over a broad range of environments is essential for sustainable management of rangeland ecosystems [146]. Although rangelands are complex ecosystems varying at multiple scales in time and space [147, 148], most management usually intends to maintain or enhance livestock production by reducing plant community variability in space and time [149, 150]. This is usually accomplished by promoting spatially uniform dominance of a few productive forage species. Although it is generally believed that improper grazing practices leading to overgrazing are responsible for bush encroachment, it is not attributed to heavy grazing alone, but is strongly influenced by seasonality, which is a characteristic of arid and semi-arid environments [42]. In combination with seasonality, the ban on fire and exclusion of browsing animals such as goats and camels may also contribute to the invasion of bush encroachment.

Rangeland management practices, particularly fire suppression and overgrazing, have been reported to increase the proportion of some native species [70]. These natives can reduce overall forage quality or quantity (e.g. *Juniperus* spp., *Artemisia tridentata*, and *Gutierrezia* spp.)

or poison livestock (e.g. *Delphinium* spp., *Astragalus* spp., and *Amsinckia menziesii* var. *intermedia*). One of the challenges of managing invasive species is that there is no particular life cycle typical to noxious weeds of rangelands reported [151]. Thus, noxious rangeland weeds can be annuals (e.g. *Centaurea solstitialis*, *Crupina vulgaris*, *Bromus tectorum*), biennials (e.g. *Carduus nutans*, *Conium maculatum*, *Onopordum acanthium*), long-lived herbaceous perennials (e.g. *Convolvulus arvensis*, *Centaurea maculosa*, *Cirsium arvense*), shrubs (e.g. *Gutierrezia* spp., *Artemisia tridentata*), or trees (e.g. *Juniperus* spp., *Prosopis glandulosa*). Although several plant families represent these species, the largest number of noxious species belongs to the Asteraceae (sunflower) family.

Effective rangeland management requires sound ecological data about the land being managed; however, obtaining such data is not sufficient to ensure the implementation of restoration practices by land users. Thus, rational decisions at the farm or community, regional and national levels, depend on researchers providing not only ecologically sound but also economical, effective alternatives for land use [152]. In addition, because natural resource depletion and recovery compound over time, it is necessary to assess the sustainability of management alternatives over decadal periods [153]. Furthermore, to determine the true advantage of restoration management, it is necessary to compare the benefits of changing management practices with the cost of not changing current practices, which, rather than maintaining productivity, may lead to loss of production through shifts in plant species composition, accelerated soil erosion, and loss of biodiversity.

4.2.2. Chemical — Herbicides

Chemical control methods are usually expensive to apply and should be considered only under specific circumstances. Thus, their nature are suited primarily to the initial thinning of bush at high density, where there is poor fuel load to support fire, where trees are above the browse line, where the bush is unacceptable to animals and where the herbicide is intended to selectively kill a specific plant [154]. However, herbicides can sometimes be used in follow up operations such as after fire where there is a need for pre-emergence herbicide application intended to kill the seedlings of a target plant in soil. Herbicides have been applied extensively on rangelands to reduce forbs that were considered undesirable, which have been assumed to lead to an increase in grass production and ultimately to an improvement in livestock performance [155]. Herbicides are the primary method of weed control in most rangeland systems [151]. In South Africa, there is a considerable effort taken by the government to address the negative impact of alien invading species on the natural and environmental resources of the country [8].

Herbicides vary in their chemical properties, that make them vary more with their mode of action under different climatic and soil conditions, and they further vary in their methods of application and their effect on the ecosystems. There are two broad groups of herbicides used in rangelands. The first type is composed of the herbicides that are applied on the soil surface and are absorbed by the roots; these are the herbicides that are based on tebuthiuron, ethidimuron or bromacil as their active ingredient [154]. The second group of herbicides is sprayed onto the plant and absorbed directly by the foliage and other above ground parts of the plants;

these herbicides have picloram as the active ingredient. The second group may also have ingredients such as 2, 4-D and 2, 4, 5-T. Soil applied formulations are marketed as granules, wettable powders or as liquid with active ingredients ranging in concentration between 20% and 70%. Granular products can be applied by hand, with some suited to aerial application. Wettable and liquid products are mixed with water and applied on the soil surface adjacent to the stem of the plant. The application rates of soil formulations vary according to clay content, organic matter and pH of the soil. These herbicides remain in the soil inactive until it rains such that the active ingredient can dissolve in water so that the roots can absorb it. Herbicides applied directly to the plant normally have an oil or water base and are applied to either the stem or the leaves of the plant.

In South Africa, of particular note are herbicides containing bromacil (5 – Bromo -3- sec – butyl – 6 – methyluracil) as the active ingredient (a. i.) which are used to control encroaching species. These herbicides include Bushwacker SC (Enviro Weed Control Systems (Pty – Ltd), Bushwacker GG (Enviro Weed Control Systems (Pty Ltd) and Rinkals 400 PA (Dow AgroSciences LLC) e.t.c [156]. These herbicides vary primarily in their bromacil concentration, thus, Bushwacker SC contains 500 g of bromacil per litre, Bushwacker GG contains 200 g of bromacil per kilogram and Rinkals 400 PA contains 400 g of bromacil per kilogram. These herbicides are usually selective within certain application rates, environmental conditions, and methods of application. Bromacil works by interfering with the photosynthetic pathway of plants [157]. Its application is usually done just before the active growth stage of plants, thus, before the wet season stabilizes. It quickly dissolves in soil water and may stay in the soil for several years [157]. Bromacil is readily absorbed through the root system [158] and is a specific powerful mobile inhibitor of photosynthesis [159]. The target plant must be undergoing active photosynthesis for the herbicide to be effective. It inhibits photosynthesis by blocking the photosystem II reaction, thereby, preventing the conversion of sunlight into chemical energy, thus, it blocks the photosynthetic electron transport [159]. Bromacil blocks electron transport from QA to QB in the chloroplast thylakoid membranes by binding to the D-1 protein at the QB binding niche. The electrons that are blocked from passing through photosystem II are transferred through a series of reactions to other reactive toxic compounds. These compounds disrupt cell membranes and cause chloroplast swelling, membrane leakage, and ultimately cellular destruction [160]. Inhibition of photosynthesis thus results in slow starvation of the target plant and eventual death. It is translocated upward via the xylem to foliage and interferes with light-harvesting complexes [159]. In the soil, there is little adsorption of bromacil to soil colloids, therefore, it moves (leaches) through the soil and it can contaminate groundwater [157]; however, it is highly susceptible to microbial degradation [161]. When used as a selective herbicide, it can persist in the soil for one year; however, if it is applied at high concentrations, it can persist for more than one year [161].

The herbicide 2,4-D [(2, 4-dichlorophenoxy) acetic acid] is also a commonly used herbicide in the rangeland vegetation management [162]. Combined estimates of 2,4-D use annually on cropland, pastureland, and rangeland could range from 12.7 to 14.9 million kg [163]. Native and exotic dicots are primary targets of many rangeland herbicide applications [162, 164]. However, these plants also contribute key structural, vegetation, and nutritional elements to

wildlife habitat [165] and livestock diets [166]. Some forbs are foraged by animals especially during the seasons when forage is scarce. Therefore, reducing forbs with herbicide might influence ecosystems across trophic levels and potentially alter ecosystem function. Furthermore, biodiversity has been proposed as a source of stability in managed ecosystems [167, 168]. Therefore, decreasing forb diversity with the use of phenoxy herbicides like 2,4-D alters arthropod habitat and reduces arthropod diversity, which influences higher trophic levels [149,169]. The decrease in forb abundance and diversity beyond normal temporal dynamics could be detrimental to wildlife because forbs also comprise key structural, vegetative, and nutritional elements [165].

Although herbicides are considered effective in controlling weeds, they are often facing the challenge with evolution of resistant weed populations [170, 171]. Thus, depending on both the population's genetic background and ecological scenario, apart from expressing herbicide resistance, weed species adapt to herbicides by phenological changes [172, 173]. Comparisons of herbicide-resistant and susceptible biotypes have shown that populations can vary not only in morphological traits but also in developmental responses, such as relative growth rate, photosynthetic rate or germination rate [174, 175]. Adjusting seed germination time and rate has been considered as one of the potential mechanisms by which annual weeds can improve their competitive ability in agricultural scenarios [173, 176]. Hence, success of annual weed species in cropping systems may be assessed through the degree of synchronization of germination (determined by factors controlling exit from dormancy), ability to germinate at high rates (determined by seed response to environmental factors, mainly temperature), and seed longevity (determined by genotype and seed response to environmental factors promoting ageing).

On the other hand, herbicides have some effects on the environment, thus, some plants and animals, which are not targeted are also exposed. The environmental fate of herbicides is related to chemical and physical properties of the products, amount, and frequency of use, methods of application, abiotic and biotic characteristics of the environment, and meteorological conditions [177]. At the recommended rates of use in agriculture, the half-life of herbicides ranges from up to 1 month (e.g. 2, 4-D), to 3-12 months (e.g. atrazine, trifluralin, metsulphuron methyl), to more than 1 year for picloram, tebuthiuron, pendimethalin, chlorsulphuron, and ethametsulphuron methyl [178, 179]. Persistence can be extended under certain use conditions, for example, high pH soils, and low soil moisture [179]. Residues can accumulate to toxic concentrations with consecutive treatments, and products and their metabolites such as atrazine and chlorsulphuron can exhibit persistent and toxic properties [179].

4.2.3. Mechanical

Mechanical control options include the physical felling or uprooting of plants, often in combination with burning [180]. Mechanical control is labour-intensive and thus expensive to use in extensive and dense infestations, or in remote or rugged areas.

4.2.3.1. Rangeland burning

Fire is regarded as the natural factor of the southern African environment; it is thought to have occurred from time immemorial, and therefore, it is part of ecosystems. Rangeland burning is an important ecological management tool in the maintenance and productivity of grasslands in Southern Africa region [181]. The burning in rangelands is practiced for a number of reasons; one of these reasons is to control bush encroachment. To use fire effectively in rangelands, it is important to understand how it behaves and to develop an insight into the way in which various factors influence such behaviour. Fire intensity is one of the important components of the fire regime [182]. Fire regime can be defined as season and frequency of burning together with type and intensity of fire [18]. The effect of fire on natural ecosystems arises from a response of living organisms to the release of heat energy generated by the combustion of plant material. Thus, it is an oxidation process involving a chain reaction during which the solar energy originally converted into carbon compounds by photosynthesis is released as heat during fire [183]. The effect of fire on vegetation, therefore, depends upon the amount of heat energy, and upon the rate and vertical level at which it is released [184]. The rate of fire is measured in terms of time taken to burn a given unit area, it is affected by a number of factors including fuel load and moisture. The vertical level at which heat energy is released during fire determines the height at which plants will be burned. The plant (tree) height is one of the important factors determining the effect of fire on bushes, thus, as the bushes become taller, the fire intensity required to cause a topkill of the stems and branches become critical. Thus, as the plant height increases, the bushes become resistant to fire [182].

Since the effectiveness of fire in rangeland to control bush encroachment depends largely on the fire intensity, which, in turn, depends on fuel characteristics such as fuel load. It is important to note that fire cannot be applied at all times, thus, there should be considerations on the suitability of the ecosystem to support fire. The high intensity fire is required to control bush encroachment at all phases, thus, controlling coppice growth and bush seedlings or maintaining bush at an available height and in an acceptable state for browsing animals [184]. Use of fire as a control method for bush encroachment, therefore, has higher potential in higher rainfall areas where the soil moisture available is reliable and sufficient to produce fuel load that can support regular fires. The use of fire has to be sustained in order to get good results; this is because the bush can recover through coppice regrowth and seedling recruitment after burning, therefore, there should be periodic follow up burn. In moist areas, the frequency of burning required to control bush encroachment depends on the rate at which the bush recovers. The recommended type of fire used in controlling bush encroachment is generally head fire (burning towards the direction of wind); this will mostly occur in the form of surface fire except in extreme conditions where it can develop into crown fire in more densely wooded areas with more flammable foliage. The season of burning should be during the early spring, after the first spring rain. This will ensure the intense fire but with minimal undue deleterious effects on the grass sward. Fire should be applied close to the commencement of the growing season as possible to minimise the length of soil exposure to potential soil erosion.

Reduction of bush encroachment with fire has positive results on herbaceous vegetation biomass production, thus, biomass production is enhanced, and therefore, forage production

increased, which is positive to livestock production. Where fire is used as a regular management tool, it changes species composition, thus, species that are adapted to fire tend to dominate while species that are not favouring fire do not persist. Thus, in South Africa, frequent burning in the False Thornveld of Eastern Cape, favours species such as *Themeda triandra* and has a negative effect on the abundance of *Cymbopogon plurinodes* [185]. Similar results have been observed at the Tall Grassveld of Kwazulu Natal, where *Tristachya leucothrix*, *Cymbopogon excuvatus* and *cymbopogon validus* became dominant with burning frequency [183]. Furthermore, where higher frequency of fire is used, for example where burning is annual, the bush will be controlled but that has an effect on the basal cover of herbaceous plants, thus, the basal cover becomes poor due to effects of fire on plant vigour. That, in turn, renders the soil susceptible to soil erosion, which is another environmental disaster. Fire remains the cheapest form of management available to conserve and perpetuate natural plant communities. However, its effectiveness is based on clear and objective application of a fire regime, thus frequency, season and intensity may be used effectively to retain the natural element and control the invasive elements in the flora of natural ecosystems [186].

4.2.3.2. Manual/Physical cutting/clearing

Manual and mechanical techniques such as pulling, cutting, and otherwise damaging plants, are used to control some invasive plants, particularly if the population is relatively small. These techniques can be extremely specific, and therefore, minimizing damage to desirable plants. However, manual techniques are generally labour and time intensive. These techniques are effective if the treatments are administered several times to prevent the weed from re-establishing. In the process, labourers and machines may severely trample vegetation and disturb the soil, thus, providing prime conditions for re-invasion by the same or other invasive species.

Bush encroachment reduces grass growth in rangeland as discussed in the previous sections and that results in reduced biomass production, which subsequently affects forage production. The approach that has been used to address the negative impacts of invading species in South Africa has been predominantly physical by clearing alien plants [187]. Clearing of the bush in encroached areas results in an increased dry matter yield and basal cover of herbaceous vegetation [184], which are good indicators for rangeland health if the functional characteristic of such an ecosystem is forage production. Furthermore, species richness of herbaceous plants and relative abundance of few of the species among the initial population that is intolerant of bush cover increase with tree cutting [142]. As a result, the reduction of bush cover can restore herbaceous plant productivity and biodiversity in rangelands [188]. However, there are herbaceous species that have a positive relationship with certain trees, and removal of such trees negatively leads to reduction of these herbaceous species. This decline indicates the shifts in the microenvironment due to the removal of ecologically important trees, thus exposing sensitive herbaceous species to increased light intensity.

It is important, however, to note that although bush cutting has positive results on forage productivity, it has high costs involved [142]. Therefore, it is more applicable on the smaller scale. On the larger scale, where bush clearing is done with heavy implements such as a

bulldozer blade, the trees are removed with their roots, which minimises resprouting of encroaching species. However, the soil disturbance generally severely affects the grass layer, but the grasses will often re-establish themselves [154]. The re-establishment of grasses will be following the secondary succession trend, thus the first colonisers are likely to be annual pioneers, which have little forage value. Furthermore, severe soil disturbance may encourage the establishment of a large number of seedlings of some woody plants. This may lead to establishment of a woody community that is denser than the original community.

4.2.4. *Biological control of encroaching and invasive species*

Biological control has been defined as the use of living organisms to reduce the vigour, reproductive capacity, or effects of weeds [189]. Biological control (biocontrol) involves the deliberate introduction of invertebrates or diseases, and is aimed at reducing the effects of ecological release. Biocontrol is aimed at arriving at a situation where the plant is returned to the status of a non-invasive naturalized alien, that is an alien plant that is able to survive, and even reproduce, but does not invade aggressively in its new habitat [6]. Biological control could be regarded as the only sustainable mechanism to prevent the spread of invasive alien species in the long term [190]. Biocontrol is potentially very cost-effective, and environmentally benign. Despite concerns to the contrary [191], the modern practice of using carefully screened and host-specific biocontrol agents is safe, and “host shifts” have not occurred in the over 350 recorded cases where weed biocontrol agents have been used worldwide [192].

Although there are some inconsistencies in terms of when biocontrol practices were established in South Africa, at least there is an agreement in that biocontrol agents have been released against 47 weed species. The disagreement in literature is such that Olckers and Hill (1999) indicated that in South Africa, biocontrol has been practiced since 1910, and that to date, 103 biocontrol agents (including invertebrates and pathogens) have been released against 47 weed species. Whilst on the other side, it has been suggested that the biological control of weeds has been practiced since 1913 and since then some 47 weed species have been subjected to the effects of approximately 85 species of biocontrol agents [190]. Therefore, based on the cited literature, there is an uncertainty about the years of establishment of biocontrol in South Africa and for this chapter the assumption will be that the biocontrol was adopted for use between 1910 and 1913. Although in South Africa physical methods of controlling the alien species are mostly used, biological control using species-specific invertebrates and pathogens from the plant’s country of origin is also a control option; however, there has been a considerable resistance to its use [180]. The seed-feeding weevil is one of the agents that have been released against *Acacia mearnsii* in areas where the wattle is not grown commercially [8]. Nevertheless, plant-attacking agents could potentially be used; however, these compared with seed-attacking agents such as weevils could kill the target plant and therefore, impact severely on commercial prospects. The impact of biological control agents on controlling invasive species vary with species controlled, biological agents introduced, mode of operation of agents and many other factors. The use of biological control measures on invasive plants have been reported in South Africa with varying rates of success. The elaborate example where the invasive plants were controlled with biological control agents was at Kruger National Park

(KNP). The impact of *S. rufinasus* on *A. filiculoides* within the Kruger National Park (KNP) has been exceptionally good. Thus, 100% clearing of the weed was achieved in a few months after release and the infestation has been maintained at that level. The insects are able to survive for long periods in the vicinity and re-establish themselves should the area become re-infested.

Biological control agents such as *Neochetina eichhorniae*, *Cercospora rodmanii*, *Orthogalumna terebrantis*, *Niphograptus albiguttalis*, *Neochetina bruchi*, *Ecritotarsus catarinensi* have been used to control invasion by *Eichhornia crassipes* (Water hyacinth). Although proven in many other instances elsewhere to be effective, the agents released within the KNP have had little impact in terms of bringing the infestation under control. This little impact has been ascribed to frequent low level flooding as well as major floods that have repeatedly washed the infestation away, and therefore, preventing large numbers of insects to build up [194]. *Lantana camara* has been cited to be one of the invasive plants at KNP and other areas of South Africa. Two biological control agents viz. *Octotoma scabripennis* (leaf-mining hispine beetle) and *Falconia intermedia* (*Lantana* sapsucker) have been introduced at KNP. However, *O. scabripennis* failed to establish and the initial trial site for *Falconia intermedia* was reported to have been destroyed by the floods and therefore, both agents have provided insignificant impact on *L. camara* [194].

Opuntia stricta (Sour prickly pear) has been identified as one of the invasive species at KNP and therefore, it was one of the species that were controlled. In an attempt to control this species, two agents have been introduced against it, the first of which being *Cactoblastis cactorum* (phycitid moth) in 1988 [195] and subsequently *Dactylopius opuntiae* (cochineal) in 1996 [196]. The structure of infestations of *O. stricta* changed after the introduction of *C. cactorum* where large plants were replaced by high densities of smaller plants. However, fruit production did not decline and therefore *C. cactorum* failed to provide the degree of control that was expected [195]. Predation and parasitism, especially ant predation of eggs, has a definite impact on the distribution and abundance of *C. cactorum*. *Dactylopius opuntiae*, which had been instrumental in the control against *O. ficusindica*, was released on at least three occasions between 1990 and 1995 yet failed to establish due to the biotype that was used. The Plant Protection Research Institute (PPRI) sourced a different biotype of *D. opuntiae* from Australia, which established well and is reported to be currently destroying large stands in the Skukuza region in South Africa [196].

Pistia stratiotes (Water lettuce) was determined to be one of the invasive species within the Kruger National Park. The snout weevil (*Neohydronomus affinis*) was introduced to control the weeds. The impact of *N. Affinis* on *P. Stratiotes* varied at different infestations throughout the KNP. The other biocontrol agent *Cyrtobagous salviniae* (snout beetle) was released to control *Salvinia molesta* (Kariba weed). The infestations of *S. molesta* at the three areas where the agent was released and established were brought under complete control and have been maintained at that level. *Trichapion latoventre*, *Rhyssomatus marginatus* and *Neodiplogrammus quadrivittatus* were used to control *Sesbania punicea* (Red Sesbania) at Kruger National Park. The impact of the three agents on plants has been reported to be exceptionally good [194]. The three weevil species have reduced the problem to such an extent that *S. punicea* is under complete control in the area, thereby requiring no further action to be taken. The biological control of *S. punicea* remains the best example of an invasive tree species control.

The use of mammals such as goats in agricultural areas to control bush encroachment has been reported in South Africa [197]. Apart from tree seedlings, which can be affected by smaller browsers, the use of browsers to execute control on woody plants largely excludes wild game [154]. However, elephants have also been reported to be effective in controlling bush encroachment [198, 199]. Nevertheless, their use is confined to large game reserves or game farms where their population should be large enough to make an appreciable impact on the woody vegetation, which could, in turn, lead to serious management problems. The control of bush encroachment by use of mammals such as goats is dependent firstly on the acceptability of plant species that are controlled to these mammals for use as browse, and secondly availability of the browse material. The acceptability relates to the palatability and nutritional value of a browse material to the browser. Browse availability relates to the height at which browse material can be accessed by browsing animals, the browse line for goats is approximately 1.5 m. Boer goats are well suited to controlling woody plants because the intensity and frequency with which they utilise the browse can be controlled. Furthermore, the Boer goats are relatively insensitive to chemical deterrents, such as high tannin levels present in many woody species [154]. Boer goats cannot be used to control dense stands of woody plants whose canopies extend above the browse line of approximately 1.5 m.

4.2.5. Integrated bush encroachment and invasion management

Integrated weed control usually involves a combination of at least three of the primary elements of control - mechanical, chemical and biological [180]. Integrated weed management (IWM) could be defined as a system for the planning and implementation of programs, using an interdisciplinary approach, to select a method for controlling undesirable plant species or group of species using all available methods. These methods generally vary between preventative and restorative domains. The success of preventative encroachment measures mostly depends on the understanding of the causes of encroachment and identification of barriers for natural recovery. Restorative measures depend on the rangeland ecosystem structure and functional characteristics to be restored. Integrated bush encroachment control is a multidisciplinary, ecological approach to managing unwanted plant species in rangeland ecosystems.

However, it is important to note that the decision to use a certain method to control the bush encroachment is informed by the cost of using that method against the benefit. Bush encroachment control methods are management systems [137] that might have varied policy implications for bush control [193]. Therefore, understanding the potential role of different bush encroachment control methods for promoting herbaceous species composition requires recognition of the objectives of resource users and policymakers [142]. The failure to recognise the long-term intended ecosystem status could lead to a subsequent failure to achieve bush encroachment control objectives and that could further lead to land use practice and policy controversy. Thus, the resource users are interested in livestock production through increased plant productivity, while the goal of policymakers is environmental preservation. Therefore, the land use practice imperatives and policy directives should be harmonised to permit both forage production and biodiversity conservation functional characteristics of the ecosystem to thrive.

The increasing invasion of non-indigenous species has become one of the top causes of global biodiversity loss and environmental change [200, 201]. Therefore, there is a need for development of intensive mechanisms to control these invasions in the ecosystems before the natural value of ecosystems is lost permanently. As part of a comprehensive remedial effort to control invasions, assessment and characterisation of invader species will serve as a foundation towards integration of efforts to control invaders species. There is urgency for more rigorous and comprehensive assessments of the impacts and risks associated with plant invasions [202]. Thus, prevention and control strategies can be targeted appropriately if sufficient assessments are conducted [203]. In the approaches toward the control of invasive alien weeds, any intervention needs to be aligned with the different stages of spread and characteristics of a desired ecosystem. The stages of spread can be divided into four broad phases: (i) arrival or entry phase; (ii) adaptation and establishment phase; (iii) an exponential growth phase; and (iv) dominance phase. It is in the exponential growth stage of weed spread that integrated control programmes find a logical relevance. Prevention, and early detection and eradication, are more appropriate for the first two stages, while options may be severely limited once weed populations reach the final stage of total ecosystem domination.

Plant invasions are interdisciplinary both by their impacts and by utility and therefore, assessments should recognize the interdisciplinary nature of the problem of species invasions. Thus, the ecosystem characteristics determine whether the appropriate conditions allow for the establishment of the invasive species, and on the other hand, economic systems affect the state of the ecosystem through its use, and through the prevention and control measures implemented to stop the invasions. Hence, accounting for the economic and ecological links and feedbacks is critical in invasion assessments [204]. It is fundamental to have a clear understanding on different functions of ecosystems, thus, an assessment of rangeland area in terms of its ability to achieve its ecosystem functions. Natural resource managers and farmers at all levels require full knowledge of ecosystem functions. This could be achieved through collating results from experiments in different fields or locations within the context of a more encompassing systems management framework that treats the rangeland ecosystem as a complete bio-economic unit. Therefore, in order to improve decision making, farmers need answers to questions at the systems level, including the biological and economic elements of the rangeland production entities they are attempting to manage.

Most often, a single method is not always effective to achieve sustainable control of the rangeland weeds. This is because of among other reasons some methods can only control bush encroachment at a certain stage and some could leave areas that are treated vulnerable to other forms of landscape hazards. For example, use of fire in rangelands depending on the intensity will burn shoots of woody plants; however, the seeds in the soil could be left to germinate and furthermore, some seeds may be stimulated to germinate by fire. It is also difficult to ascertain a complete kill of unwanted species with fire because normally the basal buds of certain trees remain unburned and therefore resprout. It is for these reasons that the introduction of biological control agents becomes important especially where complete removal of the invading species is anticipated. Use of herbivores works effectively where the intention is to maintain the current stand of encroaching species especially in the savanna where there is

coexistence between grasses and trees. There are species which are not preferred by animals for foraging, and use of biological control through herbivores would not be effective; therefore, introduction of invertebrates could be used. However, most of the invertebrates are not readily available in Southern Africa for use at the farm or landscape level. It is impractical to burn certain areas that are encroached; this is sometimes due to poor fuel load that can support high fire intensity needed to burn the woody species. Encroachment in some of these areas cannot be controlled with the use of herbivores (goats) and herbicides could be useful.

All this, therefore, suggests that there are areas and bush encroachment situations where a single method can be used; however, a combination of different methods could be used simultaneously or alternatively in subsequent approaches. Nevertheless, it is important that prior to the implementation of any selected method or any combination or any sequence to develop post encroachment treatment management plan. This is because removal of bush with any technique can leave the land vulnerable to soil erosion or further encroachment of the same or new species. Therefore, a successful long-term management program should be designed to include combinations of mechanical, biological, and chemical control techniques. Numerous mechanical and cultural options have been developed to manage noxious rangeland weeds, including mowing, prescribed burning, timely grazing, and perennial grass reseeding or inter-seeding. Furthermore, several herbicides are registered for use on rangelands and most biological control programs focus on noxious rangeland weed control. Successful management of noxious weeds on rangeland will require the development of a long-term strategic plan incorporating prevention programs, education materials and activities, economical and sustainable multi-year integrated approaches that improve degraded rangeland communities, enhance the utility of the ecosystem, and prevent reinvasion or encroachment by other noxious weed species [151].

There are a number of factors to consider in selection of the bush control method; however, the dominant consideration is the cost of the method. However, there are furthermore considerations beyond the cost of the method. The use of fire in controlling bush encroachment in rangelands is determined by a threshold amount of flammable fine fuel needed to carry fire that is sufficiently intensive to reduce woody plants. Furthermore, to effectively control woody plants with burning, fire must be applied regularly. Many rangelands occur in semi-arid environments in which forage-based livestock production is the primary agricultural activity and intermittent droughts are inevitable [205]). Therefore, accumulating sufficient fine fuel to carry fires in such environments requires the reduction in livestock numbers compared to areas where fire is not used. Hence, sustainable utilisation of semi-arid rangelands depends on complex management of animal species, stocking rates, and the vegetation composition, structure, phenology and quality [129].

The integration of bio-control agents and herbicides in a scientifically sound and rigorous management plan is the first step in a long-term approach to weed management. Such management plans should aim to maximise the benefits of all the respective control options and thereby ensure the infestation is contained and the density reduced to acceptable thresholds. Biological control is used as an important, long-term management solution to numerous weeds worldwide. When carefully integrated into management plans the combination of bio-

control and other control measures may provide effective solutions to the problem, and various methods therefore, should not be used in contradiction to one another. All available knowledge surrounding a particular invasive plant problem needs to be considered when developing such integrated programmes. No single method is likely to prevent either distribution or densification of the plant from or in its current range. Combination of the biological control and herbicides can bring remarkable results; while herbicides are used to contain the infestation to its present range, biological control (invertebrates) is being released into dense stands where it is proving destructively effective in controlling the plants. Goats used in the system that allows coppice growth to be used frequently and severely strongly influence woody plants, that is, provided that their canopies be below the browse line. Where the plants are above browse line, fire can be used to reduce plant height where fuel load is sufficient; however, where fuel load is not sufficient chemical or physical control can be used and, in both cases, goats can be used as follow up control.

In this chapter, integration in the control of invasive species is not limited to control methods themselves in isolation but in all the processes relating to bush encroachment management. Primarily, it is important as the initial stage of integration to identify and characterise invasion/encroachment of species. This should include establishment of their origin, mode of establishment and spread (seeds, cuttings etc), their phenological and morphological characteristics and assessing their favourable growth conditions. It is further important to determine the degree of invasion/encroachment, which will help setting economic and ecological thresholds of invasion. The analysis of the ecological and economic impact of invasion/encroachment in the environment should be carried out prior to any intervention. That will help in determining whether there is a need for intervention and magnitude of such intervention. The need for intervention should be assessed against the set thresholds for invasion. Setting objectives for invasion/encroachment management is very fundamental because the objectives will be used as the yardstick for the control.

A number of factors will guide selection of the approach to control bush encroachment. These factors include species to be controlled, the stage of invasion and landscape of an area. The approach to be selected would be chemical, mechanical and biological depending on the approach suited to the species to be controlled, the major landscape on which the invasion has occurred and the stage of invasion. The method that is ecologically and economically sound and practical should be selected. Integrated bush encroachment approaches may be practiced in combinations that could either be used simultaneously, alternatively or sequentially. In simultaneous integration of bush control methods, more than one method that could complement each other under the prospects of chemical, biological or physical methods used together. Some methods cannot be used simultaneously because of the danger that they can cause on other organisms and environment. For example, the methods that can be integrated simultaneously could be manual clearance and use of goats as browsers. The alternative integration could be executed through turns, thus, one method first and then the other. The alternate integration can be practiced in rotation if planned properly, for example, use fire with a given period in between goat treatment. Thus, burning can be applied every three years while goat use is continued. Sequential integration is executed in succession of methods where one

method can be used to prepare for the next method in a sequence. In this integration there should be short-term objectives relating to each method and long-term objectives, which are based on the integrated approach. Thus, mechanical control in the form of fire or physical cutting can be used to reduce plant height to facilitate the use by goats as the maintenance stage of control. Where there is high density of bush, which impairs the movement of animals, or where the bush is above the browse line of goats or where the bush is unacceptable to browsing animals yet the fuel load is poor, mechanical cutting would be useful. This would reduce the bush density, which will open up for goats to be able to browse, that will further open up for grass to grow then fire can be used as a follow up. Where the bush has higher density but there is sufficient fuel load, fire will be the most applicable method. Fire will clear up the bush faster and relatively cost effectively, therefore, where there is enough fuel load fire is recommended as the first on the integration followed by use of goats. Biological control would always be the last in the sequence and it is the approach that helps in achieving long-term bush encroachment control objectives. The use of invertebrates (Weevils) could be integrated with the use of herbivore (goats) since the weevils take care of the seeds and the goats can take of the foliage to maintain the stands.

A post treatment management plan should be part of integration in bush encroachment control, thus, there should be a clear plan on what rangeland management system will be practiced that will ensure that the control objectives are achieved. Thus, some invasion control methods such as the use of fire can leave the soil bare and susceptible to soil erosion and, therefore, there should be a clear objective plan on what practices will be taken immediately after treatment. Furthermore, on the areas that are severely encroached and grass biomass and basal cover are affected, use of herbicides will also leave the soil bare and grazing can worsen the situation and lead to soil erosion. Therefore, as part of integration, exclusion of treated areas to minimise grazing should be considered. This exclusion could be coupled with introduction of plant propagules, thus, revegetation through seeds or seedlings of the grass on the bare patches.

There is a need for periodic monitoring and evaluation as part of integration of the encroachment control. This will help in determining whether the treatment is achieving expected results within the given timeframes. That will help in realising if there is a need for the adjustment of the plan. Effective bush control monitoring and evaluation should be done according to the pre set objectives; it will help in the establishment of whether the objectives are achieved. Performance measures, monitoring, and adaptive management are necessary. Using these methods, status and trends can be tracked, analysis and accountability facilitated, and decisions adapted so that the intended balance among social, economic, and ecological concerns is achieved. Ecosystems' performance appraisal will be important at the end of the integration, this should be a pronouncement of whether the target ecosystem has been reached and should be coupled with sustainability management programme that will eliminate factors that could have lead to encroachment. Ecosystem performance measures can provide a quantitative basis for evaluating how well actions under the integrated bush control approach are meeting stated objectives. Performance measures allow for continuous learning, which broadens understanding about how ecosystems function. There are many approaches to

evaluate performance; however, performance measures should specifically address management goals and objectives and should be quantifiable, expressing status and trends of specific resource values of concern, such as unique ecosystem type.

5. Conclusions and recommendations

Bush encroachment and invasion could be attributed to a number of factors, which by their nature vary with species and locality. These factors cannot easily be ranked according to the strength of causation and/or according to the intensity of their effect on rangeland ecosystems. Factors that are blamed for bush encroachment include improper grazing practices, suppression of fire, drought, rainfall intensity and distribution and climate change. The temporal and spatial distribution of bush encroachment follows a sigmoid distribution curve. Although some invasive species are abundant, they are localised in certain areas whilst, on the other hand, certain species are widely distributed but low in copiousness. There are three major methodological guidelines; these fall under chemical, mechanical/physical and biological and depend on a number of factors within economic and ecological impressions. Bush encroachment occurrences are generally caused by different factors, at different landscapes, by different plant species and with different effects. Therefore, the invasion control methods should consider this variation for success in treatments. Thus, there are areas and invasion situations where a single method can be used; however, a combination of different methods could be used in simultaneous or alternative or subsequence approaches.

Integrated plant invasion management should have four major stages of execution; these are comprised of diagnostic, preventative, control and management. The diagnostic stage should include identification and characterisation of invasion, determination of the degree of invasion, analysis of the ecological and economic impact of invasion, determination of the need for intervention, and setting objectives for intervention. The control stage should include selection of invasion control approach or combinations. Management stage includes post-treatment management, monitoring, evaluation, and ecosystems' performance appraisal. Preventative stage is more practical on the areas that are not yet invaded; at this stage management of areas that are not yet encroached is central. Assessment and characterisation of vulnerable areas for invasion will be important in developing an encroachment prevention plan. It is also important to assess plant invasion predisposing factors; however, these may vary with species and localities. In the diagnostic stage, determination of the level of spread is very fundamental and will serve as the background for selection of the bush encroachment control and management methods. The stage of bush encroachment spread can be divided into four broad phases viz, entry phase, adaptation and establishment phase, an exponential growth phase and dominance phase. It is in the exponential growth stage of weeds spread that integrated control programmes find a logical relevance. Prevention, and early detection and eradication, are more appropriate for the first two stages, while options may be severely limited once weed populations reach the final stage of total ecosystem domination. Although there is massive literature on the plant invasion and bush encroachment, there is still a significant need for further research in establishing fundamental characteristics of bush encroachment phenomenon in

rangelands. This will lead in systematic characterisation of bush encroachment and subsequently that will lead to development of more practical and radical yet scientific bush encroachment control and management practices in rangelands.

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