

TOWARDS A POLICY ON

INDIGENOUS BUSH ENCROACHMENT

IN SOUTH AFRICA



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Authors:

Jane Turpie, Pieter Botha, Kevin Coldrey, Katherine Forsythe, Tony Knowles, Gwyneth Letley, Jessica Allen and Ruan de Wet

Project Management:

Hlengiwe Mbatha – Project Manager, Department of Environmental Affairs
Barney Kgope – Co-Project Manager – Department of Environmental Affairs
Jane Turpie – Technical Team leader, Anchor Environmental Consultants
Dumisani Nxumalo – Advisor, Climate Support Programme – Deutsche Gesellschaft für Internationale Zusammenarbeit (GIZ) GmbH
Mirko Abresch – Technical Advisor, Climate Support Programme – Deutsche Gesellschaft für Internationale Zusammenarbeit (GIZ) GmbH

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Contact Information:

Department of Environmental Affairs
Environment House, 473 Steve Biko Street
Arcadia, Pretoria, 0001
South Africa
Tel: +27 12 399 9146
Email: HMbatha@environment.gov.za

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Bush encroachment is a global phenomenon. In South Africa, bush encroachment entails increases in the abundance of indigenous woody vegetation in the grassland and savanna biomes. While an increase in above ground biomass may be seen as a benefit to carbon sequestration, this study was conceived with the view to provide policy direction in the context of three United Conventions; UNFCCC, UNCBD and UNCCD for which South Africa is signatory.

Further, to review the extent and causes of the problem in South Africa, broadly evaluate its impacts, determine the policy direction that South Africa should take towards bush encroachment, and identify appropriate policy responses. In particular unpack bush encroachment in a context of climate change, desertification, land degradation, sustainable land management, food security and greenhouse gas fluxes in terrestrial ecosystems.

Although the independent research and findings contained in this report do not necessarily represent the views, opinions or position of government, the department believes that this research is critical to enhance our understanding of bush encroachment in the country and actions that policy makers can make in dealing with impacts of bush encroachment while fulfilling obligations of the three UN conventions.

Hlengiwe Mbatha

Chief Directorate: Climate change Mitigation

Directorate: Climate Change Mitigation Research and Analysis

Department of Environmental Affairs

EXECUTIVE SUMMARY

Introduction

Bush encroachment entails increases in the abundance of indigenous woody vegetation in the grassland and savanna biomes. In South Africa, where these biomes make up 27.9% and 32.5% of the land surface area, respectively, there has been a significant increase in tree cover since national-scale aerial photography was first undertaken in the 1940s.

Given global concerns over deforestation and climate change, bush encroachment may be perceived as beneficial in terms of carbon capture and storage, and the increased provision of woody biomass. However, it also alters the structure and functioning of ecosystems, with these changes becoming increasingly irreversible as the fundamental nature of the ecosystems change. Such changes not only alter landscapes and their biodiversity, but also the nature and value of ecosystem services delivered from them. It is therefore an important policy issue to be addressed in relation to South Africa's commitments under the United Nations (UN) conventions on climate change (UNFCCC, 1992), biodiversity (UNCBD, 1992) and combatting desertification (UNCCD, 1994).

The aims of this study were to review the extent and causes of the problem in South Africa, broadly evaluate its impacts, determine the policy stance that South Africa should take towards bush encroachment and identify appropriate policy responses.

Overall approach

We reviewed available information on the nature and extent of bush encroachment and its spatial variation, as well as on its causes. We then devised a set of simple models and assumptions to estimate the impacts that bush encroachment has had across different parts of the country to date. Following this, and a review of the potential methods of controlling the problem, we undertook a scenario analysis to evaluate the potential viability and implications of attempting various forms of control in different areas. All of these analyses were conducted at a high level, using simple assumptions, in order to evaluate the situation from a broad policy perspective.

Recognising that there is considerable spatial variation in bush encroachment across the country, the extent and impacts of bush encroachment and potential remedial interventions were

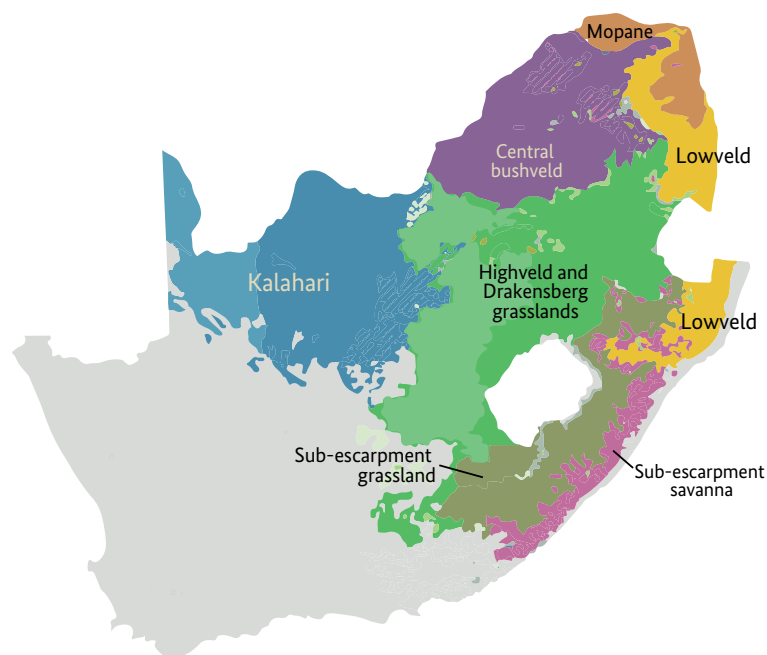


FIGURE I. Bush encroachment zones defined in this study, in relation to biome and bioregion boundaries (Mucina & Rutherford 2006)

summarised for a set of seven bioregional zones that were largely based on Mucina & Rutherford's (2006) bioregions (Figure I). Results from earlier studies were used to determine the average change in woody cover over the monitoring period for affected areas in each of the bioregional zones and the total extent of bush encroachment in each zone was estimated from recent mapping of bush encroachment.

The nature and extent of bush encroachment in South Africa

Estimates of the extent of bush encroachment have been made at various spatial scales using field studies, landscape photography (dating back to the 1800s), aerial photography (dating back to the 1940s), satellite data (which dates back to the 1970s) and modelling based on statistical analysis of existing information. Previous estimates put the land area affected by bush encroachment at between 5.7 million and 13 million ha, or up to about 10% of land area. A recently-completed study has refined this estimate to about 7.3 million ha (Warren, Hugo and Wilson 2018). In general, areas with over 500 mm mean annual precipitation have had higher rates of woodland expansion than lower rainfall areas.

Determining the extent of bush encroachment that has occurred to date requires defining a starting point, baseline or 'natural' condition. This is difficult in dynamic, disturbance-driven ecosystems. Evidence suggests that the problem has been around for over a century, but increasing exponentially, so that most encroachment has only happened in the past 30 years. Thus, the aerial photographs from the 1940s would make an adequate baseline suggesting that estimates made from more recent satellite data are not too inaccurate for the most part. The baseline

is important for informing a restoration strategy, but management goals should also take other societal values into account.

Bush encroachment involves the proliferation of woody species that naturally occur in savanna ecosystems. Over 40 species have been listed as being part of the bush encroachment problem in South Africa, and new species are being added. Bush encroachment is not caused by particular species, but is rather a change in the balance of the types of plants occurring in ecosystems. Nevertheless, some species respond to the drivers of bush encroachment more prolifically than others, and may be identified as the 'chief culprits' in a particular area. However, the same species are likely to be benign and useful at their natural densities.

Some of the most prominent encroacher species in different areas include blackthorn *Senegalia mellifera* in the Kalahari, sicklebush *Dichrostachys cinerea*, in the Central Bushveld and Lowveld, mopane *Colophospermum mopane* in the Mopane region, red bush willow *Combretum apiculatum* and silver cluster leaf *Terminalia sericea* in the Lowveld, sweet thorn *Vachellia karroo* in the Sub-escarpment Savanna and Grasslands, bankrupt bush *Seriphium plumosum*, in the Highveld and Drakensberg Grasslands and the paperbark thorn *Vachellia sieberiana* in the Sub-escarpment Grasslands.

In past bush encroachment studies, the average starting percentage woody cover varied considerably across the seven encroached zones, with the mopane zone having the highest average percentage woody cover at the start of monitoring but the lowest overall change in woody cover (Figure II). The sub-escarpment grassland zone

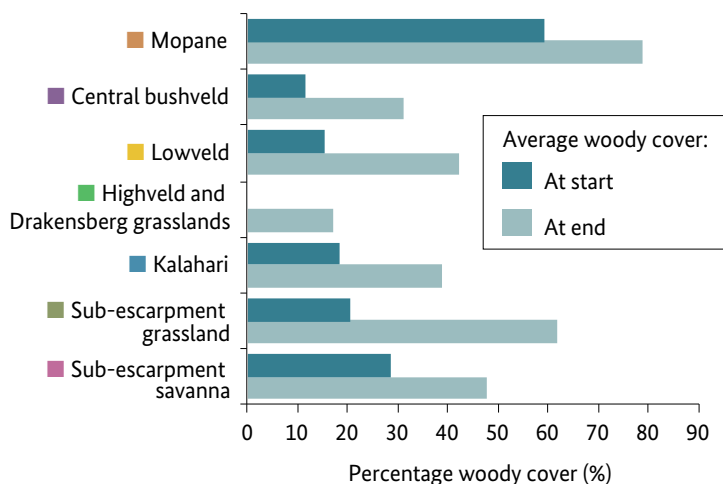


FIGURE II. The average percentage woody cover at the start and end of monitoring based on a catalogue of studies in southern Africa as summarised by O'Connor et al. (2014).

had the highest average change in overall woody cover from the start to the end of monitoring period (41%), followed by the Lowveld zone which had an average overall change of 27%. The other zones had an average overall change in woody cover of about 20%.

The causes of bush encroachment

Savannas are characterised by the coexistence of grasses and trees (or shrubs) and are the transitional biome between grasslands and forests. The grasses in these ecosystems tend to be highly shade-intolerant and fire-tolerant C_4 grasses and C_3 trees. The extent of woody cover in these systems largely boils down to how variation in rainfall and fire affect the competition between grasses and tree seedlings. Fire is a particularly important driver. Savannas occur in areas where rainfall is highly seasonal, and where fire can occur once the grass layer that has built up during the wet season becomes dry enough to burn. The C_4 grasses are flammable and form the main fuel for fires in savannas, rather than woody biomass. These grasses also recover quickly from fire, whereas tree seedlings and some adult trees will be damaged or killed by fire. Fire thus acts to keep savannas open. Increased rainfall and increased rainfall period thus move savannas towards closed forest systems, whereas lower rainfall, longer drier seasons and resulting increased fire moves them towards grassland systems. In the absence of fire under mesic conditions, forest trees, which are more shade tolerant than savanna trees, can recruit through the grass layer and ultimately outcompete grasses and savanna trees.

The occurrence of fire requires a source of ignition as well as fuel that is adequately dry, abundant and dense enough to sustain its spread across a landscape. Fire therefore typically occurs in regions where there is enough rain (>450mm) to generate a high enough fuel load, coupled with a long enough dry season in which fuel has the opportunity to dry out sufficiently to burn. At higher rainfall (>1 800mm), conditions are generally too moist for fire.

Prolonged, high grazing pressure reduces the fuel load, thereby suppressing fire. In addition, active suppression of 'runaway' bush fires to protect property is also likely to play an important role. The intensity of the ecological response to overgrazing and fire suppression is influenced by availability of resources (rainfall, soil nutrients and

atmospheric carbon dioxide), as well as positive feedbacks that develop as bush encroachment progresses. As encroachment progresses, there may be a 'tipping-point' as tree canopy cover increases over 45-50%, above which fires rarely occur.

The progression of bush encroachment is also influenced by rainfall. In South Africa, several studies found that a pulse of bush encroachment occurred in semi-arid and mesic savannas in southern Africa following the drought of the 1960s and the above-average rainfall of the 1970s. Soil characteristics may also be an important determinant of the vulnerability of landscapes to encroachment, since they influence the dynamics between grasses and trees, and the combination of low nutrients and high rainfall may favour bush encroachment.

Elevated atmospheric carbon dioxide is also likely to promote bush encroachment. Since the start of the industrial revolution, atmospheric CO_2 has increased from 278 to 390.5 ppm in 2011 (IPCC 2013, ch. 6 pp. 467). Plants using the C_4 pathway, which include the grasses in savanna ecosystems (including 'false grasslands'), are generally not constrained by available atmospheric CO_2 , whereas those using the C_3 pathways, which include the woody species in savanna ecosystems, are limited by the amount of CO_2 in the atmosphere. Therefore, the addition of more CO_2 helps C_3 plants to grow faster and outcompete C_4 species. The strength of this effect will also depend on what other factors are limiting.

Impacts on ecosystem services

The gains or losses in the supply and value of ecosystem services as a result of bush encroachment were estimated for encroached areas for each of the bioregional zones. This was differentiated into state-owned protected areas, private rangelands and communal rangelands. These estimates were based on GIS layers from a recent national mapping of ecosystem services (Turpie et al. 2017), and on recently-mapped estimates of carbon stocks analysed in relation to woody vegetation cover. Our estimates suggest that bush encroachment has resulted in carbon gains in the order of 4.3–28.5 tonnes per ha in the affected areas, worth some R23–154 per ha in terms of climate change costs avoided in South Africa (Table I).

TABLE I. Order of magnitude estimates of the value of carbon sequestration through bush encroachment in different ecological zones, based on the average recorded change in woody cover in those zones (ROW = rest of world)

	WOODY CARBON POOL GAIN (T/HA)	CO ₂ EQUIV	VALUE GAIN TO SA (R/HA)	VALUE GAIN TO ROW (R/HA)
Mopane	28.5	104.6	154	44 100
Central bushveld	7.1	25.9	38	10 900
Lowveld	13.8	50.5	75	21 300
Highveld	4.3	15.7	23	6 600
Kalahari	16.4	60.0	89	25 300
Sub-escarpment grassland	27.7	101.6	150	42 900
Sub-escarpment savanna	6.7	24.4	36	10 300

The carbon values were then combined with the estimated changes in value of ecosystem services within each land tenure type within each bioregional zone to generate a net loss or gain in ecosystem service value due to the impacts of bush encroachment for a situation equivalent to the average level of bush encroachment recorded in that zone. The impacts of bush encroachment are highly context specific (Table II). In most cases, our high-level estimates suggest that bush encroachment has led to an overall loss in the value of ecosystem services. In most communal land areas, however, bush encroachment could have yielded net positive effects, as a result of increased fuelwood. However, in these cases it is likely that bush encroachment would only be positive to the point where marginal gains are exceeded by marginal opportunity costs in terms of losses of grassland benefits such as grazing and thatch, and given the absence of information on marginal values, it is possible that these thresholds have actually been exceeded. In the arid Kalahari ecoregion, bush encroachment appears to have had a net economic benefit. This does not take impacts on biodiversity into account, however. Again, any positive impacts may only remain positive up to a point.

TABLE II. The net gain or loss per hectare for each land tenure type within each zone

ZONE	PROTECTED AREAS	COMMUNAL	PRIVATE
Mopane	70.7	94.7	-129.9
Central bushveld	-65.8	183.7	-25.6
Lowveld	-134.8	347.0	-225.9
Highveld and Drakensberg grassland	-62.4	-607.8	-198.5
Kalahari	80.1	110.1	23.4
Sub-escarpment grassland	-472.3	-755.6	-695.1
Sub-escarpment savanna	-4951.2	267.9	-1446.6

Based on this, it can be argued that the climate change adaptation benefit of addressing bush encroachment would outweigh the mitigation benefit of allowing it to proceed. While the exact amount of carbon sequestered through bush encroachment in South Africa is unknown, even if it were substantial, the risk of losing biodiversity and further degrading ecosystem services from allowing bush encroachment to continue unheeded is considered unacceptable. Moreover, the potential risk to biodiversity of allowing bush encroachment would contradict the commitments made under the UN CBD. It is clear that *bush encroachment should be considered a form of land degradation* under the UN commitments, and that other, less damaging, emission reduction opportunities should be employed to meet those targets.

Potential remedies

Bush encroachment can be avoided or reversed to some extent by better rangeland management (to maintain a healthy grass layer), including fire management. Alternatively, where it has progressed too far for this to be effective on its own, bush encroachment should be cleared or thinned manually or mechanically. Chemical spraying is harmful and should be discouraged. Best practices for active clearing include determining the appropriate degree of remediation, having a long-term management strategy and undertaking follow up treatments accordingly. In addition, it is important to introduce sound land management practices to maintain the gains made.

Income generation through wood harvesting could make clearing a viable business option in itself, but its economic viability is unknown and there are potential risks in encouraging this in order to address bush encroachment, such as overharvesting of woody biomass and decreasing long-term soil fertility.

Policy and legislation affecting management

Current policy and legislation do not deal specifically with bush encroachment. The Conservation of Agricultural Resources Act (CARA) encourages maintenance of rangelands, but if agricultural clearing occurs within an important biodiversity area or affects listed species, it can require authorisation under the National Environmental Management Act (NEMA), National Forestry Act (NFA) or Biodiversity Act (NEMBA). There is some level of ambiguity as to the treatment of bush thinning, as opposed to complete clearing, for example with a bulldozer.

Evaluation of policy scenarios

Four potential policy scenarios were evaluated using a high-level estimation of the potential costs and benefits for each of the affected bioregions, taking land tenure into account. These were:

1. *Laissez faire (no action).*
2. *Support/incentivise/regulate better rangeland management* to retard, arrest or reverse bush encroachment through better grazing and burning practices.
3. *Fund manual restoration*, for example through a Working for Land type of programme, in conjunction with #2 to maintain the gains made.
4. *Facilitate mechanical restoration*, in conjunction with #2 to maintain the gains made.

Clearing and follow-up costs were taken from the literature and key informants. The gains and losses of ecosystem services were based on the preceding analysis, with the assumption that demand for ecosystem services would increase at 3% per year. Potential benefits from use of the woody biomass in value adding activities was not quantified in this study as this will be estimated in detail elsewhere. Instead, we estimated the harvesting profit required to make clearing feasible if the ecosystem service benefits are not enough to justify action on their own.

The estimates are indicative, in that they are constructed from simple models and assumptions, based on the limited available data at the time of this study. It was assumed that the goal of restoration would be to reduce woody cover to its initial extent. This too is based on simple assumptions of relationships between the different ecosystem services and woody cover, where in reality, some of these relationships are likely to be sigmoidal (s-shaped). Furthermore, not all values are captured in this study, such as the cultural and spiritual values of biodiversity.

The results suggest that a *laissez-faire* policy (Scenario 1) would lead to further losses in welfare in most cases, apart from some communal areas and in the arid Kalahari region (Figure III). While active restoration leads to ecosystem service gains, in the absence of generating income from the biomass removed the costs outweigh these benefits in most cases (Scenarios 3 and 4). Incentivising better land management as the sole remediation action (Scenario 2) was assumed to be neutral in this analysis, in that this intervention can already be considered to be an imperative (namely at no extra cost), and because it maintains the current status quo.

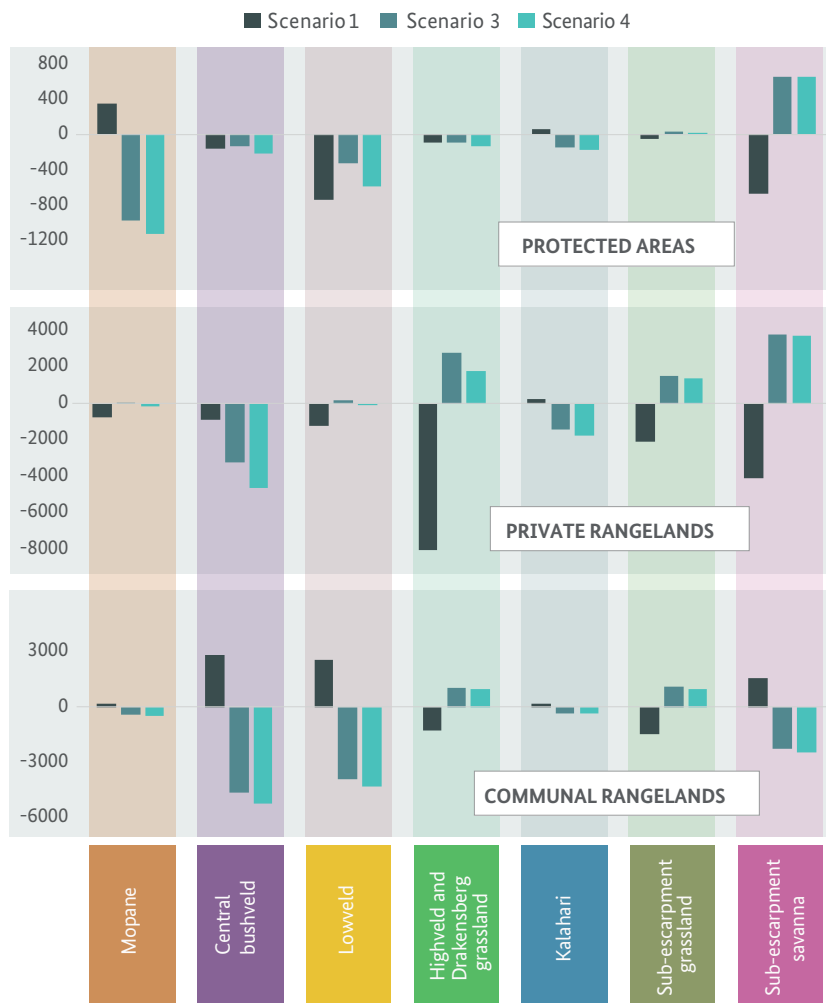


FIGURE III. The net present value (in R millions) of different policy scenarios (1 – *laissez faire*, 3 – manual clearing and 4 – mechanical clearing) for each land-tenure type (protected areas, private rangelands and communal rangelands) for each region

For some areas (Highveld and Drakensberg Grasslands, Sub-escarpment Grassland and Savannas), private benefits of clearing would exceed the costs. However, little clearing has been implemented, which suggests that certain barriers may be at play, such as lack of understanding of the benefits of or methods for taking action, lack of access to capital, or unwillingness to engage additional labourers.

Manual clearing is cheaper than mechanical clearing and can contribute to job creation. A government programme is needed for addressing excessive encroachment in communal areas. Manual clearing may be less suitable for some protected areas or in private rangelands. Regardless of whether active clearing is pursued

or which method is used, government interventions to improve rangeland management practices need to be implemented throughout as the highest priority measure.

In general, the costs of clearing are high enough that either subsidies or a significant income from the harvested biomass will be required in order to stimulate clearing action. In the cases where active clearing, without using the biomass harvested, is unlikely to yield a positive net outcome for society due to the high costs involved, we estimated how much profit would need to be made from the harvested biomass in order to make active clearing economically viable (Table III).

TABLE III. The approximate harvesting profit (R per ha) required to make manual or mechanical clearing feasible for the areas where it is not already feasible, for the average level of bush encroachment

ZONE	LAND TENURE		
	PROTECTED (R/HA)	PRIVATE (R/HA)	COMMUNAL (R/HA)
Mopane	1 330 / 1 540	- / 180	1 490 / 1 700
Central bushveld	440 / 660	670 / 960	2 230 / 2 500
Lowveld	290 / 550	- / 90	2 690 / 2 980
Highveld & Drakensberg grassland	240 / 360	-	-
Kalahari	200 / 230	140 / 1704	230 / 260
Sub-escarpment savanna	-	-	-

These figures should be compared with estimates from the forthcoming national study on this potential. Note, however, that there will be limitations on the viability of generating income from biomass, due to demand and access to markets. Encouragement of these activities would also need to go hand in hand with measures to reduce their potential negative impacts, such as clearing beyond optimal levels and encouraging regrowth.

Based on the above, it can be concluded that:

- In protected areas, better land management is the best option for addressing the problem, apart from in the sub-escarpment grasslands and savannas, where active clearing would be worthwhile, especially in the latter.
- In private rangelands, active clearing would be the best option in all but the Mopane, Kalahari and Bushveld regions, where better land management would be the best option.
- In communal rangelands, active clearing would only be worthwhile in the grassland ecoregions. For the rest, where no action may lead to a positive net gain up to a point, measures to improve land management should be introduced to the expected benefit of poor households.

RECOMMENDATIONS

- 1. Recognise bush encroachment as a form of land degradation.** Continued encroachment could have a significant negative impact on overall supply and value of ecosystem services, biodiversity and livelihoods.
- 2. Strengthen extension services and institutions for rangeland management.** Promote sustainable land management practices that reduce bush encroachment, regardless of any other strategy adopted and regardless of region. These include long-term sustainable stocking rates and implementation of rotational grazing practices that provide adequate rest for grazing areas. In communal land areas, this may need to be facilitated by the establishment of defined grazing areas for defined rights holders.

3. Identify thresholds of potential concern and develop monitoring systems. Rigorous monitoring systems should be developed to ensure that bush encroachment is managed optimally at the national, provincial and local levels, based on the latest scientific evidence. Thresholds of potential concern for biodiversity and ecosystem services need to be identified for each of the different regions.

4. Remove legal barriers and develop norms and standards for clearing/thinning encroached areas. The legal aspects of bush encroachment management should be clarified and (potential) conflicts between the different statutory bodies should be remedied. A set of norms and standards should be developed to reduce bureaucratic delays.

5. Promote sustainable income-generating bush-clearing activities in private rangelands. This should be done within a regulatory framework that avoids incentivising unsustainable practices.

6. Establish government-funded manual clearing programmes in selected communal areas. In communal rangelands, manual clearing programmes should be funded in affected areas of the grassland ecoregions and any other localised problem areas.

7. Set up a bush encroachment information and advisory service. This advisory unit would ideally develop detailed guidelines for the management of bush encroachment in each of the different regions and land use types identified in this report and provide decision-support systems and tools to land managers for bush encroachment management in South Africa.

8. Conduct further research. Further research is needed into the biodiversity impacts of bush encroachment, the potential effects of woody biomass removal on soil fertility and the possible role of woody cover in restoring degraded soils. In addition, a better understanding is needed of the barriers currently preventing active clearing by landowners in areas where the private benefits of clearing appear to outweigh the costs.

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INTRODUCTION

1.1 BACKGROUND AND RATIONALE

Bush encroachment is an increase in the abundance of woody vegetation in grassland and savanna ecosystems. An increase in woody plant biomass in grassland and savanna areas is a worldwide phenomenon that has been reported over the past 50–100 years (Sankaran et al. 2005; Buitenwerf et al. 2012; Lehmann et al. 2011). In South Africa, grasslands and savannas make up 27.9% and 32.5% of the land surface area, respectively. Since national-scale aerial photography was first undertaken in the 1940s, there has been a significant increase in tree cover within these biomes (Stevens et al. 2016).

Given global concerns over deforestation and climate change, bush encroachment may be perceived as beneficial in terms of carbon capture and storage, and the increased provision of woody biomass. However, bush encroachment alters the structure and functioning of ecosystems, gradually turning grasslands into woodlands and woodlands into forests, with these changes becoming increasingly irreversible as the fundamental nature of the ecosystems change (Hoffman & O'Connor 1999; Archer et al. 2001; Roques et al. 2001; Goslee et al. 2003; Asner et al. 2004; Walker & Meyers 2004; Van Auken 2009; Jordaan 2010, Briggs et al. 2005; Brook & Bowman 2006; Bowman et al. 2010, Wigley et al. 2010). Such changes not only alter landscapes and their biodiversity, but also the nature and value of ecosystem services delivered from them. For example, increased woody biomass can lead to reduced stream flows and ground water recharge, reduced grazing capacity and reduced tourism appeal. Thus, it is very important to refine our understanding of how bush encroachment comes about, how it can be controlled or reversed, and the costs and benefits of doing so, especially given that dealing with the problem has often proven difficult and costly.

While the ecology and impacts of invasive alien plants (IAPs) have received much attention in the past and the policy direction on these species is clear, bush encroachment by indigenous species

has not yet received much attention in the policy arena. Following increasing research and understanding in recent years, it is clear that bush encroachment is an important policy issue to be addressed in relation to South Africa's commitments under the United Nations (UN) conventions on climate change (UNFCCC, 1992), biodiversity (UNCBD, 1992) and combatting desertification (UNCCD, 1994).

1.2 STUDY OBJECTIVES AND SCOPE

The overall objective of this study was to inform the development of appropriate policy on bush encroachment in South Africa and to help government identify and formulate interventions that address bush encroachment, as informed by scientific evidence.

This project interrogates and addresses the potential conflicts in implementing South Africa's commitments to the three UN conventions. Firstly, it assesses whether or not bush encroachment is a form of land degradation, since it can significantly change the structure and function of ecosystems and the services they provide. It considers bush-encroachment in a number of typical South African landscapes and key elements for consideration in a decision to clear or not. Secondly, it explores a set of potential management scenarios, for example, maintaining bush encroachment, partial clearance to provide fuelwood to local communities and improve livestock production, or industrial-scale clearing for the production of bioenergy and biochar. The latter interventions were previously identified in the National Terrestrial Carbon Sinks Assessment (NTCSA, 2015) as part of the land-based climate change mitigation opportunities.

The study does not attempt to provide new information on the distribution of bush encroachment or highly accurate estimates of its impacts, but rather attempts to improve insights and policy direction on the problem through a critical analysis of available information, some preliminary estimates and expert opinion.

1.3 STUDY APPROACH

This study was carried out as a desktop exercise based on available information. To this end, the team members attended South African Environmental Observation Network's (SAEON's) Woody Biomass Symposium in August/September to learn about the data being compiled for mapping woody biomass throughout the country and arranged to incorporate these data as far as possible. In addition to the use and review of available data and literature, the work has also benefited from regular team discussions and from discussions with key experts in the field.

1.4 STRUCTURE OF THE REPORT

The structure of the rest of the report is as follows:

Chapter 2 describes the *nature and extent of bush encroachment* in South Africa, based on available geographic data, and scientific studies carried out in South Africa, the broader region and the global literature. We have divided the country into bioregion-based zones and describe the different nature and extents of bush encroachment in each of them.

Chapter 3 presents our understanding of the *drivers of bush encroachment*, based on a literature review. We try to tease apart how the natural dynamics of grassland and savanna ecosystems

are altered in the process of bush encroachment to provide a better understanding of the drivers. We also attempt to provide a clear explanation of a complex problem without oversimplification.

Chapter 4 presents high-level estimates of the likely order of magnitude of different *costs and benefits of bush encroachment*, taking bioregion and socio-economic settings into account as far as possible. This analysis is framed in terms of the capacity of landscapes to generate different ecosystem services. The chapter concludes with our overall *resolution* on whether bush encroachment should be considered as a means for carbon sequestration or as an ecosystem degradation.

Chapter 5 reviews the *potential remedies* for dealing with bush encroachment, and **Chapter 6** reviews *policy and legislation* pertinent to the treatment of bush encroachment.

Following this, **Chapter 7** evaluates *alternative policy scenarios* – laissez faire, land management interventions only, and manual or mechanical clearing in addition to land management, by generating ball-park estimates of the potential costs and benefits of these options in different ecoregions and under different land tenure.

Finally, **Chapter 8** presents the recommendations from the study.

THE NATURE AND EXTENT OF BUSH ENCROACHMENT IN SOUTH AFRICA

2.1 DEFINING BUSH ENCROACHMENT

The increase in woody biomass in grassland and savanna ecosystems has been defined in various ways by different authors (see Table 2.1 for a list of definitions). In this report we use the following definition:

“Bush encroachment is the process whereby the cover of indigenous woody plants (trees and shrubs) in a grassy ecosystem (savanna or grassland) increases substantially relative to the indigenous woody cover of some (historical) reference state.”

TABLE 2.1. Published definitions of bush encroachment, bush invasion and woody encroachment.

TERM	DEFINITION	REFERENCE
Bush encroachment	Stands of plants listed as indicators of woody encroachment in which individual plants are separated by a distance of less than three times the mean crown diameter.	Conservation of Agricultural Resources Act, 1983 (Act No. 43 of 1983) (DoA 1983)
	The directional increase in the cover of indigenous woody species in savanna ecosystems.	O'Connor et al. (2014)
	An increase in the biomass and abundance of woody species and the suppression of perennial grasses and herbs.	Oldeland et al. (2010)
	The transition from a grass-dominated vegetation community to a woody-dominated vegetation community.	Doudill et al. (1998)
	The conversion of savannas to dense, acacia-dominated thickets with little grass cover.	Meik et al. (2002)
	The suppression of palatable grasses and herbs by encroaching woody species.	Ward (2005)
	The invasion of grassland by woody plants or the increase in woody biomass in savanna woodlands.	Meadows & Hoffman (2002)
	The proliferation of woody plants (usually unpalatable) which suppresses herbaceous plants and reduces grazing capacity.	Smit et al. (1996); Wiegand et al. (2005)
Woody plant encroachment/bush encroachment	The proliferation of woody plants at the expense of grass in the savanna and grassland biome.	Wigley et al. (2009)
Bush/brush encroachment	The degradation of herbaceous cover and progressive increase in woody plant density.	Scholes & Archer (1997)
Bush invasion	The establishment of indigenous woody species in formerly treeless grassland ecosystems.	O'Connor et al. (2014)
Encroachment	The increase in density, cover and biomass of indigenous woody or shrubby plants.	Eldridge et al. (2011), Van Auken (2000; 2009)
Shrub encroachment	An increase in woody plant cover.	Tews & Jeltsch 2004
Woody encroachment	The increase in woody biomass, stem densities or woody cover.	Stevens et al. (2016)
	The changing balance in the proportions of trees and shrubs relative to grasses and herbs.	Devine et al. (2017)
Woody plant encroachment	The displacement of grasses by woody plants.	Hibbard et al. (2001)

Most definitions of bush encroachment refer to an increase in indigenous woody plants only and do not include invasion by alien trees and shrubs. In this study, we only consider bush encroachment by indigenous species. While the invasion of grasslands and savannas by alien woody plants can have similar consequences to those from bush encroachment by indigenous species, the causes are often different. The policy and management responses also differ. Invasion by alien woody species is known to be harmful and the problem is already being tackled through existing government and non-government programmes. However, there are divergent views about the impacts and management of indigenous bush encroachment, arising from different contexts and perspectives on the problem, and these issues will be further explored in order to determine a suitable policy response.

Bush encroachment also does not include the increase in woody cover that occurs during the natural recovery or restoration of degraded forest or thicket vegetation. For example, many parts of the Thicket Biome in the Eastern Cape have been degraded by goat farming into vegetation that often resembles savanna. The restoration of these areas to the dense, woody vegetation that existed previously provides numerous benefits and is pursued by large-scale restoration programmes (Mills et al. 2007, 2013).

2.2 EXTENT OF THE PROBLEM AND HOTSPOT AREAS

2.2.1 Estimates of spatial extent

There is considerable information on bush encroachment of indigenous species within South Africa (e.g. Devine et al. 2017; O'Connor et al. 2014; Stevens et al. 2016; Wigley et al. 2010) as well as in neighbouring countries such as Namibia, Botswana and Zimbabwe (Bester 1999; Kalwij et al. 2010; O'Connor et al. 2014). Estimates of the extent of bush encroachment have been made at various spatial scales using field studies, landscape photography (dating back to the 1800s), aerial photography (dating back to the 1940s), satellite data (which dates back to the 1970s) and modelling based on statistical analysis of existing information. Methods have also been devised for the continuous monitoring of bush encroachment using Google Earth images (Ludwig et al. 2016), and satellite data (Marston et al. 2017). No large-scale comprehensive, empirically-based estimates had been completed at the time of this study, but the preliminary results of

the Woody Biomass Project by SAEON (Warren, et al. 2018) were released as this study was being completed.

There are several localised estimates of the spatial extent of bush encroachment in South Africa (O'Connor et al. 2014). These are discussed in more detail in section 2.4. Relatively few attempts have been made to estimate or map the national scale of the problem (Table 2.2). Earlier estimates put the land area affected by bush encroachment at about 13 million ha in the 1960s (Van der Schijff 1964 in Kraaij & Ward 2006, 236), which is over 10% of South Africa's land area. The most recent estimate by Warren, et al. (2018) is 7.3 million ha, or 6% of the country.¹

More recent studies have used satellite data to estimate the extent of bush encroachment in South Africa. Stafford et al. (2017a) estimated the extent of bush encroachment at a national scale using land cover data from 2010 (previously available through SANBI BGIS, <http://bgis.sanbi.org/>). Areas of encroachment were defined as those where the percentage woody thickening was >20%, but it was not clear what method had been used to identify thickening. Their study also limited the analysis to arid savannas (<680mm rainfall), arguing that it would be difficult to distinguish bush encroachment from natural closed canopy formations in mesic savannas. Stafford et al. (2017a) produced a conservative estimate of bush encroachment of 8 million ha, which is about 6.5% of South Africa's land area. Based on their relatively simple mapping method, Stafford et al.'s (2017a) analysis suggests that bush encroachment is a particular problem in Limpopo, northern parts of the North West and Mpumalanga provinces, northern Kwazulu-Natal and parts of the Eastern Cape (Figure 2.1).

1 Warren, K., Hugo, W. and Wilson, H. 2018. Preliminary report and data on bush encroachment and land cover change, released to DEA, DEA consultants, and selected collaborators. Results are subject to quality assurance and review.

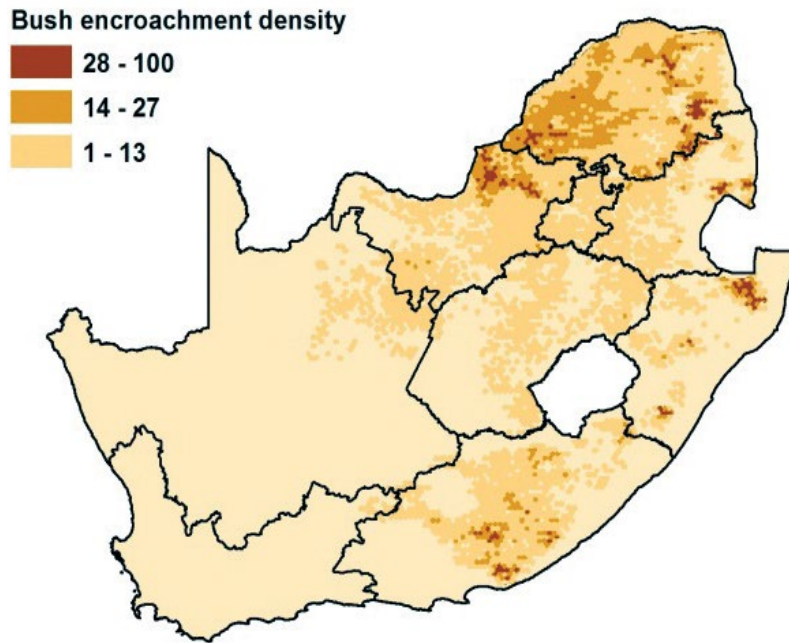


FIGURE 2.1. The extent and density (t/ha) of woody plants from bush encroachment.
 Source: Stafford et al. 2017a

TABLE 2.2. Summary of studies estimating the extent of bush encroachment in South Africa

REFERENCE	METHOD	EXTENT OF ENCROACHMENT	COMMENT
Van der Schijff 1964	Unknown*	13 million ha	Area 'affected'
Stafford et al. 2017a	Not reported	8 million ha	Areas of >20% increase in woody cover
Skowno et al. 2016	Landsat data, change in seven vegetation categories	5.7 million ha	Grassland to woodland only
Warren et al. 2018	Uses a number of sources including Skowno above	7.3 million ha	Vegetation changes

*Cited in Kraaij & Ward (2006)

Skowno et al. (2016) used multi-season Landsat data in conjunction with satellite L-band radar backscatter data to estimate the extent of woodlands and grasslands in South Africa in 1990 and 2013. Their study suggested that over this 23-year period, woodlands have replaced grasslands over about 5.7 million ha and that grasslands have replaced woodlands over about 3 million ha (Figure 2.2). Areas with over 500 mm mean annual precipitation had higher rates of woodland expansion than lower rainfall areas. Their study does not consider areas where there has been encroachment within the woodland areas, however.

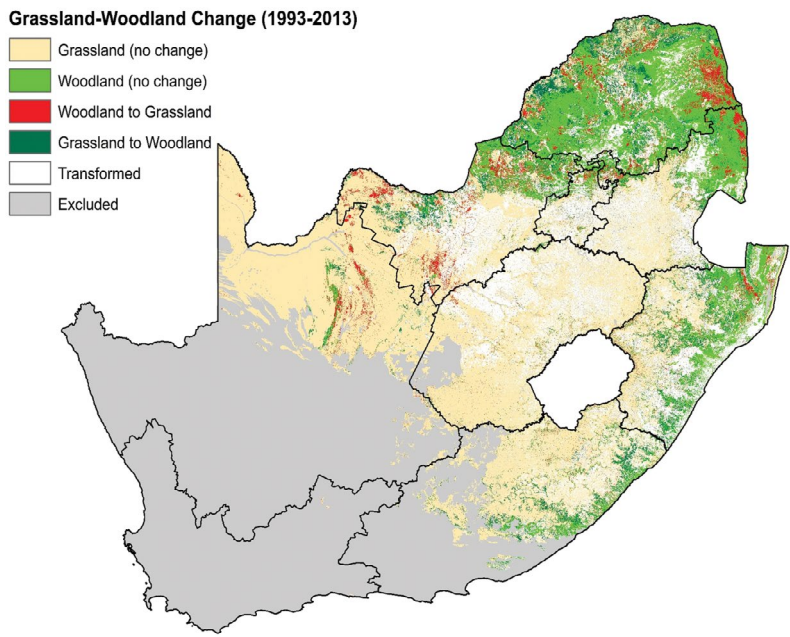


FIGURE 2.2. Estimated change in woodland and grassland extent between 1990 and 2013 in the grassy biomes of South Africa. Source: Skowno et al. (2016)

In Mucina & Rutherford’s (2006) mapping and description of South Africa’s vegetation types, their notes sometimes give an indication of where bush encroachment has been observed and where it potentially might be occurring. Skowno (in a subsequent, unpublished study; in litt) used these notes to categorise vegetation types into those where bush encroachment has been observed, is probable or possible (Figure 2.3). The overall extent of these vegetation types does not differ significantly from bush encroachment recorded from satellite analyses (Stafford et al. 2017a, Skowno et al. 2016), but there are some notable differences. The analysis based on Mucina & Rutherford (2006) suggests further extension of bush encroachment into the Kalahari and includes all savanna and grassland vegetation types. While this analysis does not have the resolution to determine actual extent within these vegetation types, it adds useful additional perspective in that it is likely to consider changes over a longer period than the satellite-based analyses.

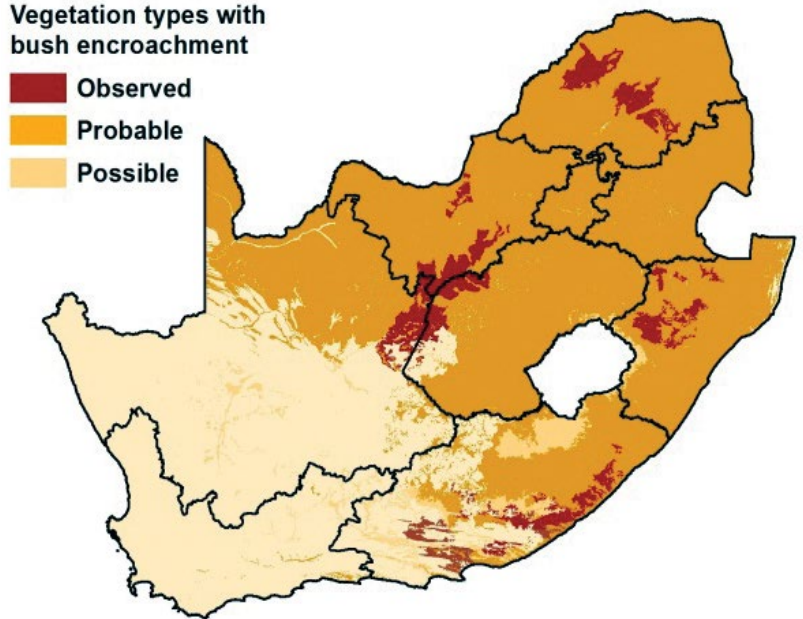


FIGURE 2.3. The potential extent of bush encroachment based on the mention of bush encroachment in descriptions of vegetation types in Mucina & Rutherford (2006), compiled by Skowno (unpublished data).

The differences between the mapped outputs of the above studies are largely related to the different approaches taken. A critique of Stafford et al's (2017a) study is difficult because of the lack of information on the mapping technique used. Their description makes it appear that they do not distinguish between the natural woody component in savanna systems and bush encroachment. As such the detail of their mapping should be used with caution. Skowno et al. (2016) were only able to detect bush encroachment where it has resulted in a shift from grassland to woodland. This technique is unable to distinguish densification within areas already classed as woodland, not shifts that do not warrant the area being classed as 'woodland' (namely, encroachment of small woody shrubs in the grassland biome and the earlier stages of bush encroachment where the canopy cover is still low. The approach based on Mucina & Rutherford (2006) used notes that were not explicitly set out to determine the extent of bush encroachment. The delineation of these vegetation types was also not based on bush encroachment, so in many cases although bush encroachment may occur in parts of given vegetation types, mapping the entire unit may over-estimate its extent. This is particularly obvious for some azonal vegetation types which occur across large parts of the country in a discontinuous manner.

More detailed mapping that considers the strengths and weaknesses of the above approaches is necessary for fine-scale mapping of the extent of bush encroachment across South Africa. In addition to SAEON's Woody Biomass

Project (see above), the CSIR is also currently mapping bush encroachment across South Africa using much higher resolution satellite data. These data, once available, will hopefully yield a much clearer understanding of where bush encroachment is occurring across South Africa. In the meantime, examination of the available data sources gives a good indication of the broad extent at national scale, although caution must be exercised when focusing in on particular areas and the exact placement of boundaries.

Some broader-scale studies have used a modelling approach to estimate the current, past and future extents of bush encroachment. Moncrieff et al. (2015) estimated the extent of grassland and savanna biomes in 1900 and 2100, from a base year of 2012. Their study suggested that woodlands have already replaced significant areas of grasslands over the past century, namely more than has been observed in recent decades, and that future changes over the remainder of this century will be significantly greater (Figure 2.4). It should be noted that the study focused on impacts of climate change, and is likely to have underestimated the role of other limiting factors in the assumptions about plant responses to increased availability of atmospheric CO₂, namely overestimating the expansion of savannas. However, the study did not consider other drivers of bush encroachment relating to land management. While the estimates are not considered particularly reliable, particularly in terms of the rates of change, they do perhaps indicate where bush encroachment might eventually occur.

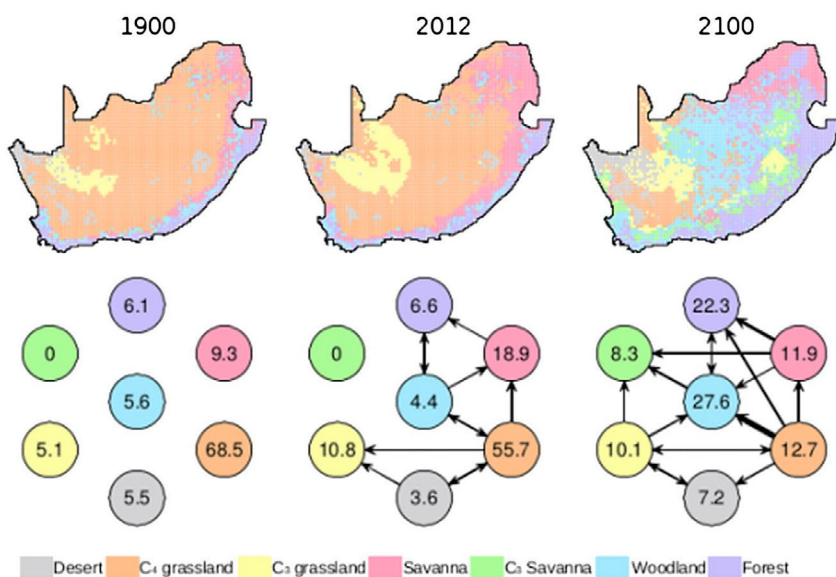


FIGURE 2.4. The distribution of South African biomes in 1900, 2012 and 2100 simulated using an adaptive dynamic global vegetation model (ADGVM) (Source: Moncrieff et al. 2015).

Graw et al. (2016) investigated the use of multiple datasets derived from earth observation data to detect and predict bush encroachment at the scale of the African continent (Figure 2.5). Their work resulted in a probability map derived from a logistic regression of land cover and other variables. Nevertheless, the authors concluded that more field data would be needed to do this more accurately.

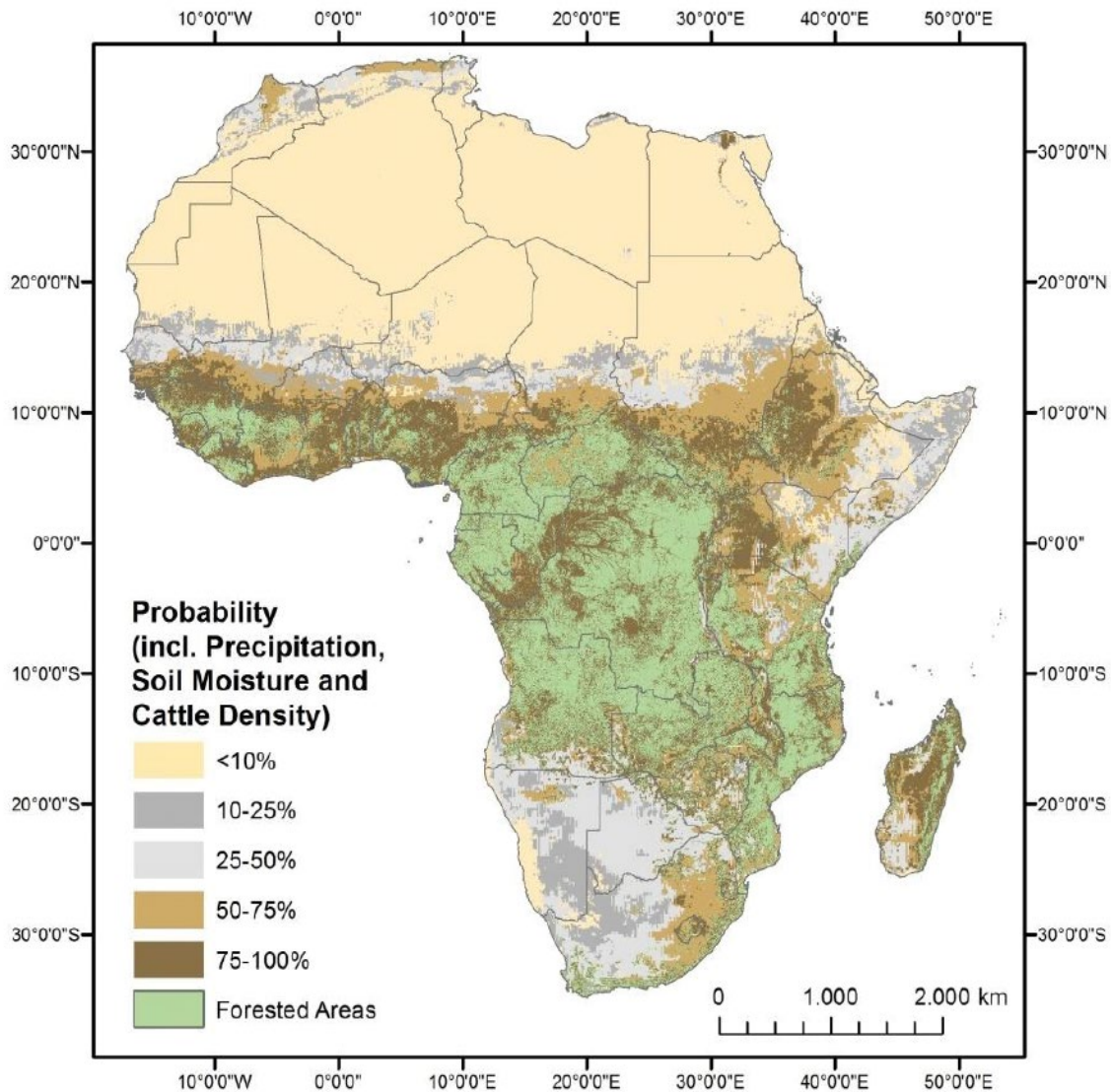


FIGURE 2.5. Probability of bush encroachment based on the prediction for woody vegetation of 2007 and 2009 including three main impact variables: precipitation, soil moisture and cattle density under consideration of land cover (Source: Graw et al. 2016)

2.2.2 Defining the baseline

Determining the extent of bush encroachment that has occurred to date requires defining a starting point, baseline or 'natural' condition. This is the case for any assessment of ecosystem health. However, the fact that savanna ecosystems are disturbance-driven and have endured many anthropogenic influences over the past centuries, makes the identification of a baseline particularly difficult (Van Langevelde et al. 2003).

The mapping studies described above all use a relatively short time period for their analyses, with a starting point in 1990 in the case of Skowno et al. (2016). Thus, their analyses only capture the extent of invasion after 1990, and give no indication of the encroachment that occurred prior to this date, despite the understanding that bush encroachment has been occurring for at least five decades. Data that precede these satellite data include local-scale data such as old photographs as well as aerial photography, of which country wide surveys exist dating back to the 1940s. Ongoing research through the rePhotoSA project (rephotosa.adu.org.za) through comparing recent photographs of landscapes to historic photographs has been able to quantify the increase in woody plant encroachment across time steps from 1940–2008 (Ward et al. 2014). Encroachment occurred at relatively low rates during 1940–1964, 1964–1973 and 1973–1993, but at much higher rates between 1993 and 2008, and with more instances of complete encroachment (Ward et al. 2014). This suggests that the problem has been increasing exponentially, and that it is possible that the bulk of the encroachment has happened in the past 30 years. It also suggests that the aerial photographs from the 1940s would make an adequate baseline, and that estimates made from more recent satellite data are fairly accurate for the most part.

Being able to define what the system was like before bush encroachment would not only help to clarify our understanding of the changes, but would be useful in setting restoration goals for the purpose of meeting biodiversity conservation targets. Beyond this, however, the way in which bush encroachment is dealt with is perhaps more of a choice for society based on our understanding of the implications of woody plant density, and is less dependent on knowing what the natural condition was. Deciding on a response to bush encroachment will therefore need to take a range of issues into account, including the demand for different types of ecosystem services.

2.2.3 Hotspot areas

Identifying hotspots of bush encroachment is fraught with difficulty in the face of incomplete and insufficient data. From the Skowno et al. (2016) study, areas with a concentration of patches of changes from grassland into woodland can be identified. These exist within the Limpopo Province, especially along the border with Botswana. Other concentrations include the Vryburg and Kuruman areas, northern KwaZulu-Natal and along the Eastern Cape coast. However, these represent hotspots in terms of encroachment into the grassland biome, and do not include densification hotspots within existing savanna areas. The areas of greatest bush encroachment identified by Stafford et al. (2017a) occur in generally similar areas, but are not always in agreement. Thus, rather than identifying hotspots per se, we have presented available data on the extent of bush encroachment throughout South Africa as well as the distribution of the main culprit species and some of the dynamics and species involved in each area.

2.3 THE MAIN SPECIES INVOLVED

Bush encroachment involves the proliferation of woody species that naturally occur in savanna ecosystems. Thus, it is not surprising that over 40 species have been listed as being part of the bush encroachment problem in South Africa (Table 2.4), and that new species keep being added to this list (for example, for the Durban area, Table 2.3). It is important to keep in mind that bush encroachment is not caused by particular species, but is rather a change in balance of the broader phenotypes of plants occurring in ecosystems. Nevertheless, some species respond to the drivers of bush encroachment more prolifically than others, and may be identified as the “chief culprits” in a particular area. What makes some species more responsive than others is likely to be linked to their life history characteristics and the spatial context. However, where ‘chief culprits’ can be identified, caution should be used in how they are tackled, since they are likely to be benign and useful in their natural densities.

In southern Africa, the main encroacher species include the mouse bush, *Vachellia hebeclad*, sweet thorn (soetdoring) *Vachellia (Acacia) karroo*, scented thorn *V. nilotica*, umbrella thorn *V. tortilis*, blackthorn *Senegalia (Acacia) mellifera* and sicklebush *Dichrostachys cinerea*, as well as driedoring *Rhigozum trichotomum* and wild camphor bush *Tarchonanthus camphoratus* in the Northern Cape (O’Connor et al. 2014). These main species tend to dominate in different areas.

Whereas sickle bush has been a conspicuous encroacher species throughout southern Africa, its dominance is replaced by sweet thorn in the Eastern Cape and much of KwaZulu-Natal, and the paperbark thorn *Vachellia (Acacia) sieberiana* in the mesic grasslands of KwaZulu-Natal. The black thorn is the main encroacher species of arid savannas areas (<400 mm of rainfall per annum; O’Connor et al. 2014). The main bush encroachment species in South Africa are highlighted in bold in Table 2.4. Among these, the most notable are the sickle bush, black thorn and sweet thorn.

Although a number of the main culprit species share many traits and respond generally to the main drivers of bush encroachment there are nuances specific to each species that make them more likely to become problem species in different parts of the country and/or respond differently to interventions. An understanding of the life-history of the species that is involved in bush encroachment at site level may help landholders to manage them more effectively. While a detailed study of every species involved in bush encroachment is beyond the scope of this study, Box 2.1 provides a summary of the distribution and life history traits of some of the more common encroacher species.

TABLE 2.3. Encroacher species listed by Ethekewini Municipality (Durban) (eThekewini ECOFILES Sheet 3)

BOTANICAL NAME	COMMON NAME	BOTANICAL NAME	COMMON NAME
<i>Pteridium aquilinum</i>	Bracken Fern	<i>Strelitzia nicolai</i>	Natal Wild Banana
<i>Dalbergia obovata</i>	Climbing Flat-bean	<i>Trema orientalis</i>	Pigeonwood
<i>Brachylaena discolor</i>	Coast Silver Oak	<i>Dichrostachys cinerea</i>	Sickle Bush
<i>Maesa lanceolata</i>	False Assegaiwood	<i>Antidesma venosum</i>	Tassel Berry
<i>Albizia adianthifolia</i>	Flat-crown	<i>Clerodendrum glabrum</i>	Tinderwood
<i>Bridelia micrantha</i>	Mitzeeri	<i>Phoenix reclinata</i>	Wild Date Palm

TABLE 2.4. Declared indicators of bush encroachment in South Africa (The Conservation of Agricultural Resources Act, 1983 (Act No. 43 of 1983); North West Provincial Government 2008. Environment Outlook: A report on the state of the environment 2008). Note that *Vachellia* and *Senegalia* were formerly *Acacia* and *Seriphium plumosum* was formerly *Stoebe vulgaris*. Western Cape not included in tables in The Conservation of Agricultural Resources Act, 1983.

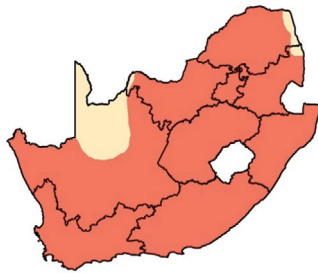
BOTANICAL NAME	COMMON NAME	PROVINCE							
		G	K	M	EC	L	NC	F	NW
<i>Asparagus spp.</i>	Wild asparagus							x	x
<i>Azima tetraantha</i>	Needle bush		x	x	x	x			
<i>Colophospermum mopane</i>	Mopane					x			
<i>Combretum apiculatum</i>	Red bush-willow	x	x	x		x			x
<i>Commiphora pyracanthoides</i>	Cork tree, Common corkwood						x		
<i>Dichrostachys cinerea</i>	Sickle bush	x	x			x			x
<i>Diospyros lycioides</i>	Blue bush		x	x		x			x
<i>Dodonaea angustifolia</i>						x			x
<i>Euclea crispa</i>	Blue guarri	x	x	x	x	x	x	x	x
<i>Euclea divinorum</i>	Magic guarri		x	x		x			
<i>Euclea undulata.</i>	Common guarri	x	x	x	x	x	x	x	x
<i>Grewia bicolor</i>	Bastard raisin bush			x			x		x
<i>Grewia flava</i>	Wild raisin, Velvet raisin	x	x	x		x	x		x
<i>Grewia flavescens</i>	Rough leaved/Sandpaper raisin		x			x			x
<i>Grewia monticola</i>						x			
<i>Leucosidea sericea</i>	Old wood	x	x	x	x	x	x	x	x
<i>Lopholaena coriifolia</i>	Lopholaena	x	x			x			x
<i>Maytenus polyacantha</i>	Kraaldoring					x			
<i>Maytenus senegalensis</i>	Red spike-thorn		x						
<i>Rhigozum trichotomum.</i>	Three-thorn rhigozum/Driedoring						x		
<i>Senegalia ataxacantha</i>	Flame thorn	x	x	x	x	x			
<i>Senegalia caffra</i>	Common hook-thorn	x		x		x			x
<i>Senegalia mellifera</i>	Black thorn	x				x			x
<i>Senegalia nigrescens</i>	Knob thorn		x	x		x			x
<i>Senegalia senegal</i>	Three-hook thorn, Three-thorned Acacia		x	x		x			
<i>Seriphium plumosum</i>	Bankrupt bush							x	x
<i>Strychnos madagascariensis</i>	Black monkey orange		x	x		x			x
<i>Tarchonanthus terilizati</i>	Camphor bush, Sagewood						x		x
<i>Terminalia sericea</i>	Silver cluster leaf	x	x	x		x	x		x
<i>Vachellia borleae</i>	Sticky thorn		x	x					
<i>Vachellia erubescens</i>	Blue thorn	x		x		x			x
<i>Vachellia exuvialis</i>	Flaky thorn			x		x			
<i>Vachellia fleckii</i>	Plate thorn					x			x
<i>Vachellia gerrardii</i>	Red thorn		x	x		x			
<i>Vachellia grandicornuta</i>	Horned thorn		x	x		x			
<i>Vachellia hebeclada</i>	Mousebush, Candle thorn	x				x		x	x
<i>Vachellia karroo</i>	Sweet thorn/Soetdoring	x	x	x	x	x		x	x
<i>Vachellia luederitzii luederitzii</i>	False umbrella thorn					x			x
<i>Vachellia luederitzii retinens</i>	Belly thorn		x	x		x			x
<i>Vachellia nilotica</i>	Scented thorn, Redheart	x	x	x		x			x
<i>Vachellia permixta</i>	Slender thorn					x			
<i>Vachellia sieberiana</i>	Paperbark thorn		x	x		x			
<i>Vachellia tenuispina</i>	Fyndoring	x				x			x
<i>Vachellia tortilis</i>	Umbrella thorn, Curly pod Acacia	x	x	x		x	x	x	x

BOX 2.1. Characteristics of some of the species most commonly associated with bush encroachment throughout its range in South Africa.



Sweetthorn /Soetdoring
Vachellia (Acacia) karroo
(Photo: Wikimedia Commons)

- Shrub to medium tree



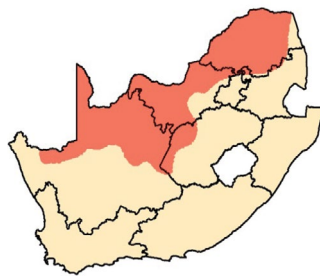
Mainly a problem in Eastern Cape and KZN – thickening and densification occurring

- Produces many long-lived seeds, can respond immediately to good rainfall
- Seedlings can emerge years after adults removed
- Seedlings struggle to establish in dense healthy grassland
- Reduced grass cover assists recruitment
- Re-sprouts following fire, but smaller individuals killed
- Fire frequency important in determining population structure
- Browsing post fire can help control regrowth



Black thorn
Senegalia (Acacia) mellifera
(Photo: Wikimedia Commons)

- Small tree



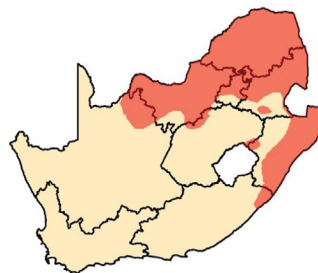
Northern Cape arid savanna

- Episodic increases in population
- Seeds not long-lived, mostly not viable during dry years
- Requires a series of good rainfall for natural recruitment
- Re-sprouts following fire, but smaller individuals killed
- Overgrazing and sparse grass may assist germination but absence of fire important in survival



Sickle bush
Dichrostachys cinerea
(Photo: Wikimedia Commons)

- Shrub to small tree



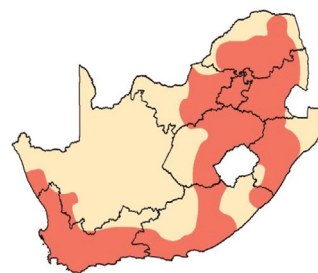
Mesic and lowland savannas North-West Province, Mpumalanga and KwaZulu-Natal

- Can reproduce by clonal growth (root suckering) as well as by seed
- Can continue to produce clonal following frequent fires for many years
- Has persistent hard seeds, dispersed by animals
- Fire alone doesn't seem to be able to control species, browsing post fire could help control regrowth



Bankrupt bush
Seriphium plumosum
(Photo: Wikimedia Commons)

- Dwarf woody shrub



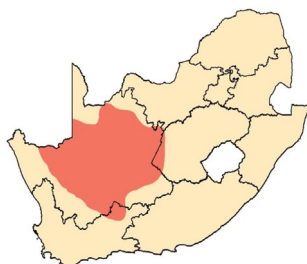
Problem in grasslands of Free State, extending into North-West province, as well as parts of Limpopo, Mpumalanga.

- Seeds dispersed long distances by wind
- Easily invades fallow areas with degraded soils where it can outcompete grasses
- Adults fire resistant and fire stimulates seed germination
- Sheep and browsers may help suppress species, but shift to cattle may help proliferate
- However, requires grass cover for seedlings and does not do well in overgrazed areas.



Driedoring
Rhigozum trichotomum
(Photo: Wikimedia Commons)

- Woody shrub



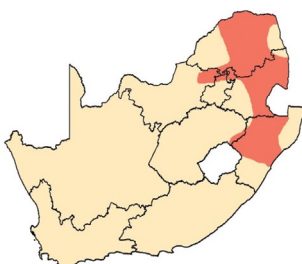
Problem around Kimberley and Gordonias on edge of savanna

- Can spread through both vegetative propagation and seeds
- Extremely drought resistant, can respond to rainfall quickly
- Can respond quickly to rainfall and overgrazing
- Mainly Karoo species, but exists on edge of dry savanna
- Related to overgrazing and loss of mixed-species perennially grass species



Paperbark thorn
Vachellia (Acacia) sieberiana
(Photo: Wikimedia Commons)

- Medium to large tree



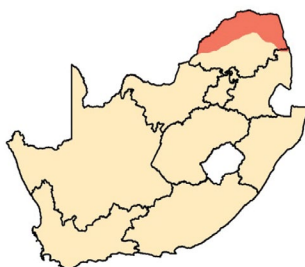
Invading mesic grassland in KZN

- High seedling emergence after high intensity fires
- Plants >2 yrs resistant to intense fire
- Grass competition important to establishment
- Removal of grass by fire or intense grazing can increase seedling recruitment



Mopane
Colospermum mopane
(Photo: Wikimedia Commons)

- Medium tree



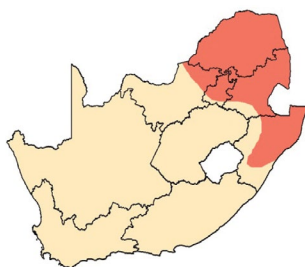
A problem throughout its distribution in SA, generally increase in density rather than expansion

- Can create monospecific stands with herbaceous understorey
- Can regrow/coppice from cut stumps
- Overgrazing decreases competition with herbaceous layer
- Can produce dwarf multi-stemmed trees in response to soils or heavy browsing by large herbivores such as elephants and eland
- Exclusion of sporadic hot fires might increase density



Red bush-willow
Combretum apiculatum
(Photo: Wikimedia Commons)

- Small to medium tree



Mesic lowland savanna including Kruger NP extending into Limpopo Province

- Unlike Vachellia and Senegalia species, is non-leguminous
- Heavy browsing can reduce bush canopy and density, but also related to climatic conditions
- Increased interval between fires leads to increased bush density
- Strong coppicing ability – becoming low-growing multi-stemmed plants after cutting

2.4 SPATIAL VARIATION IN BUSH ENCROACHMENT

Empirical studies of bush encroachment that have been carried out in southern Africa are not evenly distributed (Figure 2.6, O'Connor et al. 2014). Their distribution may in part be due to concentrations in areas where bush encroachment is particularly severe, but are also likely the result of other site selection biases. As a result, much of the research on bush encroachment and its drivers may be somewhat limited to certain types of bush encroachment such as those in the lowland and coastal savanna bioregions. More studies examining bush encroachment across its range (for example, Stevens et al. 2016) are required to ensure a thorough knowledge of bush encroachment within South Africa.



FIGURE 2.6. Location of study areas of academic papers on bush encroachment in southern Africa (O'Connor et al. 2014).

From these studies, it is clear that the nature of bush encroachment varies between different ecological regions and in different socio-economic contexts. Understanding this variation also helps to understand the drivers of bush encroachment. It is also expected that this variation will be important to consider, together with the land-use/land tenure context, when deciding on policies and remedies for dealing with it (see section 5). Thus, the areas vulnerable to bush encroachment in South Africa were divided into bioregional zones which are referred to in the subsequent parts of this report. These zones are largely based on Mucina & Rutherford's (2006) bioregions (Figure 2.7).

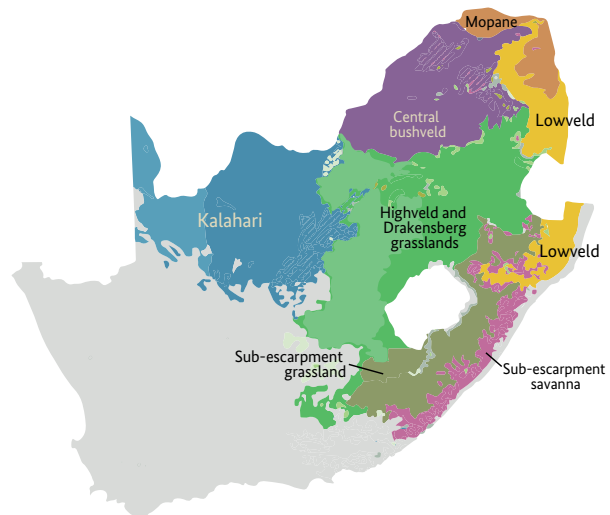


FIGURE 2.7. Bush encroachment zones defined in this study, in relation to biome and bioregion boundaries (Mucina & Rutherford 2006)

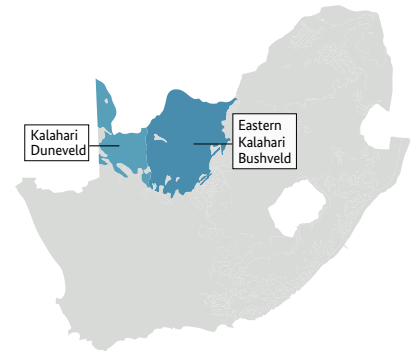
The way in which bush encroachment plays out in each of these zones is also influenced by anthropogenic factors. These include the interrelated factors of land tenure, population density and the predominant land use activities. In this regard, four types of human systems can be recognised in untransformed landscapes:

- Public and private conservation areas where there is little human encroachment and active management;
- Private, commercial rangelands carrying cattle, small stock or game.
- Communal land areas characterised by relatively high numbers of scattered homesteads, subsistence farming, natural resource harvesting activities and heavily utilised rangelands.

These contexts have an important bearing on the nature and intensity of the drivers that are described in the next sections, their consequences and also on the ways in which bush encroachment can potentially be managed. The ecological characteristics, nature and extent of bush encroachment are discussed for each of these zones below.

KALAHARI

The Kalahari xeric (arid) savanna region includes the Eastern Kalahari Bushveld and Kalahari Duneveld bioregions defined by Mucina & Rutherford (2006), and is shared with southeastern Namibia and southern Botswana. Most of this area lies within the Northern Cape Province and between 900 and 1500 metres above sea level (masl). This region is characterised by nutrient poor Kalahari sands, extreme temperatures, and low, infrequent rainfall of about 150–500 mm, with rainfall decreasing towards the west. Rainfall is mainly in the summer and autumn months, is patchy and may come in the form of violent storms. Winters are dry and prone to frosts. Some of the area is protected, with the remaining areas being heavily grazed.



In the drier areas of the Kalahari, there are relatively few trees, and these are mostly found along river courses. In the less arid areas, which are more prone to bush encroachment, the vegetation is open savanna with grasses (*Schmidtia* spp., *Stipagrostis* spp., *Aristida* spp., and *Eragrostis* spp.) interrupted by trees (for example, camelthorn *Acacia erioloba*, grey camelthorn *A. haematoxylon*, shepherd's tree *Boscia albitrunca*, false umbrella thorn *A. luederitzii*, black thorn *Senegalia mellifera*, and silver cluster-leaf *Terminalia sericea*) and shrubs (for example *Grewia flava*, *Ziziphus mucronata*, *Tarchonanthus camphoratus*, *Rhigozum trichotomum*, *Acacia hebeclada*, and *Lycium* spp., (van der Walt & le Riche 1999; Lovegrove 1993; Seymour n.d., Figure 2.8).



FIGURE 2.8. The Kalahari region is characterised by nutrient poor Kalahari sands, open arid savanna with grasses interrupted by trees such as camelthorn, false umbrella thorn, black thorn and silver cluster leaf. (www.istockphoto.com, www.flickr.com)

Over 90% of the Kalahari zone consists of untransformed land cover classes, made up almost entirely of low shrub landcover in the west transitioning to higher proportions of grassland and woodland/open bush in the east according to the 2013–2014 South African National Land-Cover Dataset (GEOTERRAIMAGE 2015). Towards the east there are also increasing pressures and land transformation through cultivated fields and some irrigated pivots near dams and river courses. Commercial agriculture makes up only 5% of the zone, with subsistence agriculture covering less than 0.5% of the land. Protected areas account for a large section of the north-west section of this zone (for example the Kalahari Gemsbok National Park), with a few smaller nature reserves occurring in the east as well as sporadically throughout the zone.

The main encroaching species in this region is the black thorn, with recorded increases in woody cover around Barkly West near Kimberley (Kraaij & Ward 2006) and another site 60 km west of Kimberley (Sirami *et al.* 2009). Black thorn tends to be the main encroacher species of savannas receiving less than 400 mm of rainfall per year on average (O'Connor *et al.* 2014). Other species include three-thorn rhizogium *Rhigozum trichotomum*, camphor bush *Tarchonanthus camphoratus* and wild raisin *Grewia flava* (Sirami *et al.* 2009, O'Connor *et al.* 2014). The three-thorn rhizogium has been recognised as an invasive shrub along the Savanna/Karoo boundary since the early 1980s (Vorster & Roux 1983) and is reportedly responsible for

thickening of an estimated 5.2 million hectares of veld in the area near Gordonia and Kimberly (Moore et al. 1988). Britz & Ward (2007) conducted a study in Pniel in the Northern Cape and found that black thorn (*S. mellifera*) cover increased by almost 35% over a period of 37 years, with an annual percentage change in woody cover of 0.25% (Table 2.5). Rohde & Hoffman (2012) studied vegetation change from 1876 to 2009 in southern Namibian communal and commercial rangelands and found that the overall percentage change in woody cover was higher on commercial rangelands with an annual change of 0.26% compared to 0.14% for communal rangelands (Table 2.5). The dominant encroaching species were *S. mellifera*, *S. erubescens*, *V. tortilis*, and *D. cinerea*.

TABLE 2.5. Summary of estimates of bush encroachment from the Kalahari zone

	ROHDE & HOFFMAN 2012	ROHDE & HOFFMAN 2012	BRITZ & WARD 2007
Study area	Orange River to Okahandja, Namibia	Orange River to Okahandja, Namibia	Pniel, Northern Cape
Land use	Communal	Commercial	Commercial
Period	1876-2009	1876-2009	1957-1993
Years	134	134	37
Starting woody cover (%)	14.7	8.9	31.1
Encroached % woody cover at end of study	33.1	43.7	40.0
Overall % change from starting to end	18.4	34.8	8.9
Annual change in woody cover (%)	0.14	0.26	0.25
Species	<i>S. mellifera</i> , <i>S. erubescens</i> , <i>V. tortilis</i> , <i>D. cinerea</i>		

CENTRAL BUSHVELD

The savannas of the central bushveld bioregion are moister than those in the Kalahari and typically occur on the plateau at 600–1800 m altitude in areas of rainfall between 500 and 900 mm per year. Rainfall occurs in summer with dry winters with infrequent frost events. The soils in these areas are generally nutrient poor, with laterites being prominent.

The vegetation of this bioregion is diverse but generally has a continuous and dominant grass layer interspersed with woody vegetation of varying heights and densities (Figure 2.9). The deciduous microphyllus trees such as *Vachellia* species (for example umbrella thorn *V. tortilis*, scented thorn *V. nilotica*, camel thorn *V. erioloba*, and sweet thorn *V. karroo*) are common and in some areas a few broadleaved tree species (for example red bush-willow *Combretum apiculatum*, wild seringa *Burkea africana*) also occur. The grass layer is dominated by tall perennial mesophytic species particularly of the Andropogoneae tribe. Moist savannas can be highly productive, but since the vegetation is of low nutritive value, indigenous ungulates tend to occur at low densities. However, South African moist savannas are less productive than their more tropical counterparts. These savannas normally burn annually during the winter dry season.



The central bushveld zone has over 80% untransformed land cover, predominantly woodland/open bush with pockets of grassland and occasional patches of dense bush and low shrubland (GEOTERRAIMAGE 2015). In this zone there are large tracts of both commercial and subsistence cultivated fields, together making up approximately 10% of the area. In the western areas of the zone there is increasing transformation from woodlots and plantations. High density urban and village settlements are located in this zone, especially near Gauteng as well as in the former homelands to the north-east. There are numerous small protected areas scattered throughout this zone. In the east, large parts of the zone fall within the Vhembe, Kruger to Canyons and Magaliesberg Biosphere Reserves.



FIGURE 2.9. The vegetation of the central bushveld is diverse and has a continuous dominant grass layer interspersed with woody elements of varying heights and densities. (<https://www.barewalls.com/posters-art-prints/bushveld.html>)

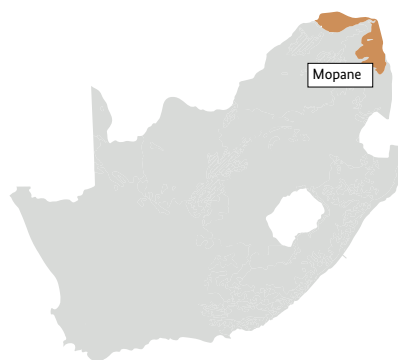
Bush encroachment in this region has mainly been attributed to sickle bush *Dichrostachys cinerea* (Hudak & Wessmann 2001). In addition, a number of other species, including black thorn, blue thorn *Senegalia erubescens*, plate thorn *V. fleckii*, and umbrella thorn, have also been recorded as contributing to increasing woody cover (van Vegten 1983). Bankrupt bush *Seriphium plumosum*, which is mainly a problem in the grassland biome, has also been recorded affecting farmlands in this zone (Avenant 2015). Van Vegten (1983) studied thornbush encroachment in the communal bushveld of eastern Botswana over a 25-year period, and found a 13% overall increase in woody cover (Table 2.6). O'Connor (2001) and Frost (1999) had similar findings for woody encroachment in *dambos* (vleis) in the savannas of Limpopo Province in South Africa and Matabeleland South Province in Zimbabwe, with an overall change in woody cover of 28% and an annual change in woody cover of 0.9% (Table 2.6). Overall change in woody encroachment was found to be lower in the Madikwe Game Reserve in North West Province compared to the other studies in this zone, with an annual percentage change of only 0.2% being recorded (Table 2.6, Hudak & Wessman 2001).

TABLE 2.6. Summary of estimates of bush encroachment from the central bushveld zone

	O'CONNOR 2001	VAN VEGTEN 1983	HUDAK & WESSMAN 2001	FROST 1999	
Study area	Venetia, Limpopo	Eastern Botswana	Madikwe, North West	Shangani Ranch, Zimbabwe	
Land use	Commercial: Dambo	Communal	Commercial / Protected	Commercial: Dambo	Savanna
Period	1955-1987	1950-1975	1955-1996	1963-1996	1963-1996
Years	33	25	42	34	34
Starting woody cover (%)	8.0	3.8	18.4	10.3	17.5
Encroached % woody cover at end of study	36.8	16.9	26.6	38.6	36.5
Overall % change from starting to end	28.8	13.1	8.2	28.3	19.0
Annual change in woody cover (%)	0.9	0.5	0.2	0.9	0.6
Species	<i>V. tortilis</i> , <i>V. nilotica</i> , <i>S. mellifera</i> , <i>S. erubescens</i> , <i>D. cinerea</i> , <i>A. fleckii</i> , <i>V. karroo</i>				

MOPANE

The Mopane bioregion occurs in the north of South Africa within the Limpopo Province and extending down to the Mpumalanga Provincial boundary within the Kruger National Park. This region lies between 200 and 600 masl, and experiences very hot temperatures in summer, but is generally frost-free during winter. This region receives between 300 and 650 mm of rainfall per year on average, mainly during summer months.



In these areas, mopane trees can form dense, mono-specific stands, or they occur in open mopane veld which also contains other species such as lowveld cluster-leaf *Terminalia prunioides*, wild raisin *Grewia flava* and red bush-willow *Combretum apiculatum*. In the more diverse parts of the bioregion, mopanes are usually found on clayey bottomlands whereas hillsides and uplands tend to be dominated by other species (Mucina & Rutherford 2006). In areas dominated by mopanes, two main types of vegetation structure can occur: open with short, multi-stemmed shrubs (1-2 m), or tall mopane woodland (Figure 2.10). These differences are related to soil characteristics and rainfall, with taller stands found in areas with deeper soil, higher rainfall, higher levels of Phosphorus (P), Potassium (K) and Calcium (Ca), and higher pH (Smit et al. 1996).



FIGURE 2.10. Two main types of mopane vegetation structure can occur in the mopane zone; tall mopane woodland or mopane woodland with short multi-stemmed shrubs (1–2 m). (www.nanzhila.com, blog.tracks4africa.co.za)

The mopane zone mostly consists of over 95% untransformed land cover classes, mainly grassland and woodland/open bush with patches of denser bush apparent (GEOTERRAIMAGE 2015). Scattered subsistence agriculture and villages exist mainly within the south eastern node of the zone. Large parts of this south-eastern section of the zone are within the Kruger National Park, with other small parks and reserves being found across the rest of the zone. The majority of the zone falls within the Vhembe and Kruger to Canyons biosphere reserves.

Only one local-level study has documented bush encroachment in this bioregion where overall percentage change in mopane cover was 20% over a 22-year period and annual change in woody cover was recorded to be 0.9% (O'Connor 1983, Table 2.7). Bush encroachment by mopane has, however, been recorded in both Namibia and Zimbabwe (Bester 1999; Clegg 1999) and mopane is listed as an encroacher species in South Africa (The Conservation of Agricultural Resources Act, 1983 (Act No. 43 of 1983).

TABLE 2.7. Summary of estimates of bush encroachment from the mopane zone

	O'CONNOR 1983
Study area	Venetia Limpopo
Land use	Commercial: Savanna
Period	1955–1977
Years	22
Starting woody cover (%)	59.0
Encroached % woody cover at end of study	78.8
Overall % change from starting to end	19.8
Annual change in woody cover (%)	0.9
Species	<i>C. mopane</i>

LOWVELD

The Lowveld bioregion spans the lower altitude areas of the east of the country from Limpopo Province down through Swaziland and into north-eastern KwaZulu-Natal. Altitudes range from 50 m in the south up to 1000 m at the base of the escarpment. Mean annual rainfall ranges from about 300 to 900 mm, and occurs mainly in summer. Winters are dry, but generally frost free. Some of the vegetation types within this zone are considered sourveld with deep, predominantly sandy to sandy-loam soils in the uplands, to clayey soils with a strong structure in the bottomlands. Vegetation consists of a mixture of open deciduous and semi-deciduous tree savanna, tall shrubland and occasionally dense thickets (Figure 2.11). Tall silver cluster-leaf *Terminalia sericea* and wild syringa *Burkea africana* are dominant in deep sandy soils. Marula *Sclerocarya birrea* and red bush-willow *Combretum apiculatum* are dominant in tall open shrubland. The herbaceous layer is dominated by relatively dense, tall, tufted grasses such as *Hyperthelia dissolute*, *Elionurus muticus* and *Hyparrhenia hirta*.

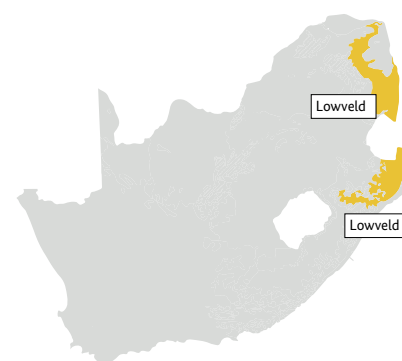


FIGURE 2.11. Vegetation in the Lowveld zone consists of a mixture of open deciduous and semi-deciduous tree savanna, tall shrubland and occasionally dense thickets. (www.kruger-lowveld-info.co.za, www.nightjartravel.com)

This zone comprises a broad mix of natural land cover classes including grassland, woodland, and dense bush which cover approximately 75% of the zone. However, a significant portion (16%) of the zone is transformed, including plantations, cultivated orchards, and sugar cane plantations. Protected areas cover large areas of the zone, including southern sections of the Kruger National Park and adjacent nature reserves. The southern parts of the Lowveld zone along the east coast of the country also contain many protected areas.

Bush encroachment, especially within the Kruger National Park, has mainly been attributed to red bush-willow *Combretum apiculatum*, silver cluster-leaf *Terminalia sericea* and sickle bush *Dichrostachys cinerea*,

which were found to comprise two thirds of the woody plants in sampled encroached areas (Smit et al. 2016). Excluding the Eckhardt et al. (2000) study, overall and annual woody cover in various protected areas (Roques et al. 2001; Gordijn et al. 2012; Wigley et al. 2010; Watson & MacDonald 1983, Table 2.8) increased from 36–47% with an average annual change in woody cover of 0.8% (Table 2.8). The overall change and annual change in woody cover in the Kruger National Park was found to be significantly lower than the other studies conducted in this zone, and in one of the areas, woody cover had in fact decreased over the study period (Eckhardt et al. 2000, Table 2.8).

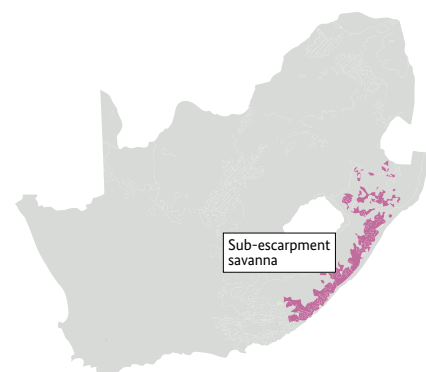
TABLE 2.8. Summary of estimates of bush encroachment from the lowveld zone.
P = Protected, CN = Communal, CM = Commercial

	ECKHARDT ET AL. 2000		ROQUES ET AL. 2001				GORDIJN ET AL. 2012	WIGLEY ET AL. 2010			WATSON & MACDONALD 1983	
Study area	Kruger National Park		Hlane Reserve in Swaziland				Ithala Game Reserve	Hluluwe-iMfolozi Park			Hluluwe-iMfolozi Park	
Period	1940–1998		1947–1997				1943–2007	1937–2004			1937–1975	
Years	59		51				65	68			39	
Land use	P		Overall	CN	P	CM	P	P	CM	CN	P	Control
Starting woody cover (%)	11.9	19.7	2.0	2.0	3.0	1.0	12.2	14.0	3.0	6.0	44.0	64.0
Encroached % woody cover at end of study	4.3	22.1	35.7	39.0	50.7	18.0	47.1	58.0	50.0	25.1	80.0	77.3
Overall % change from starting to end	-7.6	2.4	33.7	37.0	47.7	17.0	34.9	44.0	47.0	19.1	36.0	13.3
Annual change in woody cover (%)	-0.13	0.04	0.7	0.7	1.0	0.3	0.5	0.7	0.7	0.3	0.9	0.4
Species	<i>D. cinerea</i> , <i>V. karroo</i>											

SUB-ESCARPMENT SAVANNA

This bioregion occurs between the sub-escarpment grasslands and the coast, from KwaZulu-Natal into the Eastern Cape. These areas occur below 1000 m altitude and have an average annual rainfall of 550–1200 mm, mostly in summer. Frequent mist also provides an important additional source of moisture in this region. Soils in the region have minimal development and are usually shallow, especially on slopes, with some deep alluvial pockets occurring in valley bottoms.

Vegetation generally consists of deciduous trees of short to medium height dominated by *Vachellia* species. In addition to this thorny element, many areas also contain an evergreen component such as wild olive *Olea europaea*, shepherd's tree *Boscia albitruna* and blue guarri *Euclea crispata*. Some succulent plants such as *Euphorbia spp.* and *Aloe spp.* can occur and form dense stands in eroded soils. Dense, grassy undergrowth, generally considered sourveld is also characteristic of this area. Dominant grass species include *Themeda triandra* and *Astrida junciformis* (Figure 2.12).



This zone has the lowest percentage of untransformed land cover, with only 66% of the land being untransformed (GEOTERRAIMAGE 2015). This untransformed area is mainly made up of grasslands, some woodlands and dense bush land cover types. Almost 20% of the zone is occupied by village and subsistence land cover types, especially along the southern half of the coast. Along the northern half of the zone there are large, dense urban areas (especially surrounding Durban), cane farming and plantations. This zone has only a few small protected areas.

Sweet thorn *Vachellia karroo* has been identified as the most important encroaching species in the Eastern Cape and much of KwaZulu-Natal (O'Connor et al. 2014). Other species mentioned as contributing to bush encroachment in this region include *Scutia myrtina* (O'Connor & Crow 1999). Studies conducted in the Eastern Cape (O'Connor & Crow 1999; Puttick et al. 2011, 2014a; Shackleton et al. 2013; Table



Figure 2.12. Vegetation in the sub-escarpment savanna zone typically consists of short to medium deciduous trees, some evergreen trees, succulent plants and a dense grassy undergrowth. (www.ferndale4x4.co.za, Addo Elephant National Park: www.handluggageonly.co.uk)

2.9) found that overall woody cover increased on average by 15% and had an average annual change in woody cover of just 0.2%, significantly lower than the percentage change recorded by studies conducted in KwaZulu-Natal where overall change in woody cover was as high as 46% and average annual change was 0.85% (Table 2.9).

TABLE 2.9. Summary of estimates of bush encroachment from the sub-escarpment savanna zone. P = Protected, CN = Communal, CM = Commercial

	PUTTICK ET AL. 2014A		PUTTICK ET AL. 2011	O'CONNOR & CROW 1999	SHACKLETON ET AL. 2013	HOFFMAN & O'CONNOR 1999	HOGAN 1997
Study area	Grahamstown Eastern Cape		Bathurst Eastern Cape	Kei Road Eastern Cape	Willowvale Eastern Cape	Thukela River Basin KZN	Harold Johnson Nature Reserve, KZN
Land use	CM	CN	CN	CM	CN	Mixed	P
Period	1942–2004		1942–2004	1937–1986	1961–2009	1955–1998	1937–1993
Years	63		63	50	49	44	57
Starting woody cover (%)	11.7	6.4	91.9	17.0	14.0	21.0	37.0
Encroached % woody cover at end of study	18.5	11.8	98.6	35.0	29.0	59.0	82.5
Overall % change from starting to end	6.8	5.4	6.7	18.0	15.0	38.0	45.5
Annual change in woody cover (%)	0.1	0.1	0.1	0.4	0.3	0.9	0.8
Species	<i>V. karoo</i> , <i>S. myrtina</i> , <i>V. tortilis</i> , <i>V. nilotica</i>						

HIGHVELD AND DRakensBERG GRASSLANDS

Grasslands occur in summer rainfall areas above 1000 m where rainfall varies from 500 mm to over 1000 mm. C_4 grasses predominate, except at the highest altitudes where C_3 grasses are more common. Mucina & Rutherford (2006) divided South Africa's grasslands into five main bioregions: Dry Highveld grassland, Mesic Highveld Grassland, Drakensberg Grassland and Sub-Escarpment Grassland, and Coastal Belt Grasslands (Figure 2.7). This zone encompasses the first three of these bioregions.

The Dry Highveld bioregion grasslands are generally short plains, low-tussock grasslands and cover most of the western half of this zone (Figure 2.13). These grasslands fall into the 'sweet' grassland category and are dominated by chloridoid grasses. Dominant



species include *Themeda triandra* in natural conditions and *Eragrostic curvula* and *E. chloromelas* in degraded states. These areas are prone to being replaced by 'false' Karoo (namely low bushy vegetation) due to overgrazing of palatable grasses (Booyesen & Tainton 1984). The Mesic Highveld grasslands cover most of the eastern half of the zone where there is higher precipitation and are generally considered to be 'sour' grasslands dominated mainly by andropogonoides grasses (Mucina & Rutherford 2006). In these areas shrublands are generally found on rocky outcrops and the higher the surface rock cover the higher the proportion of woody cover compared with herbaceous vegetation (Mucina & Rutherford 2006).

The Drakensberg Grassland bioregion is generally associated with the escarpment near Lesotho but extends further south to the Amatole Mountains. These grasslands may contain heathlands on steep slopes and on mountain ridges and summits with high precipitations and occasionally snow. They contain significant proportions of geophytes and herbs in addition to grassy components (Figure 2.13). Dominant species include *Themeda triandra* and *Heteropogon contortus*.



FIGURE 2.13. Dry Highveld grasslands are typically low-tussock grasslands and the Drakensberg grasslands are situated on steep slopes and mountain ridges and contain significant proportions of geophytes, herbs and grassy components. (www.britannica.com/place/Highveld, www.mistypeaks.co.za)

Almost 70% of this zone remains untransformed, mostly comprising grassland with some sparser low shrubland cover classes towards the western edges. Most of the remainder has been transformed to cultivated fields and there is also a high density of urban areas in Gauteng. There are a few scattered small protected areas, mainly within the Drakensberg on the southern boundary with the sub-escarpment grasslands. In these grasslands, which mostly lie within the Free State, bankrupt bush *Seriphium plumosum* has been described as the most common encroacher species (Avenant 2015). The vegetation types affected by bankrupt bush in South Africa are widespread (Figure 2.14). Bankrupt bushes suppress the growth of natural vegetation and transform natural grassland areas into green bankrupt bush 'deserts'.

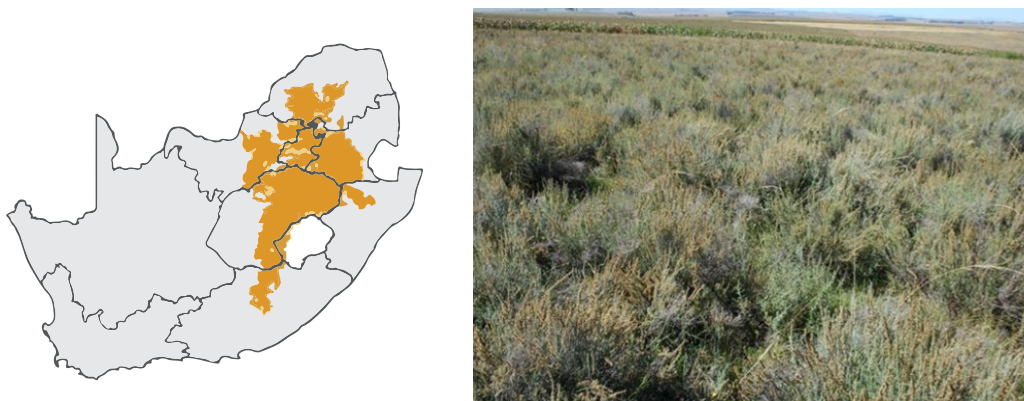
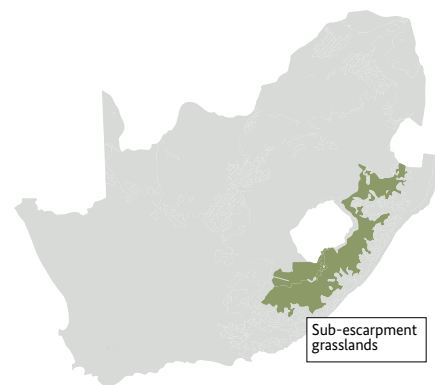


FIGURE 2.14. The area in South Africa affected by bankrupt bush *Seriphium plumosum*, and bankrupt bush encroaching into grassland. (Source: Avenant 2015)

SUB-ESCARPMENT GRASSLANDS

The sub-escarpment grasslands are found mainly in the KwaZulu-Natal midlands and Eastern Cape Province. These areas have warm, wet summers, high rainfall and frequent mists, and are also known as the mist-belt grasslands. They tend to be dominated by perennial grasses and forbs that are adapted to frequent disturbances such as fire (Figure 2.15). The soils in these areas are leached and are nutrient poor, mainly supporting sourveld. Many of the vegetation types in this region are classified as transitional vegetation between the higher altitude grasslands and lower altitude savannas. This tussock dominated sourveld is dominated by the grasses *Themeda triandra* and *Hyparrhenia hirta*. This bioregion also contains sparsely scattered shrubland and woody pockets including sweet thorn *Vachellia karroo*, scented thorn *V. nilotica* and common hook-thorn *V. caffra* and paperbark thorn *V. sieberiana*.



Approximately 75% of this zone is untransformed, and is dominated by grassland with small sections of wooded vegetation, especially towards the southern end of the zone on the boundary of the thicket biome (GEOTERRAIMAGE 2015). This zone contains numerous rural villages with subsistence agriculture covering almost 15% of the total zone area (Figure 2.15). Some commercial cultivation and plantations also exist in the northern sections of the zone.



FIGURE 2.15. Sub-escarpment grasslands of KZN and Eastern Cape tend to be dominated by perennial grasses and forbs. This zone contains numerous rural villages and mainly subsistence agriculture. (keywordsuggest.org, www.picemaps.com/south-african-eastern-cape/)

The main encroacher species in these grasslands are the paperbark thorn *V. sieberiana* and sweet thorn *V. karroo* (O'Connor et al. 2014, Russell & Ward 2016). Other woody species including (*Euclea crispata*, *Searsia spp.*, *Diospyros spp.* and *Grewia spp.*) have also been found to be encroaching along drainage lines (Russell & Ward 2016). Grellier et al. (2012) conducted a study in Potshini in KwaZulu-Natal and found that over a 65-year period woody encroachment increased by 45% with an annual change of 0.3% (Table 2.10). A study by Russell & Ward (2014) in Rorkes Drift in KwaZulu-Natal found that overall woody cover had increased by 75% over a 133-year period, with an annual change in woody cover of 0.3% (Table 2.10). In Fort Beaufort in the Eastern Cape, Puttick et al. (2014b) found that the change in woody cover was significantly higher on commercial land compare to communal land with an annual change of 0.15% on communal land and 0.8% on commercial land (Table 2.10).

TABLE 2.10. Summary of estimates of bush encroachment from the sub-escarpment grassland zone

	PUTTICK ET AL. 2014B		RUSSELL & WARD 2014	GRELLIER ET AL. 2012
Study area	Fort Beaufort, Eastern Cape		Rorkes Drift KZN	Potshini KZN
Land use	Communal	Commercial	Commercial / Communal	Communal
Period	1949–2004		1879–2011	1945–2009
Years	56		133	65
Starting woody cover (%)	48.0	28.0	6.0	0.0
Encroached % woody cover at end of study	56.0	70.0	75.4	45.3
Overall % change from starting to end	8.0	42.0	69.4	45.3
Annual change in woody cover (%)	0.15	0.8	0.3	0.3
Species	<i>V. karroo</i> , <i>V. ataxacantha</i> , <i>V. sieberiana</i> , <i>E. crista</i>			



2.5 SUMMARY OF BIOREGIONAL VARIATION

A summary of vegetation types, land use and woody encroachment in the seven zones is outlined in Table 2.11. There is considerable variation across these zones in terms of vegetation and land use variables as well as the types of encroacher species and the extent of woody encroachment (Figure 2.16, Figure 2.17).

The results from the bush encroachment monitoring studies as summarised by O'Connor et al. (2014) were used to determine average annual change in woody cover and the average change in woody cover over the monitoring period for each of the bush encroached zones (Figure 2.16, Figure 2.17). The approach used to determine woody cover in the Highveld and Drakensberg grassland zone was different to that used for the other zones, due to the nature of the dominant encroacher species, bankrupt bush *Seriphium plumosum*, which is a small shrub, unlike the common encroacher species found in the other zones. A study by Avenant (2015) on the extent and impact of *S. plumosum* (bankrupt bush) was used. The encroached land was classified in terms of plant density and plant density classes (25, 50, 75, 100% cover). The number of hectares for each density class was calculated and multiplied by the plant density class value, yielding the number of hectares encroached per each density class. By combining the number of encroached hectares, the total number of encroached hectares across the zone was estimated, and an estimate for the woody cover as a percentage of the total area based on the density of the bankrupt bush across the zone was calculated. Based on these assumptions it was estimated that the percentage woody cover of bankrupt bush has increased from 0% woody cover to 17% woody cover in the Highveld and Drakensberg grassland zones.

TABLE 2.11. Summary of vegetation types, land use and woody encroachment in the seven bush encroached zones.

BIOREGIONS	VEGETATION DESCRIPTION	PERCENTAGE UNTRANSFORMED	LAND USE	MAIN ENCROACHER SPECIES
SUB-ESCARPMENT GRASSLANDS	Mesic sourveld grasslands with orographic influences. Sparse and scattered woody patches and transitional zones towards sub-escarpment savanna.	75%	Important for commercial livestock production and large areas under communal rangelands. Afforestation and cultivation also important.	<i>Vachellia sieberiana</i> (paperbark thorn), <i>Vachellia karroo</i> (sweet thorn)
HIGHVELD & DRAKENSBERG GRASSLANDS	Sweetveld in the west of the zone and Sourveld grassland in the east. Woody components generally found on rocky outcrops.	67%	Important for commercial cultivation of crops and commercial livestock production. Small area under communal tenure or protected areas.	<i>Seriphium plumosum</i> (bankrupt bush)
SUB-ESCARPMENT SAVANNA	Moist savanna, dominated by deciduous <i>Vachellia</i> thorn trees plus some evergreen trees and succulents. Sourveld.	66%	Main land uses include communal subsistence agriculture and rangelands. Afforestation and commercial cultivation also important.	<i>Vachellia karroo</i> (sweet thorn)
LOWVELD	Moist savanna, mixture of open deciduous and semi-deciduous tree savanna, tall shrubland and some dense thickets.	76%	Large protected areas. Communal rangelands, commercial livestock/game farming. Afforestation and cultivation also important.	<i>Combretum Apiculatum</i> (red bush-willow), <i>Terminalia sericea</i> (silver cluster-leaf), <i>Dichrostachys cinerea</i> (sicklebush)
MOPANE	Moist savanna, dominated by mopane, tall, closed stands or open and mixed	95%	Large protected areas as well as communal rangelands, commercial livestock and game farming.	<i>Colophospermum mopane</i> (mopane)
CENTRAL BUSHVELD	Moist savanna, tall grasses with mainly deciduous microphyllus trees and some broad-leafed species.	83%	Predominantly commercial livestock and game farming. Some communal rangelands and scattered protected areas present.	<i>Dichrostachys cinerea</i> (sicklebush)
KALAHARI	Dry savanna, trees naturally relatively sparse, mainly thorn trees, also silver cluster-leaf and shrubs.	91%	Mainly used for commercial rangelands, including a few communal areas. Large arid protected areas.	<i>Senegalia mellifera</i> (black thorn)

The Mopane zone had the highest average rate of change in woody cover of 0.86% per year, followed by the Central Bushveld (0.61%), and then the Lowveld (0.51%). The Kalahari zone had the lowest average annual change in woody cover (0.22%).

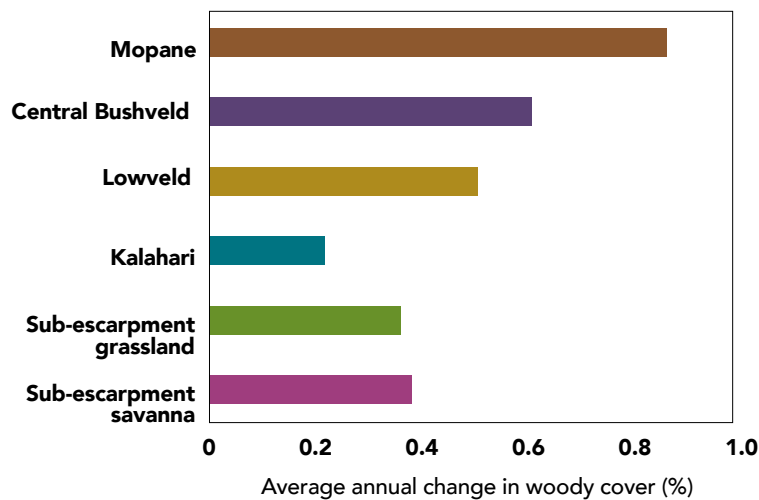


FIGURE 2.16. The average annual change in woody cover in each of the bush encroached zones based on studies summarised by O'Connor et al. (2014).

The average starting percentage woody cover varied considerably across the seven encroached zones, with the mopane zone having the highest average percentage woody cover at the start of monitoring but the lowest overall change in woody cover (Figure 2.17). The sub-escarpment grassland zone (41%) had the highest average change in overall woody cover from the start to the end of monitoring period, followed by the Lowveld zone which had an average overall change of 27%. The other zones had on average an overall change in woody cover of about 20%.

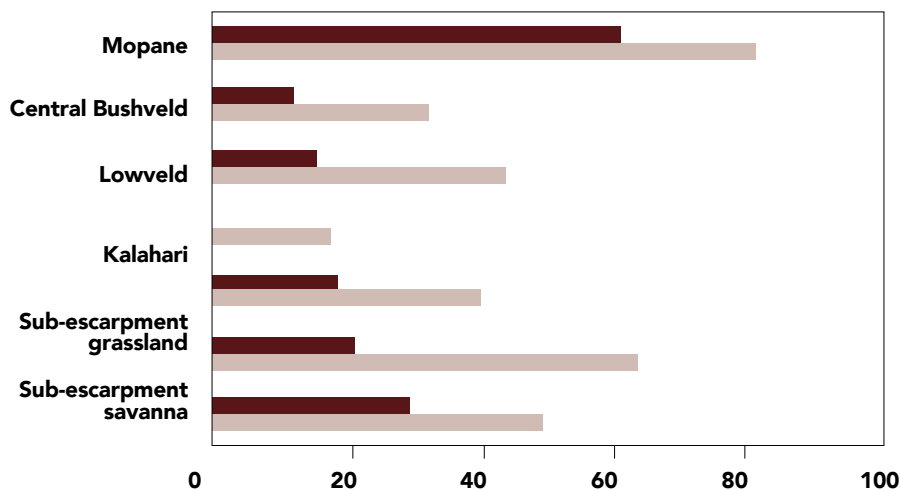


FIGURE 2.17. The average percentage woody cover at the start of monitoring compared to the average percentage woody cover at the end of monitoring based on a catalogue of studies in southern Africa as summarised by O'Connor et al. (2014).

Estimates of the area in each of the regions that has undergone bush encroachment between 1990 and 2013 (Table 2.12) was taken from preliminary data supplied by SAEON (Warren et al. 2018). These data considered shifts in land cover classes between the two periods with additional certainty estimates for different vegetation groups from Skowno et al. (2016). These data only consider recent encroachment and are therefore an underestimate of the full extent of encroachment which has been noted in all areas pre-dating 1990. Using the full extent of wooded land cover types as a maximum extent would likely yield an overestimate as there would have been an existing level of wooded areas prior to encroachment. In addition, this is likely an underestimate of encroachment in the highveld grassland region. To take this into account a preliminary estimate of the extent was taken as a mid-point between this estimate of 'recent' bush encroachment at the theoretical maximum level of encroachment (taken as percentage of area under woodland and dense bushland within each zone). For the highveld grassland region, estimates of the extent of encroachment were taken from Avenant (2015) and applied to estimated affected vegetation groups. These estimates provide a rough indication of the likely extent of the area which has undergone bush encroachment within each region, however, additional detailed mapping, using aerial photography, would be necessary to improve these figures.

TABLE 2.12. The encroached area (ha) per land tenure type and bioregion and the proportion of total bioregion area.

ZONE	ENCROACHED AREA (HA) PER LAND TENURE TYPE			PROPORTION OF TOTAL (%)
	PROTECTED	COMMUNAL	PRIVATE	
Mopane	257 558	103 732	292 517	35
Central bushveld	120 265	795 958	1 837 747	38
Lowveld	282 520	377 306	286 145	26
Highveld grassland	69 832	111 312	2 112 499	20
Kalahari	38 145	72 956	503 826	5
Sub-escarpment grassland	5 040	99 697	155 225	7
Sub-escarpment savanna	7 048	302 338	147 335	16

THE CAUSES OF BUSH ENCROACHMENT

3.1 AN OVERVIEW OF SAVANNA ECOSYSTEM DYNAMICS

Climate and soils broadly define the geographic range in which open grasslands and savannas can occur. Within these grassy ecosystems or 'rangelands' (a collective term given to open grassland, savanna and woodlands), a number of further ecological and human drivers further define the observed ratio of trees to grasses, and in certain cases, the associated phenomenon of bush encroachment. In order to understand bush encroachment, it is important to understand the dynamics of grassland-savanna ecosystems. This sub-section thus provides a brief overview of savanna ecology for the reader who may be unfamiliar with it.

While grasslands are largely treeless, woodlands exist in a continuum from scattered trees and shrubs to a dense cover of woody plants within a matrix of grass. Grasslands can be divided into 'true' or 'false' grasslands. 'True' grasslands are where grassland appears to be the climax community, whereas 'false' grasslands are where there is evidence that the climate will permit succession into shrubland, woodland or forest, and are maintained as grassland by factors such as grazing and fire or seasonal waterlogging (Booyesen & Tainton 1984). The 'true' grasslands tend to be the short, sour grasslands (where nutritive value decreases in winter) occurring at higher altitudes which are dominated by a diversity of C_3 grasses, whereas 'false' grasslands tend to be less sour grasslands at lower altitudes that are dominated by C_4 grasses. In areas of potential savanna, grasslands can be sour, mixed or sweet (nutrient value remains relatively constant through summer and winter), and tend to be sweeter towards the Karoo. Thus, while savanna has generally been equated with woodland, savanna can be considered to include any system with a continuous layer of C_4 grasses, regardless of whether trees are present (Lehmann et al. 2011).

Savannas are characterised by the coexistence of grasses (namely grasses and herbs) and trees (namely woody plants) and are the transitional biome between grasslands and forests. The grasses in these ecosystems tend to be highly shade-intolerant and fire-tolerant C_4 grasses and

C_3 trees. The co-existence of grasses and trees in savanna ecosystems and the factors that maintain this balance and stop these elements from outcompeting one another to form grasslands or forest ecosystems has been the subject of much research and modelling (for example Walker & Noy-Meir 1982; Scholes & Walker 1993; Higgins et al. 2000, House et al. 2003, Sankaran et al. 2005, Musil et al. 2009, Bond & Midgley 2012, Buitenwerf et al. 2012). As a comprehensive review of savanna ecology is beyond the scope of this report, we encourage readers to further explore the very extensive literature on the subject (for example Scheffer & Carpenter 2003; Scholes & Archer 1997; Scholes 2009).

In essence, the balance between grasses and trees largely boils down to how variation in rainfall and fire affect the competition between grasses and tree seedlings. In arid environments, grasses are better than tree seedlings in accessing water and outcompete them (Higgins et al. 2012). In less arid environments, however, tree seedlings may be able to reach deeper water than grasses through root growth. Fire is a particularly important driver. Savannas occur in areas where rainfall is highly seasonal, and where fire can occur once the grass layer that has built up during the wet season becomes dry enough to burn. The C_4 grasses are flammable and form the main fuel for fires in savannas, rather than woody biomass (Baudena et al. 2015). These grasses also recover quickly from fire (Higgins et al. 2007), whereas tree seedlings and some adult trees will be damaged or killed by fire (Sankaran et al. 2008, Higgins et al. 2000). Fire thus acts to keep savannas open. Increased rainfall and increased rainfall period thus move savannas towards closed forest systems, whereas lower rainfall, longer dry seasons and increased fire moves them towards grassland systems. In the absence of fire under mesic conditions, forest trees, which are more shade tolerant than savanna trees, can recruit through the grass layer and ultimately outcompete grasses and savanna trees. Indeed, Bond et al. (2003a) have suggested that most grassy biomes with >750 mm mean annual precipitation (MAP) in this region would switch to forest in the long absence of fire.

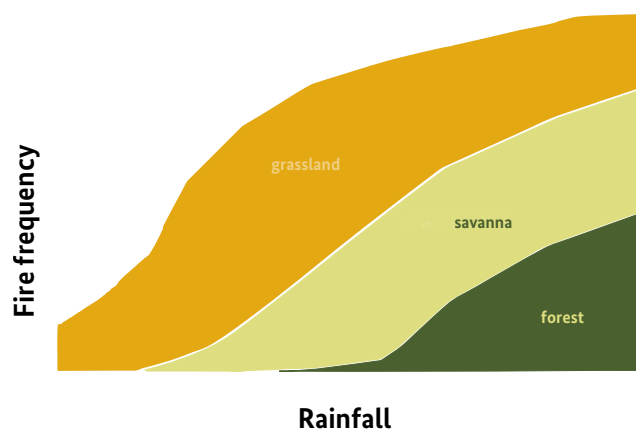


FIGURE 3.1. Simplified schematic diagram of the relationship between rainfall, fire frequency and woody biomass, all other things being equal. Fire frequency and intensity is driven by herbivory and the seasonality of the rainfall. (Source: Authors)

Thus, in general, a competitive relationship exists between trees and grasses in rangeland ecosystems (Scholes & Walker 1993). The nature of the relationship is strongly asymmetric where taller trees can have a direct suppressive effect on the grass layer below their canopy (Scholes 2009). In turn, the grass layer can affect the prevalence of trees, but indirectly through fire. Tree saplings perish if they are not large enough to escape the ‘fire trap’ created through the combustion of the grass layer.

The relative ability of the grass and tree layer to compete is further affected by a number of factors. These include the influence of soils, nutrients and atmospheric carbon dioxide (CO₂) on the relative growth rates of competing grasses and trees, and the influence of herbivory on the grass layer and fuel loads. These are, in turn, subject to a range of direct anthropogenic influences such as management of livestock and fires, as well as indirect influences such as climate change, which affects temperature, the amount and seasonality of rainfall and fire frequency.

In contemporary South Africa, there are few areas that are not affected by a degree of human intervention. An outcome of this is the observed ‘fence-line contrasts’ that are common throughout the country, which are the result of different forms of management practice in adjacent farming areas. Changes to the disturbance regime, rainfall and nutrients therefore need to be considered within the context of broader human-ecological systems, where land-use choices and associated management practices (that include the harvesting of woody biomass), lead to observed rangeland structure and the potential for bush encroachment (Scholes 2009).

3.2 HYPOTHESES ON BUSH ENCROACHMENT

Hypotheses regarding the encroachment of woody plants in grasslands and savannas build on the understanding about how the balance between trees and grasses in savannas is maintained. Modelling studies of bush-encroachment were initially resource-based (for example differential ability to compete for water and nutrients). Many of these models, such as the two-layer soil-water hypothesis (Walter 1939), were based on the separation of rooting niches for grasses and trees. There is limited support for the models based on rooting niches, since bush encroachment has been found to occur on soils too shallow to allow for root separation (Wiegand et al. 2005). In particular, root separation is unlikely to explain the initiation of bush encroachment, as germinating trees and grasses utilise the same soil layers (Ward 2005).

Following the criticism of the resource-based models, disturbance-based models were developed, including the patch-dynamic model (Ward 2005, February & Higgins 2010). These models demonstrated that drought, fire and grazing can interact to control the establishment and recruitment of trees. Models of grass and tree coexistence have been proposed based on tree seedling survival into the adult stage (Higgins et al. 2000), as well as the interaction between fire and herbivory (Van Langevelde et al. 2003). Many authors agree that overgrazing is a primary driver of bush encroachment, but they disagree as to the causal mechanism.

Some authors, such as Wiegand et al. (2005) suggest that encroachment is a naturally occurring

phenomenon in semi-arid and arid systems where distribution of rainfall and nutrients varies spatially. Disturbance creates space and leaves water and nutrients available for the germination of trees. Where soils are nitrogen-poor, nitrogen-fixing trees have a competitive advantage over other plants and may dominate areas made available by disturbance. Rainfall also varies in space and time, creating patches suited to the survival of trees. However, to explain bush encroachment, this requires that there has been a unidirectional change (on average) in these drivers over time.

In reality, bush encroachment is likely to be driven by a combination of these factors. Indeed, there is a call for integrated models which consider both resource limitation and disturbance, since their relative importance may vary across broad regions and environmental gradients (Kambatuku et al. 2011). The main drivers of bush encroachment and their interactions are summarised in Figure 3.2. The various components of this diagram are described in more detail in the following sections.

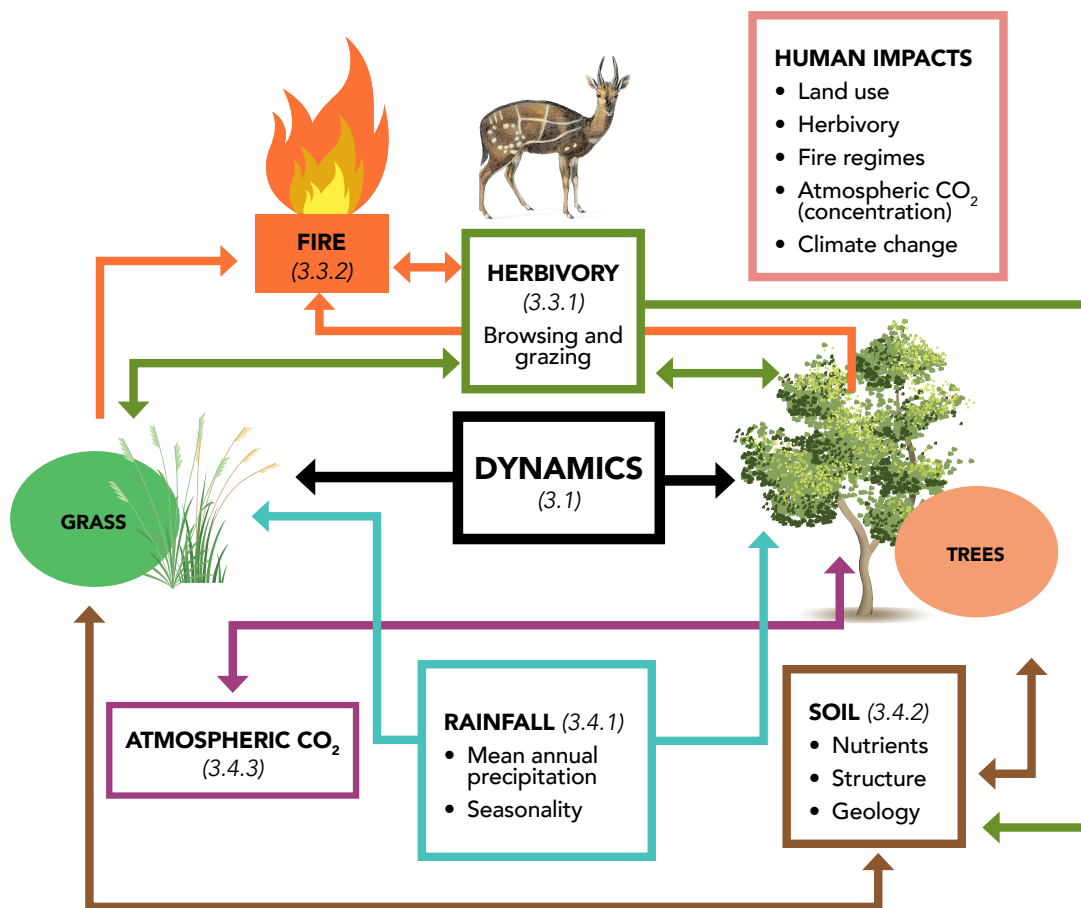


FIGURE 3.2. Overview of the mechanisms hypothesised to influence the dynamics between grass and trees, potentially affecting bush encroachment. The specific roles and linkages between abiotic components (atmospheric CO₂, rainfall and soil) and disturbance regimes (fire and herbivory) are discussed in the sections of the report indicated in parentheses. Human impacts through land use changes, control of herbivory, altered fire regimes, atmospheric CO₂ emissions, contributions to climate change, as well as active clearing of vegetation should be considered when forming a coherent picture of bush encroachment dynamics.

3.3 OVERGRAZING AND FIRE SUPPRESSION AS PRIMARY DRIVERS

3.3.1 Overgrazing

The relative magnitude of grazing *versus* browsing pressure within a savanna ecosystem has the ability to shift the observed balance between grasses and woody plants. An increase in grazing pressure has the ability to inhibit the growth of grasses as well as the opportunity for intense fires as grass fuel loads are reduced. In time this may result in the system shifting to a more bush encroached state. This scenario is often observed on commercial cattle farms, where grazing pressure is often increased significantly and sustained for a longer portion of the year. With no corresponding increase in browsing pressure, this often leads to a more bush-encroached system (Moleele et al. 2002; Scholes 2009). In a study of South African grassy biomes, Skowno et al. (2016) noted that the area of closed woodland on commercial farms and traditional rangelands increased by 0.19% per year over the period 1990–2013.

The opposite trend may also occur where browsing pressure increases disproportionately to grazing pressure. Within conservation areas, an increase in elephant browsing pressure and their ability to uproot adult trees can lead to the gradual opening up of woodlands to a more open savanna state (Stevens et al. 2016). Skowno et al. (2016) found that the area of woodlands in protected areas with elephants decreased by 0.43% per year over the 23-year period of their study. The decrease in tree canopy cover may allow grasses to flourish, which in turn may lead to more intense fires and the shift of the systems into a more open state over time. Another example is the increase in browsing pressure by goats (as well as fuel wood harvesting by humans) that often results in a decrease in woody cover in areas under communal land-tenure or municipal commonage (Puttick et al. 2014a, b).

However, to understand the impact of both grazing and browsing pressure on the structure of savanna, they need to be jointly considered with potential fire regimes that may occur in a particular area (Bond & Keeley 2005; Archibald & Hempson 2016). This is explored further below.

3.3.2 Fire suppression

Fire is one of the principle drivers of the structure of savanna ecosystems (Scholes & Archer 1997). The occurrence of fire requires a source of ignition as well as fuel that is adequately dry, abundant and dense enough to sustain its spread across a landscape. Fire therefore typically occurs in regions with between 450 and 1 800 mm of annual rainfall and in particular, a prolonged dry season in which fuel has the opportunity to dry out sufficiently to burn (Figure 3.3). On the drier end of the range at the transition into semi-arid shrubland, low annual rainfall limits the growth and accumulation of a fuel load that is adequate enough to sustain fire. On the wetter end of range, at the boundary between 'dry forests' and 'moist forests', fuel loads are generally too moist year-round to allow fire to occur. This is often not only a result of the amount of rainfall, but also its seasonality. Outside of moist forests, a single wet season and importantly a prolonged dry season of 7–8 months, provides enough time for fuel to dry sufficiently to burn.

The observed ecosystem structure and the relative abundance of trees versus grasses, is in part due to the form of fire in open African rangelands. In comparison to the boreal forests of the northern hemisphere or the *Eucalyptus* forests of Australia in which high-intensity canopy fires occur, fire in savanna ecosystems occurs in the form of low-intensity surface fires. This is a result of the nature and structure of fuel loads. Closed canopy boreal and *Eucalyptus* forests provide a substantial, tall and continuous fuel load, which is often highly flammable and provides the foundation for high-intensity canopy fires that can carry across a landscape from tree to tree. In comparison, the canopy in savanna and woodland ecosystems is generally not continuous, and often being deciduous, loses a potential leaf fuel load during the dry season in which fire may occur. These factors limit the potential for intense canopy fires in South African rangelands.

In comparison, the near-continuous grass layer in open savanna and grassland ecosystems provides a relatively small, but sufficient fuel load to sustain the spread of fire across landscapes. Although surface fires in savannas are less intense they are still more than sufficient to clear small shrubs and saplings, leading to more open savanna and grassland systems. If fire occurs at regular intervals after a few years of grass accumulation, surface fires are generally adequate to keep a rangeland in an open state.

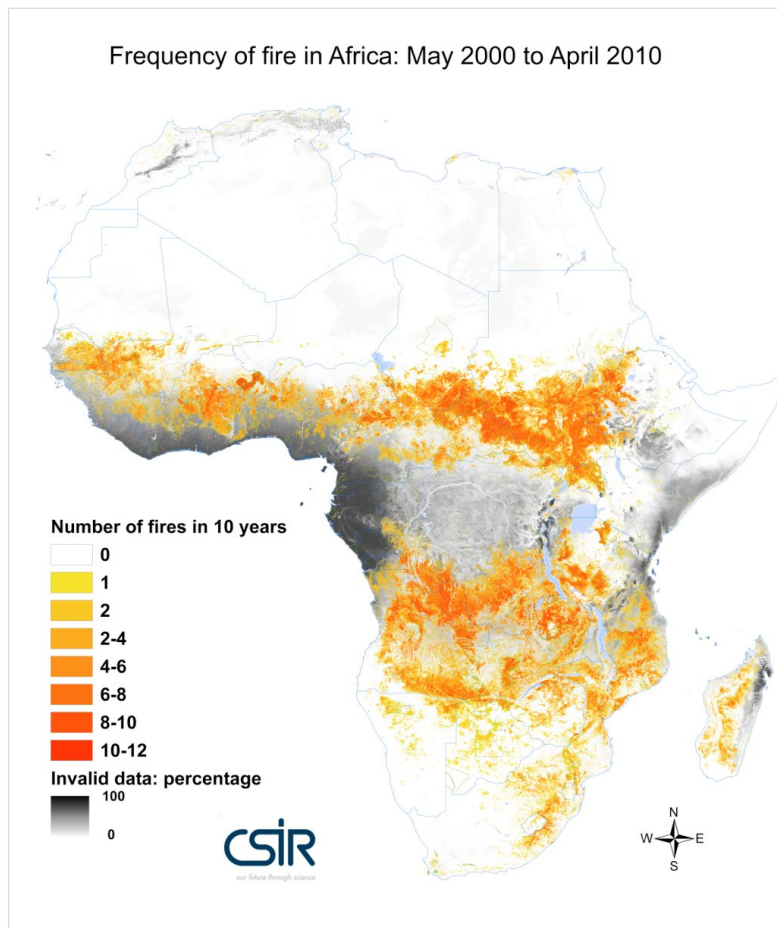


FIGURE 3.3. Frequency of fire in Africa for the period 2000 to 2010. Dividing the number of times a pixel was detected as having burned into 10 gives the approximate fire return time, in years (Source: David Roy, NASA and Sally Archibald, CSIR).

However, a number of factors can lead to a decrease in fuel loads and the intensity of surface fires, which in turn allow woody plants to survive, 'escape the fire trap' and thrive over time. Perhaps the most prominent are the consumption of the grass layer by grazing herbivores as well as early-season burns that combust the grass layer in a low-intensity manner before a substantial fuel load can accumulate, as described above. A further driver that is particularly pertinent to bush encroachment, is the suppression of the grass layer under the canopy of trees. As canopy cover increases, the amount of light and precipitation reaching the grass layer decreases,

but the impact of the effect on fire is non-linear. Staver et al. (2011) found that there may be a 'tipping-point' as tree canopy cover increases over 45–50%, above which fires rarely occur. This leads to a tree-dominated state where a lack of fire allows the further survival and growth of trees, which further suppresses the grass layer and the probability of fire. An understanding of this 'feedback loop' is important when considering bush encroachment and the type of interventions required to decouple the feedback loop and shift an encroached system into a more open state. In addition, active suppression of 'runaway' bush fires to protect property is also likely to play an important role.

3.4 INFLUENCE OF SPATIAL AND TEMPORAL VARIATION IN RESOURCES

3.4.1 Rainfall

Rainfall may be one of the strongest factors governing the extent of bush encroachment in African savannas (Ward 2005). However, there remains much debate regarding the relative importance of resource availability (for example water, nutrients) and disturbance regimes (for example fire, herbivory) as determinants of woody cover in savannas (Sankaran et al. 2005).

In a study of 854 sites across Africa, maximum woody cover in savannas receiving a mean annual precipitation (MAP) of less than ~650 mm was found to be constrained by, and to increase linearly with, MAP (Figure 3.4, Sankaran et al. 2005). These savannas were classified as arid and semi-arid, and deemed to be stable systems in which water availability constrains woody cover and facilitates the coexistence of grasses and trees. Below the maximum woody cover controlled by MAP, woody cover is reduced further by the interaction of fire, herbivory and soil properties (Sankaran et al. 2005). Unstable savannas are deemed to occur where MAP is above ~650 mm. In these systems sufficient water is available to support closed canopy cover, and disturbances are required to allow grasses to dominate in patches (Sankaran et al. 2005).

A study on savanna vegetation across Africa, Australia and South America found that rainfall and rainfall seasonality were important in determining the presence or absence of savanna vegetation

(Lehmann et al. 2011). Rainfall seasonality reduces the rate of canopy closure (Sarmiento 1984) and increases fire frequency (Archibald et al. 2009). Fires can occur more frequently since fuel drying is promoted by pronounced seasonality in rainfall, affecting the spatial distribution and temporal availability of fuel (Bradstock 2010). In addition, the presence or absence of savanna vegetation is related to inter-annual rainfall variability and the probability of drought. Drought can increase tree seedling mortality and decrease the growth rate of adult trees (Fensham et al. 2009). Rainfall seasonality therefore limits woody growth and influences the likelihood of disturbance in savanna systems. Soil fertility was also found to be important, but its effect varied depending on rainfall (Lehmann et al. 2011). In areas of high rainfall, savanna was more likely to occur in sites of low soil fertility, while in arid areas savanna was more likely to occur on fertile soils (Lehmann et al. 2011). The effects of soil nutrients are discussed further in section 3.4.2.

Decreases in decadal rainfall over Zimbabwe have been shown to coincide with reductions in the size of woody patches, likely due to increased mortality of trees and shrubs (Shekede et al. 2016). As woody patches contract, grass patches expand to exploit resources, particularly water, in the absence of competition from woody plants (Shekede et al. 2016). Transient rainfall events facilitate the growth of trees and shrubs

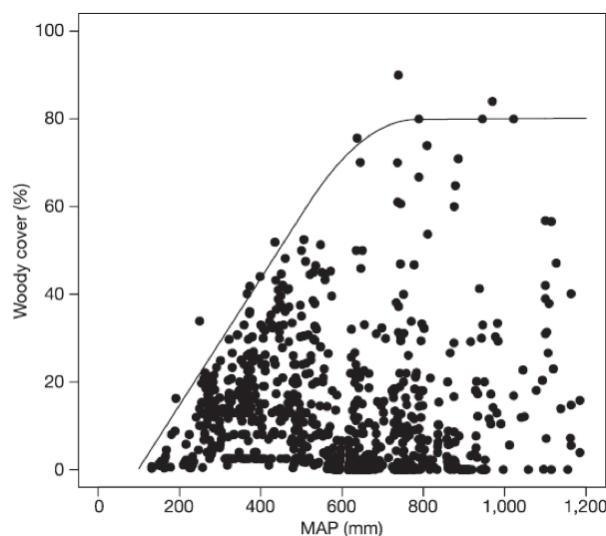


FIGURE 3.4. Woody cover of African savannas as a function of mean annual precipitation (MAP; Sankaran et al. 2005). Maximum tree cover is represented by using a 99th quantile piecewise linear regression. The regression analysis identifies the breakpoint (the rainfall at which maximum tree cover is attained) in the interval 650 ± 134 mm MAP (between 516 and 784 mm). Trees are typically absent below 101 mm MAP. The equation for the line quantifying the upper bound on tree cover between 101 and 650 mm MAP is $\text{Cover (\%)} = 0.14(\text{MAP}) - 14.2$. Data are from 854 sites across Africa.

and a long-term expansion in overall woody cover tends to occur. In contrast, decreases in rainfall facilitate the dominance of grass patches. The effect of decreased rainfall is greater than that of increased rainfall. Drought periods, it is therefore suggested, may be the optimum time to manage bush encroachment. In the Zimbabwe study, about 85% of woody cover dynamics could be explained by changes in decadal rainfall, highlighting the importance of water availability in models explaining tree-grass relationships (Shekede et al. 2016).

Further corroborating the results from Zimbabwe, inter-annual variation in rainfall along the Kalahari Transect in southern Africa has been shown to affect the interaction between grass and trees (Yu et al. 2017). In dry environments (MAP < 900–1000 mm), high inter-annual variability favours trees over shallow-rooted grasses, while the opposite is true in wetter environments (MAP > 900–1000 mm). Deeper root systems increase the competitive advantage of trees over grasses in dry environments, allowing them to access subsoil water. Whereas the relatively rapid growth rate of grasses yields a competitive advantage under wetter conditions, facilitating the exploitation of ephemeral water resources and increasing fire-induced tree mortality (Yu et al. 2017).

In South Africa, bush encroachment has not been associated with trends of either stable or decreased rainfall in the past century (Buitenwerf et al. 2012, Russell & Ward 2014). However, inter-annual rainfall variation has been linked to bush encroachment, in agreement with the above-mentioned findings from Zimbabwe and the Kalahari Transect. Several studies have found that a pulse of bush encroachment occurred in semi-arid and mesic savannas in southern Africa following the drought of the 1960s and the above-average rainfall of the 1970s (O'Connor et al. 2014).

While demonstrating a link between inter-annual rainfall variation and bush encroachment, these studies did not investigate the effects of changes in inter-annual rainfall variation. If the inter-annual variation in rainfall and the frequency of droughts in South Africa increases as a result of climate change in the coming decades, bush encroachment may be favoured as a result, but this trajectory is uncertain (O'Connor et al. 2014). Furthermore, there is substantial uncertainty in the predictions of climate change models regarding changes in rainfall amount and variability in South Africa for this century (Stevens et al. 2015).



3.4.2 Soil characteristics

Patterns of bush encroachment across a landscape are influenced to some extent by the soil nutrients and soil characteristics of that landscape, which influence responses to disturbance factors such as fire and grazing. In addition to natural heterogeneity, the alteration of soil nutrient status, whether through human intervention or the positive feedback loops created by pioneer encroaching trees, may be an important determinant of the vulnerability of landscapes to encroachment. For example, the effects of fertiliser treatments in previously cultivated soils have been shown to have decades-long effects on the establishment of woody encroaching species (Mills et al. 2017). Furthermore, encroaching species may themselves influence soil nutrient characteristics, creating positive feedback loops that facilitate recruitment of seedlings. As many of the main encroaching species are leguminous, fixation and deposition of nitrogen in soils beneath canopies can significantly alter the nutrient regimes of encroached soils. In addition, soil organic carbon content of surface soil layers can be increased by the presence of encroaching species (Li et al. 2016). When considered in the context of the interactions with rainfall and disturbance regimes, the effects of soil structure and nutrients are complex and require careful scrutiny.

Nutrient availability has been shown to influence the dynamics between grasses and trees (Mills et al. 2012, Mills et al. 2013a, Mills et al. 2013b, Devine et al. 2017, Mills et al. 2016, Mills et al. 2017). Specifically, the relative abundances of soil nutrients classified as anabolic (for example boron, magnesium and manganese) or catabolic (for example phosphorous, copper and zinc) have been linked to the presence or absence of trees in savanna and grassland systems (Mills et al. 2013, Mills et al. 2016). Magnesium, boron, manganese and copper have been identified as the soil nutrients most likely to affect tree abundance in some savanna systems (Mills et al. 2017).

Root competition for soil water and nutrients is an important determinant of the balance between grasses and trees (Pärtel & Helm 2007). The responses to, and effects of, these interactions vary considerably in time and space. Savannas typically overlay weathered, nutrient-poor soils. Indeed, poor soil fertility has been widely cited as the main determinant of the lack of trees in savanna biomes (Bond 2010). There is some empirical evidence to support this view, although specific effects may vary according to

species assemblage. For example, in a pot-based experiment, the effect of competition from six grass species was examined on *V. karroo* and *V. nilotica* (Tedder et al. 2012). Under high nutrient conditions, grasses outcompeted *V. karroo*, while little effect was seen in *V. nilotica* (Tedder et al. 2012).

Soil nutrients appear to influence rates of change in woody cover (Devine et al. 2017). On nutrient-poor soils, grasses may recover slowly from grazing pressure, compared with grasses on nutrient-rich soils, decreasing tree-grass competition. Where fires are infrequent, growth of trees may be facilitated further (Levick & Rogers 2011). The effect of the interaction between soil properties and disturbance on vegetation dynamics may be greater in dry savannas, reducing tree-grass competition (Sankaran et al. 2005, Devine et al. 2015).

In areas of low rainfall and high nutrient availability, the growth of palatable grasses is favoured, resulting in increases in herbivory which appears to maintain the savanna system (Lehmann et al. 2011). The first trees to establish in grasslands may give their seedlings a competitive advantage over herbaceous vegetation in their vicinity by creating localised patches of nitrogen-rich litter and reducing light availability (Siemann & Rogers 2003). This competitive release, in combination with other environmental factors, may account for the rapid expansion of woody vegetation into grasslands after the establishment of pioneer trees (Siemann & Rogers 2003).

The structure of the soil itself may influence the competitive dynamics between plants and trees. Kambatuku et al. (2011) demonstrated that: i) competition from grass inhibited the growth of tree seedlings on both sandy and rocky substrates; ii) the presence of tree seedlings facilitated grass growth on rock substrates where simulated herbivory was absent; iii) simulated herbivory decreased the competitive ability of grass seedlings in the presence of tree seedlings; and iv) the nutrient status of grass was positively influenced by competition from tree seedlings. Thus, bush encroachment may be facilitated by high grazing pressure on rocky substrates (Kambatuku et al. 2011).

Thorough assessment of soil types, underlying geology and an understanding of the soil nutrient status may be needed to form an effective strategy to combat bush encroachment across landscapes.

3.4.3 Atmospheric carbon dioxide

Since the start of the industrial revolution, the combustion of fossil fuels as well as release of carbon located in forests and soil, has led to a gradual increase in atmospheric CO₂. Over this 200–300-year period, the concentration of atmospheric CO₂ has increased from 278 to 390.5 ppm, which is one of the primary drivers of anthropogenic climate change (IPCC 2013; ch. 6 pp. 467). Elevated atmospheric CO₂ may also have a disproportionate effect on the growth of plants.

Whereas all plants photosynthesise,² a number of photosynthetic pathways have evolved over time. Known as C₃, C₄ and CAM photosynthetic pathways, these may influence the ability of certain types of plants to grow and thrive in a world with elevated atmospheric CO₂. Whereas C₄ plants are generally not constrained by available atmospheric CO₂, the photosynthetic rate of C₃ plants is limited by the amount of CO₂ in the atmosphere. It is therefore assumed that as the amount of atmospheric CO₂ has increased over time, the ability of C₃ plants to compete with C₄ species has improved.³

Of relevance to savanna ecology and bush encroachment, is that while the majority of grass species in drier South African savannas follow a C₄ photosynthetic pathway, woody species use a C₃ process. It is therefore proposed that as atmospheric CO₂ increases over time there will be an associated increase in the ability of woody species to thrive and compete with C₄ grasses (Bond & Midgley 2000, Bond et al. 2003b). Woody species may not only be able to grow relatively faster, but may be able to escape the 'fire trap' more frequently as a result, thereby leading to landscapes more dominated by woody species.

Experiments on the effect of elevated atmospheric CO₂ on two typical African savanna tree species – the sweet thorn *Vachellia karroo* and the scented thorn *Vachellia nilotica* – clearly illustrate the substantial effect that elevated atmospheric CO₂ can have on increasing the growth rate of woody C₃ plants versus C₄ grasses (Kgope et al. 2010).

However, the outcome may be more complex. Based on photosynthetic pathways, C₃ plants should perform better in relative terms under elevated atmospheric CO₂, but the effect may be limited by nutrient and water availability as well as seasonal variations in climate (Reich et al. 2014; Zelikova et al. 2014). Whereas soil nitrogen limitations may be addressed to a certain extent by the nitrogen-fixing ability of typical encroaching species (the leguminous *Vachellia* spp. or *Dichrostachys cinerea*), the net effect on multispecies communities located on a variety of soil types and exposed to seasonal variations in climate may be more complex (Potvin et al. 2006).

While early experiments clearly illustrate the significant potential impact of elevated atmospheric CO₂ on the growth of C₃ species, further experimentation and modelling may be required to fully understand the impact on multispecies communities at landscapes scales, especially in a region that is predicted to become warmer and drier over time.

2 The process of using solar energy, water and atmospheric CO₂ to make sugars and the building blocks of plants.

3 A more detailed introduction to each photosynthetic pathway can be found at: <https://www.nature.com/scitable/knowledge/library/the-ecology-of-photosynthetic-pathways-15785165>.

TRADE-OFFS AND POLICY IMPLICATIONS

4.1 SOUTH AFRICA'S INTERNATIONAL COMMITMENTS

Climate change, desertification, and the loss of biodiversity were identified as the greatest global challenges to sustainable development during the 1992 Rio Earth Summit. As a result, the United Nations Framework Convention on Climate Change (UNFCCC), the United Nations Convention on Biological Diversity (UNCBD) and the United Nations Convention to Combat Desertification (UNCCD) were established. South Africa is a signatory to all three of these conventions (Box 1).

BOX 1. South Africa's commitments under the UN conventions on Biological Diversity, Combating Desertification and Climate Change

UNCBD (ratified by South Africa in 1995)

This brings countries into collective action to conserve biological diversity, to promote sustainable use of its components, and to promote the fair and equitable sharing of benefits arising from the use of genetic resources. Signatories are accountable for conserving biological diversity by, amongst other things, conserving the natural biodiversity of the country.

UNCCD (ratified by South Africa in 1997)

This brings countries into collective action to combat desertification, the degradation of land in arid, semi-arid, and dry sub-humid areas. Desertification is caused primarily by human activities, such as over-exploitation and inappropriate land use, as well as climatic variations. Signatories are accountable for combating desertification by, amongst other things, addressing unsustainable land management practices.

UNFCCC (ratified by South Africa in 2016)

This brings countries into collective action to address climate change threats within the context of sustainable development and eradication of poverty. The Paris Agreement sets the goal of holding the increase in global warming to below 2 degrees Celsius and pursuing efforts to limit global temperature increase to 1.5 degrees Celsius. Signatories are accountable for reducing greenhouse gas emissions by, amongst other things, increasing the carbon sink capacity of the country.

The UNCBD and UNCCD, focussed on conserving biodiversity and halting degradation, provide a clear mandate to government to address bush encroachment, either by inhibiting further encroachment or by restoring encroached land. However, the UNFCCC has two primary aims, climate change adaptation and mitigation, which may be contradictory in the context of managing bush encroachment. Whereas the sequestration of carbon in woody biomass following encroachment may be viewed as a form of climate change mitigation, bush encroachment is likely at the same time to reduce water services and the productive capacity of land, therefore reducing the ability of local residents and downstream economies to adapt to climate change.

Similarly, the vision of South Africa's National Climate Change Response Policy (NCCRP) is a 'transition towards a lower-carbon and climate-resilient South Africa'. As for the UNFCCC, this would be achieved through both climate change adaptation and mitigation measures.

The NCCRP aims to 'manage inevitable climate change impacts through interventions that build and sustain South Africa's social, economic and environmental resilience' (adaptation), and at the same time, to 'make a fair contribution to the global effort to stabilise greenhouse gas concentrations' (mitigation) (DEA 2012: 5).

4.2 IMPACTS ON ECOSYSTEM STRUCTURE, FUNCTIONING, CONNECTIVITY AND BIODIVERSITY

Bush encroachment is characterised by an increase in density and cover of native bush (or shrubs), resulting in a transformation of grasslands into shrub/tree savannas and shrub/tree savannas into woodlands (NNF 2016, Archer et al. 2017). Not only does this widespread habitat change have an impact on livestock production and game carrying capacity through losses in grazing capacity, but it also has significant implications for numerous other ecosystem services (Archer et al. 2017). The effects of encroachment on biodiversity and ecosystem functioning are not well understood (Eldridge & Soliveres 2014). The challenge is thus to quantify the many associated impacts of bush encroachment so that trade-offs can be accurately evaluated at both a spatial and temporal scale (Archer et al. 2017). This information can then be used to make informed decisions around bush management with the aim of 'devising approaches for creating or maintaining woody-herbaceous mixtures in spatial arrangements that negotiate and balance competing land use and conservation objectives' (Archer et al. 2017: 65).

Woody plant encroachment has an impact on grassland and savanna ecosystems, their ecological functioning, biodiversity, and their ability to deliver ecosystem goods and services, such as nutrient cycling, grazing, hydrological services and carbon storage. This, in turn, has an impact on productive economic sectors and society as a whole. Changes to ecosystem integrity and ecological functioning have an impact on biodiversity, which is strongly correlated with the delivery of many ecosystem services. For example, the diversity of plants, animals and various other organisms can affect aesthetic, recreational and tourism values, and can reduce the provision of certain wild foods, medicines and raw materials. Losses in biodiversity can also have

an impact on non-use and option values. Stevens et al. (2016) found that the widespread change in woody cover in the savanna biome of South Africa is likely to affect users of communal lands, private lands and conservation areas, resulting in costly economic and ecological consequences.

Feedback loops that reinforce the drivers of bush encroachment, such as grazing, fire dynamics and soil nutrients, also have further impacts on biodiversity and ecosystem services. For example, the impact of an increase in tree canopy cover on fire is non-linear and there may be a 'tipping-point' as tree canopy cover increases over 45–50%, above which fires rarely occur. Fire is therefore suppressed in bush encroached areas and is acknowledged as a critical factor when it comes to understanding the consequences and trade-offs associated with bush encroachment on the delivery of ecosystem services and the types of interventions that may be required to shift encroached systems into open state systems.

The effects of encroachment on ecosystem structure and functioning can be highly variable across the landscape (Eldridge et al. 2011). Benefits generally include the potential gains in carbon storage, as well as increased provision of wood fuel, poles and timber and the value added from these, and the potential increase in agricultural support services. Costs generally include the loss of grazing capacity, tourism appeal, and impacts on water supply. A list of ecosystem services included in the analysis, their relevance to bush encroachment and the expected direction of change in the service as a result of bush encroachment is shown in Table 4.1. However, changes in ecosystem services as a result of bush encroachment may not always be consistent in their direction or magnitude, depending on the social and ecological context, and can result in a combination of gains and losses.

TABLE 4.1. A list of ecosystem services and expected direction of change in the service as a result of bush encroachment

CATEGORY	SERVICES	DIRECTION OF CHANGE	BRIEF EXPLANATION
Provisioning services	Livestock and game fodder	-	Reduction in the supply of grazing used as an input to free-ranging livestock and game farming.
	Harvested renewable resources	-/+ overall +	Some losses in certain wild foods, medicines & non-woody raw materials, but increased provision of wood fuel, poles and timber and value added from these (energy, furniture etc.). Overall change in value of harvested resources likely positive as woody resources have a higher value compared to non-woody resources.
Cultural services	Amenity values (aesthetic, tourism)	-	Reduced aesthetic, recreation and tourism use values via impacts on biodiversity.
	Existence and bequest (non-use) values	-	Reduced values via impacts on biodiversity.
Regulating services	Carbon storage	+	Increases in above ground biomass results in increases in carbon storage.
	Water yield	-	Interference with hydrological services through increased evapotranspiration, decreased infiltration & decreased surface runoff lead to reductions in water yield.
	Sediment retention & water quality amelioration	-/+	Context-specific: The difference in soil erosion and water quality amelioration under thicket and woodland versus grassland likely to be negligible.
	Pollination	-/+	Context-specific: In some instances, there will be a positive impact with higher bee densities due to higher vegetation biomass. Or it could be negative if bee density is also correlated to plant diversity which may decrease with increasing bush encroachment.

4.3 APPROACH TO ASSESSING COSTS AND BENEFITS

In this study, we have attempted to generate an idea of the magnitude of the costs and benefits of bush encroachment in different zones, based on the limited available data. Where possible, we have made order-of-magnitude estimates of the gains or losses in ecosystem service values generated per unit area in state-owned protected areas, private rangelands and communal rangelands in each of the bioregional zones, for the average level of encroachment in that zone. These estimates are inaccurate on many levels but serve as indicative starting estimates which will hopefully be refined in future. In some cases, the possible changes in ecosystem services were only discussed qualitatively.

'Baseline' (namely starting point) estimates of woody cover and the average change in woody cover over time were extracted from the literature for each zone using studies that had estimated the impacts of bush encroachment in each of these regions over time (see section 2.5). The average estimates of woody cover for each zone at the start and the end of monitoring were combined to generate an average start (baseline) and end (current) point for each bush encroached zone (Figure 2.17), thus providing a means of assessing

change in habitat and thus a change in the value of ecosystem services. It is acknowledged that there is sample selection bias with the use of these studies and that the values generated will therefore likely reflect above average estimates for the impact of bush encroachment (namely worst-case scenario for the costs and best-case scenario for the benefits). However, without detailed/accurate spatial estimates of changes in woody cover, these studies provide a starting point for estimating the economic impacts of bush encroachment.

The current value (ZAR, 2015) of each ecosystem service was extracted for each land tenure type (communal, private and protected area) in each zone using the spatial dataset generated by Turpie et al. (2017). Using the overall change in percentage woody cover for each zone, simple ratios were applied to the current value of the service and adjusted accordingly to generate a first-order estimate of the impact of bush encroachment on these values (namely an estimate of what the value would have been before the impact of bush encroachment). More detail about the methodology has been included

in the next section under each ecosystem service. It should be noted that these values are ballpark estimates only, providing insight into the magnitude of the change in ecosystem service value as a result of the change in woody cover across the landscape, associated trade-offs and bioregional variation in the impacts associated with bush encroachment.

The approach used to determine the impact of bush encroachment on ecosystem services in the Highveld and Drakensberg grassland zone was different to that used for the other zones, due to the nature of the dominant encroacher species, bankrupt bush *Seriphium plumosum*, which is a small shrub and unlike the common encroacher species found in the other zones. A study by Avenant (2015) on the extent and impact of *S. plumosum* (bankrupt bush), found that 35.5% of the land area covered by the survey was encroached. The encroached land was then classified in terms of plant density and plant density classes (25, 50, 75, 100% cover). The results indicated that 1.69% of the surveyed land was in fact 100% encroached, 21% was 75% encroached, 45% was 50% encroached, and 33% was 25% encroached. The number of hectares for each density class was then calculated and multiplied by the plant density class value, yielding the number of hectares encroached per each density class. Combining the number of encroached hectares provided the total number of encroached hectares, providing an estimate for the woody cover as a percentage of the total area based on the density of the bankrupt bush across the zone. Based on these assumptions it was estimated that the percentage woody cover of bankrupt bush has increased from 0% woody cover to 17% woody cover in the Highveld and Drakensberg grassland zone. This change in woody cover was then used in the same way as the estimates of change in woody cover for the other encroached zones.

4.4 IMPACTS OF BUSH ENCROACHMENT ON ECOSYSTEM SERVICES

4.4.1 Carbon sequestration

The nature of carbon terrestrial carbon stocks and fluxes

Dry organic matter, either in the form of plant biomass, surface litter or soil organic matter, is approximately 50% carbon (IPCC 2013). Woody plants, grasses and forbs breathe in and out a substantial amount of carbon dioxide (CO₂) on a daily basis as they photosynthesise and respire. When they grow, a small fraction of the CO₂ that is breathed in is fixed in organic material and woody biomass in the form of carbon that makes up the building blocks of plants. Over the years, as trees and grass grow, they may lose leaves, branches or fine roots, which break down and either provide inputs into soil organic matter or are released back into the atmosphere through the process of respiration. This broad process of transferring carbon from the atmosphere into plant- or soil-organic matter is known as carbon sequestration.⁴ (IPCC 2007)

At the same time, changes in land cover (for example from forest to cropland) can lead to the release of sequestered carbon into the atmosphere. The clearing and combustion of biomass or the ploughing and turnover of soils releases carbon sequestered in wood and soils back into the atmosphere. Globally the release of carbon from terrestrial ecosystems accounts for approximately one-fifth of all human-generated carbon emissions and is one of the principal sources of GHG emissions from the African continent (IPCC 2007).

As the phenomenon of bush encroachment is prolific in South African landscapes, important national land-use and climate change policy questions are: (i) How does bush encroachment potentially affect terrestrial carbon stocks? (ii) Does encroachment lead to the net sequestration of carbon over time or the net release thereof? (iii) How does encroachment affect fluxes of other important greenhouse gasses associated with land-use and agriculture, for example, methane or nitrous oxide? (iv) How the potential effect varies depending on scale and location?

4 The carbon pools are in a constant state of turnover, but the consistent growth and existence of terrestrial carbon pools (woody, litter, soil) over the long-term (>30 years) is viewed as 'sequestration'.

To articulate the flow and storage of carbon in terrestrial ecosystems and to answer questions such as these, ecologists often adopt a conceptual model of carbon ‘pools’ and ‘fluxes’ (Figure 4.1). Terrestrial carbon is separated into a set of four general pools: the woody biomass, herbaceous biomass, litter and soil organic carbon pools due to their nature and magnitude. As the phenomenon of bush encroachment is likely to affect each of the carbon pools in a particular manner, each pool is considered separately below.

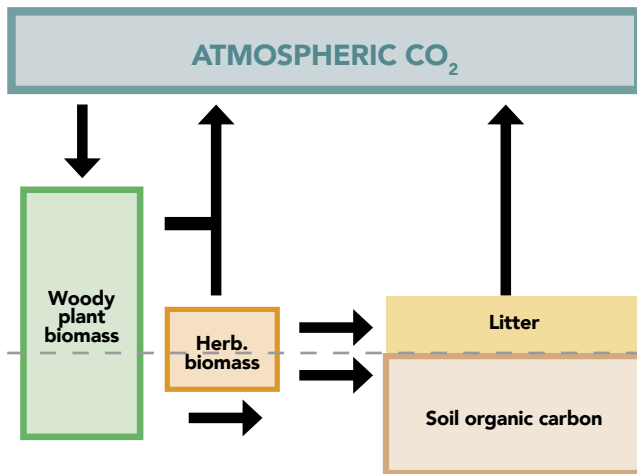


Figure 4.1: A basic illustration of terrestrial carbon pools and fluxes – see Scholes et al. 2013, 3 for a more detailed illustration and description of each pool and flux in a South African context.

The change in biomass carbon stocks (above- and below-ground)

The gradual encroachment of open grassland and savanna landscapes by woody plants is expected to increase both the total amount of carbon sequestered in the woody carbon pool as well as the rate of sequestration. An assessment of South African sites by Hudak et al. (2003), indicated that woody carbon stocks could increase by 20 tC.ha⁻¹ in drier semi-arid areas (Ganyesa, Kalahari sands) and up to 100 tC.ha⁻¹ in more mesic sub-humid areas (Madikwe, red clay loam), when considering the sum of both above- and below-ground woody carbon. Interestingly,

the study found that while carbon stocks had increased on 26 year old ‘severely’ encroached sites to approximately 100 tC ha⁻¹, the amount of biomass and associated carbon decreased to 70 tC ha⁻¹ in older ‘chronically’ invaded 60 year sites where self-thinning may have occurred.

Aside from the study by Hudak et al. (2003), there is limited field data on changes in carbon stocks due to bush encroachment in South Africa. A further indication of the potential impact can be drawn from Smit et al.’s (2016) assessment of 28 sites in northern Namibia (MAP 457 mm) where stocks increased by an average of 44 tC.ha⁻¹, but in general there is limited empirical data on which to attempt to understand the nuances of how carbon stocks under bush encroachment may vary with rainfall, soil type and time since encroachment.

In terms of a change in the rate of carbon sequestration, although there are few local assessments of how the rate of carbon sequestration may change following bush encroachment, observations in the United States and elsewhere indicate that the rate of woody plant growth and associated carbon sequestration can increase significantly for a number of years as an area becomes encroached (Barger et al. 2011; Archer et al. 2017; Figure 4.2). The magnitude of the increase is, however, dependent on annual rainfall, having little effect in areas with less than 400 mm in annual precipitation (Figure 4.4).

Within South African wooded landscapes, the mean annual increment in wood mass is usually approximately 4% of the wood standing crop in a mature state, ranging from 6% in young stands to 2% in mature woodland (Scholes & Walker 1993; Shackleton 1997; Scholes 2004). As many encroaching species are nitrogen-fixing, it is likely that sequestration rates will be around the upper bound of this range. However, in a similar manner to understanding the impact of bush encroachment on standing carbon stocks, there is clear need for further empirical research in South Africa to inform national policies and management and utilisation strategies.

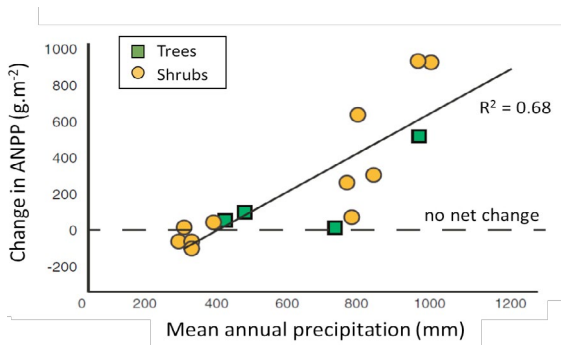
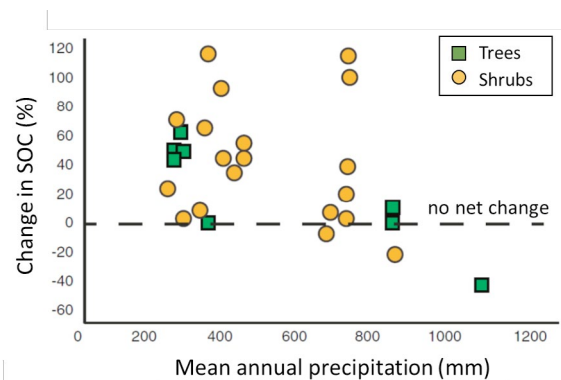


FIGURE 4.2. The change in above ground net primary productivity ($\text{g biomass m}^{-2}\cdot\text{yr}^{-1}$) due to bush encroachment as a function of mean annual precipitation. Source: Barger et al. 2011; Archer et al. 2017.

The change in soil organic carbon pool

Observed soil organic carbon (SOC) stocks are broadly defined by climate, soil type and associated rates of plant growth, litter production and decomposition (Ojima et al. 1993; Scholes & Walker 1993). The encroachment of woody species has the potential to change both the type and rate of production, including the nature of litter inputs into soils (Hudak et al. 2003). The establishment of woody cover can generate a larger litter load under canopy, and in addition, decrease near-ground solar radiation that in turn reduces soil evaporation and the decomposition of litter (Breshears 2006).

However, the magnitude and direction of the effect on SOC under canopy can vary relative to annual rainfall. An assessment of a set of encroached sites across the United States observed that SOC increased in dry sites (MAP 200–800mm), while there was a decrease in SOC in more mesic sites (Jackson et al. 2002; Barger et al. 2011, Figure 4.3). In wetter areas (MAP > 800 mm), the decrease in the size of the SOC pool was larger than the increase in the woody pool leading to a net reduction in ecosystem carbon stocks relative to a non-encroached state.



When attempting to estimate the effect of bush encroachment on SOC at a landscape scale, understanding the scalability of under-canopy effects and confounding factors, such as overgrazing, is crucial. Whereas there may generally be a positive effect on a suite of soil attributes under canopy at low tree densities (for example soil C and N), this effect may become less positive or even negative at higher tree densities (Eldridge & Soliveres 2014). In addition, the direction and magnitude of the effect may be specific to the type of encroaching species, for example, in the drylands of southern Ethiopia, Belay & Kebede (2010) found that while SOC increased under *Vachellia mellifera*, soil carbon decreased under the canopy of *V. deradrepnolobium*.

Differences in SOC under-canopy compared to inter canopy areas may be confounded by further ecological drivers that are associated with bush encroachment, particularly overgrazing. As a landscape becomes more encroached, the overgrazing of remaining open areas within the mosaic may lead to a reduction in SOC in the inter-canopy areas. Irrespective of the abundance or density of woody plants, overgrazing can cause a decrease in grass biomass in grasslands, which if increased to the point of overgrazing, can lead to a decrease in basal cover, litter loads and soil organic carbon (Abril et al. 2005; Eldridge & Soliveres 2014).

For these reasons, changes in SOC due to bush encroachment need to be compared at a field or landscape scale, rather than an under-canopy / intercanopy scale, comparing an entire bush encroached area, with one that has not been encroached at all. In these future assessments, the case needs to be taken up for measuring SOC to a depth of 1 meter and reporting the change in carbon stocks and not only the change in concentration. Many past studies only evaluate the impact of bush encroachment on SOC in the upper layers of soil (for example to a depth of 10 cm) and often do not measure bulk density that may lead to the inappropriate reporting of impacts (Peichl et al. 2012).

FIGURE 4.3. (left) The change in the soil organic carbon pool due to bush encroachment as a function of mean annual precipitation (source Barger et al. 2011; Archer et al. 2017). The green blocks indicate where an increase in woody cover was predominantly through an increase in the prevalence of taller trees, while the yellow circles illustrate where encroachment occurred through an increase in smaller woody shrubs.

The initial intention of this assessment was to provide an estimate of the change in carbon stocks due to bush encroachment at a regional or national scale. Limited availability of suitable datasets as well as empirical studies inhibited this initial aim, but an attempt has been made to at least provide a broad indication of the potential change in the size of the woody carbon pool due to bush encroachment over the past 25 years (Table 4.2).

TABLE 4.2. Order of magnitude estimates of the value of carbon sequestration through bush encroachment in different ecological zones, based on average recorded change in woody cover in those zones.

	WOODY CARBON POOL GAIN (T/HA)	CO ₂ EQUIV (T/HA)	VALUE GAIN (R/HA)	VALUE GAIN TO REST OF WORLD (R/HA)
Mopane	28.5	104.6	154	44 100
Central bushveld	7.1	25.9	38	10 900
Lowveld	13.8	50.5	75	21 300
Highveld	4.3	15.7	23	6 600
Kalahari	16.4	60.0	89	25 300
Sub-escarpment grassland	27.7	101.6	150	42 900
Sub-escarpment savanna	6.7	24.4	36	10 300

The initial estimation is based on a recently published report by SAEON (Warren et al. 2018), which evaluated the change in above-ground woody carbon stocks over the period 1990–2013. The SAEON analysis is based on the observed changes between the 1990 and 2013 national land cover maps and an estimate of the average carbon stock per land-cover class based on the aboveground woody carbon stock data produced during the National Terrestrial Carbon Sink Assessment (Scholes et al. 2013). It should be noted that there is good opportunity to calibrate this initial indication and to further consider changes in the soil, litter and deadwood carbon pools. A full consideration of the change in carbon stocks would also need to consider pre-1990 encroachment. This would require extensive mapping, which is currently not readily available. The estimated sequestration was then valued based on the conversion of carbon to CO₂ equivalents, and the social cost of carbon from a South African perspective (see Turpie et al. 2017). The avoided social cost to the rest of the world is also provided for comparison. Note that these are very high-level estimates.

4.4.2 Provision of natural resources

Millions of South Africans harvest wild foods, medicines, and woody and non-woody resources for both subsistence and commercial use (Shackleton et al. 2001; Shackleton & Shackleton 2004; Lannas & Turpie 2009; Peterson et al. 2012), particularly where there are limited economic opportunities (Dovie et al. 2006). The total sustainable provisioning value of terrestrial natural

resources across South Africa has been estimated to be R7.5 billion per year, with woody resources accounting for R5.5 billion of this (Turpie et al. 2017).

The harvesting of natural resources is particularly important in communal areas where a variety of plants are harvested for food and medicinal purposes, raw materials such as thatch and building poles, and fuel. The use of these plants depends on household income, and the availability and cost of alternatives. Based on a meta-analysis of studies across the country, Shackleton and Shackleton (2004) estimated that on average, 95% of rural households draw fuelwood from neighbouring landscapes, 62% collect fence poles, and 49% of households obtain structural timber for housing and other needs, worth a total of R400 to R1 680 per household per year, depending on location. Woody resources are often drawn from species associated with bush encroachment, such as sickle bush *Dichrostachys cineria* and knob thorn *Senegalia nigrescens*.

An increase in woody cover is expected to increase the availability of woody resources. For example, mopane trees are considered to be excellent fuelwood and timber for crafts and furniture, and certain *Vachellia* species are considered good for charcoal and fuelwood (Palgrave 1977; Van Wyk & Van Wyk 2011). However, the provision of certain wild foods and medicines, and non-woody raw materials such as thatching grass are expected to decrease. While increasing tree density could be associated with an increase in fruits, seeds

and medicinal bark, the common encroacher species (for example sweet thorn, sickle bush and red bushwillow) are not usually utilised for these purposes. It is therefore expected that there is a net loss in the provision of non-woody resources due to bush encroachment. Woody resources have a significantly higher aggregate value than non-woody resources, and therefore the overall net impact is likely to be a gain in provisioning value as grasslands are converted to woodlands and thickets. Furthermore, the additional biomass provided by bush encroachment has the potential to generate electricity (Stafford et al. 2017a, b).

Based on conservative biomass stock values (4, 8, 12 t/ha for light, medium and dense encroachment), Stafford et al. (2017a) estimated that the total amount of biomass within encroaching woody plants could amount to about 58 million tonnes. Of this, 0.5% could be used for sawn timber (gross output net present value of \$14 million), 3% for poles (\$21.1 million), 32.5% for firewood and charcoal (\$560.2 million), 53% for electricity (\$702.9 million) and 10% would remain as residues.

There are few published feasibility assessments on the use woody biomass from encroachment for the generation of electricity (Stafford et al. 2017b). There are a small number of assessments on relative biomass-energy opportunities, for example, Mugido et al. (2013) who evaluated the cost of generating power using alien invasive species in the Eastern Cape (R62 per GJ compared to a R49 per GJ coal-based alternative), but a detailed analysis of the full cost and feasibility of the opportunity remains to be undertaken, together with field assessments of standing biomass stocks in order to calibrate initial supply models.

In this study, we used data from a national spatial assessment of the value of ecosystem services (Turpie et al. 2017) disaggregated to the different ecoregions to evaluate the potential impacts of bush encroachment in these different areas. The values were estimated based on the minimum of estimated sustainable yields and estimated demand of woody (fuelwood and poles) and non-woody (grass, wild plant foods and medicines) resources at the census sub-place (~village area) level. Data on sustainable yields, demand and prices were drawn from the regional literature (Shackleton 1993; Dovie et al. 2004; Barnes et al. 2005; Twine et al. 2003; Turpie et al. 2007; Mmopelwa et al. 2009; Mander 1998; Dovie et al. 2006; Shackleton et al. 2002, 2007).

We used the relationship between average provisioning values and average woody cover of four broad land cover classes (grassland, low shrubland, woodland/open bushland, and dense bush/thicket, Figure 4.4) to estimate the impact of changes in woody cover. The current average provisioning value (R/ha) for each zone was extracted from the Turpie et al. (2017) spatial dataset. The corresponding average woody cover was estimated on the basis of data extracted from the National Land Cover 2013/14 report (GEOTERRAIMAGE 2015). The change in value between the different vegetation types could be used to infer the change in harvesting value due to bush encroachment as the habitat changes across a continuum from grassland into dense thicket (see Figure 4.4). The change in overall woody cover for each bush encroached zone was then used to determine the change in provisioning value as a result of bush encroachment.

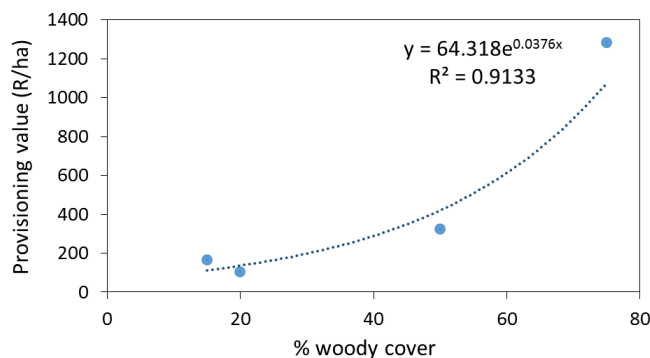


Figure 4.4. The relationship between provisioning value (R/ha) and percentage woody cover of different land cover classes.

These calculations suggest that in all areas, bush encroachment has probably resulted in net gains in the value of provisioning resources, because of the importance of fuelwood (Table 4.3). On communal lands, the highest per hectare gains in the provision of natural resources are likely to have been in the Lowveld zone with an estimated increase in value of about R600 per hectare per year, followed by the Sub-Escarpment Savanna zone and Sub-Escarpment Grassland zone. In these zones, the demand for natural resources is higher due to the higher number of rural villages where people rely on natural resources for their livelihoods. The gains in provisioning value were lowest in the Kalahari zone. A similar pattern was seen for the value gained on private land (Table 4.3).

TABLE 4.3. Overall change in provisioning value (2015 R/ha/y) on communal and private land resulting from average levels of change in woody cover in each zone.

ZONE	CHANGE IN VALUE ON COMMUNAL LAND (R/HA/Y)	CHANGE IN VALUE ON PRIVATE LAND (R/HA/Y)
Mopane	364.2	29.5
Central bushveld	243.0	26.6
Lowveld	601.9	103.3
Highveld & Drakensberg grassland	163.5	12.4
Kalahari	70.7	5.1
Sub-escarpment grassland	471.6	49.8
Sub-escarpment savanna	565.3	78.7
Average per hectare gain in value	354.3	43.6

4.4.3 Livestock and game fodder

In commercial rangelands, livestock overgrazing has resulted in an increase in woody encroachment which in turn has lowered the overall grazing capacity of the land. This loss in grazing capacity could have significant economic consequences for the livestock and game industries. However, bush encroachment would also potentially favour browsers, which may offset some of these losses for the game farming industry.

In the Highveld and Drakensberg Grassland zone, the encroachment of bankrupt bush on commercial rangelands has been found to decrease livestock production by about 65% (Avenant 2015). In the same study, it was found that the majority of farmers in the study area had lost between 30–60% of their grazing capacity, and a total of 35% of the commercial farmland included in the survey was encroached (Avenant 2015).

Based on estimates of changes in woody cover (for example, the displacement of grass by woody elements) the per hectare loss in fodder production value as a result of the average level of bush encroachment recorded in each

zone was estimated to be higher in commercial rangelands than in communal rangelands (Table 4.4). However, the difference in value between these two land tenure types is not consistent across zones, with the per hectare loss in value on communal lands being higher than the per hectare losses on commercial land in the Mopane and Central Bushveld zones. This is because the current per hectare fodder production value is higher in communal areas compared to commercial areas in these two zones. Per hectare losses were highest on communal lands in the Sub-Escarpment Grassland zone where average recorded change in woody cover has been close to 42% and where there are areas of high grazing capacity. Similarly, per hectare losses were highest on commercial land in the Sub-Escarpment Grassland zone, followed by the Sub-Escarpment Savanna and Lowveld zones (Table 4.4).

The results presented here only consider the change in grazing capacity and the impacts of bush encroachment on the production of livestock and game carrying capacity. They do not consider the potential gains associated with increased browsing. However, these are not likely to be significant.

TABLE 4.4. Estimated loss in value of livestock and game fodder production (2015 R/ha/y) on communal and private land as a result of average measured changes in woody cover in each zone.

ZONE	% INCREASE IN WOODY COVER	LOSS IN VALUE ON COMMUNAL LAND (R/HA/Y)	LOSS IN VALUE ON PRIVATE LAND (R/HA/Y)
Mopane	20.0	-74.6	-43.2
Central bushveld	19.4	-91.3	-88.0
Lowveld	26.8	-137.8	-207.6
Highveld & Drakensberg grassland	17.0	-183.8	-197.1
Kalahari	20.8	-44.9	-63.2
Sub-escarpment grassland	41.5	-600.0	-715.9
Sub-escarpment savanna	19.6	-250.2	-366.9
AVERAGE LOSS IN VALUE (R/HA/Y)		-197.5	-240.3

4.4.4 Water supply

The proliferation of higher-biomass plants generally results in higher levels of evapotranspiration, which means that a reduced proportion of rainfall ends up as stream flow and/or infiltrating into groundwater (Smit & Rethman 2000; Jackson et al. 2005; Nosetto et al. 2005; Colin 2010).

The impacts of bush encroachment on water resources have not been estimated in South Africa, but the impacts of alien invasive plants have (Le Maitre et al. 2016, 2001, 2000). Le Maitre et al. (2016) used taxon-specific annual water flow reduction factors for 26 alien invasive plant taxa, the average of which was 75%. Stafford et al.

(2017a) assumed that water use by encroaching plants is 40% less than that of alien invasive plants. Therefore, an average reduction factor of 45% was used in this study. This factor was applied to the change in woody cover observed within the different regions and multiplied by the mean annual runoff (MAR) for each region to yield the reduction in MAR due to bush encroachment. The water value used to calculate the loss in value from water flow reduction was the tier-1 water resource management charge of R1.50/m³ in 2008 (Blignaut et al. 2008), inflated for a present value figure of R2.36/m³. Estimated losses were highest in sub-escarpment savanna protected areas, sub-escarpment savanna private land and sub-escarpment grassland in communal land (Table 4.5).

TABLE 4.5. Loss in value from water flow reduction (2015 R/ha/y) on communal, private and protected areas as a result of changes in woody cover in each zone.

ZONE	% INCREASE IN WOODY COVER	LOSS IN VALUE ON COMMUNAL LAND (R/HA/Y)	LOSS IN VALUE ON PRIVATE LAND (R/HA/Y)	LOSS IN VALUE ON PROTECTED AREA LAND (R/HA/Y)
Mopane	20.0	-349.2	-270.5	-41.6
Central bushveld	19.4	-6.2	-2.3	-99.4
Lowveld	26.8	-191.5	-196.1	-161.5
Highveld & Drakensberg grassland	17.0	-610.7	-36.9	-85.6
Kalahari	20.8	-4.2	-7.1	-2.2
Sub-escarpment grassland	41.5	-777.1	-178.9	-544.9
Sub-escarpment savanna	19.6	-83.3	-1194.4	-4866.7
AVERAGE LOSS IN VALUE (R/HA/Y)		-288.9	-269.4	-828.8

4.4.5 Sediment retention

Soil erosion is a result of water or wind removing the nutrient-rich upper soil layers, the implications of which can be felt both on-site and off-site. The main on-site impacts of soil erosion are the reduction in soil quality, structure, stability and texture, as well as a reduced water-holding capacity, which can exacerbate flooding. The main off-site effect is increased sediment loads in downstream aquatic ecosystems, impacting on ecosystem health and leading to sedimentation of rivers and reservoirs.

Vegetative cover, in the form of either canopy or ground cover, prevents erosion by stabilising soil and by intercepting rainfall, thereby reducing its erosivity (De Groot et al. 2002). The removal of vegetation increases the rate of surface erosion, but the extent to which soil is eroded is determined by numerous factors including soil type, rainfall patterns (amount and intensity), slope and the type and amount of vegetative cover. Landscape degradation is one of the leading causes of sediment loss, mainly through decreases in vegetation cover and erosion of soil through overgrazing.

One of the commonly-used models for predicting soil loss from rainfall is the revised universal soil loss equation (RUSLE). These RUSLE models take vegetation into account by ascribing a C-factor to each of the different land cover classes. The C-factor considers influencers such as prior land use, canopy density, ground cover, and surface roughness in determining the relative effectiveness of the vegetation/land cover in preventing soil loss. These factors are then combined with other influencing factors such as slope, rainfall, and soil type to determine soil loss. Species ability and plant functional traits will influence the exact C-factor for any given landscape (Burylo et al. 2012) but in general, studies to determine the relative C-factor value for grassland and woodland vegetation types in good condition indicate that there is a negligible difference between the two (Fernandez et al. 2003; El-Hassanin et al. 1993; Yan et al. 2003; The Natural Capital Project 2015). This is largely because canopy cover from woody plants and ground cover from herbaceous plants play a similar role in preventing sediment loss. These studies do, however, indicate significant differences between the C-factor of bare or sparsely vegetated land cover types and those with more extensive ground or canopy cover.

Ground cover plays a significant role in controlling the rates of runoff and sediment loss in savanna

landscapes (Bartley et al. 2006). The degradation of grassland through overgrazing exposes soils leading to increased soil damage and erosion. Furthermore, trampling by livestock leads to soil compaction, which increases soil surface erosion, especially via water runoff. Therefore, bush encroachment related to anthropogenic changes to fire regimes or elevated levels of CO₂ will have a negligible effect on sediment retention, while bush encroachment as a result of overgrazing will result in increased soil erosion as overgrazing drives both of these processes.

Thus, while accelerated erosion is usually a result of *decreased* vegetative cover, it is also likely to be higher in situations of bush encroachment than under more natural conditions. Unfortunately, there is not enough information on this to be able to estimate the likely magnitude of these impacts.

4.4.6 Agricultural support services

Pollination and pest control services are provided by wild, free-living organisms living in surrounding natural habitats. This improves crop yields and/or saves on input costs. There is likely to be a link between the abundance of these organisms and above-ground biomass, but there is little evidence of this in the literature. In a study by Blaum et al. (2009) in which arthropod abundance and richness was assessed against bush encroachment in the southern Kalahari, Northern Province of South Africa, the results indicated that total arthropod abundance increases with shrub cover, but that different species responded differently with some increasing, some decreasing and some exhibiting a hump-shaped response to increased shrub cover. Bees were not assessed in the study, however. There are insufficient data to produce any estimates of this impact.

4.4.7 Tourism value

In southern Africa, savanna ecosystems and their megafauna contribute significantly to nature-based tourism value. Bush encroachment can affect mammal densities and, therefore, wildlife tourism. Indeed, recent studies have provided evidence that changes in vegetation structure and increases in vegetation density may decrease tourist satisfaction and attractiveness of protected areas (Arbieu et al. 2017; Gray & Bond 2013).

Arbieu et al. (2017) collected data on vegetation structure and perceived mammal densities along road transects and conducted a tourism survey across four different protected areas in three southern African countries (Botswana, Namibia and South Africa). They showed that the ease

of spotting animals was significantly reduced when shrub cover exceeded about 30%, and that increased vegetation density and height had a negative impact on tourists' wildlife viewing experience (Arbieu et al. 2017). Similarly, Gray & Bond (2013) found that herd sizes and densities of animals in Hluhluwe-iMfolozi Park were much reduced in woody areas, and that about 40% of potential future visitors to the Hluhluwe-iMfolozi Park may be lost if woody cover increased and animals became more difficult to see (Gray & Bond 2013).

Based on the results of visitor surveys conducted by Gray & Bond (2013) and Arbieu et al. (2017) we assumed that an increase in woody cover beyond the threshold of 30% would result in a decrease in tourism demand and a loss in tourism value associated with conservation areas. For

each zone, the percentage change in woody cover beyond the 30% encroachment level was multiplied by the maximum 40% change in tourist demand as a result of bush encroachment, to get a percentage change based on the severity of bush encroachment in each zone.

Tourism is unlikely to have been significantly affected in the Highveld and Drakensberg grassland zones where average recorded woody cover is below 30%. Similarly, impacts in the Central Bushveld are likely to have been relatively small, as average recorded woody cover in this zone has only reached 31%. Zones which have experienced more significant changes in woody cover, namely the Sub-Escarpment Savanna, Sub-Escarpment Grassland, Lowveld and Mopane zones, were estimated to have experienced substantial losses in tourism value (Table 4.6). This would be important for protected areas in these zones.

TABLE 4.6. Loss in tourism value (2015, R/ha/y) in protected areas as a result of changes in woody cover in each zone and the assumption that tourism demand decreases at a threshold of 30% woody cover in conservation areas.

ZONE	ESTIMATED LOSS IN TOURISM (%)	LOSS IN TOURISM VALUE ASSOCIATED WITH PROTECTED AREAS (R/HA/Y)
Mopane	8.00	-42.0
Central bushveld	0.40	-4.6
Lowveld	4.80	-47.8
Highveld & Drakensberg grassland	0.00	-
Kalahari	3.60	-6.3
Sub-escarpment grassland	12.80	-77.3
Sub-escarpment savanna	7.20	-120.6
AVERAGE LOSS IN VALUE (R/HA/Y)		- 42.7

4.5 OVERALL TRADE-OFFS AND THEIR SPATIAL VARIATION

For each tenure type within each bush encroached zone the estimated impacts on values of different ecosystem services were summed to generate a net loss or gain in ecosystem service value due to the impacts of bush encroachment for a situation equivalent to the average level of bush encroachment recorded in that zone. The results suggest that the impacts of bush encroachment are highly context specific (Table 4.7). In most cases, our high-level estimates suggest that bush encroachment would have led to an overall loss in the value of ecosystem services, but in a few cases, mostly on communal land areas, bush encroachment could yield net positive effects. The breakdown of these values is shown in Table 4.8 and depicted graphically in Figure 4.5.

TABLE 4.7. The net gain or loss (R per ha) for each land tenure type within each zone.

ZONE	PROTECTED AREAS (R/HA)	COMMUNAL (R/HA)	PRIVATE (R/HA)
Mopane	70.7	94.7	-129.9
Central bushveld	-65.8	183.7	-25.6
Lowveld	-134.8	347.0	-225.9
Highveld & Drakensberg grassland	-62.4	-607.8	-198.5
Kalahari	80.1	110.1	23.4
Sub-escarpment grassland	-472.3	-755.6	-695.1
Sub-escarpment savanna	-4951.2	267.9	-1446.6

TABLE 4.8. Order-of-magnitude estimates of the gains and losses of different ecosystem service values (R per ha) within different land tenure areas and different ecological zones for average levels of encroachment of each zone

ZONE	LAND TENURE	CARBON (R/HA)	HARVESTED RESOURCES (R/HA)	TOURISM (R/HA)	FODDER (R/HA)	WATER SUPPLY (R/HA)	NET BENEFIT (R/HA)
Mopane	Protected	154.3		-42.0		-41.6	70.7
	Communal	154.3	364.2		-74.6	-349.2	94.7
	Private	154.3	29.5		-43.2	-270.5	-129.9
Central bushveld	Protected	38.2		-4.6		-99.4	-65.8
	Communal	38.2	243.0		-91.3	-6.2	183.7
	Private	38.2	26.6		-88.0	-2.3	-25.6
Lowveld	Protected	74.5		-47.8		-161.5	-134.8
	Communal	74.5	601.9		-137.8	-191.5	347.0
	Private	74.5	103.3		-207.6	-196.1	-225.9
Highveld & Drakensberg grassland	Protected	23.2		0.0		-85.6	-62.4
	Communal	23.2	163.5		-183.8	-610.7	-607.8
	Private	23.2	12.4		-197.1	-36.9	-198.5
Kalahari	Protected	88.5		-6.3		-2.2	80.1
	Communal	88.5	70.7		-44.9	-4.2	110.1
	Private	88.5	5.1		-63.2	-7.1	23.4
Sub-escarpment grassland	Protected	150.0		-77.3		-544.9	-472.3
	Communal	150.0	471.6		-600.0	-777.1	-755.6
	Private	150.0	49.8		-715.9	-178.9	-695.1
Sub-escarpment savanna	Protected	36.0		-120.6		-4866.7	-4951.2
	Communal	36.0	565.3		-250.2	-83.3	267.9
	Private	36.0	78.7		-366.9	-1194.4	-1446.6

4.6 CONCLUSION: LAND DEGRADATION OR CARBON SINK?

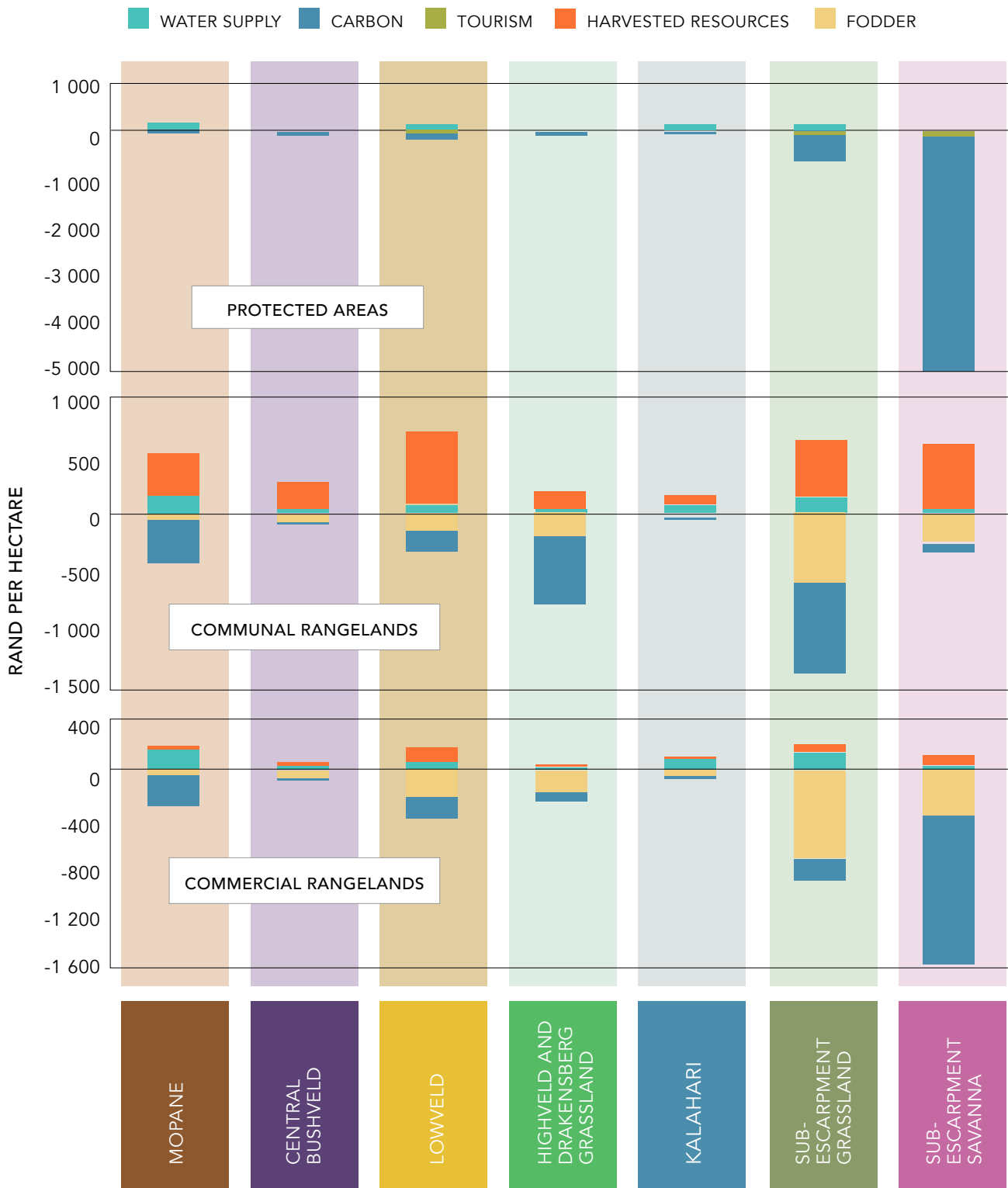


FIGURE 4.5. Order-of-magnitude estimates of the gains and losses of different ecosystem service values (R per ha) within different land tenure areas and different ecological zones for average levels of encroachment of each zone. Note that the Y-axis is truncated at -3 000.

Based on the evaluation of impacts above (Tables 4.7 and 4.8, as well as Figure 4.5), it can be concluded that, in general, bush encroachment results in an overall reduction in the value of ecosystem services to South Africans and **should be considered a form of land degradation**, despite the benefits of increased biomass, including carbon sequestration. Bush encroachment should not be accepted as a carbon sink. Rather, a better managed landscape with a healthy perennial grass layer and associated reductions in soil loss should be the target from a carbon perspective.

In some regions communal land areas stand to gain from bush encroachment, mostly as a result of increased access to fuelwood and charcoal for energy use. However, this gain in energy services must be weighed against the loss in grazing capacity and grassland provisioning, such as thatching grass. The gains will likely only be positive up to a point where the marginal gain in energy services starts to be exceeded by the marginal opportunity cost of a loss in grazing capacity and grassland provisioning services. In other words, these will likely turn out to be short-term gains.

Bush encroachment has occurred as a result of anthropogenic influences on ecosystem functioning, notably poor land management. In other words, it is largely a result of land owners' own actions, unlike in the case of invasive alien plants which spread onto land as a result of past introductions elsewhere in the landscape.

The clearing of bush encroachment will fulfil the climate change adaptation objective of

the NCCRP by restoring and maintaining healthy ecosystem functioning. Allowing bush encroachment will not achieve a fair contribution to global mitigation efforts, especially given the magnitude of the likely effect, the loss of resilience and the fact several other more efficient and effective means of reducing atmospheric carbon dioxide exist. The NCCRP advocates finding a balance between the need to respond to climate change and the development needs of the country. The principles of the policy and its strategic approach highlight the need for climate change response measures to be economically and environmentally rational, realising upliftment and development opportunities where possible. Whereas allowing bush encroachment may fulfil a narrow mitigation goal, clearing and preventing bush encroachment may lead to a far broader and more rational set of adaptation and national development outcomes.

While the exact amount of carbon sequestered through bush encroachment in South Africa is unknown, even if it were substantial, the risk of losing biodiversity and further degrading ecosystem services from allowing bush encroachment to continue unheeded is considered unacceptable. Moreover, the potential risk to biodiversity of allowing bush encroachment would contradict the commitments made under the UN CBD. It is clear that **bush encroachment should be considered a form of land degradation** under the UN commitments, and that other, less damaging, emission reduction opportunities should be employed to meet those targets.

POTENTIAL REMEDIES

5.1 ACTIVE MANAGEMENT OPTIONS

Bush encroachment can be addressed in three main ways. The first approach is to take **no action**. Bush encroachment will likely then increase in severity over time, or at least the existing bush encroachment will persist. This approach may be justified in contexts where the benefits of bush encroachment outweigh the costs (Eldridge et al. 2011), and/or where active management is not feasible. A hypothetical example of such a context is a communal area where wood harvesting is more important than cattle grazing.

The second approach is **reactive management**, namely reducing woody cover (for example by clearing or spraying) and/or preventing bush encroachment from increasing (containment). Reactive management is appropriate in contexts where the negative impacts of bush encroachment outweigh the benefits. Management interventions may seek to reduce negative impacts on ecosystem services, or focus on a specific service that is highly valued, for example grazing capacity. Possible interventions include: i) controlling woody plants by manual, mechanical or chemical means ii) using grazing management and/or fire to reduce woody cover or to stop it from increasing further.

The third approach is to take **proactive** steps to prevent bush encroachment from occurring in the first place. As is the case for many other land management problems, for example invasive alien plants, preventing bush encroachment – or controlling it at an early stage – will generally be less costly than managing it once it occurs (De Klerk 2004; Stafford et al. 2017). Bush encroachment may be prevented by manipulating the drivers of bush encroachment that are under the control of land managers, namely fire and herbivory. However, the feasibility of prevention will depend on the relative importance of the local drivers that can be directed by land managers versus global drivers such as increasing atmospheric CO₂ concentration.

The decision about which management approach to take will depend on the management goals (desired state of the landscape, priority ecosystem goods and services), as well as the local context. The decision-making process should be guided by specialised advisory services and a decision-support system, and should be underpinned by

clear policy and regulation on the management of bush encroachment across South Africa.

5.1.1 Reactive management approaches

Land managers have several options for reducing existing bush encroachment. Woody plants can be cleared by hand or with machines, controlled chemically, and/or managed using fire and herbivores. Cleared woody biomass may also be harvested and used for several purposes to generate income. These management options can be used separately or in various combinations. Below we discuss the effectiveness, impacts, costs and viability of the different options.

Clearing trees and shrubs manually or mechanically is a common way of controlling bush encroachment. In most savanna contexts, ‘clearing’ should not involve the removal of all trees and shrubs, but rather reducing the woody cover to an appropriate level for the specific context, namely ‘thinning’ (Smit 2004). It should be noted that in this respect ‘clearing’ bush encroachment differs from clearing invasive alien plants, where the goal is always to clear the plants completely. However, in grassland vegetation subject to bush encroachment⁵ it is appropriate to remove all the invading indigenous woody plants. Here we use the term ‘clearing’ to mean killing woody plants by hand or with machines, regardless of the level of woody cover that the clearing aims to achieve.

Manual control

Woody plants can be removed manually using a variety of tools. Felling (also known as stumping) is the removal of woody plants with tools such as handsaws, axes, mattocks, chainsaws or hand-held revolving blades. Trees and shrubs can be cut either above or below the ground level. The height at which a tree/shrub is cut is important, since many woody plant species will resprout or coppice (regrowth of the stump to form many branches) when cut above ground-level, unless herbicide is applied to the cut stumps. For example, in Namibia it was observed that 90% or more of woody plants, of various encroacher species,⁶ resprouted when cut above-ground

5 Namely bush invasion, the establishment of indigenous woody species in formerly treeless grassland ecosystems.

6 For example, *Senegalia (Acacia) mellifera*. For *Dichrostachys cinerea*, 100% regrowth was observed (De Klerk 2004)

without herbicide application to stumps or other follow-up treatments (De Klerk 2004). When woody plants were cut ~10 cm or more below the ground most species did not coppice.⁷ While effective, it is time-consuming to cut stumps below-ground and not always feasible (De Klerk 2004). Besides the cutting height, the season in which plants are cut may also influence how much regrowth occurs. Woody plants are most sensitive to defoliation after their first growth flush, if they use stored carbohydrates to produce new growth (Menke & Trlica 1981). Correct application of herbicides along with manual control is crucial. In many cases in South Africa, manual clearing without proper herbicide application has resulted in extensive regrowth and great financial loss (K. Kellner, pers. comm.) In certain cases, regrowth may be desirable. If the management goal is only the sustainable harvesting of wood, rather than for example rangeland restoration, then plants may be allowed to coppice. In Namibia, an area of coppicing woody plants can be harvested again after 10–15 years (De Klerk 2004).

Manual control is labour intensive. It can thus create jobs, provided that the economic benefits derived from clearing are greater than the costs in the specific context. The potential for manual control to create jobs is an important consideration, especially in communal areas. However, the need for job creation should not override considering the overall effectiveness and ecological soundness of any management option. Other advantages of manual control include that: i) specific encroaching species and size-classes of plants can be targeted precisely ii) soil disturbance is very limited relative to using heavy machines.

Mechanical control

Bush encroachment can also be cleared using heavy machines such as bulldozers, tractors and custom-built bush cutting machines. Bulldozing woody plants is faster than manual control, but it causes substantial soil disturbance, which favours the establishment of woody seedlings (De Klerk 2004). As a result, bush encroachment may reoccur in bulldozed areas, possibly more intensely than before clearing. For instance, bulldozed areas in Namibia became bush encroached again within six years (Zapke 1986). Bulldozing may also fail to kill a small proportion of adult shrubs and trees. For this reason, bulldozing needs to be followed by aftercare treatments (for example manual/

chemical control, burning or grazing/browsing) to control woody seedlings and any resprouting adult plants. The running cost of bulldozing and other mechanical methods is comparable to that of manual clearing (Table 5.1), however the capital cost of acquiring the machines should also be considered. The costs of clearing should be weighed up against the benefits it can provide (see Section 5.1). Some land managers will not be able to afford their own heavy machinery and will thus have to hire machines or rely on contractors (Rothauge 2014a). In certain contexts, for example many communal areas, mechanical control may be prohibitively expensive, unless funded by the state.

Some custom-built machines for clearing bush encroachment have been developed in South Africa and in Namibia.⁸ These machines typically have large rotating blades that cut trees and shrubs. They are able to cover large areas, can move relatively nimbly and can be used to selectively cut unwanted woody plants. As with manual control, cut stumps need to be treated with herbicide to prevent coppicing, and effective aftercare treatments need to be applied in the first few years after clearing. Using such machines will cause some soil disturbance, but to a much lesser extent than bulldozing, since plants are not uprooted and the machines have tires instead of tracks. Trains of machines can be used in an integrated manner, namely a cutting machine, followed by a stacking machine, followed by a chipping machine. Machine trains are particularly useful for harvesting woody biomass on a large scale and have been used in Namibia (De Wet 2015). These include machines with hydraulic grabbing claws that can cut and lift trees in a single action. A drawback of the cutting machines is that the large rotating blades may be dangerous if not carefully monitored (De Klerk 2004). As a result of their specialised nature, they will mostly be used by contractors, rather than owned by individual land owners. An example of an integrated bush clearing operation in South Africa is a company that uses custom-built cutting machines and Bell stackers, combined with the manual cutting of small shrubs with hand-held rotating blades, followed by manual application of herbicides and monitoring by drones (Fig. 5.1; Fig 5.2).⁹

7 An important exception is *Dichrostachys cinerea*.

8 See for example <http://bushclearingsolutions.co.za/index.php/machinery/> in South Africa and the 'Bosvark' and other machines in Namibia (De Wet 2015).

9 <http://bushclearingsolutions.co.za>



FIGURE 5.1. A) Custom-built cutting machine for clearing bush encroachment. B) Bell skid steer loader used to stack trees after cutting. Photo credit: <http://bushclearingsolutions.co.za>



FIGURE 5.2. Hand-held rotating blade. Photo credit: <http://bushclearingsolutions.co.za>

Chemical control

The chemical control of bush encroachment entails the application of herbicides to the foliage or stems of woody plants, or to the soil around plants. Herbicides can be applied manually, by spraying tractors or by aeroplane. Chemical control can be used as the primary treatment for bush encroachment, or it can be combined with manual/mechanical clearing or burning. In this section we discuss using chemical control as the primary treatment.

Chemical control should be considered when: i) a large area needs to be treated; ii) the woody component is so dense that other methods are too expensive or impractical; iii) the majority of trees have grown out beyond the reach of browsing animals; iv) the tree density is such that animal access is severely restricted; v) the woody component is largely unpalatable; or vi) it is not practical to incorporate browsers in the livestock system (De Klerk 2004).

While chemical control can be effective and

economical in these contexts, its drawbacks should also be considered. Firstly, woody plants killed by arboricides are 'virtually unharvestable', namely their biomass cannot be used to generate income (De Wet 2015). Secondly, many of the chemicals typically used have some negative environmental impacts (discussed below). Thirdly, chemical control is costly.

In southern Africa, the most popular chemical control method among farmers is the use of systemic, photosynthesis-inhibiting arboricides (Harmse 2016). These arboricides are typically applied in granular form, either by hand or by aeroplane. The active ingredient is usually Tebuthiuron, which infiltrates the soil after rain, where it is absorbed by the roots of woody plants and leads to death once carbohydrate reserves are exhausted (Bezuidenhout et al. 2015). Tebuthiuron is highly effective in killing woody species at relatively low dosage rates. However, the accumulation and persistence of Tebuthiuron in the soil is a major concern. For instance, Du Toit and Sekwadi (2012) found that Tebuthiuron-treated

soil excluded any plant recovery for at least eight years after application in semiarid South African grassland. In contrast, Haussmann et al. (2016) observed immediate re-infestation of chemically cleared sites by an undesirable perennial shrub in the highland savanna of Namibia. There is no clear consensus about the persistence of Tebuthiuron in soils (K. Kellner, pers. comm.). Besides its persistence in the soil, Tebuthiuron is not selective. While it does not kill some broadleaf woody species, its application by aeroplane often kills non-target species. For this reason, aerial application of arboricides should only be used in dense stands of encroaching species. In addition, anecdotal reports indicate that arboricides such as Tebuthiuron can kill non-target trees and shrubs some distance away from the application area and long after the application as it spreads through the groundwater. For example, in the Dordabis area of Namibia, large riverine trees were dying off 15 years after upstream farms had been treated intensively with arboricides from the air (Rothauge 2014b). Indeed, negative examples of 'aerial overkill' are so common in Namibia that some rangeland experts are in favour of a blanket ban on the aerial application of arboricides (Rothauge 2014a). This option is not considered a good policy option for South Africa and is not considered further here.

5.1.2 Proactive land management

Indirect (through overgrazing) and direct suppression of fire are the main causes of bush encroachment, the effects of which vary spatially as a result of local factors such as rainfall and soil characteristics, as outlined in Section 3. Thus, it follows that bush encroachment should be preventable, and reversible, if it has not progressed too far, through better land management. As follow-up treatments, appropriate management of grazing and fire are also crucial for preventing bush-encroachment from re-occurring. This section will assess the principles and primary considerations of the common land management approaches under different land tenures.

If the conditions of a given landscape favour tree- or shrub-dominance, then any bush encroachment intervention applied will either require constant resource allocation and re-application, or else will be temporary (Wiegand et al. 2006). Sustainable land management practices manipulate landscape-level factors to promote the desired vegetation structure. In recognition of the dynamic nature of ecosystems, these practices follow adaptive management principles. These principles prioritise the maintenance of ecosystem

integrity through ongoing monitoring and evaluation, allowing for the informed adjustment of management practices (Lee 1999).

Disturbance factors can be manipulated to create conditions favourable to grass-dominance, thereby reducing the impact of bush encroachment in grasslands and savannas. Woody species proliferate when grass competition is compromised, so combating bush encroachment depends partly on the productivity of the herbaceous layer. See Section 3 for a brief description of the factors affecting tree- and grass-dominance. The primary disturbance factors that can be controlled are fire and herbivory, but the strategies employed are highly dependent on, among other things, land use type and available resources. As a general principle, areas that are over-utilised require rest from disturbance to recover while under-utilised areas require some form of disturbance to promote palatable grass species and remove moribund, flammable biomass (Stuart-Hill 1999).

Grazing management

There are many different approaches to land management and to grazing management in particular.¹⁰ A common thread is that it is important to allow adequate rest for the veld through carefully-managed rotational grazing, rather than continuous grazing, which is cheaper for livestock farmers in the short-term but degrades vegetation.

Continuous grazing is a system by which animals graze in one area without any barriers to their movement. This system is prevalent in communal areas, game ranches and national parks. The comparatively low cost of supporting infrastructure, management input and labour advantages this approach over more intensive methods. The disadvantage, though, is that maintenance of ecosystem integrity can easily be compromised. In the presence of predators, grazers gather in large herds and uniformly utilise the grass layer. In the absence of predators, however, grazers selectively utilise the palatable grass while avoiding the unpalatable climax and pioneer species, resulting in reduced productivity and competitive advantage of the grass layer. This reduced competitive advantage provides the opportunity for woody species to encroach, further degrading the grass layer. Selectively

¹⁰ For a comprehensive guide to land management practices, see Bothma & Du Toit 2010, Tainton 1999 or Van Oudtshoorn 2015.

grazed land is more difficult to rehabilitate as both rest and disturbance degrades the land further by increasing the climax and pioneer species, respectively, with little advantage provided to the palatable, productive forage species. Degradation under this system is further accelerated due to the erosion of footpaths leading to preferred grazing areas.

Rotational grazing is usually applied using a system of fenced camps. There are different ways of managing the rotation. Low stocking rates and frequent rotations, for example, allow for high-productivity grazing with only modest infrastructure investment, rapid recovery of palatable species and the accumulation of the critical biomass required for a high-intensity controlled fire or to save forage banks for periods of drought (Hempson et al. 2015). High utilisation grazing with high to ultra-high stocking rates, by contrast, requires substantially more resources to manage but if applied correctly can improve water infiltration, increase soil nutrient status and biology, remove unpalatable climax species and improve the productivity and vigour of the grass layer (Acocks 1966). A disadvantage of this system is that it requires substantial management inputs through constant monitoring of the veld condition and appropriate adjustments to the rotational strategy. Irrespective of strategy, the system needs to be carefully monitored in order to avoid overgrazing (and thereby allowing bush encroachment). This can be very difficult to achieve in communal grazing lands, depending on the strength and philosophy of the institutions involved.

Management of grazing in protected areas is achieved using watering points. Grazers can be kept away from over-utilised areas by closing artificial water points in the vicinity and by placing salt licks and feed in under-utilised areas. High-selectivity grazers are unable to utilise tall grasses so simulating bulk grazers by clearing tall grass attracts more grazers to the area due to the increased visibility and the availability of new growth. Grazers are attracted to areas that have recently burned for similar reasons, and because of the high nutrient concentration in the ash. These imprecise and indirect methods of manipulating grazers often lead to uneven forage utilisation, which can result in under- and over-utilised grass patches on a scale of a dozen meters. Patches of degradation in the grass layer are potential sites for woody encroachment, which may serve as nucleation points for greater woody cover increases.

Using fire as a management tool

Fire can also be used as a supplementary measure to grazing management, to prevent bush encroachment and/or speed up the recovery of the vegetation. Here, the objective is to stimulate relatively high-intensity fires at the right time of year in order to prevent woody plant seedlings from surviving.

To effectively combat bush encroachment, late dry season fires are optimal, when fuel loads will have built up, and because dry foliage burns hotter (Case & Staver 2017) "type" : "article-journal", "volume" : "54" }, "uris" : ["http://www.mendeley.com/documents/?uuid=05fc44df-29fd-4914-bf91-41b960a9bdd3"] }], "mendeley" : { "formattedCitation" : "(Case & Staver 2017. Burning earlier also carries more risk of soil erosion. Intense fires require a high fuel load (>4 tonnes/ha), high ambient temperature (>25°C) and low humidity (<40%). Thus, it can only be effective in conjunction with well-managed, rotational grazing systems in which fuel loads are allowed to build up.

Active post-burn management is also important, in order to prevent an increase in woody vegetation through coppicing and resprouting. This requires the protection of the burned area from over-grazing and stocking with browsers to utilise the coppice. Schutz et al. (2011) found this decreased the resprouting rate of *Vachellia (Acacia) karoo* due to the depletion of their root carbohydrate reserves. The post-burn manipulation of herbivores is more easily achieved in rotational systems but can also be achieved in continuous systems by using a patch mosaic burn pattern; burning a large area or simultaneously burning several areas across the property to limit the grazing pressure on a single site. Browser utilisation of the resprouting woody plants can be promoted by application of molasses (Esterhuizen & Myburgh 2016) or simulated in the absence of browsers by manual clearing as discussed in Section 5.1.1.

Appropriate land management can be costly, especially the capital costs of setting up rotational grazing systems. However, considering the benefits of avoiding productivity losses as a result of bush encroachment and other forms of degradation, investing in proper land management practices is sound. The cost of administering controlled burning is relatively small, but carries considerable risk and requires technical capacity.

5.2 GENERAL BEST PRACTICES AND MANAGEMENT CONSIDERATIONS

Regardless of which approach is taken, there are several general best practices for dealing with bush encroachment to consider. First, the management of bush encroachment should be context-specific, considering the region, the encroaching species, current and historical land-use and land ownership (De Klerk 2004; Smit 2004; O'Connor et al. 2014). Second, the costs and benefits, trade-offs, potential conflicts of interest and possible unintended consequences of management actions should be assessed. Third, not all woody plants should be removed from savanna landscapes, since woody plants play important ecological roles (De Klerk 2004; Smit 2004; Eldridge & Soliveres 2014). Rather, the appropriate density of woody plants for the specific context should be determined (discussed below). Fourth, a long-term management strategy should be developed, as once-off actions against bush encroachment rarely achieve lasting benefits. Such a strategy should include monitoring and adaptive management. Follow-up (aftercare) treatments are crucial to prevent bush encroachment from recurring. Lastly, the causes and not just the symptoms of bush encroachment should be addressed. Sound land management practices are vital for preventing bush encroachment (O'Connor et al. 2014), as well as for managing the impacts of bush encroachment where it cannot be prevented.

Along with the above-mentioned best practices, there are a number of other important considerations for the effective management of bush encroachment. Firstly, land managers need to **define management goals**, including what the appropriate percentage of woody cover is for a particular context. The appropriate level of cover will vary across regions and vegetation types and for different encroaching species and land uses. Woody cover in savannas also varies naturally at the landscape scale for various reasons (see Section 3). Under different land uses, the appropriate level of cover may vary both among properties, for example a protected area versus a cattle farm, and within properties, for example woodlots versus grazing areas.

Since savannas are dynamic ecosystems with the tree-grass ratio varying in space and time, it is not always simple to determine what **the baseline or reference state** (namely the 'natural' percentage woody cover/abundance and diversity of woody species) is for a particular area (Section 2.2.2.). Aerial photography is one way to determine what

the state of an area was previously. South Africa's aerial photographs go back to the 1940s, when the entire country was photographed for the first time, and are a valuable record of the historical vegetation. However, it should be noted that the vegetation in some areas had already been modified extensively by people before the 1940s, for example through changes in herbivory and fire frequency. There is also evidence that bush encroachment had already occurred in some areas before the 1940s (O'Connor et al. 2014). Nevertheless, aerial photographs indicate that the rate of bush encroachment has increased markedly since 1993, suggesting that the bulk of bush encroachment may have occurred in recent decades. For this reason, satellite data from this period may also provide an indication of the reference state. In addition to aerial photos and remote sensing, repeat landscape photography,¹¹ past vegetation surveys¹² and even living memory may also be used to determine the reference state. In some contexts, the reference state will also be the desired state of the ecosystem pursued by land managers, for example in protected areas. In other contexts, the management goal will be to achieve a state that maximises particular ecosystem goods and services, for example grazing potential on commercial cattle farms.

The management goals for bush encroached areas may also be informed by the concept of **thresholds of potential concern**,¹³ namely 'upper and lower levels of change in selected biotic and abiotic variables which act as indicators of the acceptability of ecosystem condition' (Biggs et al. 2011: 2). It is possible to define thresholds of potential concern for bush encroachment, namely the approximate level of woody cover above which ecosystem functioning changes dramatically. This may be the percentage woody cover at which grass cover is insufficient for frequent fires.

Another factor that should be considered is the **scale** at which management is undertaken. For example, if woody plants are limited at the local scale by seed production or seed dispersal, then bush encroachment should be managed

11 See the rePhotoSA project (rephotosa.adu.org.za).

12 For example, surveys done by Acocks to define vegetation types across SA. Acocks, JPH (1988) Veld types of South Africa. (Third edition) Mem. Bot. Surv. S. Afr. No. 57: 1–146

13 This concept is used within the framework of 'strategic adaptive management', and in South Africa is most commonly applied in national parks and for integrated water resource management.

on a large enough scale to avoid the continued dispersal of seeds from encroaching species into cleared areas (Midgley & Bond 2001; Moleele et al. 2002). Management costs may also depend on scale, for example land managers may reduce costs by sharing bush clearing machines. In some cases, individual farms or reserves will be large enough to form management units, while in other cases neighbouring land managers will need to collaborate. Conflicts of interest may arise where land managers have different goals. For instance, game farms stocked with browsers such as kudu may opt not to manage bush encroachment at all, while the adjacent commercial cattle farms may want to reduce bush encroachment to increase grazing potential.

5.3 POTENTIAL FOR REALISING CO-BENEFITS FROM THE HARVESTED BIOMASS

The extent and impacts of bush encroachment across South Africa necessitate solutions that are scalable, sustainable, long-term and pragmatic. Income generation through wood harvesting is one strategy that is potentially sustainable and scalable, and that could make clearing a viable business option in itself.

Biomass can be harvested when clearing bush encroachment and potentially used for a variety of purposes. It can be used on the land that is cleared, or sold and processed to generate income that can help cover the costs of clearing operations. This approach holds promise in South Africa and is indeed the centrepiece of Namibia's strategy for managing bush encroachment (Rothauge 2014a; Birch et al. 2016). A range of products can be made from woody biomass, including: i) firewood; ii) charcoal; iii) wood chips; iv) woodfuel briquettes;¹⁴ v) building panels; vi) fence posts; vii) wood pellets for burning in industrial furnaces and power plants; viii) animal feed;¹⁵ ix) biochar.

Two reviews of the economic viability of wood harvesting in the southern African context are presented below. The first focuses on lessons learned from the failure of the wood-pellet industry in South Africa between 2008 and 2013. The second assesses the economic opportunities

of five different woody biomass utilisation strategies in Namibia. The reviews below do not take into consideration the potential unintended ecological consequences of wood harvesting. For example, Zimmermann et al. (2017) suggest that the removal of woody biomass can result in a significant, long-term decrease in overall soil fertility.

5.3.1 The South African wood-pellet industry

A review by Bowd et al. (2018) outlines the opportunities for woody biomass in sub-Saharan Africa to contribute to the much-needed stabilisation and expansion of electricity supply, for which South Africa makes up the largest proportion of power demand in sub-Saharan Africa. The benefits of using the wood-pellet industry for energy production are, in theory, considerable. Of the renewable energy sources available in South Africa, woody biomass is the only one that is not weather-dependent and has acknowledged additional ecological, social and economic benefits (Bowd et al. 2018).

Adoption of wood-pellet combustion:

- emits significantly less SO₂ and NO_x compared to South Africa's coal-based electricity production
- has the potential to recover wood-waste that would otherwise be incinerated or be disposed of in landfills
- can utilise existing coal-based infrastructure so requires less capital investment and operational training than other renewable sources
- allows for the diversification of energy sources on a local- and national-scale
- has 80% lower combustible boiler maintenance costs compared to coal and heavy oil
- can create secure job provision throughout the supply chain and ancillary services industry.

While the woody biomass sector in Europe and North America is already mature and growing, South Africa has had comparatively little success in the development of this sector. Four pellet plants were established in South Africa between 2008 and 2010, but all four plants had closed down by 2013. An extensive systems-approach review of the woody biomass sector was undertaken by Bowd et al. (2018), assessing the failure of the South African wood pellet industry. Based on the lessons learned and through consultation

14 See for example Bushblok in Namibia, <http://bushblok.com>.

15 For example, Boskos developed in SA (<http://www.wesenterprises.co.za/downloads/BoskosBrochure.pdf>) and 'bush feed' mixtures used by some Namibian farmers (Rothauge 2014).

with representatives of the former pellet plants, Bowd et al. (2018) made recommendations to re-establish a resilient South African wood-pellet industry. The proposed strategies incorporate ecological, social and economic aspects that weren't considered previously. The intention behind the review was to improve the resilience of potential wood-biomass harvesting efforts for the wood-pellet industry in South Africa.

5.3.2 Opportunities for woody biomass utilisation in Namibia

The Namibian Nature Foundation, supported by GIZ and funded by the Economics of Land Degradation Initiative, assessed the economic opportunities for woody biomass utilisation in Namibia as a means to combat bush encroachment (Birch et al. 2016, 2017). They calculated the costs and benefits of utilising encroacher biomass for the production of charcoal, firewood and animal feed, as well as for thermal power and electricity generation.

The net benefits of electricity generation from woody biomass were found to provide more than five times the income than for firewood harvesting, and more than double that of charcoal production. The cumulative net benefits for each use, their multiplier effects and the improved ecosystem services are presented in Figure 5.4 under central, best and worst case 25-year scenario models. The cost/benefit resilience models predict that net positive returns from wood harvesting would require at least 5–10 years of substantial investment, if they are to be achieved at all, but long-term sustainability and profitability is possible. The amount and distribution of woody biomass available for harvesting in South Africa will be estimated in the forthcoming SAEON Bio-Energy Atlas. We have, therefore, not included any such estimates in this study.

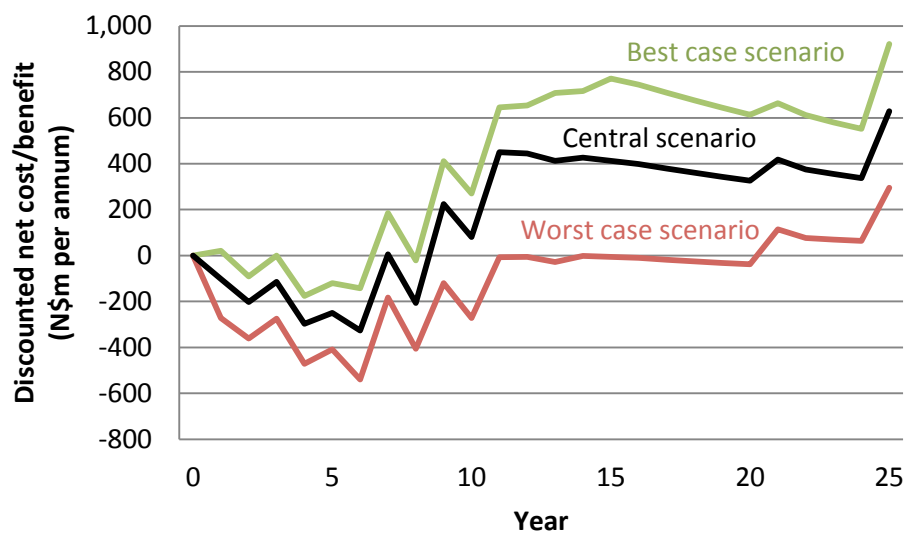


FIGURE 5.4. Discounted net cost/benefit of electricity generation from biomass under different scenarios in Namibia (Birch et al. 2017).

POLICY AND REGULATION AFFECTING MANAGEMENT

The management of indigenous landscapes in South Africa falls under the remit of a substantial set of policies and national departments. This ranges from the work of environmental and conservation departments (DEA), to forestry and agricultural agencies (DAFF) as well as rural development and land reform (DRDLR) and spatial planning at national, provincial and municipal scales. This situation often leads to an 'overlap' in mandate, especially in 'worked' indigenous landscapes that are being used for the production of livestock. While many of these areas have substantial conservation value, they are also viewed as productive agricultural land.

Based on an analysis of pertinent land-use policy undertaken during the National Terrestrial Carbon Sink Assessment (Boardman et al. 2014), together with input from stakeholders who are working on the subject in the field and academia, a set of principle national Acts was identified as pertinent to bush encroachment (Table 6.1). All of these focus on some aspect of land management, be it from a conservation, agriculture or water production (ecosystem service) point of view. Whereas the word 'encroachment' does not occur in any of the texts, most share a common spirit with the intention to better manage landscapes in a sustainable manner in order to achieve ecological, social and economic outcomes. None of the policies explicitly states whether they support or discourage bush encroachment.

The Conservation of Agricultural Resources Act, 1983 (Act No. 43 of 1983) (CARA) aims to conserve the natural agricultural resources of the country by protecting vegetation, halting erosion and combating weeds and invader plants. It aims to ensure 'the maintenance of the production potential of land'. In the context of livestock production, especially the production of grazing animals, this may include the maintenance of rangelands in a relatively open state that enhances fodder production and quality. The implementation of this goal often includes clearing encroaching woody cover as part of grazing and fire management. However, if agricultural clearing operations occur within an important biodiversity area or affects threatened species, it can lead to the implementation of enforcement measures under the National Environmental Management Act, 1998 (Act No. 107 of 1998) (NEMA) and other legislation.

The National Forests Act, 1998 (Act No. 84 of 1998) (NFA), NEMA and the National Environmental Management: Biodiversity Act, 2004 (Act No. 10 of 2004) (NEMBA) ensure the long-term conservation of biodiversity and forests. Through associated regulations and listing notices (Table 6.2) they provide an instrument through which to halt the clearing of indigenous landscapes and especially areas and individual species that may be of conservation importance (For example critically endangered or threatened species).

TABLE 6.1. Pertinent national Acts that have relevance to the consideration and management of bush encroachment in South African landscapes

ABBREVIATION	DATE	TITLE	PURPOSE
CARA	1983	Conservation of Agricultural Resources Act	The objects of this Act are to provide for the conservation of the natural agricultural resources of the Republic by the maintenance of the production potential of land, by the combating and prevention of erosion and weakening or destruction of the water sources, and by the protection of the vegetation and the combating of weeds and invader plants.
NFA	1998	National Forests Act	To: 'a) promote the sustainable management and development of forests for the benefit of all; b) create the conditions necessary to restructure forestry in State forests; c) provide special measures for the protection of certain forests and trees; d) promote the sustainable use of forests for environmental, economic, education, recreational, cultural, health and spiritual purposes; e) promote community forestry; f) promote greater participation in all aspects of forestry and the forest products industry by persons disadvantaged by unfair discrimination.'

ABBREVIATION	DATE	TITLE	PURPOSE
NWA	1998	National Water Act	'To ensure that the nation and water resources are protected, used, developed, conserved and controlled ... protecting aquatic and associated ecosystems and their biological diversity, reducing and preventing pollution and degradation of water resources....'
NEMA	1998	National Environmental Management Act	'To provide for co-operative, environmental governance by establishing principles for decision-making on matters affecting the environment, institutions that will promote co-operative governance and procedures for co-ordinating environmental functions exercised by organs of state; and to provide for matters connected therewith.'
NEMA	2002, 2004	National Environmental Management Amendment Acts	To amend certain sections of NEMA (1998). Including '...to provide for the prohibition, restriction or control of activities which are likely to have a detrimental effect on the environment; and to provide for matters connected therewith.'
NEMBA	2004	National Environmental Management: Biodiversity Act	'To provide for the management and conservation of biological diversity within the Republic and of the components of such biological diversity.' As such the focus of this Act is on the preservation of species (a widely defined term) and ecosystems irrespective of whether or not they are situated in protected areas. Biodiversity is defined as the 'variability among living organisms from all sources including, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part and also includes diversity within species, between species, and of ecosystems.'

TABLE 6.2. Regulations and listings relevant to realisation of the Acts noted in Table 6.1

ACRONYM	DATE	TITLE	PURPOSE
EIAREG	2006	NEMA: EIA* Regulations	'To regulate procedures and criteria as contemplated in Chapter 5 of the Act for the submission, processing, consideration and decision of applications for environmental authorisation of activities and for matters pertaining thereto.'
NFAREG	2009	Regulations on the National Forests Act, 2009	The document outlines the steps to follow if applying for a license and what is allowed under the license, such as applying for the protection of trees and forests, the Minister's authority to collect information from owners and enforcement, offences and penalties.
NEMA LN 1	2010	Listing Notice 1:	'to identify activities that would require environmental authorisations prior to commencement of that activity and to identify competent authorities in terms of sections 24(2) and 24D of the Act' [NEMA]
NEMA LN 2	2010	Listing Notice 2:	'to identify activities that would require an environmental authorisation prior to the commencement of that activity and to identify competent authorities in terms of sections 24(2) and 24D of the Act' [NEMA]
NEMA LN 3	2010	Listing Notice 3:	'to list activities and identify competent authorities under sections 24(2), 24(5) and 24D of the Act, where environmental authorisation is required prior to commencement of that activity in specific identified geographical areas only' [NEMA]

* Environmental impact assessment

Thus, if clearing of woody plants occurs within a gazetted 'natural forest' or leads to the removal or pruning of protected tree species, further punitive measures may be implemented in line with the National Forestry Act (NFA), unless approval is sought from the relevant authority.

Stakeholders are often under the impression that CARA and the environmental management acts have juxtaposed positions on the potential clearance of bush encroachment. This is not necessarily true. Whereas CARA can be interpreted to support the clearing of woody encroachment to maintain the productive potential of land, NEMA, NEMBA and the NFA do not necessarily forbid it completely, but rather aim to reduce the impact of disturbances on the long-term survival of important species and conservation areas. For example, NEMA (1998) states in its principles:

(4)(a)(i) That the disturbance of ecosystems and loss of biological diversity are avoided, or where they cannot be altogether avoided, are minimised and remedied:

NEMA, NEMBA and associated listing notices provide a systematic process through which to evaluate the conservation importance of particular areas and levels of assessment and authorisation that may be required before clearance is allowed to occur, if at all (Figure 6.1, Table 6.3).

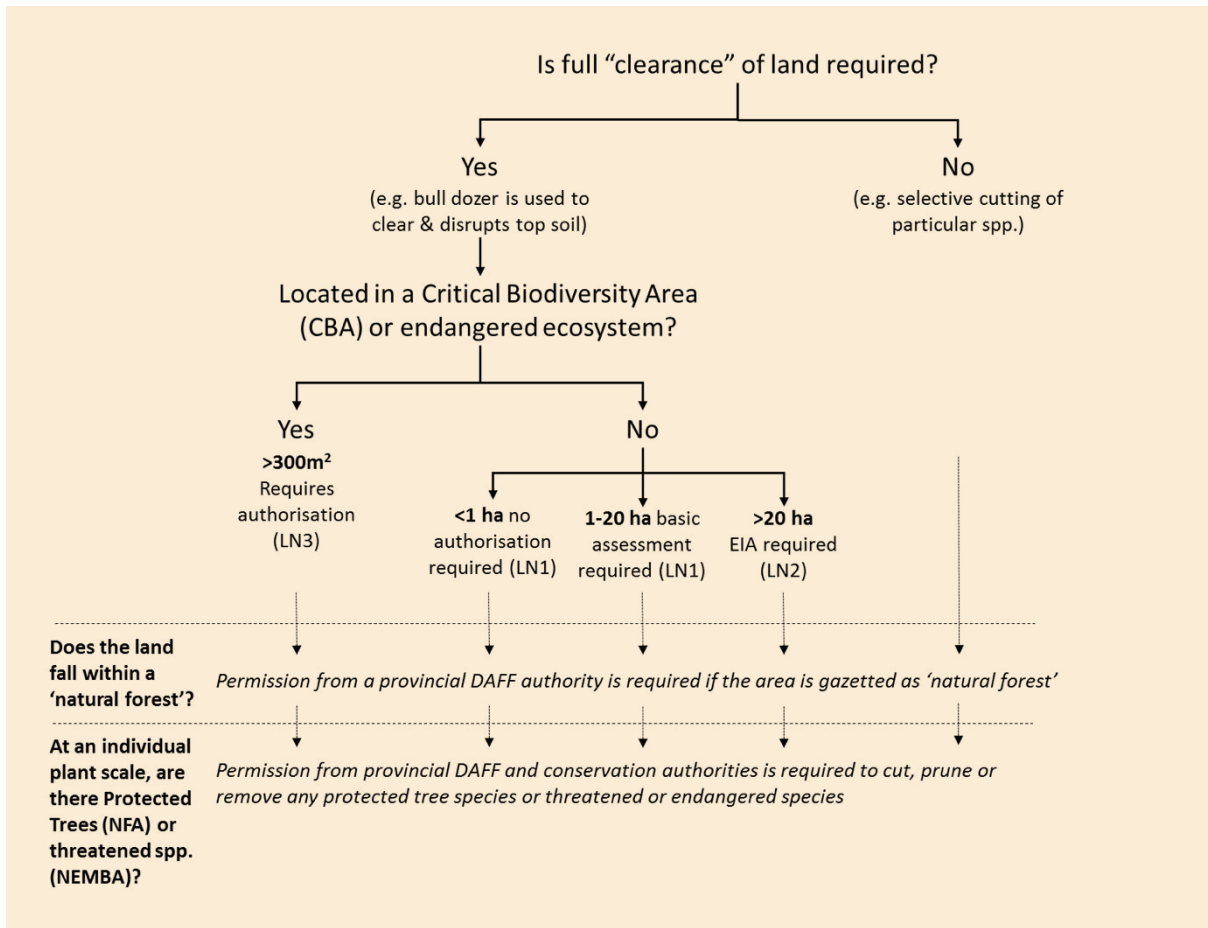


FIGURE 6.1. The process of understanding the level and type of authorisation needed to clear land and particular species

The first consideration is the type of woody plant removal envisioned. If implementation is done through the selective cutting of individual woody plants, without disturbance to topsoil, it may not trigger NEMA and NEMBA regulations. If, however, 'clearance' involves the full clearing of vegetation and disturbance of topsoil, for example, when using a bulldozer, permission may be required under NEMA and NEMBA depending on the size of the area, its conservation status and the occurrence of threatened or endangered species (Figure 6.1). Details of requirements are in Listing Notices 1, 2 and 3 to NEMA (Table 6.3).

TABLE 6.3. Particular NEMA policy and regulatory text

DOCUMENT	TEXT
NEMA 2004 (section 2)	<p>'Environmental authorisations'</p> <p>24. (2) The Minister, and every MEC with the concurrence of the Minister, may identify -</p> <ul style="list-style-type: none"> (a) activities which may not commence without environmental authorisation from the competent authority; (b) geographical areas based on environmental attributes in which specified activities may not commence without environmental authorisation from the competent authority; (c) geographical areas based on environmental attributes in which specified activities may be excluded from authorisation by the competent authority; (d) individual or generic existing activities which may have a detrimental effect on the environment and in respect of which an application for an environmental authorisation must be made to the competent authority:

DOCUMENT	TEXT
NEMA 2004 (section 3)	3. The following sections are hereby inserted in the principal Act after section 24: 'Procedure for listing activity or area 24A. Before identifying any activity or area in terms of section 24(2), the Minister or MEC, as the case may be, must publish a notice in the relevant Gazette- (a) specifying, through description, a map or any other appropriate, the activity or area that it is proposing to list; (b) inviting interested parties to submit written comments on the proposed manner listing within a period specified in the notice.
NEMBA 2004 (section 40)	Bioregions and bioregional plans 40. (1) The Minister or the MEC for environmental affairs in a province may, by notice in the Gazette - (a) determine a geographic region as a bioregion for the purposes of this Act if that region contains whole or several nested ecosystems and is characterised by its landforms, vegetation cover, human culture and history...
NEMBA 2004 (section 52)	(1) (a) The Minister may, by notice in the Gazette, publish a national list of ecosystems that are threatened and in need of protection. [See Section 52 of the Act, these may include critically endangered ecosystems, endangered ecosystems, vulnerable ecosystems and protected ecosystems]
NEMA LN 1 (activity 27)	[States the need for a basic assessment in the case of] The clearance of an area of 1 hectare or more, but less than 20 hectares of indigenous vegetation, except where such clearance of indigenous vegetation is required for - (i) the undertaking of a linear activity; or (ii) maintenance purposes undertaken in accordance with a maintenance management plan.
NEMA LN 2 (activity 15)	[States the need for an Environmental Impact Assessment in the case of:] The clearance of an area of 20 hectares or more of indigenous vegetation, excluding where such clearance of indigenous vegetation is required for— (i) the undertaking of a linear activity; or (ii) maintenance purposes undertaken in accordance with a maintenance management plan.
NEMA LN 3 (activity 12)	[Environmental authorisation is required in the case of] The clearance of an area of 300 square metres or more of indigenous vegetation except where such clearance of indigenous vegetation is required for maintenance purposes undertaken in accordance with a maintenance management plan. This activity only applies in specified geographical areas: - Within any critically endangered or endangered ecosystem listed in terms of section 52 of the NEMBA or prior to the publication of such a list, within an area that has been identified as critically endangered in the National Spatial Biodiversity Assessment 2004; - Within critical biodiversity areas identified in bioregional plans;

The definition of 'clearance' can therefore have significant implications and certainly requires further assessment and clarification. A generally-adopted definition is included in Table 6.4 below. Whereas it may be obvious that complete clearing of land using a bulldozer, removing all vegetation and disturbing topsoil, may be viewed as 'clearance', there is a certain level of ambiguity on whether less intrusive measures, for example, the cutting of woody plants above ground level and poisoning of root stock, is 'clearance' if topsoil and the herbaceous layer are left intact.

TABLE 6.4. Pertinent concepts and definitions

Indigenous vegetation

'refers to vegetation consisting of indigenous plant species occurring naturally in an area, regardless of the level of alien infestation and where the topsoil has not been lawfully disturbed during the preceding ten years' (as defined in NEMA Listing Notice 1).

Endangered species and ecosystems

- 'critically endangered ecosystem' means any ecosystem listed as a critically endangered ecosystem in terms of section 52(2); (NEMABA 2004)
- 'critically endangered species' means any indigenous species listed as a critically endangered species in terms of section 56; (NEMABA 2004)
- Definition of a 'forest' and associated considerations as per the National Forests Act, 1998 (Act No. 84 of 1998)
- 'forest' includes-- a. a natural forest, a woodland and a plantation; b. the forest produce in it; and c. the ecosystems which it makes up;
- 'natural forest' means a group of indigenous trees-- a. whose crowns are largely contiguous; or b. which have been declared by the Minister to be a natural forest under section 7 (2);
- 'woodland' means a group of indigenous trees which are not a natural forest, but whose crowns cover more than five per cent of the area bounded by the trees forming the perimeter of the group.
- 'protected tree' means a tree declared to be protected, or belonging to a group of trees, woodland or species declared to be protected, under section 12 (1) or 14 (2);

What is 'clearance'?

- 'Ploughing of land, bulldozing of an area, eradication or removal of vegetation cover with chemicals, amongst others, constitutes clearance of vegetation, provided that this will result in the vegetation being eliminated, removed or eradicated.'
- Burning of vegetation (e.g. fire-breaks), mowing grass or pruning does not constitute vegetation clearance, unless such burning, mowing or pruning would result in the vegetation being permanently eliminated, removed or eradicated. i.e. eradication of weeds or plant types not occurring naturally within the specific area by means of selective chemical application would not constitute clearance of indigenous vegetation (Pienaar 2017)

What is a linear activity?

'An activity that is arranged in or extending along one or more properties and which affects the environment or any aspect of the environment along the course of the activity, and includes railways, roads, canals, channels, funiculars, pipelines, conveyor belts, cableways, power lines, fences, runways, aircraft landing strips, firebreaks and telecommunication lines'

What is a 'maintenance management plan'

'maintenance management plan' means a management plan for maintenance purposes defined or adopted by the competent authority (as defined in NEMA Listing Notice 1)

Together with NEMA, the stipulations of the NFA need to be considered simultaneously (Table 6.5). If the forest or woodland is gazetted as a 'natural forest' under the act, clearance or even selective cutting may be prohibited unless permission is sought from the relevant authority. In addition, individual trees on the national list of protected trees species may not be removed or even cut without permission.

TABLE 6.5. Particular NFA policy and regulatory text

DOCUMENT	SECTION	TEXT
NFA 1998	12	<p>Protection of trees Part 3 allows the Minister to declare a tree, a group of trees, a woodland or a species of trees as protected. The procedure for and the effect of this declaration are set out. An emergency procedure is included to protect trees threatened with immediate harm.</p> <p>12. Declaration of trees as protected. (1) The Minister may declare:</p> <ul style="list-style-type: none"> a. a particular tree; b. a particular group of trees; c. a particular woodland; or d. trees belonging to a particular species, to be a protected tree, group of trees, woodland or species.
NFA 1998	3	<p>Principles to guide decisions affecting forests</p> <ul style="list-style-type: none"> a. natural forests must not be destroyed save in exceptional circumstances where, in the opinion of the Minister, a proposed new land use is preferable in terms of its economic, social or environmental benefits; c. forests must be developed and managed so as to-- <ul style="list-style-type: none"> i. conserve biological diversity, ecosystems and habitats; ii. sustain the potential yield of their economic, social and environmental benefits; iii. promote the fair distribution of their economic, social, health and environmental benefits; iv. promote their health and vitality; v. conserve natural resources, especially soil and water;
NFA 1998	7	<p>Part 1 prohibits the destruction of indigenous trees in any natural forest without a licence. Prohibition on destruction of trees in natural forests.--(1) No person may cut, disturb, damage or destroy any indigenous, living tree in, or remove or receive any such tree from, a natural forest except in terms of--</p> <ul style="list-style-type: none"> a. a licence issued under subsection (4) or section 23; or b. an exemption from the provisions of this subsection published by the Minister in the Gazette on the advice of the Council.

EVALUATION OF ALTERNATIVE MANAGEMENT OPTIONS

This section presents a high-level exploration of the potential costs and benefits of a range of possible policy scenarios in each of the affected bioregions or bush encroachment zones, taking land tenure into account.

7.1 SCENARIOS CONSIDERED

A set of government policy scenarios for managing bush encroachment was devised as follows:

Scenario 1. Laissez faire (no action)

This scenario would result in bush encroachment continuing unabated, potentially leading to 100% woody cover.

Scenario 2. Incentivise/regulate better rangeland management

Under this scenario, government incentives or legislation are put in place to encourage landowners/custodians to implement more effective land management practices such as rotational grazing, stocking to carrying capacities and undertaking controlled burns where necessary. The aim of this intervention would be to retard, arrest or reverse bush encroachment, depending on the starting condition, through better grazing and burning practices. This policy option would require the expanded deployment and improved effectiveness of agricultural extension services, with additional trained extension service officers employed to achieve the standard of one officer to every 450 landowners (National Council of Provinces: Land and Mineral Resources 2014), and appropriate incentives and institutional changes implemented to facilitate better management in communal land areas.¹⁶ In the case of protected areas, this would require more scientific input to develop better management systems. The cost of implementation would therefore vary considerably across the three land-tenure types.

¹⁶ Options include creation of property rights systems, and applying taxes or fees. There is a wealth of literature on this issue, which is not elaborated here.

Scenario 3. Fund manual restoration, followed by better land management

Under this scenario, manual restoration would be funded by government, for example through a Working for Land type programme. This restoration would be followed by incentivisation of more effective land management practices (as per Scenario 2) to maintain the gains made. This policy option would create jobs and could also generate revenue during the clearing phase to offset the costs of clearing through the harvesting of biomass by targeting value-adding industries such as fuelwood and charcoal production, biomass to energy production or biochar production.¹⁷

Scenario 4. Facilitate mechanical restoration, followed by land management

Under this scenario, incentives are introduced for mechanical clearing by landowners or private companies, followed by land management as per Scenario 2. The extent of investment required by government would depend on the extent to which the landowners' or companies' private gains outweigh the costs they have to incur. If the benefits outweigh the costs, then government might provide or encourage the provision of clearing services at cost, paying attention to removing social or other obstacles to farmers. If the costs outweigh the gains, then interventions would be required either to subsidise the clearing costs directly,¹⁸ or to facilitate the creation of viable business opportunities for private entrepreneurs who then clear at low cost to farmers.¹⁹

¹⁷ SAEON are currently investigating the feasibility of these different value-adding industries spatially and their findings will contribute to this study.

¹⁸ For example, low equipment rental rates and less-stringent part-time labour laws. A state-owned parastatal or agricultural organisation could provide low rental rates for clearing equipment to private landowners, allowing them to manage their own clearing operations and keep the harvested biomass for fodder and fuelwood or sell it to value-adding industries.

¹⁹ For example, soft loans and preferential procurement of biomass for state-owned value-adding industries, such as biomass to energy or biochar production. The preferential procurement of biomass by state-owned entities would stimulate demand for biomass and therefore bush clearing.

7.2 METHODS AND ASSUMPTIONS

7.2.1 Level of restoration

For the active clearing scenarios (3 and 4), it was assumed that restoration involves clearing of bush to the level observed at the start of monitoring level for each region (2.16 on page 35), hereinafter termed the 'initial extent'. Natural levels would likely be lower than this, but are unknown at this stage.

7.2.2 Costs²⁰

For the active clearing scenarios (3 and 4), costs were estimated to clear the woody cover from the encroached extent to the initial extent for the encroached hectares in each region, with these extents being the average situation reported from studies in the literature within each zone (2.16 on page 35). Note that we have not attempted to extrapolate to present day extents, but this possible resulting underestimate is balanced by the fact that studies are likely to have been biased towards more heavily impacted areas.

The cost for manual clearing (scenario 3) was assumed to be R3 500 per hectare²¹ to achieve a 25% to 75% reduction in woody cover. In some regions, restoration would require clearing less than 25% woody cover and we have assumed the cost to be 60% of the above cost (R2 100 per hectare), since a portion of the costs will be fixed. A further R500 per hectare once-off herbicide treatment cost is incurred one year after clearing.²² Similar assumptions were used for mechanical clearing (scenario 4), but with cost assumptions of R4 500 per hectare (Table 5.1) and R2 700 per hectare,²³ respectively, as well as the once-off cost of R500 per hectare.

All the scenarios, except for scenario 1 (no action), include more effective land management practices as a measure to curb the continuation of bush encroachment. The cost of implementing more effective land management practices will be borne by both government and land owners/custodians. The cost to government would include increased numbers of extension

officers (at about R250 000 per annum) for private lands, which is negligible at the per ha level, resources to bring about appropriate changes for communal areas, and research costs for protected areas, if not across the board. Since improving land management is the mandate of government in any case, in order to maximise productivity, ensure sustainability and resilience to climate change, we do not include these as a cost pertaining specifically to addressing bush encroachment. It was assumed that these policies would be implemented in any case.

7.2.3 Benefits

In estimating the benefits under each scenario, we have assumed that economic growth, population growth and increasing scarcity of resources will result in an annual 3% increase in demand for ecosystem services.

The potential benefits from adopting more effective land management practices were not quantified in this analysis as the rate at which bush encroachment is slowed or reversed, and therefore the timing of the delivery of benefits, is unknown.

The potential additional benefits from productive use of the woody biomass removed through clearing were not quantified in this analysis. The ongoing study by SAEON is expected to shed more light on this. We have, however, estimated the harvesting profit required to make clearing feasible if the ecosystem service benefits are not enough to justify action on their own.

7.2.4 Time to restoration

The time to restoration for manual clearing, which alters both the intervals for when costs are incurred and for when benefits are delivered, is based on the annual budget allocated to the Working for Water programme. The annual budget in 2016 was ~ R1.5bn from which we have deducted R0.5bn for overhead costs specific to that programme, yielding an annual operational budget of R1bn. This budget was used to estimate the number of hectares that could be manually cleared per year in each region by firstly calculating the budget per working day (assumed to be 220 days per year) and then calculating the number of hectares that could be cleared per day on that budget (assuming that manual clearing costs are R3 500/ha) and then calculating how many days it would take to clear all the encroached land. The estimated number of hectares required to be cleared for each land-tenure type within each bioregion was based on preliminary data

20 For an update on costs reported here, see the EcoRestore/Debushing Support database on bush control in South Africa and Namibia which was due to be released in February 2018 but was still outstanding when this report was published. (K. Kellner, pers. comm.)

21 Based on discussions with key informants.

22 R310–750 / ha for once-off follow-up herbicide treatment one year after cutting, based on discussions with key informants.

23 R1 600–3 000, based on discussions with key informants.

supplied by SAEON (2.12 on page 33). Lessons learnt from the Working for Water programme suggest that the actual cost to clear may be far higher than initial estimates which would impact the time to restoration and the overall feasibility of manual clearing. We have assumed that the gain in benefits begin a year after the clearing has taken place and is calculated on the number of hectares cleared in the previous year/s.

The time to restoration for mechanical clearing is assumed to be within the first year and therefore all costs, except for the once-off herbicide follow-up cost, are incurred in the first year. The full benefits are therefore gained from the second year. The benefit of clearing to society as well as the private landowner was computed, where the benefit to the landowner was estimated as the gain in grazing capacity less the loss in provisioning services. The reason for this approach is to determine whether it is already in the landowner's

best interests to clear or whether government would be required to subsidise private clearing when it is in society's interest to do so (when the benefit to society of clearing is positive but the benefit to the landowner of clearing is negative).

7.3 RESULTS

Extending the analysis of Chapter 4, the results of a no-action policy would be expected to lead to further losses in welfare in almost all cases, apart from protected areas in two of the bioregions, communal areas in three of the bioregions and private land in one of the bioregions (Figure 5.5). Incentivising better land management as the sole remediation action cost was assumed to be neutral in this analysis, in that this intervention can already be considered to be an imperative (namely at no extra cost), and because it maintains the current status quo, or achieves slow restoration gains.

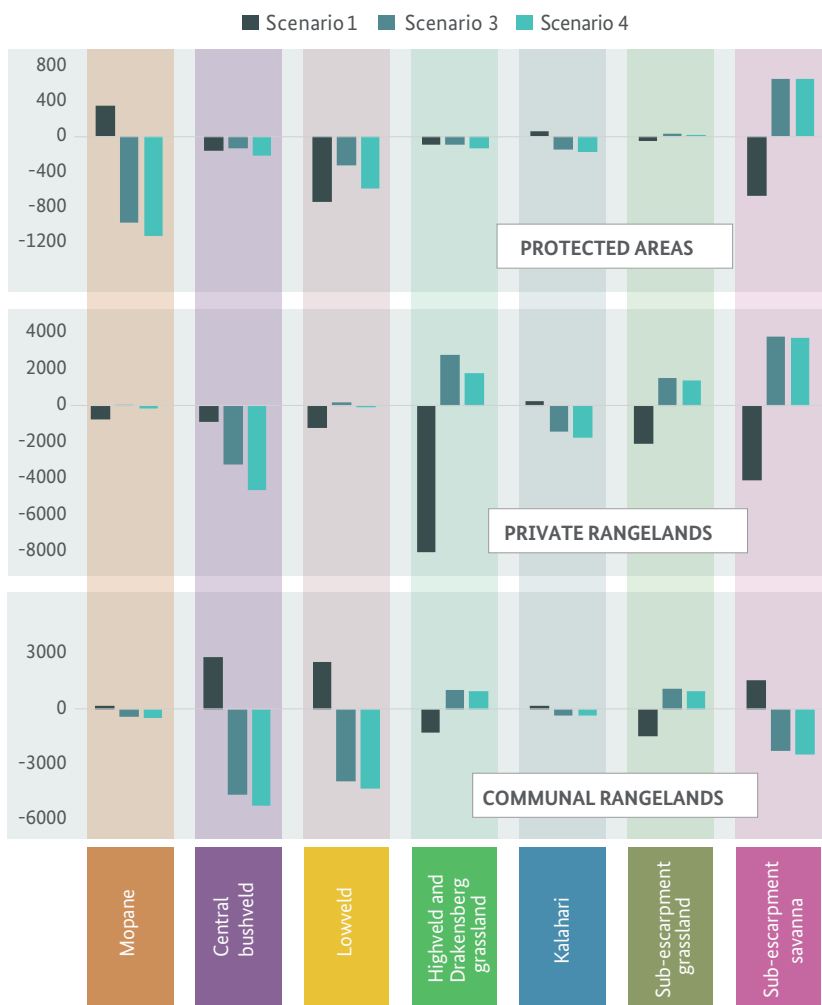


Figure 7.5. The net present value (in R millions) of different policy scenarios (1 – *laissez faire*, 3 – manual clearing and 4 – mechanical clearing) for each land-tenure type (protected areas, private rangelands and communal rangelands) for each region.

The results suggest that while active restoration leads to ecosystem service gains, in the absence of generating any income from the biomass removed, the costs are likely to outweigh these benefits in most cases (Figure 5.5). There are, however, clear cases where such policies would probably have a positive welfare outcome, regardless of method and without the potential benefits from harvesting biomass, such as the entire sub-escarpment grassland region, the protected areas and private rangelands in the sub-escarpment savanna region, and the private and communal rangelands in the highveld and Drakensberg grassland region. In the cases where it is feasible to actively clear, it is more feasible to clear manually than mechanically, owing to the lower cost and the manageable number of

hectares needing to be cleared which limits the time to achieve restoration. However, lessons learnt from the Working for Water programme suggest that the actual cost to clear may be far higher than initial estimates.

In the cases where active clearing without using the biomass harvested is unlikely to yield a positive net outcome for society due to the high costs involved, we estimated how much profit would need to be made from the harvested biomass in order to make active clearing economically viable (Table 5.9). These figures can later be further evaluated based on estimates of biomass from the forthcoming national study. Until this is known, it is difficult to estimate the extent to which subsidies might be required.

TABLE 7.9. The approximate harvesting profit required (R/ha) to make manual / mechanical clearing feasible for the areas where it is not already feasible, for the average level of bush encroachment.

ZONE	LAND TENURE		
	PROTECTED (R/HA)	PRIVATE (R/HA)	COMMUNAL (R/HA)
Mopane	1 330 / 1 540	- / 180	1 490 / 1 700
Central bushveld	440 / 660	670 / 960	2 230 / 2 500
Lowveld	290 / 550	- / 90	2 690 / 2 980
Highveld & Drakensberg grassland	240 / 360	-	-
Kalahari	200 / 230	140 / 1704	230 / 260
Sub-escarpment savanna	-	-	-

7.4 DISCUSSION

7.4.1 Limitations of the analysis

The purpose of this desktop analysis was purely exploratory and must be viewed with caution. The above estimates are purely indicative, in that they are constructed from simple models and assumptions, based on few available data at the time of this study. The woody cover estimates also only pertain to the average situations for each bioregion as reflected in published studies, which are likely to have been biased towards the more heavily encroached areas. Thus, our estimates cannot be extrapolated to each region's encroached area as a whole. It should be borne in mind that the results reflect a higher-than-average bush encroachment situation within each zone, that there will be areas of much lower and much higher extents of the problem within each zone, and that the typical or average situation will also vary between private, protected and communal lands within each zone, unlike in this

analysis. Further research should focus on refining the models used here to take geographical variation in environmental factors and woody cover estimates at a finer scale into account, as well as to incorporate better estimates of costs and benefits, including benefits from the use of harvested biomass.

In interpreting the results, this study assumed that the goal of restoration would be to reduce woody cover to its initial extent. In fact, this might not be the best solution for biodiversity or society. In estimating the changes in ecosystem services, we have simplistically assumed linear or exponential relationships between the different ecosystem services and woody cover. In reality these relationships are likely to be sigmoidal (s-shaped) due to substitution and trade-off dynamics inherent in demand for ecosystem services and due to the sigmoidal relationship between

biomass and woody cover. Apart from removing the inherent inaccuracy, a better handle on these relationships would allow the determination of the level of woody cover that would maximise benefits and therefore determine the optimal degree of restoration.

It is also important to note that not all values are equal, since some ecosystem services are more important for sustaining local livelihoods than others. For example, the gain in carbon sequestration from increased woody cover has a substantial long-term benefit for South Africans, however, when weighed against a loss in water yields for catchment areas for example, the latter should dictate policy more. Furthermore, not all values are captured in this study, such as the cultural and spiritual values of biodiversity.

7.4.2 Inaction may be risky

Although the 'cheapest' option for government, the financial savings from not intervening and not introducing more effective land management practices would be offset by the loss in ecosystem services in most regions and across most land-tenure types. The exceptions would be all the communal rangelands except for the grassland regions, protected areas in the Mopane and Kalahari regions, as well as private rangelands in the Kalahari region where increased bush encroachment delivers gains in ecosystem services. In these cases, it may prove beneficial to allow bush encroachment to continue up to a point. However, it should be borne in mind that (a) the rate of ecosystem service gains is likely to slow down as bush encroachment progresses, (b) households in communal areas have multiple sources of income, some of which require grasslands, and (c) allowing bush encroachment to continue may carry a significant risk to biodiversity. Thus, a policy of inaction, even if only applied to these areas, would not be recommended. At the very minimum, policy measures leading to more effective land management should be implemented (scenario 2).

7.4.3 Active clearing will need to be incentivised

The results indicate that while in most areas the costs of clearing would exceed the benefits, for some areas (the highveld and Drakensberg grassland region and the sub-escarpment grassland and savanna regions), landowners should already have the incentive to clear their land based on their private costs and benefits of clearing. However, very little active clearing is actually being implemented on private

rangelands. This suggests that in addition to the direct costs, certain barriers may be at play, such as lack of understanding of the benefits of or methods for taking action, lack of access to the capital to tackle the problem, or lack of willingness to engage additional labourers. In general, the costs are high enough that a significant income from the harvested biomass will probably be required in order to stimulate clearing action. This potential is unknown at this stage, but if the biomass generated is sufficient to make value-adding activities profitable as a stand-alone venture, then bush clearing may take place even where other benefits to society might be relatively small. In other words, the outcomes depicted in Figure 5.5 could be far more positive.

The viability of value-adding activities will depend not only on the account of woody biomass yielded but on location in relation to centres of demand. Thus, the reality is that this would only be a partial solution, and that some additional assistance may be required in areas beyond those benefiting from such business opportunities. Applied at scale, these value-adding activities will ultimately be subject to diminishing returns after some point. Thus, while stimulating value-adding activities will clearly be an important strategy for addressing bush encroachment, this cannot be seen as the panacea.

While reliance on value-adding activities seems unavoidable, encouragement of these activities would need to go hand in hand with measures to reduce their potential negative impacts. Where bush clearing does prove to be viable and lucrative, private clearing companies and landowners may have a perverse incentive to stimulate bush encroachment, stimulate regrowth or clear beyond optimal levels. The incentive for private companies to use cheaper and quicker clearing options could also result in substantial soil disturbance and increased potential for regrowth.

In areas where active clearing would be worthwhile, choices will need to be made between manual and mechanical clearing, and these choices are also likely to be influenced by the degree to which the costs of clearing can be offset by value-adding activities that make use of the biomass harvested. Manual clearing is cheaper and can contribute significantly to the creation of unskilled jobs in a Working for Land type programme. This would also give government complete control over the bush clearing operations, which will mitigate the perverse incentives that value-adding activities

(scenario 4) could potentially create in a private-sector scenario (namely clearing of indigenous woody vegetation that should not be cleared). However, there are some risks to this approach. Government programmes are typically less efficient, leading to a lower rate of return and a higher risk of failure. Manual clearing may be less suitable for protected areas where free-roaming predators may pose a health and safety risk, or in private rangelands, where landowners may be resistant to having de-bushing teams on their land for social and environmental reasons, as has been experienced in Namibia (NNF 2016). Mechanical clearing has the appeal of being faster and less labour-intensive, but is costlier. However, value-adding activities may make mechanical clearing viable in some areas, and where this is not the case, it would be advisable for government to subsidise this form of clearing in private land areas where public benefits (such as water provision) are likely to be high.

7.4.4 Improving land management is imperative

Regardless of whether active clearing is pursued, or which method is used, government interventions to improve land management practices in private and communal rangeland areas should be prioritised. This will prevent or reduce bush encroachment in areas where levels of encroachment are still relatively low, and in areas where active clearing has been undertaken. This will avert substantial losses in the future, either from the loss of biodiversity and ecosystem services, or the much higher costs of addressing the problem at a later stage. This policy is not sufficient for areas that have high levels of encroachment, however, since this cannot be

easily reversed through grazing and burning at the scale possible by regular landowners.

While imperative and urgent, it will take some time to train and deploy additional extension officers and it will take time for landowners and custodians to adopt new land management practices. Bush encroachment will continue in the meantime. This has the potential to push more areas beyond the thresholds of irreversible biodiversity loss.

7.4.5 The need for spatially-directed measures

The results of the study suggest that no single policy response would be optimal, but rather that spatially-directed measures should be applied as appropriate. Based on all of the above, it can be concluded that:

- In protected areas, better land management is the best option for addressing the problem, apart from in the sub-escarpment grasslands and savannas, where active clearing would be worthwhile, especially in the latter.
- In private rangelands, active clearing would be the best option in all but the Mopane, Kalahari and Bushveld regions, where better land management would be the best option.
- In communal rangelands, active clearing would only be worthwhile in the grassland ecoregions, and for the rest, where no action may lead to a positive net gain up to a point, measures to improve land management should be introduced to the expected benefit of poor households.

RECOMMENDATIONS

8.1 RECOGNISE BUSH ENCROACHMENT AS A FORM OF LAND DEGRADATION

Bush encroachment is the result of anthropogenic influences on ecosystem functioning, notably poor land management. It should be considered as a form of land degradation, despite the benefits of increased biomass, including carbon sequestration. Continued encroachment could have a significant negative impact on overall supply and value of ecosystem services, biodiversity and livelihoods.

8.2 STRENGTHEN EXTENSION SERVICES AND INSTITUTIONS FOR RANGELAND MANAGEMENT

Extension services for rangeland management should be strengthened and expanded. These should promote sustainable land management practices that prevent, or reduce the rate of bush encroachment, regardless of any other strategy adopted and regardless of region. These practices include long-term sustainable stocking rates, reduction of stock in times of drought, and implementation of rotational grazing practices that provide adequate rest for grazing areas, allowing grass biomass to recover fully. In communal land areas, this may need to be facilitated by the establishment of defined communal grazing areas for defined communities or grazing rights holders.

8.3 IDENTIFY THRESHOLDS OF POTENTIAL CONCERN AND DEVELOP MONITORING SYSTEMS

Rigorous monitoring systems should be developed to ensure that bush encroachment is managed optimally at the national, provincial and local levels, based on the latest scientific evidence. Furthermore, thresholds of potential concern for biodiversity and ecosystem services need to be identified for each of the different regions.

8.4 REMOVE LEGAL BARRIERS AND DEVELOP NORMS AND STANDARDS FOR CLEARING/ THINNING ENCROACHED AREAS

The legal aspects of bush encroachment management should be clarified and (potential) conflicts between the different statutory bodies should be remedied. A set of norms and standards should be developed to reduce bureaucratic delays.

8.5 PROMOTE SUSTAINABLE INCOME-GENERATING BUSH-CLEARING ACTIVITIES IN PRIVATE RANGELANDS

In private rangelands, active clearing should be promoted in all but the Kalahari and Bushveld regions, where better land management would be the best option. To help offset the costs of active clearing, activities that add value to bush biomass should be promoted, but within a regulatory framework that avoids incentivising unsustainable harvesting and other activities with negative impacts. The SAEON study should therefore complement this analysis to help determine potential net benefits and feasibilities spatially.

8.6 ESTABLISH GOVERNMENT-FUNDED MANUAL CLEARING PROGRAMMES IN SELECTED COMMUNAL AREAS

In communal rangelands, manual clearing programmes should be funded in affected areas of the grassland ecoregions and any other localised problem areas.

8.7 SET UP A BUSH ENCROACHMENT INFORMATION AND ADVISORY SERVICE

Because bush encroachment is a complex phenomenon and many land managers may not have the specialised knowledge, or access to experts, to determine the best course of action for their land, we recommend that a national bush encroachment information hub /advisory service should be established. This advisory unit would ideally develop detailed guidelines for the management of bush encroachment in each of the different regions and land use types identified in this report and provide decision-support systems and tools to land managers for bush encroachment management in South Africa.

8.8 CONDUCT FURTHER RESEARCH

Further research is needed into the biodiversity impacts of bush encroachment, the potential effects of woody biomass removal on soil fertility and the possible role of woody cover in restoring degraded soils. In addition, a better understanding is needed of the barriers currently preventing active clearing by landowners in areas where the private benefits of clearing appear to outweigh the costs.

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PERSONAL COMMUNICATIONS

1. Skowno, A. 26 September 2017. Written communication with Andrew Skowno from SANBI regarding unpublished data on vegetation types that potentially contain bush encroachment.
2. Kellner, K. February 2018. Personal communication with Klaus Kellner from North West University regarding general bush encroachment best practice and management considerations and bush clearing costs.



Department of Environmental Affairs

Environment House
473 Steve Biko
Arcadia
Pretoria, 0083
South Africa

DEA call centre:
+27 86 111 2468