

**Restoration of degraded subtropical thickets in the
Baviaanskloof Megareserve, South Africa**

The role of carbon stocks and *Portulacaria afra* survivorship

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Abstract

The semi-arid forms of subtropical thicket in the Eastern and Western Cape have been heavily degraded through unsustainable pastoralism over the last century or more. The degraded areas exhibit a significant loss of above-ground and below-ground carbon stocks, and consequently provide an opportunity for restoration through the formal and informal carbon markets. A prerequisite for the attainment of carbon credits is to ensure sound carbon stock baselines prior to effecting restoration. I report on the carbon stocks (including sub-pools) for a number of intact subtropical thicket types, as well as the differentials between the intact and degraded states (including the sub-pools). Total carbon stocks (TCS in t C ha⁻¹) for intact vegetation (to a soil depth 0–25 cm), ranged from 87.73±6.51 to 70.64±17.24. For degraded vegetation (including old lands), TCS (t C ha⁻¹) ranged from 34.05±3.61 to 21.03±2.70. For all vegetation types, the differentials in TCS along the degradation gradient (0–25 cm) are highly significant and strengthen the possibility for carbon credit financing to catalyse the restoration of the degraded semi-arid subtropical thickets. This study has shown a mean loss of 57.23 t C ha⁻¹ in Baviaanskloof spekboom thickets, when measured to a soil depth of 25 cm.

Portulacaria afra is a key species within the semi-arid subtropical thickets, being a canopy dominant and a driver of soil nutrient status, but has been largely lost from the degraded landscapes. Degraded semi-arid subtropical thicket vegetation lacks regeneration via seedling recruitment; restoration therefore requires the manual replanting of *P. afra* using cut truncheons. Survivorship trials were undertaken in-field in 2005 to establish restoration protocols for *P. afra*, with survivorship being recorded in 2006 and 2008. Overall survivorship for all treatments was found to be 43.2±2.8 % in 2006, dropping to 35.8±2.7 % in 2008. Planting posture (flat or upright) showed the most significant results of all the factors tested in 2006 and

2008. Micro-damming also influenced survivorship in that micro-damming was associated with marginally higher survivorship (47.4 ± 2.0 % with damming vs. 39.1 ± 1.5 % without damming in 2006, and 39.3 ± 1.9 % vs. 32.3 ± 1.5 % in 2008). Higher planting density only showed a significant positive impact on survivorship in 2008. Neither stem diameter nor clumping significantly affected degree of survival. A key finding in the study has been the non-static nature of *P. afra* truncheon survivorship, even after being well established (three years since planting). The results from the study will guide the restoration protocols for the restoration of degraded subtropical thickets, where *P. afra* requires replanting.

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Acronyms and Abbreviations

ANOVA	Analysis of Variance
BMR	Baviaanskloof Megareserve
BNR	Baviaanskloof Nature Reserve
C	Carbon
CAM	Crassulacean Acid Metabolism
CDM	Clean Development Mechanism
CO ₂	Carbon dioxide
CO ₂ e	Carbon dioxide equivalent
DST	Degraded subtropical thicket
ECP	Eastern Cape Parks
ECPB	Eastern Cape Parks Board
GDP	Gross domestic product
GHG	Green House Gases
GIB	Gamtoos Irrigation Board
GIS	Geographic information system
GPS	Global positioning system
Herb C	Herbaceous carbon
IRR	Internal rates of return
IUCN	International Union for the Conservation of Nature
Litter C	Litter carbon
MAP	Mean annual precipitation
NPP	Net Primary Productivity
OL	Old lands
pa	Per annum
PDD	Product Design Document
PES	Payment for ecosystem services
PMU	Project Management Unit

ppm	parts per million
RDB	Red data book
RNC	Restoring natural capital
Root C	Root carbon
SE	Standard error
Soil C	Soil carbon
ST	Intact subtropical thicket
STEP	Subtropical Thicket Ecosystem Planning Project
STRP	Subtropical Thicket Restoration Programme
TAGC	Total above-ground carbon
TBGC	Total below-ground carbon
TCS	Total carbon stocks
UNFCC	United Nations Framework Convention on Climate Change
WfW	Working-for-Water
Woody C	Woody and succulent shrub and tree carbon

Chapter One

General Introduction



1.1. Climate change and the carbon economy

Anthropogenic climate change has steadily received increasing media coverage since the 1992 Earth Summit in Rio de Janeiro, which precipitated the United Nations Framework Convention on Climate Change (UNFCCC) in 1994. Subsequently, the proclamation of the Kyoto Protocol in 1997 became the primary international policy tool in arresting the rate of anthropogenic global climate change caused by Greenhouse Gases (GHGs). Due to the pervasiveness of the GHG problem, world policy plays an important role towards regulating the global carbon balance (Lal 2002). Evidence for climate change is well documented in the current literature – substantiated through rises in mean global air and ocean temperatures; rising sea levels; melting glaciers; melting permafrost and melting ice caps, as well as abnormal climatic events (IPPC 2007).

The accumulation of long-lived GHGs in the earth's atmosphere is responsible for the steady rise in temperature. Principal amongst the GHGs is carbon dioxide (CO₂), while methane, nitrous oxide and halocarbons are listed as the other key anthropogenic gases (IPCC 2007). Current carbon dioxide levels in the atmosphere are at the highest level recorded in the last 420 000 years (Prentice *et al.* 2001). The start of the industrial revolution triggered a significant increase in GHGs with CO₂ levels rising from 280 ppm to 379 ppm between 1750 and 2005 (IPCC 2007).

Climate change is projected to have significant impacts on the global economy and is currently estimated to be responsible for a loss of one percent of the global GDP. Should mean global temperatures rise by 5° C, it could culminate in a 5–10 percent loss of global gross domestic product (GDP), (Stern 2006). At a local level, van Jaarsveld & Chown (2001) predict major shifts in rainfall distribution and intensity, bringing associated implications for agriculture and biodiversity. The boundaries of South Africa's biomes are predicted to shift radically and, in the worst case scenario,

the possible loss of the entire succulent Karoo biome is foreseeable (van Jaarsveld & Chown 2001). The fynbos biome could have a 51–65 % loss of surface area by 2050, precipitating an extinction of 10 % of the endemic Proteaceae (Midgley *et al.* 2002).

Carbon dioxide is a dominant GHG – hence the evolution of the carbon economy, governed by emission targets and carbon credits. The Kyoto Protocol has set emission reduction targets (in metric tonnes of CO₂ equivalent, or CO₂e) for all developed countries listed in annex one to the Kyoto Protocol. Where countries and institutions are able to surpass these targets, each ton of CO₂e below the set target equates to a carbon credit. Similarly, every ton of CO₂e that can successfully be sequestered may qualify for a carbon credit, subject to a number of conditions. Article 12 of the Kyoto Protocol articulates the rules of the Clean Development Mechanism (CDM), whereby developing countries (annex 2) that are able to reduce emissions or sequester CO₂e may qualify for carbon credits. In principle, this allows for developed countries that are unable to meet their reduction targets (or wish to continue to pollute) to purchase carbon credits from developing countries through funding afforestation, reforestation and restoration initiatives. Biosequestration in soils and vegetation has a significant role to play in the carbon economy (Lal 2002); however, Lackner (2003) argues compellingly that, in the short term, a more effective solution to the GHG problem lies in utilising existing technologies. Lackner (2003) demonstrates the issues of sequestration time (10¹ to 10⁵ yrs) and sequestration capacity (10¹ to 10⁵ GtC) in the context of terrestrial carbon sinks, through use of the currently approved CDM methodologies. This possibly implies a limited window of opportunity for restoration ecologists to employ the carbon economy in financing large restoration projects.

Concurrent to the conception of the carbon economy (at the Rio Summit in 1992), the Convention on Biodiversity was produced. Any restoration programme that employs the carbon economy should be acutely aware of the potential tradeoffs that

exist between meeting the obligations of the Kyoto Protocol and the Convention on Biodiversity (Caparrós & Jacquemont 2003). Hunt (2008) demonstrated that, in Australia, at the current carbon credit prices, reforestation with monocultures is financially more attractive to investors than restoration of native rainforest. Furthermore, in a fledgling and rapidly evolving industry (particularly in developing countries) such as the carbon economy, a real danger exists for a policy vacuum in terms of the biodiversity-carbon sequestration trade-off. In terms of restoration via the carbon economy (i.e., afforestation and reforestation), South Africa cannot afford negative publicity, as is outlined in case studies for technology-based carbon projects reported by Bond *et al.* (2007). As the public become more enlightened on the issues of climate change and carbon sequestration, consumer pressure will favour projects and institutions that are able to mitigate climate change with minimal negative social and ecological impacts. "Carbon colonialism" (Bond *et al.* 2007) is one example wherein land is expropriated from communities in developing countries to plant monocultures (mostly exotic species), bringing the associated negative impacts of biodiversity loss, purely to offset the carbon emissions in the first world.

1.2. Restoration and the carbon economy

One of the explicit aims of the Kyoto Protocol's CDM is to facilitate the restoration of degraded forests. However, despite ratification in 2005, very few working examples of CDM-financed restoration programmes have been reported in the literature. Barriers have been cited to be high transaction costs (Knowles 2005, Marais *et al.* 2007); complex and costly accounting (Schlamadinger *et al.* 2007, Lippke & Perez-Garcia 2008); and bureaucracy, leading to the marginalisation of the very communities for which CDM was designed in terms of the carbon markets (Corbera *et al.* 2009). In truth, CDM in terms of afforestation and reforestation can be seen as a dismal failure, with only one project (in China) being approved by the CDM

executive board by the end of May 2008 (Bayon *et al.* 2007 as cited in Corbera *et al.* 2009).

The voluntary carbon market has experienced growth as a consequence of the impediments to the CDM process (Corbera *et al.* 2009). Certification is provided by projects such as Plan Vivo (www.planvivo.org) and the Voluntary Carbon Standard (www.v-c-s.org), with functional projects in Mexico, Uganda and Mozambique.

The potential for restoration, reforestation and land use change to sequester carbon dioxide is widely reported. Iceland, for example, has experienced a significant loss of carbon in the last 100 years through deforestation and degradation, but has implemented measures to meet the Kyoto targets (Arnalds 2004). Mexico has been implementing carbon sequestration projects for a number of years, as part of a Payment for Ecosystem Services (PES) system. Corbera *et al.* (2009) conducted an institutional analysis of Mexico's Programme of Payments for Carbon, Biodiversity and Agro-forestry Services; they concluded that rural people had benefited financially; forests had recovered; and institutional capital had been built. However, on private land in Hawaii, the restoration of forests via the carbon economy is currently not viable, and requires State subsidy (Pejchar *et al.* 2007). A similar situation is reported in Australia, where the income from biosequestration credits failed to provide more than a fraction of the restoration costs (Hunt 2008).

Schlamadinger *et al.* (2007) concede that the current system of the Kyoto Protocol, while not perfect, has provided valuable lessons nonetheless, and that land use activities will still play a prominent role in the second commitment period (post 2012). There can be no doubt that the opportunistic employment of the carbon economy for restoration practitioners is a complex process. In an economic appraisal of investment potential for three Canadian timber species, Yemshanov *et al.* (2007) reported that carbon credits would raise the internal rates of return (IRRs) by merely 0.4–0.6 % (investment for the wood production alone produced

IRRs of 3.4–4.6 %). The question arises whether restoration will be able to provide favourable IRRs, taking into account all the transaction costs, restoration implementation, and increasing oil prices worldwide.

Perhaps more relevant to strategists for restoring degraded landscapes is the growth and interest in the Reduced Emissions from Deforestation and Degradation or REDD projects (Laurance 2008). Whereas restoring semi-arid and degraded landscapes requires massive labour and capital inputs, REDD projects merely require a change in human behaviour. Even at current carbon prices, it has been shown that Cameroonians will find it more profitable to refrain from crop farming and receive compensation from REDD carbon credits (Bellassen & Gitz 2008). REDD projects are likely to provide stiff competition to labour-intensive restoration programmes.

1.3. Carbon sequestration in subtropical thickets

Carbon sequestration is viewed as a crucial ecosystem service in subtropical thickets and has been valued at R80/t in 2002 terms (Turpie *et al.* 2003). The high total landscape-carbon stocks in subtropical thickets, despite the semi-arid landscape (Mills *et al.* 2005a, b, Mills & Cowling 2006, Mills & Cowling submitted), has been somewhat of an enigma for ecologists. Upper ranges of 168 t C ha⁻¹ compete favourably with temperate forests that receive a substantially more favourable water balance (Mills *et al.* 2005a). The evidence of vast losses of carbon stocks during the degradation process in subtropical thickets is overwhelming (Lechmere-Oertel 2003, Lechmere-Oertel *et al.* 2005a, Mills *et al.* 2005a, b; see also Chapter 3). Within the semi-arid subtropical thickets, *Portulacaria afra* (L.) Jacq., as a dominant species, plays a crucial role in both above-ground carbon stocks and below-ground carbon pools, through the processes of litter production and carbon sequestration, the degradation process and, ultimately, the restoration agenda.

1.4. Restoration, carbon credits and subtropical thicket

It has been estimated that approximately 4 000 000 hectares of subtropical thicket in South Africa have been degraded; 1 300 000 million moderately and 2 700 000 severely (Lloyd *et al.* 2002). A significant component of the degradation has been the loss of carbon from various pools (Mills *et al.* 2005b, see also Chapter 3). The vast majority of the damage was prevalent prior to 1990, probably during the economic boom of the mohair industry in the 1980s (Nel *et al.* 2007), or even earlier. The intact vegetation height is generally greater than two meters and, in many areas, more than 70 % canopy cover has been lost. These features make restoration of this vegetation eligible for carbon credits under the current rules of CDM. Furthermore, in the semi-arid or xeric forms of subtropical thickets, natural succession of severely degraded area is non-existent (Stuart-Hill 1991a), hence passing the test for additionality under CDM rules. Depending on the price of carbon credits and the complexities of accreditation and verification, carbon sequestration as a land use and farming practice could rival or surpass the profitability of pastoral farming (Mills *et al.* 2007).

1.5. Subtropical thickets

Understanding subtropical thickets has proved a challenge to South African vegetation scientists (Hoffman & Everard 1987, Cowling *et al.* 2005), partly due to the complex biogeography of the vegetation (Cowling & Holmes 1991). Early authors deemed the vegetation to have karroid origin (Bews 1925 as cited in Cowling *et al.* 2005, Acocks 1975, 1988), although Cowling *et al.* (2005) concede that Acocks (1953) did intimate possible connections to subtropical forest for mesic forms of subtropical thickets. Cowling (1983a) originally believed subtropical thickets to have originated relatively recently in geological time (Holocene), from adjacent biomes, although later, Hoffman & Cowling (1991) allude to the fact that a number of species (e.g., *Atalaya capensis*, *Encephalartos* spp. *Smelloyphyllum*

capense and others) indicate a much longer evolutionary history for subtropical thicket. Recently, using fossil records and phylogenetic data, Cowling *et al.* (2005) conclude that subtropical thicket has an ancient evolutionary history (Eocene) and is the progenitor or “mother” of the other South African biomes. Procheş *et al.* (2006), also using phylogenetic techniques, support the hypothesis that subtropical thickets are an ancient vegetation type.

The classification of subtropical thickets has progressed considerably since the comprehensive work of Acocks (1953) and subsequent revisions in 1975 and 1988. The concept of the veldtype has become entrenched in agricultural and conservation circles. Cowling (1984) introduced the term ‘transitional subtropical thicket’, incorporating Kaffrarian thicket and Kaffrarian succulent thicket, and Everard (1987) expanded the classification to include two more subclasses; mesic succulent thicket, and xeric succulent thicket. Vlok *et al.* (2003) credit Tinley (1975) as the first to introduce the concept of subtropical thicket in terms of being a biome in its own right. Acocks (1988) recognised the vegetation as valley bushveld (in a number of forms), noorsveld and spekboomveld. Within valley bushveld, Acocks (1988) discerned a northern form that stretched from the Kei River mouth northwards, a southern form from the Kei River to Kabeljous River, Fish river scrub, Addo Bush, Sundays River scrub and Gouritz River scrub.

Clearly the work of Everard (1991) played a substantial role in Low & Rebelo (1996) finally introducing the subtropical thicket biome as a recognised biome, consisting of dune thicket (Lubke 1996a); valley thicket (Lubke 1996b); xeric succulent thicket (Lubke 1996c); mesic succulent thicket (Lubke 1996d) and spekboom succulent thicket (Lubke 1996e). Despite the recognition of the biome status in 1996, the publication of “Vegetation of South Africa” (Cowling *et al.* 1997a), finds considerable areas of subtropical thicket curiously mapped amid the savanna vegetation types (Scholes 1997), the succulent karoo (Milton *et al.* 1997) or Nama-karoo (Palmer & Hoffman 1997).

The commissioning of the Subtropical Thicket Ecosystem Planning Programme (STEP) in 2000 culminated in a vast knowledge bank (see Cowling *et al.* 2003, Lloyd *et al.* 2002, Vlok *et al.* 2003) and a quantum leap in the understanding of subtropical thicket. Vlok *et al.* (2003), using the prodigious fieldwork and initial findings of Vlok & Euston Brown (2002), provide a comprehensive summary of the evolution of subtropical thicket classification, as well as the spatial distribution and evolutionary history of subtropical thickets. A major improvement in this work was the inclusion of “mosaics”, which are areas where thicket coalesces with adjacent biomes. In total, 112 various subtropical thicket types were described and mapped. More recently Mucina & Rutherford (2006) revised the vegetation classification for South Africa, coining the term Albany Thicket Biome (Hoare *et al.* 2006), but coalesced STEP’s 112 subtropical thicket vegetation types to 14, and abandoned the thicket mosaic vegetation types. The nomenclature in this thesis will follow Euston-Brown (2006), who remapped and classified the vegetation in the Baviaanskloof Megareserve (BMR) planning domain at a scale of 1:50 000. The various vegetation classifications pertaining to the study area and culminating in Euston-Brown’s vegetation classes (2006) are described in more detail in Chapter 2.

Reports of the overall endemism for plant taxa within subtropical thickets are varied; e.g., 10 % (Beentje *et al.* 1994), 9.2 % (Cowling & Holmes 1991, Everard 1987), 7.3 % (Hoffman & Cowling 1991) and 20 % (Vlok & Euston Brown 2002, Vlok *et al.* 2003). The endemism is skewed towards succulents and geophytes (Cowling & Holmes 1991); specifically the Mesembryanthemaceae, Crassulaceae, Euphorbiaceae and the *Haworthia* spp. (Hoffman & Cowling 1991) as well as Aizoaceae, Asphodelaceae and Apocyanaceae (Vlok *et al.* 2003).

The physiognomy and growth forms within subtropical thickets are highly variable (Hoffman & Everard 1987, Hoffman 1989). Numerous authors report subtropical thickets to be a dense, impenetrable, closed, evergreen, low shrub-land (0.5–8 m), with high levels of succulence and spinescence, creepers and vines, and low

herbaceous cover (Everard 1987, 1991, Acocks 1988, Lubke 1996f). The semi-arid forms have an under-storey of dwarf succulents, dwarf shrubs and geophytes, while the mesic forms support a forb and grass layer (Everard 1991). The herbaceous cover is limited; unreliable from an agricultural viewpoint but crucial to the soil moisture (Stuart-Hill 1989a). Because the climate is generally semi-arid, trees and shrubs have deep root systems; storage organs are common; and succulence and sclerophylly are also common (Hoare *et al.* 2006). The prevalence of numerous species of Mesembryanthemaceae, Crassulaceae, Cotyledons and *P. afra* infer the additional benefit of a plant community that can employ Crassulacean Acid Metabolism (CAM) to photosynthesise in times of water stress. These features combine to make semi-arid subtropical thickets incredibly drought tolerant (Kerley 1996). The woody species, which exhibit a remarkable degree of phenological and physiognomic plasticity (Kruger *et al.* 1997), are predominantly represented by members of the Ebenaceae, Celastraceae and Sapindaceae families (Vlok *et al.* 2003). Midgley (1996) predicts that short vegetation types will have a higher incidence of re-sprouters, and the incidence of coppicing in subtropical thicket supports this theory.

Subtropical thickets, specifically the succulent forms found in the Eastern and Western Cape, have evolved without fire (Kerley *et al.* 1999a) and the shrub components form the forage and production base (Stuart-Hill 1989b). However, Vlok & Euston Brown (2002) believe that fire is the ultimate herbivore and that thickets have been shaped by fire and herbivory. As with other characteristics (e.g., carbon stocks), there is a common tendency to generalise regarding various features of subtropical thickets. Fire will have a high influence in a number of thicket types, such as Gamtoos valley thicket (Vlok & Euston-Brown 2002), especially in mosaics, but clearly has a negligible role in the highly succulent and impenetrable vegetation in other areas.

Indigenous animals are an integral component in the ecology of subtropical thickets, specifically due to zoochory (Kerley 1996, Kerley *et al.* 2004.). Skead (1987, 2007) reports a wide diversity of mammals historically recorded in the vegetation. Herbivores are crucial to seed dispersal (Sigwela 1999) and hence gene flow and long-term evolutionary development of the vegetation. Strangely, no work has been reported of the role of primates in subtropical thicket ecology. Birds, however, have been shown to play a key role in subtropical thicket (Cowling *et al.* 1997b); primarily as seed dispersers (Bruton 1992, Watson 2002) and pollinators (Botes *et al.* 2007). With 307 species recorded in subtropical thickets (excluding mosaics), the biome houses well over half the bird species for the Eastern Cape Province (Dean 2002). Subtropical thicket areas feature prominently in the national "important bird areas" (Barnes 1988).

The recently updated list of international biodiversity hotspots (34) includes the Maputaland-Pondoland-Albany hotspot (Mittermeier *et al.* 2004). To qualify for hotspot status, a region must harbour a significant percentage of the world's biodiversity, as well as be under threat from habitat conversion and the concomitant biodiversity loss. Van Wyk & Smith (2001) describe the botanical diversity and endemism for the Albany Centre, citing the incidence of 4 000 plant species, with levels of endemism at around 15 %. Although the authors excluded the Kouga, Baviaanskloof and Groot Winterhoek mountain ranges and valleys (effectively the entire Baviaanskloof Nature Reserve - BNR) from the Albany Centre, one needs to consider the subtropical thickets within the BNR as intimately linked with this centre of plant endemism. Victor & Dold (2003), in a review of the threatened plant taxa (IUCN categories) of the Albany centre of floristic endemism, listed 126 species, with six extinctions. Clearly subtropical thickets are a conservation priority (Kerley *et al.* 1999b).

1.6. *P. afra* in subtropical thickets

P. afra has been widely recognised as a crucial component of the semi-arid subtropical thickets (Beentje *et al.* 1994) and is probably the most important species economically (Aucamp 1979, von Maltitz 1991). Lechmere-Oertel (2003) has called the plant a keystone species, while Vlok & Euston-Brown (2002) call it a miracle plant. This productive, succulent tree can dominate the subtropical thicket vegetation in terms of biomass, cover and relative abundance (Stuart-Hill 1991a). It also has the ability to provide steady forage through drought periods (Palmer & Pitman 1961), making it of special importance to pastoralists (Aucamp 1979, Stoltz 1991, Stuart-Hill 1989a), and technically should preclude farmers from receiving drought subsidies (Stuart-Hill 1991c).

The plant is widely distributed (Palmer & Pitman 1961, van Wyk & van Wyk 1997, Coates Palgrave 2002), favours north aspects and employs a skirt of low branches when isolated (von Maltitz 1991). The low skirt branches eventually take root, which greatly contribute to the plant's reputation as a soil binder (Palmer & Pitman 1961). The species reproduces asexually and sexually, but the success of sexual reproduction is limited (Lechmere-Oertel 2003), as recruitment via seedlings is rare (von Maltitz 1991). It is worth noting that La Cock (1991) reported that *P. afra* was the most prolific producer of seedlings in the Great Fish River, but that the results were preliminary. Although the plants have prolific seed rain, the seed bank is negligible and seed dormancy is limited (Whiting 1991).

Despite being a cornerstone of the agricultural economy in the semi-arid subtropical thickets and providing valuable ecosystem services (e.g., carbon sequestration), the plant has been subjected to extreme levels of browse pressure and has been removed from vast areas of the landscape (Aucamp 1979, Acocks 1988). Once lost, *P. afra* has very poor re-colonisation capabilities, even after the complete removal of browse pressure (Stuart-Hill 1999), facilitating the invasion of karroid elements

(Hoffman & Cowling 1991) and culminating in a steady state pseudo-savanna (Lechmere-Oertel 2003, Lechmere-Oertel *et al.* 2005a). In summation, *P. afra* does not have the ability to withstand moderate browse pressure from goats (Stuart-Hill 1991b). In fact, it has been hypothesised that pastoralism in subtropical thickets may be ecologically and economically unsustainable (Kerley *et al.* 1999a)

P. afra has unique photosynthetic abilities (Ting & Hanscom 1977) and remarkable drought tolerance (Guralnick & Ting 1986, 1987), providing a competitive advantage in the semi-arid environments. The relevant ecology and ecophysiology of this keystone species is dealt with in greater detail in Chapter 4. The wide-scale loss of this species from the semi-arid thickets; its high pre-colonial relative abundance in the vegetation communities (Lloyd *et al.* 2002); high carbon accrual rates (Mills & Cowling 2006) and the ease of propagation (Swart & Hobson 1994) make *P. afra* a crucial component in this study and in the restoration of semi-arid subtropical thickets in general (Mills *et al.* 2007).

1.7. Degradation and resilience of subtropical thickets

Desertification and degradation affect more than one billion people globally (Bainbridge 2007). In South Africa, Hoffman & Ashwell (2001) provide a comprehensive summary of the pervasive levels of degradation at a magisterial district level. Aucamp (1979) specifically records the degradation of the semi-arid subtropical thickets as a national tragedy, and various authors have alluded to the degradation of subtropical thickets since, e.g., Hoffman & Everard (1987), Stuart-Hill (1989a), La Cock (1990), Vlok & Euston-Brown (2002) and Palmer *et al.* (2004).

Hoffman & Todd (1999) reviewed the reasons for vegetation degradation and concluded that loss of vegetation cover is the primary cause in subtropical thickets, with severe degradation defined as a loss of ecosystem functionality, closely related

to severe loss of phytomass. Lloyd *et al.* (2002) used Vlok & Euston-Brown's (2002) vegetation classification and remote sensing to estimate degradation within the STEP planning domain, and estimated that 60.3 % of original surface area of the inland semi-arid subtropical thickets have been severely degraded, with 88.3 % of the associated mosaics having been severely degraded. Within semi-arid subtropical thickets, severe overgrazing has a significant impact on endemic plant species (Cowling & Holmes 1991), verging on desertification in some areas (Kerley *et al.* 1999a, Turpie *et al.* 2003). It should be noted that in a review by Dean *et al.* (1995) they cautioned on the adoption of the emotive term desertification, preferring to call the degradation in the karoo rangelands dryland degradation. Conflicting evidence and lack of data prevented the authors from categorically declaring the karoo rangelands irreversibly degraded, the vast areas of degraded spekboomveld in the Succulent Karoo perhaps being an exception.

Most authors cite unsustainable pastoralism as the greatest contributing factor to degradation of semi-arid subtropical thicket (Aucamp & Tainton 1984, Kerley *et al.* 1995, Kerley 1996). Pre-colonial societies probably kept large livestock in the Cape 1 000 years ago, and small stock considerably earlier (Smith 1998), although the distribution of these societies may have shifted with medium-term change in rainfall patterns (Bousman 1998). Interestingly, very little evidence exists for goats (*Capra hircus*) in either the archaeological or ethnographical records (Badenhorst 2006). Despite the long association between the vegetation, people and livestock, very little evidence exists for degradation within the subtropical thickets prior to European settlement and there can be little doubt that the greatest contributing factor to the wide-scale degradation has arisen from more recent, improper management of small stock in the semi-arid forms of the vegetation (Kerley *et al.* 1995). Lechmere-Oertel (2003) reports anecdotal evidence of the State promoting 20-fold stocking rates in the 1950s relative to today's recommended carrying capacities.

Differences in feeding behaviour between indigenous and domestic herbivores have compounded the problem. Lechmere-Oertel (2003) reports that goats browse in groups, resulting in highly localised impacts. Goats also negatively affect plant species through trampling, removal of nurse plants, destruction of geophytes and reduction of seed set (Vlok 1989), although Palmer *et al.* (2004) speculate that the threat from goat pastoralism has abated in recent years. Dean & MacDonald (1994) have also demonstrated that livestock units have decreased in recent years in the semi-arid rangelands. It has also been hypothesised that domestic herbivores have greater water demands compared to indigenous animals, creating localised degradation through their restrictions in movement across the landscape (Adams & Kerley 1996). Haschick & Kerley (1999) and Wilson & Kerley (2003a) demonstrated that spinescence, although irrelevant to mega-herbivores, significantly affected bite and forage intake rates in other herbivores (Wilson & Kerley 2003b), implying a measure of co-evolution. More importantly, the authors showed that goats exhibited bite rates greater than indigenous herbivores and are therefore superior foragers (Haschick & Kerley 1996).

Kerley *et al.* (1999b) point out that the vegetation has co-evolved with a high diversity and abundance of indigenous herbivores, making them the primary agents of disturbance (Midgley 1991). Indigenous herbivores, even at high densities, do not bring irreversible degradation (Stuart-Hill 1991a, 1992), although thresholds can be exceeded. Cowling & Kerley (2002) cite elephants (*Loxodonta africana*) as exerting a strong influence on the evolution of subtropical thicket vegetation. Penzhorn *et al.* (1974) reported “conspicuous” changes in the vegetation – as well as significant differences in biomass and a loss of species diversity – as a result of elephant impact. Other authors have reported similar results for elephant herbivory in subtropical thickets (Moolman & Cowling 1994, Lombard *et al.* 2002a, Cowling & Kerley 2002). However, Landman *et al.* (2008) dispute the impact of elephant herbivory *per se*, and believe “knock-on effects”, such as trampling and zoochory, to be more important.

Generally speaking, subtropical thickets are stable (Midgley 1991) but have poor levels of resilience (Hoffman & Everard 1987, Kerley *et al.*, 1995, Stuart-Hill 1999). This is partly ascribed to poor seedling recruitment (Midgley 1991, Stuart-Hill 1989b). La Cock (1992) only found seedlings under the canopies of shrubs and bush clumps, concluding that regeneration of bare open patches is unlikely. This is supported by Sigwela's (2004) findings. In the absence of wide-spread seedling recruitment, re-sprouting or asexual reproduction provides an alternative strategy for many species.

With regard to the crucially important *P. afra*, Aucamp (1979) demonstrated that the species requires exceptionally long periods of rest after a moderate defoliation event. Swart & Hobson (1994) cite *P. afra*'s palatability and slow growth rate as contributing factors in explaining poor resilience in semi-arid subtropical thickets.

Degradation can be a slow and subtle process and the severity will be manifested in a number of forms. Generally where canopy cover is intact, degradation is limited. Canopy cover loss implies a loss of phytomass and species diversity (Lechmere-Oertel *et al.* 2005a) and, ultimately, a loss of total landscape carbon. Moderate levels of disturbance increase the abundance and diversity of herbaceous species and change the microclimate within bushclumps (Henley 2001). The tendency for ancient canopy emergents (e.g., *Pappea capensis*, *Schotia afra*, and *Boscia oleoides*) to succumb to the altered microclimate and perish, vanishing from the landscape, is of grave concern for restoration ecology and conservation (Lechmere-Oertel 2003, Lechmere-Oertel *et al.* 2005a). The longevity of these species and slow growth rates accentuate the need to preserve the plants *in situ*, to avoid the need for restoration of this guild.

The loss of vegetation cover, followed by the loss of litter and later of soil horizons, has profound implications for subtropical thicket soils. Soils in degraded subtropical thickets have a tendency to crust (Mills & Fey 2003, 2004a, c). This, in conjunction

with the loss of the greater part of the topsoil (Lechmere-Oertel *et al.* 2005b) and reduced water infiltration (Mills & Fey 2003, 2004c, Lechmere-Oertel *et al.* 2005b), is predicted to have dire consequences for the production of phytomass as a function of water evapotranspiration, i.e., water-use efficiency (Lechmere-Oertel *et al.* 2005b). Rangeland degradation in other semi-arid vegetation in South Africa has been shown to negatively affect root production, litter production and water-use efficiency (Snyman 2005). Degraded subtropical thickets also show an overall reduction in soil fertility (Mills & Fey 2004a) and lower soil moisture retention capabilities, culminating in a homogenisation of the landscape (Lechmere-Oertel *et al.* 2005b).

Fabricius (1997) describes a step-wise process of degradation for semi-arid thickets in the Great Fish River valley. In essence, the process centres on the loss of phytomass, evidenced in the reduction of bush clump size, homogenisation of the landscape and an overall loss in plant and arthropod species diversity. The end result is a desertified landscape. Some arthropod genera are favoured, with xeric-adapted species exploiting the more degraded areas (Fabricius 1997, Fabricius *et al.* 2003). In the same study, Fabricius *et al.* (2002) established that habitat heterogeneity was more prevalent in intact vegetation than in a highly degraded communal land. The authors contend that the disturbance activity of indigenous herbivores would promote heterogeneity, while homogeneity is a function of overstocking and browse pressure. In cases of highly degraded environments that will require restoration, the significant loss of mycorrhizal activity (Allsopp *et al.* 1996) is of particular concern (Fabricius 1997).

Le Maitre *et al.* (2007) studied the land use practices in the Little Karoo, which has experienced levels of over-stocking in the semi-arid subtropical thickets similar to the rest of the biome. The authors concluded that the current land use practices are not sustainable and have precipitated degradation from the patch through to the catchment scale. The nature of the degradation (principally overgrazing in

subtropical thickets) has significantly changed the hydrology at a landscape level. Interestingly, Turpie *et al.* (2003) highlighted the importance of vegetation cover to the overall landscape hydrological functioning (i.e., improved water infiltration and reduced soil erosion), but concluded that the impact on downstream activities was not a major issue.

1.8. Economic value of subtropical thickets

Sims-Castely (2002) provides a review of income generated from a range of land uses within subtropical thickets. Although the values for a range of goods and services are reported from thicket, pastoralism (see Lloyd *et al.* 2002) and alternative land uses are the primary concerns with regard to degradation and restoration. The incomes reported need to be qualified with measures of sustainability (economic, social and ecological). Ironically, pastoralism which is reported to be unsustainable (Kerley *et al.* 1995, 1999a, b, Hoffman & Everard 1987, Stuart-Hill 1992) provides the lowest incomes: R100–400 ha⁻¹ yr⁻¹ (Sims-Castely 2002), R69 ha⁻¹ yr⁻¹ (Turpie *et al.* 2003), R70–150 ha⁻¹ yr⁻¹ (Kirkman 2006). However, a thorough ecological and economic appraisal of subtropical thicket by Turpie *et al.* (2003) found that the intrinsic value of the thicket (including carbon sequestration) was heavily outweighed by the agricultural values derived. The major limiting factor was the slow growth rate of plants, with recovery periods of hundreds of years for severely degraded thicket. The authors concluded that converting the natural capital into financial capital would be more profitable, in the short term, than farming sustainably, and that the total existence value for intact subtropical thickets for all South Africans was in the order of R1.20 ha⁻¹ yr⁻¹.

While eco-based tourism could potentially provide a much higher income, R1 944 ha⁻¹ yr⁻¹ (Sims-Castely 2002) or R976 ha⁻¹ yr⁻¹ (Langholtz & Kerley 2006), localised feasibility studies do not model a whole-scale or biome level change of land use to

eco-tourism (Kirkman 2006). The author calls for more conservation-management information and better incentives from the State. Smith & Wilson (2002) report that 2.5 % of the surface area of the STEP planning domain had been converted (1996–2002) from pastoralism to game-based (market-driven) farming.

Perhaps worthy of more attention is the change in quality of life for all rural citizens under various land use options. Recent studies indicate that upmarket eco-tourism developments bring a fourfold increase in the number of employment opportunities and a fourfold increase in the average annual salary (Langholtz & Kerley 2006). These results need to be contextualised due to the bias in sampling only the top-end (income earnings) of the spectrum (i.e., upmarket). Figures are likely to be considerably lower for eco-tourism ventures which occupy the lower end of the market spectrum.

The value of subtropical thicket plants in the medicinal trade is considered to be significant (Dold & Cocks 2002): of all the plants traded in their survey, 23 % originated from subtropical thicket and a further 13 % were from both forest and subtropical thicket. Pote *et al.* (2007) have calculated the importance of fuelwood in a mesic subtropical thicket, demonstrating the use of a wide range of species. No studies have as yet quantified the importance of fuelwood in semi-arid succulent thickets.

Milton (2003) defines “emerging ecosystems” as those that have a completely new biotic assemblage (and one that persists for decades) following a radically altered environment, facilitated through cultural, economic or social factors. One could argue that the pseudo-savanna state of the semi-arid subtropical thickets (Lechmere-Oertel 2003) is a prime example. Milton (2003) goes on to conclude that the future state of our emerging ecosystem (degraded subtropical thicket) is a societal decision that still needs to be made. This process, the author argues, needs to factor in inter-generational equity and tradeoffs between inaction and

action, as well as to adopt a comprehensive multidisciplinary cost-benefit approach. Lechmere-Oertel (2003) states that many farmers value the degraded vegetation more than the intact form, citing higher tick loads and inaccessible forage for the intact vegetation. Taking all the above into account would bring into play numerous contentious (and complex) issues: land rights, conservation ethics and intergenerational equity, which are well beyond the scope of this study.

1.9. Restoration background

The resuscitation of major ecosystem functions (e.g., control of soil erosion and reinstatement of nutrient cycles) is the primary motivation behind most restoration projects (Holmes & Richardson 1999). The Society for Ecological Restoration defines ecological restoration as a process of reversing the degradation of an ecological system (SER 2002). Clewell & Aronson (2007) adopt a more holistic approach, taking into account the ecological aspect (restoring species, community structure and ecological function) and the socio-economic aspect (restoring ecosystem goods and services), as well as the cultural aspect (restoring connectivity to nature).

Restoration ecologists warn of the dangers of attempting restoration without a comprehensive understanding of the causal processes. Having an intimate knowledge of ecosystem structure and function is advantageous to understanding cause and effect (Bainbridge 2007). The premise of this research is that unsustainable pastoralism has created a complicated and step-wise process of degradation, ultimately exceeding a threshold of recovery and resulting in an irreversible steady state of desertification, i.e., Lechmere-Oertel's (2003) pseudo-savanna. The assumption is that, once the driver of degradation (i.e. unsustainable pastoralism) has been removed, and restoration has been implemented (cost benefit analysis notwithstanding), it will be possible to return vital ecosystem

functions (e.g., carbon pools and water-use efficiency) to a point where natural regeneration is feasible.

Although currently, even in developed countries, restoration is generally deemed expensive (as high as R350 000 ha⁻¹), it will become a viable economic investment as resources become more limiting and ecosystem services become better quantified (Bainbridge 2007). Clewell & Aronson (2007) have gone as far as calling restoration an “elixir” to socio-economic issues and sustainable economic development, stressing the need to have restoration demonstrably linked to socio-economic upliftment (Aronson *et al.* 2006a).

In South Africa, Milton *et al.* (2003) made the point that while some progress has been made with restoration, it has been highly localised. Regional-scale restoration has not come to fruition. Few published South African restoration protocols (Holmes & Richardson 1999) are available outside of the gambit of post-mining restoration. Even the reporting of restoration trials in semi-arid rangelands is limited to a few cases (e.g., Beukes & Cowling 2003).

Within the context of subtropical thicket, Kerley *et al.* (1999b) make the assertion that, due to the scale of degradation and the mechanisms that arrest the process at the point of desertification, “considerable management input” will be required to effect restoration. The launch of the Subtropical Thicket Restoration Programme (STRP) in 2004, through Working-for-Woodlands within the Department of Water Affairs and Forestry (DWAF), seeks to demonstrate that a bioregional restoration programme is feasible (Powell *et al.* 2004, 2006, Mills *et al.* 2007).

Restoration as a socio-economic/conservation activity alone will be fraught with many challenges. Conservation planning has matured in recent years, with South Africa playing a leading role in many ways (e.g., Rouget *et al.* 2006). In a review of the integration of ecosystem services into conservation planning, Egoh *et al.* (2007)

make the distinction between 'conservation assessments' and 'conservation planning', alluding to frequent conflation of the two. The former implies the largely desktop process of spatially orientated plans for implementation. Only with the inclusion of stakeholder involvement and practical strategies for implementation, does the label 'conservation planning' truly become applicable. The authors further highlight the need for inclusion of more complex facets into both of the above (e.g., evolutionary process and ecosystem services). I would argue that, in the realm of subtropical thicket and given the extent of degradation, it is crucial to include restoration as a major spatial, social and economic factor in bioregional conservation plans. Referring to the extensive degradation in the Little Karoo, Le Maitre *et al.* (2007) point out that restoration costs will be beyond the affordability of landowners, and that a coordinated and systematic approach is needed.

1.10. Restoration in subtropical thickets

Detailed restoration studies in subtropical thickets have been limited. Graham Slater, a farmer on Krompoort farm in the Kirkwood district, re-vegetated a highly degraded north facing slope with *P. afra* truncheons in the late 1970s. Although this effort formed near mono-specific stands after 30 years, truncheon survivorship was good and carbon accrual rates were promising ($3.4 \text{ t C ha}^{-1} \text{ yr}^{-1}$) for restoring the carbon balance and soil nutrient status (Mills & Cowling 2006). This is discussed in more detail in Chapter 3.

Another re-vegetation exercise using *P. afra* has been reported by Swart & Hobson (1994), in which a number of treatments were examined. The authors reported moderate success but stressed the importance of rest from herbivory for extended periods following planting (see Chapter 4).

Hall *et al.* (2003) reported on the natural succession of a 16-year old post-mining site in the Gamtoos River valley, Eastern Cape. Despite the topsoil being carefully stored, replaced and containing propagules, the species composition relative to intact thicket was only moderate (46 % similarity). A key finding was the total absence of geophyte and succulent components. The authors reported reasonable, although highly variable (range 0–100 %), success for survivorship for planted woody saplings. The combination of a favourable climate (MAP 600–700 mm) and a substantial and viable soil seed bank indicate that restoration in these thickets would be considerably more achievable than in semi-arid subtropical thickets. In 2005, Weatherall-Thomas re-sampled the same study site and found that, although there had been a continuation of the restoration trajectory (albeit at a slower pace in terms of species accumulation), percentage cover had also increased. The return of succulents (although not thicket species) as well as two species of geophytes, was noteworthy (Weatherall-Thomas 2005).

Todkill's (2001) study, in the subtropical thickets of the Addo Elephant National Park, explicitly aimed to test whether increasing the patchiness in the degraded landscape (principally through creating various brush piles) would accelerate normal succession. The treatments were hoped to reverse the homogenisation of nutrients in the landscape (Kerley *et al.* 1999b, Lechmere-Oertel *et al.* 2005b). The rationale employed was that increased patchiness would increase localised litter and soil accumulation, improve soil properties, and thereby improve the chances for successful germination of woody species. Todkill (2001) reported that the brush-packing trials were unsuccessful in that the total vegetation cover under the brush piles decreased. The author concludes that establishment of new recruit seedlings using brush piles and seed rain is ill-advised. Todkill (2001) contends that herbivory from mammals is a major factor in seedling survival, which is contrary to Sigwela's (1999) findings. Interestingly, Bainbridge (2007) highlights the danger of ascribing a lack of seedlings to factors other than herbivory. Had the author not witnessed the incredibly short event of invertebrates eating the young seedlings, it would have

been supposed that seed inviability, desiccation or other factors may have been responsible. In addition, the Todkill (2001) study highlighted that the longevity of the seedlings was so short that it indicates that micro-climatic habitats are very unfavourable. Certain soil characteristics (such as electrical conductivity) rapidly recovered after the treatments. Phosphorus levels also increased dramatically within less than one year. Todkill *et al.* (2003) have shown that it is possible to start the process of reversing the homogenisation of subtropical thicket resources through the creation of fertility islands. When considering that the subtropical thickets have experienced 50–120 years of resource leakage, these are significant achievements.

Bontveld is a unique form of thicket or thicket-mosaic (see Vlok *et al.* 2003) where isolated clumps of thicket vegetation occur in a matrix vegetation of an adjacent biome. Watson (2002) investigated the restoration of this vegetation in a grassland matrix, following open cast mining operations near Grassridge, Addo, in the Eastern Cape. Success was achieved in the restoration of the grassland areas, but the subtropical thicket clumps proved problematic. The author concluded that considerably more intervention would be required, principally in restoring the micro-topography and soil depths needed for subtropical thicket species. The whole-scale propagation of woody species was also advocated.

1.11. Aim of the study

The focus of this thesis will be on the restoration of degraded semi-arid forms of subtropical thicket, as a starting point for the restoration of other forms of the thicket biome. The evolution of the formal and informal (or voluntary) carbon economy as a result of the Kyoto Protocol and CDM, requires rigorous baselines to be established in order to quantify any carbon sequestered (see <http://cdm.unfccc.int/EB/index.html> and <http://www.v-c-s.org/>). The establishment

of the carbon stock differentials between degraded and intact subtropical thickets is a key step in accessing the carbon markets that would fund the restoration initiative and forms the first component of this study. Concomitant with this objective would be an assessment of the carbon pool variations as a function of land use for climate mitigation purposes (Ordóñez *et al.* 2008).

The key question for the first component of the study was:

- **Is there significant differential in total carbon stocks between the various subtropical thickets in the intact state, and**
- **in total carbon stocks between the intact and degraded states?**

Chapter 3 presents the results from destructive harvesting of key plant guilds and representative plant species for the subtropical thicket vegetation in the BNR, in the form of allometric relationships and regression equations. From these data, estimates for above ground carbon stocks for four vegetation types (and across the degradation gradient) within the study area, are presented. Below ground data from Mills & Cowling (submitted) is also used to complete the calculations for total landscape carbon stocks.

As previously outlined, within the semi-arid forms of the subtropical thickets *P. afra* is considered the primary agent for restoration initiatives. The second component of the study was to investigate restoration protocols for the optimum survivorship of this keystone species. The key question for the second component of the study was:

- **What are the most effective treatments for optimal survivorship for the *en masse* propagation methods of *P. afra* cut truncheons in field?**

Chapter 4 reports on survivorship data for *P. afra* truncheons planted in June and July 2005. The survivorship for a number of multi-factorial treatments is compared in 2006 and 2008, with a view to the establishment of draft restoration protocols.

Chapter Two

Study Site



2.1. Introduction

The Baviaanskloof Nature Reserve (BNR) was recognised as only one of three protected areas within the fynbos biome which was of sufficient area (199 986 hectares, Boshoff 2008) to warrant the status of a megareserve (Boshoff 2005) i.e. Baviaanskloof Megareserve (BMR). With a planning domain of 1 000 000 hectares, and the inclusion of nearly 762 000 hectares of private and communal land adjacent to the BNR, the BMR is a significant conservation initiative. As a component of the Cape Floral Kingdom, the BNR received World Heritage status in 2004 (Boshoff 2008) in recognition of the high levels of botanical diversity and endemism (ECPB 2007). Considering that it is South Africa's fifth largest protected area (Boshoff 2005) and the largest area of contiguous declared wilderness zone, the probability for ecological processes functioning optimally would be greater than when compared to many fragmented smaller reserves. Boshoff (2005) contends that the BNR has the capacity to sustain and maintain biodiversity indefinitely. It follows that this optimality would be a function of the ecological condition of the nature reserve and that, where degradation has occurred, restoration would be a strategic management intervention. The BNR plays a crucial role in terms of water security for the Kouga Dam; the downstream irrigation for citrus and vegetable farming; and the citizens of the Nelson Mandela Bay Metropol.

2.2. Location

The BNR straddles the Eastern and Western Cape provinces between the towns of Uniondale (33° 38' 48" S and 23° 07' 58" E) in the west and Uitenhage (33° 47' 32" S and 25° 23' 46" E) in the east, with the vast majority of the surface area lying in the Eastern Cape (Figure 2.1). The BNR falls mostly under the local municipal authority of the Baviaans and Kou-Kamma Municipalities and the Cacadu District Municipality.

Although the BNR, with all the associated satellite reserves, traverses 36 quaternary catchments, the work reported in this thesis corresponds with the BNR core zone and deals with primarily the tertiary catchments L81, L82, L70 and all the subsidiary catchments.

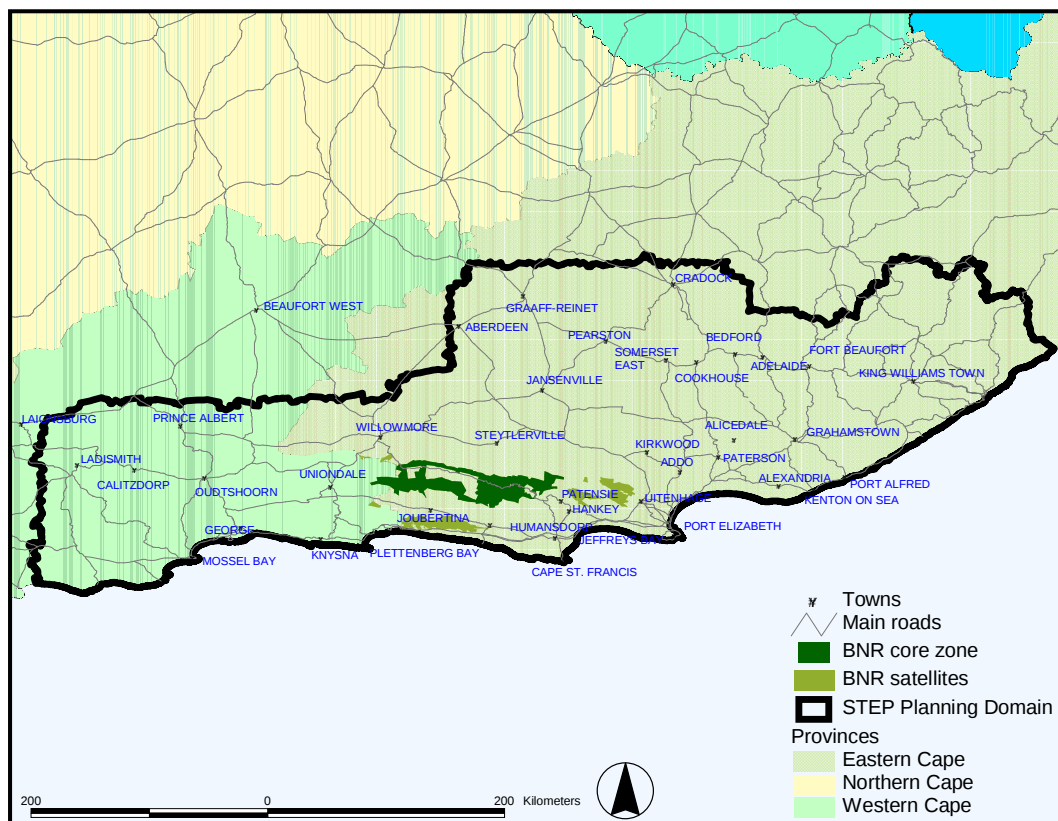


Figure 2.1: Locality of the BNR and the STEP planning domain.

The dominant land use for the Baviaanskloof area has been pastoralism (Teague *et al.* 1989). Consequently, the subtropical thickets have been extensively degraded in many areas (Euston-Brown 1995). From Figure 2.2 it becomes clear that the vast majority of highly degraded subtropical thicket falls outside of the BNR border, especially north of the Baviaanskloof Mountains. Nevertheless, a considerable surface area of degradation within the subtropical thickets of the BNR still exists (see Tables 2.2 and 2.3).

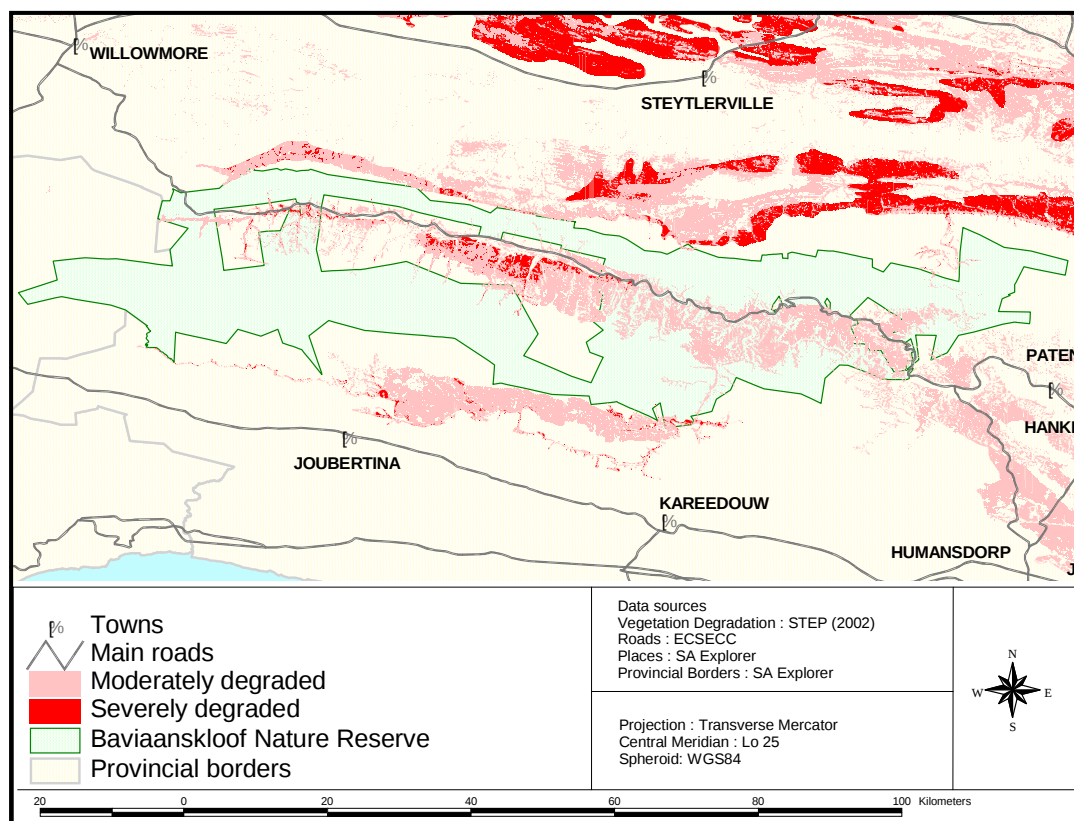


Figure 2.2: Degradation of subtropical thicket vegetation in the Baviaanskloof area (adapted from Lombard *et al.* 2002b).

The degradation of the subtropical thicket biome has been extensive. Lloyd *et al.* (2002) estimate that 42 % of the original surface area of the solid thicket types, as well as 77 % of the mosaic thicket types, has been severely degraded. The nature and drivers of the degradation vary. In the semi-arid rural areas, a loss of biomass and functionality is prevalent and is directly related to over-stocking with livestock. In the coastal areas and around cities, the vegetation is permanently transformed due to urbanisation, cultivation and mining (Lloyd *et al.* 2002). Farming with goats has been cited as being particularly responsible for the loss of vegetation cover (Kerley *et al.* 1999b, Lechmere-Oertel *et al.* 2005a); the loss of biomass and species diversity (Lechmere-Oertel 2003, Lechmere-Oertel *et al.* 2005a) and lowered soil

fertility (Lechmere-Oertel *et al.* 2005b, Mills & Fey 2004a) over much of the semi-arid areas of subtropical thickets.

There is no reason to believe that the drivers of degradation within subtropical thickets in the BNR would have been significantly different from those operating elsewhere.

2.3. Climate

Rainfall is highly variable in the subtropical thickets of the Eastern Cape (Aucamp *et al.* 1982) and long-term drought periods are not infrequent (Aucamp 1979). The rainfall variability is compounded by high temperatures (Stuart-Hill 1989a) with high daily and seasonal fluctuations (ECPB 2007). Aucamp & Tainton (1984) highlight that rainfall is both unreliable and unevenly distributed, with 25 % chance of any area not receiving 80 % of the average annual rainfall. Generally speaking, the BNR is a semi-arid region with a Mediterranean climate (Teague *et al.* 1989) although daily summer temperatures can be extreme $\sim 45^{\circ}$ C (Buckle 1989, ECPB 2007). Precipitation is predominantly winter rainfall in the west and summer rainfall in the east, with spring and autumn peaks. Two aridity gradients exist in the BNR. The north-south aridity gradient is a function of the successive precipitation from orographic rain (Euston-Brown 1995) on the parallel mountain ranges (Tsitsikamma, Langkloof, Kouga and Baviaanskloof). The frontal showers move inland from the south, depositing rain on the successive mountain ranges and leaving the northern slopes of the Baviaanskloof and Groot Winterhoek mountains semi-arid. The mean annual precipitation (MAP) in the Kouga Mountains is 547 mm pa, while that in the Baviaanskloof Mountains is 451 mm per annum (Buckle 1989). The lower altitude (800 m) of the mountains in the east of the BNR allows more precipitation to move northwards and helps to explain the east-west aridity gradient (Euston-Brown 1995). The western section of the BNR has mountain peaks ranging from (1000–1600 m). The vegetation in the valleys in the western extreme of the reserve (near

Willowmore) receives in the region of 200 mm pa, while the vegetation in the valleys in the eastern region receives approximately 700 mm pa (Schulze 1997). Vlok (1989) states that the low-lying areas (<350 m amsl) receive in the region of 250 mm rain pa, while the higher lying areas receive substantially more.

Snow frequently falls on the higher lying peaks (Buckle 1989, ECPB 2007), generally in the fynbos environments and not in the subtropical thickets. Subtropical thickets are subjected to regular light frost events where the vegetation fringes the riparian zones. Irregular (20-year interval) and heavy frost may occur in the valleys, with damage reaching well beyond the riparian zone and causing superficial damage to intact subtropical thicket. Frost is generally light within subtropical thickets (Aucamp & Tainton 1984).

2.4. Geology, geomorphology and soils

The Baviaanskloof in general is topographically extremely rugged (de Villiers 1941), and the study area is restricted to the valley slopes between the parallel ranges of the Kouga and Baviaanskloof mountains. Altitude varies from 100 m to the highest peaks in the ranges, Smutsberg (1 757 m) and Scholtzberg (1 625 m) respectively. The earliest records of geological work come from Schwartz (1903) as cited in de Villiers (1941), who explained the Enon formations as fault pits into which deposits were laid. This theory has since been refuted (de Villiers 1941). Massive uplifting and faulting are viewed as being responsible for shaping the current geology and topography of the area (de Villiers 1941, Theron 1968). The intensity of the upliftment is evidenced in the overfolding patterns and the brecciation of the sandstones (de Villiers 1941, Theron 1968). The mountain ranges that border the sample sites (Kouga and Baviaanskloof) are composed of erosion resistant Table Mountain quartzites. Within the Table Mountain Group, the BNR is dominated by the Peninsula, Skurweberg and Goudini Formations (Euston-Brown 1995). The

valleys are composed of Bokkeveld shales. The Enon Conglomerates are a unique feature of the BNR and are lower Cretaceous gravel deposits (densely packed pebbles and rocks occur in a sandy matrix), often reddish in colour (Truswell 1970) as a result of iron oxides (Euston-Brown 1995). The poor soil development on the steep slopes is due to a combination of erosion rates (Rust & Illenberger 1989), altered fire regimes and overstocking with livestock in the past, which may have accelerated erosion rates. Table Mountain quartzites provide nutrient poor soils, while Bokkeveld shales provide nutrient richer soils (Vlok 1989, Rust & Illenberger 1989). Mills & Cowling (submitted) list Ferrasols, Cambisols, Leptisols, Luvisols, and Calcisols as the dominant soil types in the BNR, being highly variable in spatial distribution due to the topography.

2.5. Vegetation

The vegetation of the BNR is a complex mix of phytochoria, which changes rapidly over short distances (Vlok 1989) with species diversity being a function of the climatic and edaphic variability of the area (Boshoff 2005). The BMR's Project Management Unit (PMU) commissioned Euston-Brown (2006) to remap the vegetation for the entire planning domain of the BMR. This provided a medium-scale (1:50 000) vegetation map, with indices and degradation status. Figure 2.3 depicts only the vegetation contained within the BNR core area (my study site). Euston-Brown (2006) retained the mosaics concept of Vlok & Euston-Brown (2002).

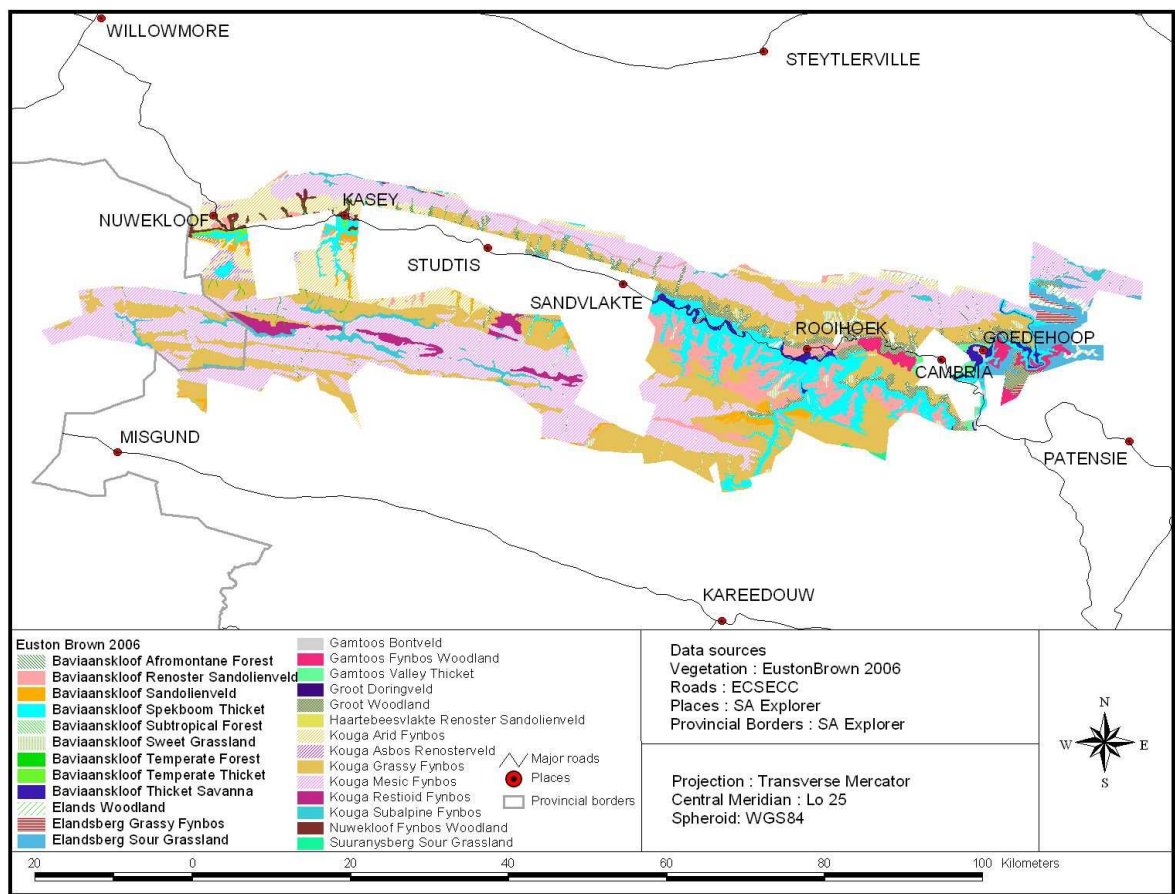


Figure 2.3: Euston-Brown's (2006) vegetation classes for the study site.

The subtropical thickets of the BNR core area occupy approximately nine percent and five percent of the surface area (adapted from Euston-Brown 2006) in the form of solid thickets and thicket mosaics respectively. Subtropical thickets have been described as a dense, evergreen, impenetrable, spinescent tangle of dwarf trees, succulent shrubs, vines and creepers (Cowling 1984, Everard 1987, Kerley *et al.* 1995). Others have referred to subtropical thickets as a two-phase mosaic, with bush clumps interspersed in a matrix of paths and open ground (Fabricius *et al.* 2003). The variability in thicket physiognomy and complexity is typified in the BNR. Figure 2.4 is an example of a dense impenetrable form of subtropical thicket (Gamtoos valley thicket), while Figure 2.5 is an example of more open, patchy vegetation.



Figure 2.4: Gamtoos valley thicket.



Figure 2.5: Baviaanskloof spekboom thicket.

The subtropical thickets are largely restricted to the valleys of the Baviaanskloof, Groot and Gamtoos Rivers, with isolated occurrences in the Kouga river catchment. The subtropical thicket classes that were sampled stretched from Sandvlakte in the west to Goedehoop in the east.

Table 2.1: Relative proportion of Euston-Brown's vegetation classes in the Baviaanskloof Nature Reserve.

Vegetation Class	Hectares	Percentage
Baviaanskloof afromontane forest	581	0.3
Baviaanskloof renoster sandolienveld	11 016	5.8
Baviaanskloof sandolienveld	2 183	1.1
Baviaanskloof spekboom thicket	16 463	8.6
Baviaanskloof subtropical forest	1 327	0.7
Baviaanskloof sweet grassland	1 246	0.7
Baviaanskloof temperate forest	296	0.2
Baviaanskloof temperate thicket	353	0.2
Baviaanskloof thicket savanna	1 972	1.0
Elands woodland	1 353	0.7
Elandsberg grassy fynbos	818	0.4
Elandsberg sour grassland	4 385	2.3
Gamtoos bontveld	238	0.1
Gamtoos fynbos woodland	2 126	1.1
Gamtoos valley thicket	756	0.4
Groot doringveld	0	0.0
Groot woodland	7 391	3.9
Haartebeesvlakte renoster sandolienveld	110	0.1
Kouga arid fynbos	14 189	7.5
Kouga asbos renosterveld	1 088	0.6
Kouga grassy fynbos	46 063	24.2
Kouga mesic fynbos	68 611	36.0
Kouga restioid fynbos	3 194	1.7
Kouga Subalpine fynbos	3 572	1.9
Nuwekloof fynbos woodland	965	0.5
Suuranyesberg sour grassland	117	0.1

Note: Bold text indicates the four vegetation types that are discussed in further chapters.

From a restoration perspective it is important to understand the spatial distribution of the vegetation prior to anthropogenic disturbance. The summary statistics of the

vegetation mapping of Euston-Brown (2006), presented in Table 2.1 provide guidance for the restoration protocols, specifically with regard to restoration trajectories and community assembly rules.

Personal observations while sampling raise some measure of doubt regarding the accuracy of Euston-Brown's (2006) mapping of flammability index within the degraded subtropical thickets. The dataset in all other respects proved to be extremely valuable. Indicator species for the four focus vegetation classes are listed in Euston-Brown (2006) as follows:

- a) Baviaanskloof spekboom thicket: *Portulacaria afra*, *Pappea capensis*, *Schotia latifolia*, *Aloe speciosa*, *Panicum maximum*, *Putterlickia pyracantha*, *Cenchrus ciliaris* and *Stipa dregeana* subsp. *dregeana*.
- b) Gamtoos bontveld: *Pteronia incana*, *Aloe africana*, *Haworthia cooperii* and *Digitaria eriantha*.
- c) Gamtoos valley thicket: *Euphorbia grandidens*, *Aloe speciosa*, *Cussonia gamtoosensis*, *Sideroxylon inerme* and *Ptaeroxylon obliquum*.
- d) Baviaanskloof thicket savanna: *Acacia karroo*, *Plumbago auriculata*, *Lycium* sp., *Rhus tomentosa*, *Celtis africana*, *Cenchrus ciliaris*.

2.6. Degradation gradient

The vegetation classes highlighted in Table 2.1 are the focus of this thesis, particularly with regard to the carbon stock assessments. For each vegetation class, a degradation gradient exists: with intact vegetation at one end of the scale, and degraded or old lands at the other end. The vision at the onset of the project was to have sufficient sampling to describe at least three classes of vegetation condition for each vegetation type: intact, degraded and old lands. A limiting factor in the quantification of the carbon stocks was the policy of restricting sampling to areas of

state land. For this reason it was not possible to find suitable examples of three degradation classes in all four vegetation classes (see Chapter 3).

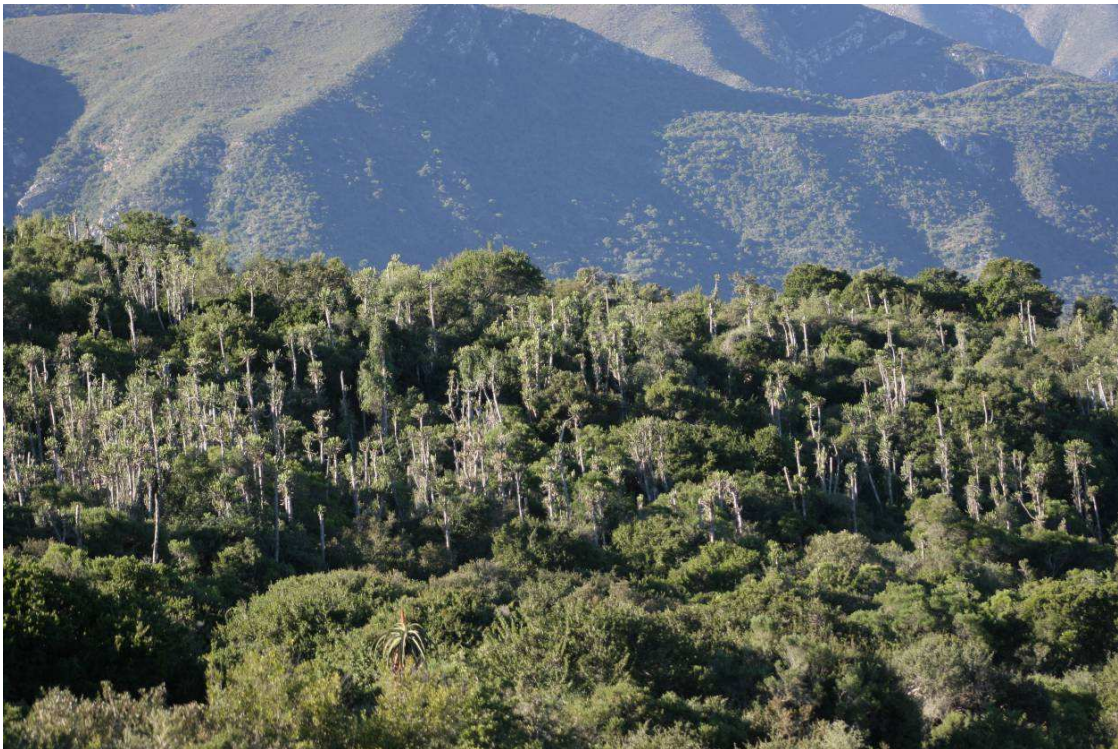


Figure 2.6: An example of intact vegetation, Gamtoos valley thicket (24° 36' 16.582" E, 33° 40' 32.702" S).

Figure 2.6 gives an indication of the complexity of the vegetation structure and, to a limited extent, the diversity of the woody and succulent species. Clearly evident are the charismatic canopy emergent *Euphorbia grandidens* and *Aloe* species. The canopy cover is near-closed (especially on south-facing slopes), and vines and creepers such as *Rhoicissus* proliferate. Grasses (e.g., *Panicum maximum*) are limited to the shade of canopies and canopy fringes, and grasses in general do not constitute a significant percentage of the canopy cover. Succulents (*Crassula* sp. and *Cotyledon* sp.) are common. Bulbs such as *Drimia* sp., *Tritonia* sp, and *Ledebouria* sp. are common, but flower infrequently, making them cryptic to the

casual observer. The litter layer is deep and soils have biotic crusts which are not prone to erosion when the vegetation is in good condition.

Excessive overgrazing in the past has changed the vegetation structure, standing biomass and species composition (Figure 2.7). Entire guilds have been restructured. The shrub succulent guild has almost completely been replaced by the grass guild. Dwarf succulents and bulbs are largely lost from the ecosystem. The high biomass of grass brings the spectre of fire to ill-adapted subtropical thicket species. The large litter layer and soil A horizon have been significantly reduced and in some places totally lost.



Figure 2.7: An example of highly degraded vegetation, previously Baviaanskloof spekboom thickets, at Akkerdal (24° 17' 37.939" E, 33° 37' 55.135" S).

The drastic loss in phytomass and vegetation complexity in old fields is clearly evident in Figure 2.8. The only woody species that is visible in the foreground, *Zygophyllum morskana*, has occupied less than five percent of the surface area,

despite agricultural activities being suspended more than 15 years ago. Isolated individuals of the highly palatable *Grewia robusta* struggle to escape the browse line. The balance of the vegetation comprises invaders from the karroid guild (Mesembryanthemaceae), various grasses that respond to rainfall pulses, and weeds (*Salsola kali*, *Atriplex* spp., *Felicia muricata*, and *Galenia pubescens*).

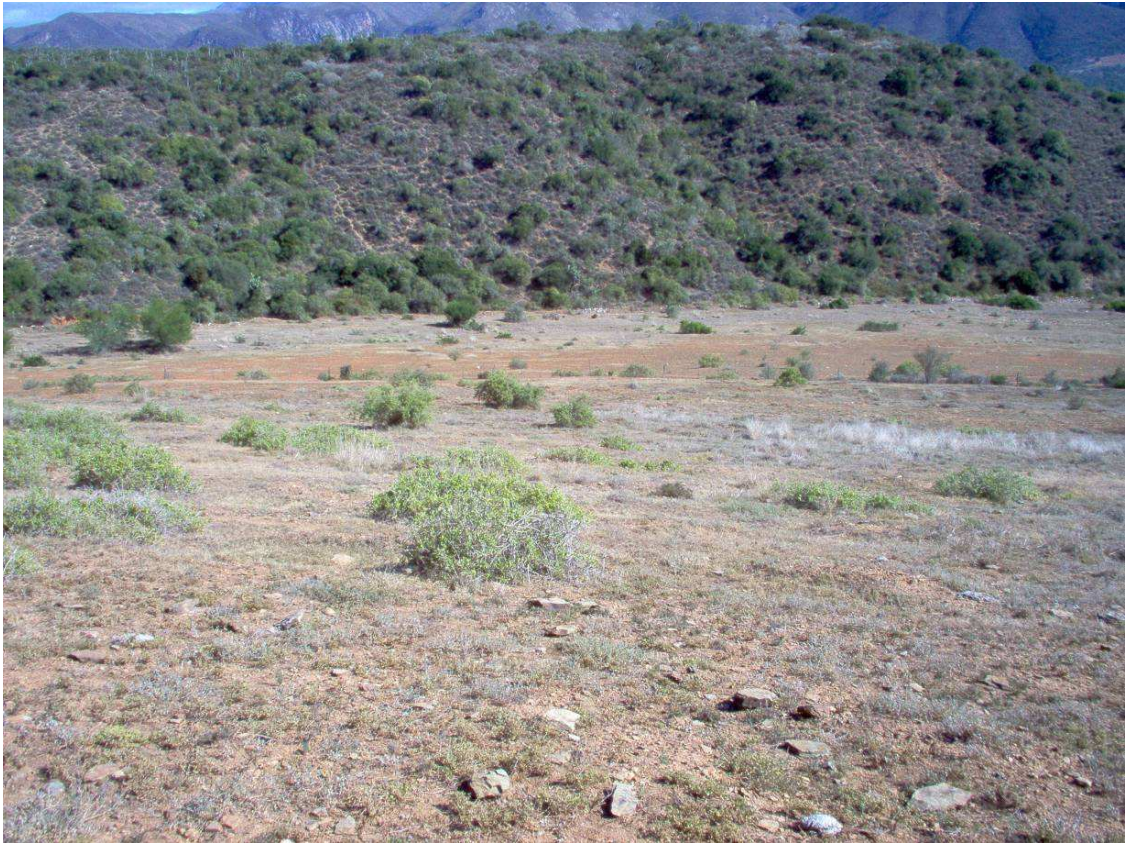


Figure 2.8: An example of abandoned old lands (previously Gamtoos valley thicket) showing no significant plant succession at Cambria (24° 34' 27.378" E, 33° 40' 49.359" S).

The data adapted from Lombard *et al.* (2002b) indicates that, in total, approximately 504 hectares of subtropical thicket (entirely Baviaanskloof spekboom thicket) have been severely degraded, and 10 200 hectares have been moderately degraded (Table 2.2). Data from Euston-Brown (2006) reveals that considerably more surface area has been degraded within the BNR (Table 2.3).

Table 2.2: Relative proportion of moderately degraded subtropical thicket vegetation in the Baviaanskloof Nature Reserve (adapted from Lombard *et al.* 2002b).

Vegetation Class	Hectares	Percentage
Baviaans spekboom thicket	1 816	17.8
Baviaans valley thicket	6 516	63.8
Gamtoos bontveld	640	6.3
Gamtoos valley thicket	1 241	12.2

Table 2.3: Relative proportion of degraded subtropical thicket vegetation in the Baviaanskloof Nature Reserve (adapted from Euston-Brown 2006).

Baviaanskloof spekboom thickets	Hectares	Percentage
Pristine	119	0.7
Good condition	537	3.3
Partially degraded	1 181	7.2
Severely degraded	14 627	88.8
TOTAL	16 463	
Baviaanskloof thicket savanna		
Heavily degraded	1 020	51.7
Severely degraded	952	48.3
TOTAL	1 973	
Gamtoos bontveld		
Heavily degraded	15	6.1
Partially degraded	102	42.6
Severely degraded	122	51.3
TOTAL	238	
Gamtoos valley thicket		
Good condition	352	46.5
Partially degraded	405	53.5
TOTAL	757	

Note: degradation classes used are those as defined and reported by Euston-Brown (2006). Where the author reports "pristine" vegetation it could be construed as having suffered the least in terms of degradation from pastoralism.

In summary, Euston-Brown (2006) estimates that, of the 19 000 hectares that constitute the subtropical thickets of the BNR, merely 119 hectares are in pristine

condition; with 888 in good condition. This implies that none of the Gamtoos bontveld, Gamtoos valley thickets or Baviaanskloof thicket savannas is in either pristine or good condition, bar 352 hectares of Gamtoos valley thicket which is in good condition. This has a direct bearing on the restoration strategy for the BNR, and also makes the choice for reference sites problematic.

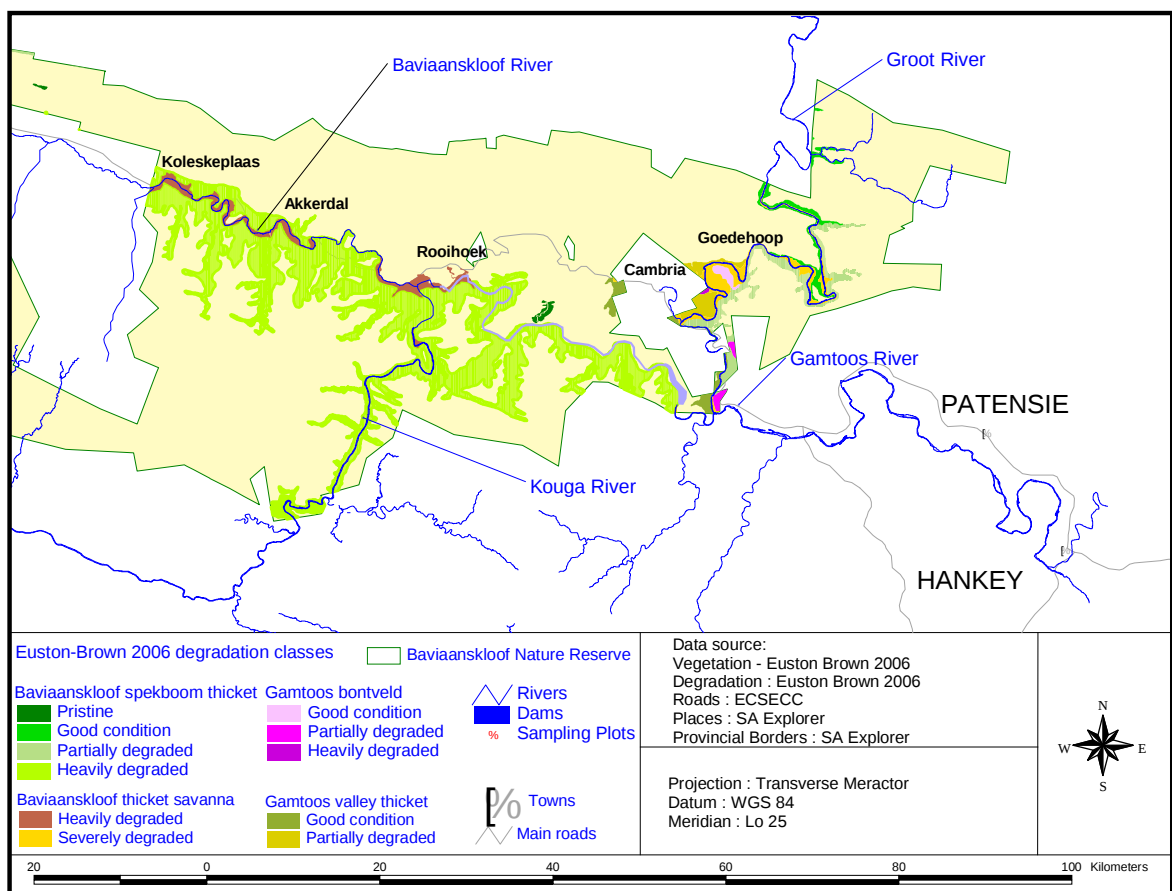


Figure 2.9: Degradation classes for subtropical thickets according to Euston-Brown (2006).

Of particular interest to the restoration strategy in the BNR that focuses heavily on the replanting of *P. afra* is the approximately 15 000 hectares of severely degraded spekboom thickets. Figure 2.9 gives an estimation of the spatial extent of the degradation as quantified by Euston-Brown (2006).

2.7. Fauna

One would expect a nature reserve of this size (and harbouring a complex mix of phytochoria) to support a considerable faunal biodiversity – however, Fabricius (1989) reports low herbivore diversity and biomass in BNR, listing kudu (*Tragelaphus strepsiceros*), grey rhebuck (*Pelea capreolus*), mountain reedbuck (*Redunca fulvorufula*), steenbok (*Raphicerus campestris*), klipspringer (*Oreotragus oreotragus*) and eland (*Taurotragus oryx*). Fabricius (1989) does make a case for the introduction of a number of extirpated species, including bushbuck (*Tragelaphus scriptus*) and black rhino (*Diceros bicornis*), both relevant to thicket ecology and restoration. It is highly improbable that bushbuck required re-introduction. Of special interest for restoration is the report that kudu only moved into BNR in 1979 (Buckle 1989) and black rhino have been reintroduced. These herbivores are extremely destructive towards newly planted *P. afra* truncheons, as are baboons (*Papio ursinus*).

2.8. Management history and planning

The BNR is currently managed by the Operations Directorate of the Eastern Cape Parks (ECP), formerly Eastern Cape Parks Board (ECPB). Land parcels are protected under the National Forest Act (Act 84 of 1998), with proclamations of provincial nature reserves in progress (ECPB 2007). The earliest management came from the Department of Forestry (1920s–1970s) as mountain catchment areas. Much of the higher lying areas were always under State control or crown land (Boshoff 2005, 2008). In the 1970s, private agricultural land was expropriated for catchment management purposes, partly for the building of the Kouga Dam. Land parcels were added post 1986 as part of a consolidation strategy. Policy change in 1987 saw the area revert from the national Forestry Department to the provincial government – Cape Nature Conservation (Boshoff 2005, 2008). With the advent of democracy in 1994 and the evolution of nine provinces, the management was

transferred to the newly formed Eastern Cape Government, and specifically the Department of Economic Affairs, Environment and Tourism. Final management transfer to the ECPB took place in 2004 (ECPB 2007).

The Eastern Cape Parks Board has produced a first draft integrated management plan for the BNR (ECPB 2007). The explicit purpose of this document is to provide focus on clearly defined objectives and activities, thereby providing the framework for prioritised operations (resources, management, reserve use, etc.) for a five year period. Restoration and rehabilitation are itemised under Key Result Area Two. Although restoration of subtropical thickets is mentioned and key performance indicators are listed, a spatially explicit restoration plan with a monitoring plan is lacking.

2.9. Conservation value

The BNR is of national and international conservation importance. It harbours good examples of seven (Boshoff 2005) of South Africa's nine biomes (Mucina & Rutherford 2006) and is situated in the transition zone between two of the world's 34 biodiversity hotspots – The Cape Floristic Region Hotspot and the Maputaland-Pondoland-Albany Hotspot (Mittermeier *et al.* 2004). The BNR has more than 1 100 plant species (ECPB 2007), of which 20 are endemics and 52 are listed as red data book (RDB) species (Boshoff 2005, ECPB 2007). The BNR also featured prominently in the spatial outputs from the STEP conservation planning process and forms a core component of their Megaconservancy Networks (Knight & Cowling 2003, Knight *et al.* 2003, Rouget *et al.* 2006). Unfortunately, many of the areas within the BNR had not been under formal state protection before their expropriation in the 1970s, or are recent inclusions through the ECPB land acquisition programme. Injudicious pastoralism prior to amalgamation into the conservation estate has rendered these areas severely degraded.

2.10. Conclusions

The vegetation of the BNR is exceptionally complex; even within the subtropical thickets there is considerable variability in growth form and vegetation complexity. The subtropical thicket areas are particularly degraded and in urgent need of restoration if this area is to retain its world heritage status, and regain its lost biodiversity and full natural capital. This could provide a significant opportunity for much needed employment in the western region of the BNR and also provide income for the BNR (and, later, the adjacent communities) in the form of carbon credits and other potential PES opportunities. The complexity and variability in the vegetation (both vegetation structure and species diversity) within the BNR needs to be taken into account in the formulation of a restoration master plan, as well as the specifics of restoration protocols.

Chapter Three

Carbon stocks in the intact and degraded subtropical thickets of the Baviaanskloof Nature Reserve



3.1. Introduction

Within the Albany Thicket Biome of South Africa (Mucina & Rutherford 2006), the more arid forms of subtropical thickets, *viz.* spekboom thickets and spekboomveld (Acocks 1988, Vlok & Euston Brown 2002, Vlok *et al.* 2003) are often dominated by *P. afra* (Penzhorn *et al.* 1974, Lubke 1996e, Acocks 1988 Vlok *et al.* 2003). Noorsveld (Acocks 1988), although included as a semi-arid form of subtropical thicket (Vlok & Euston-Brown 2002, Vlok *et al.* 2003 and Hoare *et al.* 2006), is not dominated by *P. afra*, but rather by the dwarf succulents *Euphorbia bothae* and *Euphorbia coerulescens*.

Pastoralism within the semi-arid subtropical thickets, especially with goats (*Capra hircus*), has been the dominant land-use for a number of decades (Kerley *et al.* 1999a, b). The spekboomveld and spekboom thickets have experienced extensive degradation (Aucamp 1979, Hoffman & Everard 1987, Lloyd *et al.* 2002, Esler *et al.* 2006), resulting in a loss of

- 1) phytomass (Lechmere-Oertel 2003, Lechmere-Oertel *et al.* 2005a, Mills *et al.* 2005a, b);
- 2) species diversity (Lechmere-Oertel *et al.* 2005a, Sigwela *et al.* 2003);
- 3) soil carbon (Mills & Fey 2004a, b, Mills *et al.* 2005a,b);
- 4) soil fertility (Mills & Fey 2004a, Lechmere-Oertel *et al.* 2005b),
- 5) water penetration (Kerley *et al.* 1999b, Mills & Fey 2004a);
- 6) litter production (Lechmere-Oertel 2003, Lechmere-Oertel *et al.* 2008).

It is highly probable that there has also been a loss of invertebrate species diversity in the extreme cases of desertification, similar that reported in the adjacent succulent karroo biome (Seymour & Dean 1999).

With *P. afra* being a principal browse component in semi-arid subtropical thickets (in excess of 50 % of the phytomass (Aucamp 1979), it follows that severe degradation includes the loss of this species from the plant community (Swart & Hobson 1994,

Lechmere-Oertel *et al.* 2005a). It is well recognised that once semi-arid subtropical thickets are degraded, they remain degraded (Stuart-Hill 1989a, 1999, Kerley *et al.* 1995), with seedlings being rare, even in intact vegetation (von Maltitz 1991, Midgley & Cowling 1993). Plant recruitment is also extremely low to non-existent (Hoffman & Everard 1987). Intact subtropical thickets have a thick litter layer (~10 cm) (Lechmere-Oertel *et al.* 2005b), which is vital for seedling establishment (Sigwela *et al.* 2003), but is lost in the degraded states due to the absence of leaf litter production. This litter loss also greatly affects water infiltration rates and hence sediment transports (Scheltema as cited in Stuart-Hill 1999) and could therefore play a key role in *P. afra* truncheon survivorship in the context of restoration plantings in degraded landscapes.



Figure 3.1: Degradation contrast in spekboom thicket from pastoralism across a fence line in the Kirkwood area.

P. afra reproduces both sexually and asexually (with seed rain being prolific) but because it exhibits low levels of seed dormancy, no significant seed bank exists (Whiting 1991). Seedlings *per se* are also rare (von Maltitz 1991), all of which implies that restoration requires active replanting. Despite *P. afra* having remarkable abilities to produce roots from cuttings, *P. afra* does not re-colonise these degraded areas (Stuart-Hill 1999) (Figure 3.1.). However, it has been clearly demonstrated that the manual planting of the species can recoup a significant portion (@ mean

rate of $4.1 \text{ t C ha}^{-1} \text{ yr}^{-1}$, Mills & Cowling 2006) of the total landscape carbon of 209–250 t C ha^{-1} (Mills *et al.* 2003, Mills *et al.* 2005a, b) that has been lost ($\sim 80\text{--}95 \text{ t C ha}^{-1}$, Mills *et al.* 2005a, b, and 86 t C ha^{-1} , Mills & Cowling submitted).

Article 12 of the Kyoto Protocol makes allowance for the reforestation of degraded forest areas (degradation prior to 1990), whereby metric tonnes of carbon sequestered can be verified and traded as carbon credits. Thus, the carbon economy could hypothetically provide the funding for large-scale restoration of semi-arid subtropical thickets. The work completed by the Subtropical Thicket Ecosystem Planning (STEP) team greatly advanced knowledge in terms of the spatial distribution of thickets (Vlok & Euston Brown 2002, Vlok *et al.* 2003), as well as the levels of degradation (Lloyd *et al.* 2002). The results and publications from the Conservation Farming Project funded by the World Bank, laid much of the foundation for the results presented below (Lechmere-Oertel 2003, Lechmere-Oertel *et al.* 2005a, b, Lechmere-Oertel *et al.* 2008, Sigwela *et al.* 2003, Turpie *et al.* 2003, Mills & Fey 2004a, b, Mills *et al.* 2005a, b).

The primary objective of the South African Government's Subtropical Thicket Restoration Programme (STRP) has been to develop cost-effective baseline methodologies, and to calculate the carbon stocks. In this chapter I describe the approach to and the results of the carbon stocks in intact versus degraded states, for four subtropical thicket vegetation types in the BNR.

3.2. Methods

3.2.1. Above-ground carbon

The vegetation classes sampled were stratified according to three degradation levels: Old lands (**OL**), degraded subtropical thicket (**DST**) and intact subtropical thicket (**ST**). For each vegetation class, ARCVIEW 3.2 ESRI software was employed

to spatially capture homogenous zones for each degradation class, using 2004 1:30 000 ortho-rectified digital photography. Within these zones, grids were overlaid (5mx5m in ST and DST, and 25mx25m in OL). Using scripts in ARCVIEW 3.2, grids were randomly selected for sampling. A total of 65, 50 and 76 grids were selected for old lands, degraded subtropical thicket and intact subtropical thicket, respectively. Grids were marked out in the field using steel poles, red danger tape and 100 m tape measures and spatially captured into GIS using sub-meter TRIMBLE PROXRS real-time GPS. Grids were marked out in a nested plot design (all corners and all plots marked with permanent steel pegs and danger tape) with the 25 m² and 625 m² being the outer grid for the DST/ST and OL, respectively. The outer grid was used to employ allometry, within which 4 m² grids were laid for sampling all herbaceous biomass, and a 0.25 m² plot was laid out for litter sampling. All herbaceous and litter material was collected on site, labelled and sent for drying to constant mass, and then converted to C by a 0.48 conversion ratio. Tree fall and large dead wood on the forest floor is not a feature of semi-arid thickets and all dead wood encountered was deemed negligible and incorporated into the litter fraction. All woody and succulent tree and shrub species were measured indirectly in field (canopy dimensions, stem number and stem diameters), and guild specific regressions (Table 3.3) employed to estimate standing carbon stocks, hereafter referred to as "woody C". All carbon stocks were calculated for 25 m² (ST and DST) and 625 m² for OL, and extrapolated to one hectare.

Allometric data were obtained from 14 species in the Cambria area of the BNR. The species were chosen as representatives from the dominant guilds found in the intact subtropical thickets (Table 3.1). Only a limited number of guilds chosen are similar to those used by Euston-Brown (1995), working on the same study area. This work excludes the "restiod", "proteaoid", "sedge" and "ericoid" growth forms, which are not prevalent in the Albany thicket biome. A significant deviation from the Euston-Brown's (1995) growth form classes was the separation of "elytopappoid" and "karroid" guilds in this study. Grasses were also not separated into C3 and C4, but

were pooled with the entire “herbaceous” C. Trees were divided into canopy emergents and succulent trees (stem and leaf succulents or stem succulents). Shrubs were divided into multi-stem and mega-multi-stem and stem succulents. Geophytes, although identified in the species diversity assessments, were not separated for biomass quotients and included in the non-allometric “herbaceous” biomass methods. Unknown woody species were lumped into the multi-stem guild. Allometric relations for *P. afra* were taken from destructively harvested mean stem basal area-C relationships for size classes 10 mm, 20 mm, 30 mm, 40 mm and 50 mm (n= 100, 5 397, 100, 5 397, and 200, respectively). *Pteronia incana* (karroid shrub) was included later in the allometric methods, and consequently stem diameters were not measured nor cumulative basal stem area tested for regression purposes. Allometric data for *P. capensis* (n=5) from Skowno (2003) supplemented the data collected in this study and improved the predictive power of the relationship.

Table 3.1: Species destructively harvested for allometric regressions.

Species	Family	Guild	Sample No.
<i>Acacia karroo</i> Hayne	Fabaceae	Canopy tree	15
<i>Aloe ferox</i> Mill.	Asphodelaceae	Succulent Tree: stem and leaf succulent	25
<i>Crassula ovata</i> (Mill.) Druce	Crassulaceae	Succulent shrub: stem and leaf succulent	21
<i>Ehretia rigida</i> (Thumb.)	Boraginaceae	Multistem shrub	24
<i>Euphorbia grandidens</i> Haw.	Euphorbiaceae	Succulent Tree: stem and leaf succulent	25
<i>Grewia robusta</i> Burch.	Tiliaceae	Multistem shrub	37
<i>Jatropha capensis</i> (L.f) Sond.	Euphorbiaceae	Succulent Shrub: stem succulent	21
<i>Lycium ferocissimum</i> Miers	Solanaceae	Multistem shrub	35
<i>Pappea capensis</i> Eck. & Zeyh.	Sapindaceae	Canopy tree	22
<i>Plumbago auriculata</i> Lam.	Plumbaginaceae	Mega-Multistem Shrub	21
<i>Portulacaria afra</i> Jacq.	Portulacaceae	Succulent shrub: stem and leaf succulent	5
<i>Pteronia icana</i> (L.) Szyszyl.	Celatraceae	Karroid bush	49
<i>Putterlickia pyracantha</i> (Burm) DC.	Asteraceae	Multistem shrub	46
<i>Rhus longispina</i> Eck. & Zeyh.	Anacardiaceae	Multistem shrub	24

The plants were measured *in situ* for canopy dimensions (height and diameter), as well as number of stems and basal stem diameter at the soil surface. Mean canopy diameter was assessed by averaging the length of the longest axis and the axis perpendicular to it. Basal stem diameters were measured to two decimal places (mm) using digital callipers (Mitutoyo, series 500, model CD6CX), always above the basal swelling/buttress. Where stems tapered (e.g., *Aloe* species), three measurements were taken (basal, midsection and upper) and averaged. Harvested plants were transported to a sorting facility, where each tree was stripped into leaves, fruits, flowers, thorns, branches and stems. The wet material was weighed and labelled for drying and archiving. Subsamples were taken for each species and each component (leaves, stems, thorns, etc.) and dried at 60^o C to a constant mass. Drying was undertaken in a standard industrial fruit drying oven (2mx1mx1m). The species-specific and component specific (leaves, thorns, etc.) dry:wet ratios were employed to convert the remainder of the wet biomass to dry biomass. All dry mass to carbon conversions were made using 1:0.48 conversion ratio. All C data and tree architecture data was log₁₀ transformed. The C (kg) data for each species was tested in a range of regression models using combinations of canopy and stem dimensions for suitably robust relationships using Statistica 8.0 (StatSoft Inc. 2008). The independent variables included: log₁₀ canopy height, log₁₀ mean canopy diameter, log₁₀ canopy area, log₁₀ stem number, and log₁₀ cumulative basal stem area (CBSA). With multiple regressions, all independent variables were first tested for correlations, and the multiple regression discarded where correlations >0.5. In all cases single regressions were eventually employed. Species (not harvested) encountered in the field were allocated to a specific guild based on growth form and the appropriate regression analyses employed to predict the standing C accordingly (Table 3.2).

Table 3.2: Guild-specific regression equivalents (Table 3.1) for species not destructively harvested.

Species	Equivalent	Species	Equivalent
<i>Aloe africana</i>	<i>Aloe ferox</i>	<i>Ficus sur</i>	<i>Acacia karroo</i>
<i>Aloe comptonii</i>	<i>Aloe ferox</i>	<i>Gnidia sp</i>	<i>Putterlickia pyracantha</i>
<i>Aloe lineata</i>	<i>Aloe ferox</i>	<i>Grewia occidentalis</i>	<i>Ehretia rigida</i>
<i>Aloe speciosa</i>	<i>Aloe ferox</i>	<i>Gymnosporia polyacantha</i>	<i>Grewia robusta</i>
<i>Aloe striata</i>	<i>Aloe ferox</i>	<i>Gymnosporia buxifolia</i>	<i>Ehretia rigida</i>
<i>Anthospermum galioides</i>	<i>Plumbago auriculata</i>	<i>Helichrysum sp.</i>	<i>Pteronia incana</i>
<i>Aspalathus sp</i>	<i>Lycium fercissimum</i>	<i>Indigofera spp.</i>	<i>Putterlickia pyracantha</i>
<i>Azima tetracantha</i>	<i>Putterlickia pyracantha</i>	<i>Lantana rugosa</i>	<i>Plumbago auriculata</i>
<i>Boscia oleoides</i>	<i>Pappea capensis</i>	<i>Melia azedarach</i>	<i>Acacia karroo</i>
<i>Brachylaena ilicifolia</i>	<i>Ehretia rigida</i>	<i>Nymanian capensis</i>	<i>Lycium fercissimum</i>
<i>Bromus sp.</i>	<i>Lycium fercissimum</i>	<i>Olea europea subsp. africana</i>	<i>Pappea capensis</i>
<i>Capparis sepiaria</i>	<i>Putterlickia pyracantha</i>	<i>Passerina vulgaris</i>	<i>Putterlickia pyracantha</i>
<i>Cotyledon sp</i>	<i>Jatropha capensis</i>	<i>Rhigozum obovatum</i>	<i>Lycium fercissimum</i>
<i>Diospyros dichrophylla</i>	<i>Ehretia rigida</i>	<i>Rhus glauca</i>	<i>Rhus longispina</i>
<i>Diospyros lycoides</i>	<i>Ehretia rigida</i>	<i>Rhus lucida</i>	<i>Rhus longispina</i>
<i>Diospyros scabrida</i>	<i>Grewia robusta</i>	<i>Rhus refracta</i>	<i>Rhus longispina</i>
<i>Dodonae angustifolia</i>	<i>Lucium ferocissimum</i>	<i>Rhus rehmanniana</i>	<i>Rhus longispina</i>
<i>Elytropappus rhinocerotis</i>	<i>Lycium fercissimum</i>	<i>Scotia latifolia</i>	<i>Pappea capensis</i>
<i>Eriocephalus sp</i>	<i>Plumbago auriculata</i>	<i>Scutia myrtina</i>	<i>Ehretia rigida</i>
<i>Euclea schimperii</i>	<i>Pappea capensis</i>	<i>Solanum sp</i>	<i>Putterlickia pyracantha</i>
<i>Euclea undulata</i>	<i>Pappea capensis</i>	<i>Unknown</i>	<i>Putterlickia pyracantha</i>

3.2.2. Below-ground carbon

Methods used are those of Mills & Cowling (submitted). Sampling sites were randomly selected using Arcview software (ESRI 2004). Seventy sites were selected in intact thicket, 47 in degraded thicket and 104 in old lands. Soils were sampled at the following intervals: 0–3 cm (n=3 or 4, a mean value was used for statistical analyses), 3–10 cm, 10–25 cm, 38–42 cm, 58–62 cm, 78–82 cm and 98–102 cm (all n=1). The top three layers (i.e., 0–25 cm) were taken under a canopy of trees/shrubs as well as in the open (i.e., outside of the canopy). The bottom four layers (i.e., 38–102 cm) were only taken in the open. Before excavation, the soil

surface was carefully cleared of all leaf litter and live plant material. Soil samples of 10 by 10 cm for each depth layer were carefully removed using chisels and spoons from the horizontal surface of soil pits during their excavation (i.e., not from the vertical wall of the pits). This method of sampling from a horizontal surface was used so that the volume of the excavated hole could be measured and the volume of sample taken per unit depth in a particular hole was similar. All samples were air-dried, sieved (<2 mm) and analysed for organic carbon (hereafter referred to as soil carbon) using the Walkley–Black method (Walkley 1947) and a subsample of 3–10 cm samples were analysed for particle size (3 fraction, hydrometer method), exchangeable cations by atomic absorption, extractable P (Bray II), resistance and pH (1M KCl). In the study by Mills *et al.* (2005b) bulk density of the soil was estimated using soil texture and carbon content, and rock volume in the soil was not quantified. Due to the fact that soil carbon stocks are based on estimates of mean carbon content and soil mass per land area, excluding rock volume from the calculations of soil mass can lead to overestimates in soil carbon stocks. In this study we took separate samples from each layer to calculate bulk density and rock volume. These bulk density soil samples were dried in an oven at 60 °C until constant mass, weighed and then wet-sieved (<2 mm). The volume of the residual rock fragments was determined by water displacement in a measuring cylinder, and the volume of the excavated hole was determined by carefully filling it with sand of known bulk density. Soil samples for determining root biomass were taken from the above soil pits in the same manner as described for soil carbon samples. Roots were separated from the soil matrix by wet-sieving through a 2 mm sieve, and were then dried at 60° C until constant mass. The carbon content of the roots was assumed to be 50 % of the dry root mass (Birdsey 1996).

3.3. Data analyses

Data were pooled for total C stocks (TCS) = **total below ground C (TBGC, i.e., root C and soil C from Mills & Cowling submitted), + total above ground C (TAGC = herb C, litter C and woody C)** and tested for normality (Shapiro–Wilk’s test). The data were tested for homogeneity of variances using Levene’s test and consequently differences were tested using a Kruskal–Wallis ANOVA using Statistica 8.0 (Statsoft Inc. 2008). Differences are reported at the 0.05 significance level. Correlation matrices were established using Spearman’s test, seeking to establish relationships between various carbon pools and possible predictors. The experimental design of nested plots required the geographic matching of the data pools to establish means and SE for various subtropical thicket types and degradation levels. Time and costs limited the number of sampling sites for the 26 cm to 110 cm soil depths. As a result, these data are used as supplementary input and, unless specifically stated, all results pertain to carbon stocks to a soil depth of 25 cm.

3.4. Results

3.4.1. Allometry

All species specific allometric regressions were highly significant, $p = 0.0024$ to $p < 0.000001$ (Table 3.3). The predictive power of the regressions varies across the species and guilds (R^2 ranged from 0.51 to 0.9). $\log_{10}$ cumulative basal stem area (m^2) and $\log_{10}$ canopy area (m^2) were equally distributed between the species in terms of precision, for regression purposes. $\log_{10}$ stem number, $\log_{10}$ stem diameter, $\log_{10}$ canopy height, and $\log_{10}$ mean canopy diameter for all species did not provide the effective suitable allometric predictor of above ground biomass C.

3.4.2. Total carbon stocks

The intact BNR subtropical thickets (four vegetation types pooled) partition carbon 37.5 % above ground and 62.6 % below ground. Root C and soil C account for TCS in the ratio of 5.6 % and 57.0 % respectively, whereas litter C, herb C and woody C reflect TCS as 4.4 %, 1.1 % and 32 % respectively. All carbon pools to a depth of 25 cm (bar for herb C) showed highly significant differences when comparing all vegetation classes (as a whole) between the three degradation states (i.e. $p < 0.0001$ bar for root C, $p = 0.0002$) (Table 3.4).

Table 3.3: Allometric relationships predicting above ground dry plant carbon (kg) for species destructively harvested in the subtropical thickets of the Baviaanskloof Nature Reserve (CBSA = cumulative basal stem area).

Species	n	R equation	R ² value	F	df	p	SE
<i>Acacia karroo</i>	15	$\text{Log}_{10} y \text{ (C (kg))} = 2.034(\text{Log}_{10} \text{ canopy area (m}^2\text{)}) - 1.20113$	$R^2 = 0.9513$	253.72	(1,13)	<0.000001	0.18367
<i>Aloe ferox</i>	25	$\text{Log}_{10} y \text{ (C (kg))} = 1.4306(\text{Log}_{10} \text{ CBSA (m}^2\text{)}) + 3.6975$	$R^2 = 0.7780$	80.60	(1,23)	<0.000001	0.39567
<i>Crassula ovata</i>	21	$\text{Log}_{10} y \text{ (C (kg))} = 1.1337(\text{Log}_{10} \text{ CBSA (m}^2\text{)}) + 1.9764$	$R^2 = 0.9672$	559.53	(1,19)	<0.000001	0.19500
<i>Ehretia rigida</i>	24	$\text{Log}_{10} y \text{ (C (kg))} = 0.9623(\text{Log}_{10} \text{ CBSA (m}^2\text{)}) + 2.485$	$R^2 = 0.6343$	38.16	(1,22)	<0.000001	0.35008
<i>Euphorbia grandidens</i>	25	$\text{Log}_{10} y \text{ (C (kg))} = (\text{Log}_{10} \text{ CBSA (m}^2\text{)})$	$R^2 = 0.9249$	135.47	(1,23)	<0.000001	0.19868
<i>Grewia robusta</i>	37	$\text{Log}_{10} y \text{ (C (kg))} = 1.0044(\text{Log}_{10} \text{ canopy area (m}^2\text{)}) - 0.6259$	$R^2 = 0.8502$	198.58	(1,35)	<0.000001	0.39335
<i>Jatropha capensis</i>	21	$\text{Log}_{10} y \text{ (C (kg))} = 0.9067(\text{Log}_{10} \text{ canopy area (m}^2\text{)}) - 0.7349$	$R^2 = 0.5728$	25.47	(1,19)	0.000072	0.43507
<i>Lycium ferocissimum</i>	35	$\text{Log}_{10} y \text{ (C (kg))} = 0.8615(\text{Log}_{10} \text{ CBSA (m}^2\text{)}) + 1.7706$	$R^2 = 0.7676$	108.98	(1,33)	<0.000001	0.48157
<i>Pappea capensis</i>	22	$\text{Log}_{10} y \text{ (C (kg))} = 1.3355(\text{Log}_{10} \text{ canopy area (m}^2\text{)}) + 0.1357$	$R^2 = 0.9265$	251.99	(1,20)	<0.000001	0.24783
<i>Plumbago auriculata</i>	21	$\text{Log}_{10} y \text{ (C (kg))} = 1.0821(\text{Log}_{10} \text{ CBSA (m}^2\text{)}) + 2.7320$	$R^2 = 0.9296$	250.93	(1,19)	<0.000001	0.16392
<i>Portulacaria afra</i>	5	$\text{Log}_{10} y \text{ (C (kg))} = 1.1043(\text{Log}_{10} \text{ CBSA (m}^2\text{)}) + 2.4464$	$R^2 = 0.9696$	96.47	(1,3)	0.002240	0.12412
<i>Pteronia incana</i>	49	$\text{Log}_{10} y \text{ (C (kg))} = 1.4032(\text{Log}_{10} \text{ canopy area (m}^2\text{)}) - 0.4224$	$R^2 = 0.9679$	1419	(1,47)	<0.000001	0.15833
<i>Putterlickia pyracantha</i>	46	$\text{Log}_{10} y \text{ (C (kg))} = 1.0622(\text{Log}_{10} \text{ CBSA (m}^2\text{)}) + 2.7834$	$R^2 = 0.7784$	154.58	(1,44)	<0.000001	0.33364
<i>Rhus longispina</i>	24	$\text{Log}_{10} y \text{ (C (kg))} = 1.1012(\text{Log}_{10} \text{ canopy area (m}^2\text{)}) - 0.2938$	$R^2 = 0.5077$	22.68	(1,22)	<0.000001	0.45575

Table 3.4: Carbon pools ($\text{t C ha}^{-1} \pm \text{SE}$) to a soil depth of 25 cm for pooled Baviaanskloof Nature Reserve thickets in three states of degradation.

Degradation State	Woody C*	Herb C	Litter C	Root C	Soil C	TCS
OL (n=56)	3.46 $\pm$ 0.84	0.67 $\pm$ 0.08	0.86 $\pm$ 0.22	2.28 $\pm$ 0.43	21.38 $\pm$ 1.42	28.66 $\pm$ 1.95
DST (n= 44)	4.00 $\pm$ 0.72	0.93 $\pm$ 0.11	1.39 $\pm$ 0.31	2.62 $\pm$ 0.63	21.56 $\pm$ 1.67	30.50 $\pm$ 2.05
Intact ST (n=47)	26.50 $\pm$ 3.85	0.90 $\pm$ 0.16	3.69 $\pm$ 0.72	4.65 $\pm$ 0.67	47.35 $\pm$ 4.43	83.08 $\pm$ 5.75
% OL (n=56)	12.1	2.3	3.0	7.9	74.6	100
% DST (n= 44)	13.1	3.1	4.6	8.6	70.7	100
% Intact ST (n=47)	31.9	1.1	4.4	5.6	57.0	100

* includes succulent trees and succulent shrubs

OL=old lands; DST=degraded subtropical thicket; ST=subtropical thicket; TCS=total carbon stocks

The degraded states reflect a higher percentage of the total C in the below-ground pools (approximately 8–9 % for root C and 70–75 % for soil C of the TCS), relative to intact states. The herb C pool for degraded states approximated 2–3 % of the TCS, while litter C (3–5 % of TCS) was proportionately similar to intact subtropical thickets. Woody C pools showed the greatest change in proportionate loss (percentage of TCS) and approximated 12–13 % for the degraded state. When comparing the four subtropical thicket types across the degradation gradient and the variation between the C pools (Table 3.5), some interesting patterns arise.

3.4.3. Comparing C pools for old lands

Old lands in Baviaanskloof thicket savanna exhibited mean TCS of $21.03 \pm 2.70 \text{ t C ha}^{-1}$ (n=14) to a soil depth of 25 cm, while Baviaanskloof spekboom thickets were estimated at $34.05 \pm 3.61 \text{ t C ha}^{-1}$ (n=15) to the same depth. The highly significant differences in root C ($p < 0.0001$) but not soil C ($p = 0.2444$) between old lands from the four subtropical thicket types, is notable. Herb C varied significantly between the various subtropical thicket types ($p = 0.0082$). TAGC and TBGC stocks for old lands between all vegetation types showed no significant differences ($p = 0.8244$ and 0.088 , respectively).

Table 3.5: Mean C pools $\pm$ SE in t C ha⁻¹ for the four subtropical thicket types in the BNR to a depth of 25 cm.

Degradation State	Carbon Pool	Baviaanskloof Spekboom	Gamtoos Bontveld	Gamtoos Valley Thicket	Baviaanskloof Thicket	Significance	
		n=25	n=6	n=11	n=14	n = 56	P
Old Lands	Litter C	0.66 $\pm$ 0.25	0.34 $\pm$ 0.12	1.69 $\pm$ 0.95	0.81 $\pm$ 0.16	7.43	p=0.0594
	Herb C	0.59 $\pm$ 0.10	0.40 $\pm$ 0.11	1.15 $\pm$ 0.27	0.55 $\pm$ 0.10	11.79	p=0.0082
	Woody C*	5.40 $\pm$ 1.58	3.86 $\pm$ 3.12	0.52 $\pm$ 0.27	2.14 $\pm$ 1.18	5.63	p=0.1312
	Root C	3.34 $\pm$ 0.78	3.43 $\pm$ 1.92	1.31 $\pm$ 0.27	0.64 $\pm$ 0.39	25.29	p<0.0001
	Soil C	24.06 $\pm$ 2.34	18.22 $\pm$ 2.06	22.75 $\pm$ 3.50	16.89 $\pm$ 1.57	4.16	p=0.2444
	TCS	34.05$\pm$3.61	26.25$\pm$3.25	27.41$\pm$2.95	21.03$\pm$2.70	7.65	p=0.0539
Degraded Subtropical Thicket		n = 44					
	Litter C	1.39 $\pm$ 0.31					
	Herb C	0.93 $\pm$ 0.11					
	Woody C*	4.00 $\pm$ 0.72					
	Root C	2.62 $\pm$ 0.63					
	Soil C	21.56 $\pm$ 1.67					
	TCS	30.50$\pm$2.05					
Intact Subtropical Thicket		n=32	n=11	n=4		n=47	
	Litter C	4.85 $\pm$ 0.99	1.11 $\pm$ 0.40	1.43 $\pm$ 0.82		9.98	p=0.0068
	Herb C	0.61 $\pm$ 0.17	1.04 $\pm$ 0.18	2.84 $\pm$ 0.88		13.62	p=0.0011
	Woody C*	29.00 $\pm$ 3.32	23.95 $\pm$ 13.92	13.59 $\pm$ 7.43		7.13	p=0.0283
	Root C	3.60 $\pm$ 0.58	7.84 $\pm$ 2.12	4.24 $\pm$ 1.41		6.11	p=0.0472
	Soil C	49.68 $\pm$ 6.21	40.14 $\pm$ 4.94	48.55 $\pm$ 11.97		1.09	p=0.5800
	TCS	87.73$\pm$6.51	74.08$\pm$15.62	70.64$\pm$17.24		3.90	p=0.1422

* includes succulent trees and succulent shrubs

3.4.4. Comparing C pools within degraded subtropical thicket

Degraded subtropical thickets in Gamtoos bontveld and Gamtoos valley thicket were not available for sampling within the study area. Degraded forms of the Baviaanskloof thicket savannas were not sampled, due to the fact that *P. afra* is not present in the plant community.

The carbon partitioning in DST for Baviaanskloof spekboom thickets is similar to that of old lands. Litter C, herb C and root C holding the smaller fractions (5 %, 3 % and 9 % respectively). Residual soil C is in the region of 70 % and woody C 13 %, proportionate to TCS.

3.4.5. Comparing C pools between subtropical thicket types

The three intact subtropical thickets sampled (Table 3.5) showed significant differences in all carbon pools, except for soil C ($p=0.58$) and TCS ($p=0.14$). Litter C pools in Baviaanskloof spekboom thickets were 3.7 t C ha^{-1} and 3.4 t C ha^{-1} more than Gamtoos bontveld and Gamtoos valley thickets, respectively. Herb C for Gamtoos valley thickets was $\sim 2 \text{ t C ha}^{-1}$ higher than the other subtropical thickets. Soil C pools of Gamtoos valley thickets and Baviaanskloof spekboom thickets are nearly 10 t C ha^{-1} higher than Gamtoos bontveld. Woody C was found to be 5.1 t C ha^{-1} and 15.4 t C ha^{-1} higher in Baviaanskloof spekboom thickets than Gamtoos bontveld and Gamtoos valley thickets, respectively. TCS in Baviaanskloof spekboom thickets were 13.7 and 17.1 t C ha^{-1} greater than in Gamtoos Bontveld and Gamtoos valley thickets, respectively.

3.4.6. Landscape carbon losses

The results for the various pools (Table 3.5) show only net gains for herb C in Baviaanskloof spekboom thickets (DST vs. ST) and litter C (OL vs. ST) in Gamtoos valley thickets. All other pools show net losses. The pooled subtropical thickets showed a net loss in TCS (0–25 cm) of $53.03 \text{ t C ha}^{-1}$ for degraded

subtropical thicket vs. intact subtropical thicket and 54.42 t C ha⁻¹ for old lands vs. subtropical thicket

The three subtropical thicket types showed net losses of TCS to a depth of 25 cm of 61.2 %, 61.2 % and 64.6 % for Baviaanskloof spekboom thicket, Gamtoos valley thickets and Gamtoos bontveld, respectively (old lands vs. intact subtropical thickets). Intact forms of the Baviaanskloof thicket savannas were not sampled and therefore the net loss of TCS is unknown. TCS loss between the intact state and old lands for Baviaanskloof spekboom thickets, Gamtoos bontveld and Gamtoos valley thickets are 53.68 t C ha⁻¹ 47.83 t C ha⁻¹ and 43.23 t C ha⁻¹, respectively (to a depth of 25cm).

3.4.7. Correlations

In this study I found *P. afra* presence (percentage canopy cover) for pooled intact subtropical thickets to be positively correlated with litter C ($R^2 = 0.387$), woody C ($R^2 = 0.537$), TAGC ($R^2 = 0.516$), and TCS ($R^2 = 0.406$), but negatively correlated with herb C ($R^2 = 0.592$). When viewed in isolation, Baviaanskloof spekboom thickets surprisingly produced only a single significant correlation: *P. afra* percentage cover negatively correlated with herb C ($R^2 = 0.474$). For all subtropical thickets as a whole, as expected, woody C was positively correlated with TAGC ($R^2 = 0.99$) and soil C was positively correlated with TBGC ($R^2 = 0.95$). For all subtropical thickets as a whole, root C was significantly and negatively correlated with succulent shrub percentage cover ($R^2 = 0.36$), as well as soil C ($R^2 = 0.32$). Litter C, woody C, TAGC and TCS were also significantly and negatively correlated with percentage grass cover ($R^2 = 0.56, 0.63, 0.63$, and 0.50 respectively).

For Baviaanskloof spekboom thickets, litter was strongly correlated with both woody C and TAGC ($R^2 = 0.68$ and 0.75, respectively). Root C was again negatively and significantly correlated with soil C ($R^2 = 0.37$).

3.5. Discussion

3.5.1. Measurement

Allometry

Allometric equations are a rapid, accurate and non-destructive technique to assess above ground plant carbon (e.g., Northup *et al.* 2005, Vallet *et al.* 2006). Opportunities to use innovative techniques such as described by Montès *et al.* (2000) are limited in subtropical thickets due to the impenetrability of the vegetation and cohabitation of adjacent woody and succulent species. Allometric relationships for most semi-arid subtropical thicket species are unknown. Skowno (2003) provides one for *P. capensis* ($R^2 = 0.953$, $n=5$). Ntshontsho (2006) reports R^2 of 0.64 for *Acacia kosienses* (a species similar in growth form to *A. karroo*), but limited to the subtropical coastal dune forest in northern KwaZulu-Natal. The allometric regressions reported in this study could aid in future biomass and carbon studies in subtropical thickets. The regressions for *P. afra* and *Pteronia incana* may have value in rangeland condition assessments. More work is needed on species and guilds that were not identified at the onset of the project (e.g., *Cussonia* spp.). Problematic species (poor R^2 values), such as *Jatropha capensis*, may require further investigation, with a view to finding the key parameters that would increase the precision of the regressions. For all species investigated, canopy volume, lower canopy height, and leaf area index (not investigated in this study) may significantly improve the precision of allometrically estimated above ground biomass C. Standing dead wood and coarse dead wood were assumed to be insignificant, and were therefore included in the woody C or litter C pools, respectively. This assumption needs to be tested. A number of species that occur in subtropical thickets, but not in the study area (especially in the more mesic, coastal and dune thickets), may require investigation. A major limiting factor for allometric techniques to determine carbon stocks is the complicated vegetation structure in subtropical thickets. Many species co-habit the same canopy volume, or have peculiar growth forms

due to cohabitation. More work is needed to quantify these effects and establish carbon partitioning strategies for co-habiting species.

Correlations

Whereas carbon storage in savannas and woodlands is largely governed by the number and size of the trees (Scholes 2004), this is unlikely to be the case for the semi-arid subtropical thickets that are dominated by *P. afra*. Unlike other studies (Cairns 1997, Robinson 2004) that found strong relationships between above ground biomass C (excluding litter C and herb C) and root C, these results for intact subtropical thickets failed to do so. No other studies have employed allometric relationships to estimate root carbon in subtropical thickets. Pooled results (all thicket types and all degradation levels) showed poor results ($R^2 = 0.07$) for predicting root C from woody C, and root C from TAGC ($R^2 = 0.06$). Taking intact subtropical thickets in isolation, $R^2 = 0.003$ root C vs. woody C and $R^2 = 0.009$ root C vs. TAGC.

Since root C is negatively and significantly correlated with soil C (0.37), TBGC stocks in subtropical thickets of the BNR may largely be a function of litter fall, and not root necromass or root exudates. This requires more investigation.

Regional carbon stock assessments will require the employment of remote sensing techniques to speed up the process and reduce transaction costs. Until recently, remote sensing techniques for the estimation of biomass C have not been reliable (Mitchard *et al.* submitted). The authors have found favourable results using radar ($R^2 = 0.84$) for prediction of biomass C in the forest-savanna boundary of central Africa, with accuracy decreasing towards the 200+ t C ha⁻¹ range. Challenges may still exist to establishing robust relationships with the other carbon pools that will negate intensive field sampling.

3.5.2. Total carbon stocks

Glenday (2007) reports TCS (to a soil depth of 30 cm) for dry valley thicket in KwaZulu-Natal province of $121 \pm 9 \text{ t C ha}^{-1}$ (MAP ranged from 551–970 mm pa due to topography, altitude and distance from the sea). Soil C accounted for 63 %, root C 8 %, litter C 2 %, herb C 1 %, woody C 24 %, and dead wood 2 % of the TCS. TCS (to a soil depth 40 cm) in the Sahel (280–400 mm rainfall) (Woomer *et al.* 2004) ranged from $12.0\text{--}31.2 \text{ t C ha}^{-1}$ (mean $20.6 \pm 1.8 \text{ t C ha}^{-1}$), with 87 % in the form of soil (60 % of which was retained in the first 20 cm of topsoil). Glenday (2006) estimated TCS (to a soil depth of 60cm) in African tropical forests (rainfall 2 000 mm pa) to be $360 \pm 63 \text{ t C ha}^{-1}$ (Table 3.6).

Table 3.6: Mean $\pm$ SE carbon stocks (t C ha^{-1}) and relative % of C pools to Total carbon stocks (TCS) for intact Baviaanskloof spekboom thickets and pooled Baviaanskloof Nature Reserve (BNR) thicket types compared to other vegetation types in Africa.

Carbon pool	Baviaanskloof spekboom thickets	% of TCS	Baviaanskloof subtropical thickets	% of TCS	Glenday (2007) dry valley thicket	% of TCS	Glenday (2006) tropical forest	% of TCS
Litter C	4.85 ± 0.99	5.5	3.68 ± 0.72	4.4	2.6 ± 0.3	2	5.4 ± 0.9	1.5
Herb C	0.61 ± 0.17	0.7	0.89 ± 0.16	1.1	1.08 ± 0.4	1	0.8 ± 0.4	0.2
Woody C	28.99 ± 3.32	33.0	26.50 ± 3.85	31.9	29 ± 3	24	200 ± 36	56.1
Deadwood C	within litter C	NA	within litter C	NA	1.5 ± 0.3	2	1.2 ± 0.4	0.3
Root C	3.60 ± 0.58	4.1	4.64 ± 0.66	5.6	9.2 ± 0.9	8	49 ± 9	13.7
Soil C	49.67 ± 6.21	56.6	47.34 ± 4.43	57.0	77 ± 8	63	100 ± 17	28.1
TCS	87.73 ± 6.51		83.08 ± 5.75		121 ± 9		360 ± 63	

The salient result from Table 3.6 is that the various carbon pools in this study correspond strongly to those for the dry valley thicket reported in Glenday (2007). This is particularly reassuring for the woody C pool, where potential error is maximal relative to other pools. This is largely due to the complex vegetation structure in thicket, the atypical growth-forms of many of the dwarf trees and shrubs, and the high stem numbers per square meter. The major disparities lie in higher litter C for this study, and elevated root C and soil C in the KwaZulu-Natal dry thickets; all possibly linked to high precipitation in

KwaZulu-Natal or deeper soils. Litter C for the intact BNR subtropical thickets is similar to east African tropical forests, but root C and woody C carbon pools are close to an order of magnitude less (Glenday 2006).

The TCS (0–25 cm soil depth) figures reported here are lower (65–71 %) than those reported for similar thickets in the Eastern Cape: Fish River Reserve ~ 127 t C ha⁻¹ (0–30 cm, Mills & Cowling (2006), Kirkwood, 209 ± 28 t C ha⁻¹ (0–30 cm, Mills *et al.* 2005b), 245 ± 28 t C ha⁻¹ (0–50 cm, Mills *et al.* 2005a). Data from Mills & Cowling (submitted) indicate that, on average, BNR thickets (pooled vegetation types) could store an additional 4.6 t root C ha⁻¹ and 28 t soil C ha⁻¹ in the 26–110 cm depth, equating to a TCS of ~ 117 t C ha⁻¹. Estimates for TCS that include TBGC to soil depths of 110 cm for Baviaanskloof spekboom thickets and Gamtoos bontveld (Mills & Cowling submitted) approximate 120 t C ha⁻¹ and 105 t C ha⁻¹, respectively. Mills & Cowling (submitted) provide possible insights into these discrepancies, citing as possible reasons: 1) not including rock volume in some calculations, 2) differentials in soil nutrient status between sites, 3) soil sampling depth differences, and 4) not factoring in soil bulk density.

The TCS (to a soil depth of 50 cm) for other vegetation types range from 21–50 t C ha⁻¹ in Karoo, 97–164 t C ha⁻¹ in grassland, and 34–65 t C ha⁻¹ in fynbos (Mills & Fey 2004b). Mills *et al.* (2005a) estimate TCS (to a soil depth of 50 cm) in intact grasslands to sequester 172 t C ha⁻¹, fynbos 42–81 t C ha⁻¹ and Karoo ~ 30 t C ha⁻¹. From this one can deduce that there is a reasonable degree of variation in TCS within a biome or particular vegetation type. Given the phenomenal work of Vlok & Euston-Brown (2002) and Vlok *et al.* (2003) it may be imprudent to extrapolate from TCS figures presented in this study. For carbon sequestration credibility in subtropical thickets, each specific thicket type needs to be considered in isolation.

3.5.3. Total above ground carbon

Focussing on spekboom thickets, Mills & Cowling (2006) report 34.2 t C ha⁻¹ for the Great Fish River Reserve. Lechmere-Oertel *et al.* (2005a) report 40.03 t C ha⁻¹, while Mills *et al.* (2005a,b) working in the same Sundays River spekboom thickets found 51.6 t C ha⁻¹ (40 t C ha⁻¹ biomass and 11 t C ha⁻¹ in litter). This compares with 29.60 t C ha⁻¹ for biomass C and 4.85 t litter C ha⁻¹ (a total of 34.46 t C ha⁻¹) for this study (Baviaanskloof spekboom thickets). The reason for the threefold increase in litter C between the Fish-Baviaanskloof spekboom thickets vs. Sundays River spekboom thickets could be a function of mean annual rainfall differentials (~250 mm vs. ~400 mm pa). Increasing aridity favours *P. afra* percentage cover, which is a copious litter producer (Lechmere-Oertel 2003, Lechmere-Oertel *et al.* 2008). Mills & Fey (2004a) speculate that the high percentage soil C in subtropical thicket is due to the high percentage vegetation cover and litter layer build-up, with associated slow decomposition rates.

Gamtoos valley thickets showed an increase in litter C in the degraded state. I hypothesise that this is related to some sites being located on an east aspect, which allowed for accumulation of grass sward necromass (fire having been excluded for a number of years).

3.5.4. Total below ground carbon

The total below ground carbon stocks reported for the intact subtropical thickets in this study (0–25 cm) are considerably less than those reported for similar semi-arid thickets close to the study site, 168±26 t C ha⁻¹, 0–50 cm (Mills & Fey 2004b, Mills *et al.* 2005a), 133±27 t C ha⁻¹, 0–30 cm (Mills *et al.* 2005b), but are very similar to the intact subtropical thickets of the Great Fish River Reserve, i.e., 85.3 t C ha⁻¹, 0–30 cm (Mills & Cowling 2006). Soils in the Great Fish River Reserve sequester 16±3.3 t C ha⁻¹ (0–50 cm) in root C and 69±6 t C ha⁻¹ (0–50 cm) in soil C. Mills & Cowling (submitted) estimate the soils of a similar vegetation type (Baviaanskloof spekboom thicket) to exhibit TBGC of 85.7

($7.7 \pm 1.6 \text{ t C ha}^{-1}$ in root C and $78 \pm 6 \text{ t C ha}^{-1}$ in soil C). It is well established that both root C and soil C diminish rapidly with increasing soil depth (Mills & Fey 2004b). In similar thickets, Mills & Fey (2004b) record TBGC of $71 \pm 7 \text{ t C ha}^{-1}$, in the first 10 cm.

Soil C percentage has been found to average 6.9 % for under canopy-intact vegetation, 3.8 % in the open-intact vegetation, 5.7 % under canopy-degraded and 4.2 % in the open-degraded subtropical thicket (Mills & Cowling 2006), and 5.6 %, 0–10 cm (Mills & Fey 2004a,b) for Sundays River spekboom thickets. Soil C percentage (Mills & Cowling submitted) for this study area range from $0.57 \pm 0.31 \text{ % (SD)}$ in Baviaanskloof thicket savanna, to $5 \pm 0.85 \text{ % (SD)}$ in Baviaanskloof spekboom thickets, for the intact subtropical thickets. Mills and co-workers mention that the soil C percentage values are remarkably high for a semi-arid environment. Mills & Cowling (submitted) have demonstrated the importance of ascertaining soil bulk density and rock volume between degraded and intact subtropical thickets. Degradation brings soil compaction or crusting (Mills & Fey 2004c) and greater bulk density, emphasising the need to calculate soil carbon as a function of soil mass and not soil depth *per se*. The authors also demonstrate the critical value of determining site-specific and depth specific rock volume quotients. Mills & Fey (2004a) speculate that the relatively high percentage soil C in intact subtropical thicket, relative to other biomes (Mills & Fey 2004b) may be related to vegetation canopy influences on rainfall. This remains to be tested.

An inconsistency exists within Mills & Cowling's data (submitted) whereby soil C for DST (Baviaanskloof spekboom thickets) for the 0–25 cm is reported as $18\text{--}25 \text{ t C ha}^{-1}$ and 0–110 cm is recorded as the same (25 t C ha^{-1}). Root C is reported as $1.4\text{--}3.8 \text{ t C ha}^{-1}$ for DT and 4.1 t C ha^{-1} . This begs the question of why the soil C (and most of the root C) in the lower depths of the DST has been completely lost, while the upper soil (0–25 cm) has only experienced 50 percent leakage.

3.5.5. Ecosystem carbon loss

Lechmere-Oertel (2003) and Lechmere-Oertel (*et al.* 2005a) found 40 ± 12 t C ha⁻¹ (n=5) biomass C lost (excludes soil C but includes root C) in DST for Sundays River spekboom thickets. Mills *et al.* (2005a) found a similar result, ~ 43 t C ha⁻¹. This is similar to the results obtained in this study for a similar vegetation type (Baviaanskloof spekboom thickets), ~ 29 t C ha⁻¹ (to a depth of 0–25 cm) and ~ 32 ha⁻¹ t C ha⁻¹ (to a depth of 110 cm). A key factor that needs to be taken into account is the relative proportion of *P. afra* biomass C (i.e., excludes soil C) to the TAGC stocks, as well as TBGC contribution, in the various subtropical thickets across the degradation gradient. In rectifying the carbon balance, a TAGC or TCS net loss can not be solved by the replacement with *P. afra*. A myriad other plant species have been lost – together with all the associated pollination syndromes, genetic diversity and contributions to ecosystem functioning.

Incorporating data from Mills & Cowling (submitted), TCS losses (to a soil depth of 110 cm) in the Baviaanskloof are ~ 67 t C ha⁻¹ (old lands vs. intact) and 84 t C ha⁻¹ (degraded vs. intact) for Baviaanskloof spekboom thickets. These results are very similar to Mills *et al.* (2003) 89 t C ha⁻¹, Mills *et al.* (2005a) 95 t C ha⁻¹, Mills *et al.* (2005b) 80 t C ha⁻¹ and Mills & Cowling (submitted) 86 t C ha⁻¹. Irrespective of the sampling depth, TCS loss of 57 t C ha⁻¹ (0–25 cm), and 84 t C ha⁻¹ (0–110 cm), (equating to 65 % and 71 % respectively) induces a major disruption to nutrient cycling and ecosystem functioning.

The most useful results from this study in terms of carbon trading revolve around the differentials in the various carbon sub-pools, (degraded subtropical thicket vs. intact subtropical thicket) in Baviaanskloof spekboom thickets. All carbon pools were significantly different (ST vs. DST). This vegetation type is more suitable for carbon sequestration through the planting of *P. afra* truncheons, due to the historical dominance of spekboom (mean cover 40.8 ± 2.53 %, n=49). This vegetation is also more widely degraded in the rural

areas, due to livestock farming. Degraded spekboom thickets are well known for supporting a higher percentage cover of grasses that replace the succulent component, culminating in a pseudo-savanna (Lechmere-Oertel 2003). This is supported in this study, in that the DST herb C pool showed a net increase (0.32 t C ha^{-1} , 52.8 %) relative to the intact ST. All other pools showed losses: litter C 71.3 % (3.46 t C ha^{-1}), woody C 86.2 % ($25.00 \text{ t C ha}^{-1}$), root C 27.3 % (0.98 t C ha^{-1}), soil C 56.6 % ($28.16 \text{ t C ha}^{-1}$) and TCS 65.2 % ($57.23 \text{ t C ha}^{-1}$). Mills & Cowling (submitted) expect that combined losses in soil C and root C (26–110 cm) for this thicket type will approximate 27.7 t C ha^{-1} . In summary, the potential to restore lost carbon in degraded spekboom thickets may be as high as 85 t C ha^{-1} , albeit not all in the form of *P. afra* alone.

Soil C residuals in the old lands in BNR are relatively high, implying that TBGC leakage is ongoing. Soil C for old lands in Baviaanskloof spekboom thickets, Gamtoos bontveld and Gamtoos valley thicket all approximate 50 % relative to the intact state. The mean soil C differential between DST and ST for Baviaanskloof spekboom thickets is $28.12 \text{ t C ha}^{-1}$. However, old lands, as a proportion of the total area of degraded subtropical thicket, are likely to form a very small percentage (see Lloyd *et al.* 2002), hence all carbon offset feasibility studies should not use these carbon stocks as baselines for restoration purposes. Furthermore, old lands typically occur in the bottom lands and closer to drainage lines. This implies deeper soils and, consequently, greater root and soil carbon stocks. The areas are often invaded with pioneer species (e.g., *Acacia karroo*, *Rhus* spp.) which confounds the TAGC differentials between intact natural vegetation and degraded vegetation. The vegetation structure of these old lands is markedly different from intact thicket. The invader species also have different carbon partitioning strategies compared with the suite of climax species (e.g., *Pappea capensis*, *Euclea undulata* and *P. afra*).

Despite abandonment in the BNR for at least 15–30 years, and consequent low levels of above ground plant biomass C, soil C values remained high relative to

TCS (80.3 % Baviaanskloof thicket savanna, 69.4 % Gamtoos bontveld, 51.6 % Gamtoos valley thickets and 70.7 % for Baviaanskloof spekboom thickets).

3.5.6. Potential carbon gains through restoration

The Baviaanskloof thicket savanna is, strictly speaking, a riparian form of subtropical thicket and dominated by species such as *Acacia karroo*; it does not harbour *P. afra*, due to high prevalence of frost, and is therefore irrelevant to carbon sequestration in the semi-arid succulent thickets. Riparian thickets do qualify for restoration and carbon sequestration, but they may require a separate methodology, and a more tailored marketing strategy to avoid the high transaction costs. Similarly, Gamtoos bontveld and Gamtoos valley thickets have a minor role in the carbon trading and restoration agenda, and typically will have less *P. afra* dominance when compared to spekboom thickets and spekboomveld. Intact vegetation exhibited mean *P. afra* canopy percentage cover of 15.7 ± 3.60 % ($n = 15$) and 2.7 ± 1.06 % ($n = 11$) for Gamtoos valley thicket and Gamtoos bontveld respectively. The Gamtoos valley is an important citrus producing area and currently, income from carbon credits (as a function of the price of carbon and the SA exchange rate) is no threat to the export orientated citrus industry in the valley thickets of the Eastern Cape.

Considering that the strategy for restoration, funded through carbon credits, rests heavily on the results for Baviaanskloof spekboom thickets from this study, two key pieces of information are required: 1) The net TCS differential between intact and degraded vegetation, and 2) the rate of C accrual.

The carbon sequestration potential for degraded semi-arid thickets, reported in the literature, as well as results from this study, are presented above (3.5.5). These differentials include the carbon contributions from all plant species.

P. afra percentage cover in the Baviaanskloof varies according to vegetation type, slope aspect and geology. Its relative importance as a carbon pump

(Lechmere-Oertel 2003, Lechmere-Oertel *et al.* 2008) will be a function of its percentage cover. *P. afra* has been shown to be the principal producer of leaf litter, making total annual leaf litter production (2.5 t C ha^{-1}) in subtropical thickets on par with other ecosystems that receive considerably more precipitation (Lechmere-Oertel 2003, Lechmere-Oertel *et al.* 2008). In this study, *P. afra* percentage cover in the spekboom-dominated thickets reached a mean of ($40.8 \pm 2.5 \%$). The gains of $4.1 \text{ t C ha}^{-1} \text{ yr}^{-1}$ (Mills & Cowling 2006) for a similar vegetation type with less rainfall were produced with a *P. afra* percentage cover of 93 %. Their sampling, to 100 cm depths for most of the pits, was also done on lower slopes (deeper soils and possibly less rock volume). The authors reported that soils in the Great Fish River Reserve were considerably shallower (50 % were less than 75 cm deep) (Mills & Cowling 2006). Extrapolations to t C ha^{-1} without correction factors for catena or soil depth may invoke over-estimates. Within the context of conservation areas where biodiversity conservation is the primary objective, the ecological impact of super-saturated *P. afra* landscapes needs to be assessed properly. Closed canopy *P. afra* is likely to deprive succulents, forbs and bulbs of water and light. Should super-saturation of *P. afra* be taken as a strategy to replenish the soil nutrient status, conservation authorities need to understand the implications for biodiversity.

3.5.7. Rates for carbon accrual

The rates of carbon return to the system have not been quantified for this study area or for the vegetation types in the Baviaanskloof. Williams *et al.* (2008) found that rested (no slash and burn) miombo woodlands in Mozambique were able to sequester the lost woody C (19 t C ha^{-1}) within a few decades at a rate of $0.7 \text{ t C ha}^{-1} \text{ yr}^{-1}$ but soil C accumulation was exceptionally slow to recover. Ntshontsho (2006) records TCS of 91 t C ha^{-1} ($@1.3 \text{ t C ha}^{-1} \text{ yr}^{-1}$) for a coastal forest restored site (19 years old). A dire knowledge shortage exists for carbon accrual rates or growth rates for subtropical thicket species (Turpie *et al.* 2003). Mills & Cowling (2006) report $4.1 \text{ t C ha}^{-1} \text{ yr}^{-1}$ using *P. afra* truncheons at Krompoort in the Sundays River spekboom thickets. The same study reports a

1.2 t C ha⁻¹ yr⁻¹ carbon return rate and similar rainfall; with the same species, but in the Great Fish River Reserve near Grahamstown.

3.5.8. Potential error margins

To ensure credibility for the long-term, carbon baseline assessments need to highlight potential sources for error, which may come from a variety of sources and, when compounded, could negatively affect carbon investor confidence. Glenday (2007) highlights the importance of selecting the most suitable allometric regressions. Standard carbon assessment techniques employ a conversion factor of 0.5 relative to dry biomass. It has been shown in some cases that this may produce further error in the estimation of total carbon stocks. Carbon content for 41 North American species ranged from 46.3 % to 55.5 % (Lamlom & Savidge 2003). Nabuurs *et al.* (2008) also highlight C density as a key parameter for TAGC estimations. Peltoniemi *et al.* (2006) cite carbon density in Finnish forest vegetation as a key factor influencing the accuracy of TAGC stocks, although estimations were relatively precise. They reported that Soil C estimations were much less precise, citing a number of causal factors (initial state, forest cover change, temperature, etc.). Non-species specific, standardised root:shoot ratios (not employed in this study) are also a potential source of error (Cairns *et al.* 1997).

It is highly probable that soil C and root C are strongly correlated with slope/catena position (soil depth being generally auto-correlated with catena position). A major limitation in the extrapolation of previous studies for carbon stocks in subtropical thickets has been the stratification on the landscape for catena position. In tropical rainforests of Ecuador, vegetation canopy height decreases with increasing altitude (Leuschner *et al.* 2007). Using this established principle, total vegetation volume/biomass is also likely to decline. This will reduce litter input through leaf fall, and should produce a significantly reduced litter layer. Leuschner *et al.* (2007) report that increased altitude saw a drop in leaf area index, as well as a 50 % loss of tree height and total biomass

C, but a net increase in root biomass (both fine and coarse). These patterns need to be tested for subtropical thickets.

The prevalence of wind and water erosive forces is likely to be higher at elevation, and could bring about proportionately lower carbon stocks in these soils (see Jacinthe *et al.* 2004). Annual loss in litter fall can potentially reach 75 % (Kumada *et al.* 2008). Some subtropical thickets have a mosaic (bush clump vs. bare ground) structure (Fabricius 2003, Lechmere-Oertel 2003) and this may change with slope/catena position as well. Should there be more canopy gaps higher up the slopes, there is likely to be increased solar radiation, increased loss of litter input, and increased water infiltration that drives microbial action, as well as increased net transport of labile carbon down slope (Jacinthe *et al.* 2004, Kumada *et al.* 2008). Jacinthe *et al.* (2004) cite rainfall intensity and energy as being critical factors driving net C loss.

The potential cumulative margins for error outlined above, need to be recognised and accounted for (conservatively) in carbon accounting, where restoration hopes to restore lost carbon pools. If active restoration can recoup the lost carbon (Mills & Cowling 2006) and hence qualify for the Clean Development Mechanism (CDM) under the Kyoto protocol, it is imperative that margins for error are minimised. The restoration of 1000 000 or more hectares of degraded subtropical thicket is likely to provide an opportunity for carbon offset investments for the foreseeable future. Investors, carbon brokers and projects may become disillusioned with subtropical thicket restoration in the short term, if the gains or total carbon stocks are unrealistic or over-estimated.

3.6. Conclusions

The scientific basis for restoring degraded semi-arid subtropical thickets via carbon trading has been significantly increased by this study, as well as by the work of Mills & Cowling (submitted). Total carbon differentials (above ground

and to 110 cm soil depth) for Baviaanskloof spekboom thickets approach 85 t C ha⁻¹. These degraded vegetation types have lost 65 % of the TCS to a depth of 25 cm. Carbon accrual rates following restoration (specifically for *P. afra*) for various subtropical thicket types and bioclimatic regions are required. There is considerable potential for carbon mitigation investments via subtropical thicket restoration, both in terms of landscape carbon and geographical range. The full potential can only be realised when accurate below ground carbon stocks are also taken into account (Mills & Cowling submitted). Variability in the reported carbon stocks for subtropical thickets indicate the need for site specific, not bioregional, carbon stock assessments. TBGC leakage for degraded landscapes, as a function of time since degradation took place, needs to be better understood. Should a robust correlation exist (site specific and related to environmental factors) between time since degradation and residual TBGC, this will greatly reduce the transaction costs for baseline carbon assessments.

Carbon credits for Reduced Emissions, Degradation and Deforestation (REDD) projects may soon become a reality (Laurance 2008), engendering the need for accurate carbon stocks for all 112 subtropical thicket types as classified by Vlok *et al.* (2002). The TCS for the intact subtropical thickets reported in this study will prove invaluable in building the case for REDD carbon credits accrued, both in the formal and voluntary carbon markets. The rate of carbon accruals are needed for a number of thicket types, especially the spekboom thickets and spekboomveld types.

The methods employed in this study are exceptionally time-consuming, and remote sensing support is needed for carbon stock assessments in subtropical thickets. The physiognomy of the vegetation precludes the employment of allometry as a non-destructive, indirect technique, bar in extreme cases of degradation. A robust and systematic classification system is required to provide categories for degradation, possibly using stem number and woody canopy cover as surrogates. Carbon content for key subtropical thicket species needs to be

established, and riparian forms of subtropical thicket (e.g., Baviaans thicket savanna) also require carbon stock baselines.

Given that by their nature, many of the semi-arid subtropical thickets and valley thickets are restricted to river valleys (Acocks 1975), one can expect TCS to vary significantly with catena position. A major failing in this study has been to establish these relationships, which needs to be addressed if carbon returns are to be predicted accurately. In the interim, restoration efforts and carbon offset projects should focus on lower slopes with deeper soils.

Chapter Four

Portulacaria afra truncheon survivorship in a restoration trial in the Baviaanskloof Nature Reserve



4.1. Introduction

Portulacaria afra (L.) Jacq. Commonly known as spekboom, is a keystone species (Lechmere-Oertel 2003, Lechmere-Oertel *et al.* 2005a) and a miracle plant (Vlok & Euston-Brown 2002) in the semi-arid subtropical thickets of the Eastern Cape, and western parts the Western Cape province of South Africa. Agriculturally, *P. afra* is an important species (Palmer & Pitman 1961, Aucamp 1979, von Maltitz 1991) within these vegetation types as a major supplier of browse (Oakes 1973, Aucamp 1979, Stuart-Hill 1989a), even during periods of drought (Stoltz 1991). Ironically, land management in the form of pastoralism, which has been the dominant land-use for a number of decades (Kerley *et al.* 1999a), has resulted in extensive degradation (Aucamp 1979, Lloyd *et al.* 2002, Esler *et al.* 2006) throughout much of the former range of these semi-arid vegetation types, resulting in a reduced abundance of this keystone species.

Resilience to severe degradation in semi-arid subtropical thickets is low (Stuart-Hill 1989a, Stuart-Hill 1999, Vlok *et al.* 2003). Subtropical thicket seedlings are generally rare, even in intact vegetation (von Maltitz 1991, Midgley & Cowling 1993). Although *P. afra* is known to produce copious seeds, dormancy is limited (Whiting 1991), and seedlings are rare (von Maltitz 1991). Where mass seedlings have been observed, no record exists of these developing into saplings and ultimately being incorporated into the canopy. Germination trials in laboratory settings have yielded very poor results (Oakes 1973). Despite *P. afra* having remarkable abilities to produce roots from cuttings (Palmer & Pitman 1961, Swart & Hobson 1994), and the growth of a skirt/apron at ground level (von Maltitz 1991), *P. afra* does not re-colonise highly degraded areas (Stuart-Hill 1991, Stuart-Hill 1999). Furthermore, it has been reported that birds rodents and animals transport seeds, leaves and small branches (Oakes 1973). In summary, the harsh micro-climate, precipitated through degradation, coupled with low seedling recruitment capabilities, may prevent the natural return of *P. afra*; consequently, manual planting is required. Mills & Cowling (2006) have shown that undertaking such manual replanting can result in carbon accrual rates of 3.4

t C ha⁻¹ yr⁻¹. Article 12 of the Kyoto Protocol makes allowance for the reforestation of degraded forest areas (degradation prior to 1990), whereby the amount of carbon sequestered can be verified and traded as carbon credits. The carbon economy (both mandatory and voluntary) could thus provide the funding for large-scale restoration of semi-arid subtropical thickets.

Internationally ecological rehabilitation or restoration is a complex and multi-faceted discipline, encompassing the restoration of cultural landscapes in Chile (Ovalle & Aronson 2007), expansive tree planting programmes for carbon sequestration gain in China (Chen in press), restoring lost vegetation on oceanic islands (Micol & Jouventin 1995), to restoring practically every form of terrestrial habitat on earth. The removal of alien invasive trees can be construed as a form of restoration (Woodworth 2006a). Restoration has traditionally been a by-product of mining industry (e.g., Parsons et al. 2007). Increasingly more case studies are focussing on holistic rangeland restoration (Aerts 2007), reforestation (e.g., Pal & Sharma 2001, del Campo *et al.* 2006) and combating soil erosion (Zhang *et al.* 2004).

Most restoration work that involves the use of cuttings and truncheons is restricted to water ways and riparian habitats (Bainbridge 2007) where the micro-climate and soil moisture are more forgiving for the development of roots. The planting of newly cut truncheons in a semi-arid environment with high levels of solar radiation, degraded soils and regular drought periods is novel.

In December 2003, the Department of Water Affairs and Forestry commissioned a pilot Subtropical Thicket Restoration Project in the Baviaanskloof Nature Reserve (BNR), Eastern Cape. The purpose of the initiative was to investigate the feasibility of employing emerging markets or payments for ecosystem services (PES), principally the carbon economy, to restore the degraded areas of subtropical thicket (Lechmere-Oertel 2003, Mills *et al.* 2003, 2007, Lechmere-Oertel *et al.* 2005a, Powell *et al.* 2004, 2006). Within this context, this chapter

reports on a multi-factorial experiment with the aim of elucidating the most effective methods (in terms of survivorship) to propagate *P. afra* cut truncheons.

4.2. Study site and methods

The study was undertaken within the BNR in the Eastern Cape, South Africa (24° 25' 51"E, 33° 39' 41"S). The site was specifically chosen for the level of degradation within an area of subtropical thicket, and the trial was located exclusively in old lands that were historically Gamtoos bontveld thicket type. The degraded soils within the study area are derived from shales, and have a high clay percentage and thus typically exhibit soil capping (Mills & Fey 2004a, c). The rainfall for the BNR is all-year round (Vlok 1989), with an approximate rainfall in the region of 450 mm pa. Daily temperature ranges are large, ranging from -3° C to 44° C, and frost is not uncommon in the winter months (ECPB 2007).

In 2005 a herbivore/primate exclosure plot (24° 37' 53.767" E, 33° 39' 57.861" S) of 7.6 hectares was erected to investigate the survivorship rates (and ultimately growth rates) for cut *P. afra* truncheons in field conditions within the BNR. The exclosure fence was erected according to standard game-farming specifications: the height of the fence measured 2.4 meters, with the upper 1.2 meters having five strands of galvanised high tensile steel wire and three strands of electrified wire to exclude primates. Jackal proof, wire mesh fencing was attached to the first meter above the soil, with approximately 200 mm covered with large rocks to prevent burrowing animals from entering. Prior to final enclosure, all herbivorous animals (including tortoises) were removed from the enclosure, in order to prevent animals disturbing the planted truncheons.

The area was selected for uniformity, based on degradation (phytomass lost), slope and aspect. An adjacent area of equivalent size was selected, but not fenced, to investigate the impact of herbivory on the establishment

success/survivorship and, ultimately, the growth rate of planted *P. afra* truncheons. The same experimental design as described below was applied.

Within the enclosure and outside, a grid matrix of 25 m² was created in ArcviewTM 3.2 software (ESRI 2004) and permanently marked out using steel pegs. The corners of all marked plots were captured spatially into GIS using a Trimble PROXRS sub-meter GPS with a spatial accuracy of sub-meter in the x and y coordinates. Each 25 m² plot was surrounded by a buffer of five metres where no planting occurred. Where large woody species (e.g., *Acacia karroo*) occurred within the plots or adjacent to the plots, these plots were not planted. Forty 25 m² plots were randomly selected using ARCVIEWTM 3.2 GIS scripts (ESRI 2004) and serve as controls. Plant species diversity and percentage cover data were collected for the controls. Thirty two treatments were devised using a multi-factorial design (Table 4.1) and each treatment received ten replicates. All treatments and all replicates were randomly allocated to the 400 25 m² plot matrix using ARCVIEWTM 3.2 GIS (ESRI 2004) scripts. Contractor teams from the Working-for-Woodland's Subtropical Thicket Restoration Programme (STRP) were tasked to plant out the treatments by manually digging holes (to a depth of ~15cm) and inserting the cut truncheons into the soil, following two days of drying in shade. No rooting hormone was applied. The cut truncheons were sourced from intact subtropical thicket nearby.

Prior to planting in July 2005, each truncheon's wet mass (nearest 0.1 g), height and stem diameter (nearest 0.1 mm) were recorded using digital scales and digital callipers. Each truncheon was labelled using an aluminium tag. Following planting, each truncheon's basal stem diameter at the soil surface was measured using digital callipers (nearest 0.1 mm). Mortality counts were conducted in June 2006 and August 2008. Key factors investigated are listed below in Table 4.1.

Table 4.1: Factors investigated for *P. afra* survivorship

Factor	Class 1	Class 2
Stem diameter (mm)	20	40
Micro-damming	Yes	No
Clumping per 25m ² plot	3 clumps	6 clumps
Density (truncheons/m ²)	0.72	1.44
Planting posture	Upright	Flat

4.3. Data analysis

A total of five factors were investigated, each with two categories. Stem diameter was deemed crucial, due to the trade-off between cost of planting and the possible rate of carbon accrual; the assumption made is that larger stems would require more energy to locate, transport and plant; thus equating to higher restoration costs. However, the benefit of planting larger truncheons may include a better survival chance, as well as faster growth rate and less time to a closed canopy situation. Micro-damming involved the creation of a shallow depression (~5 cm) at the base of the truncheon for the upright treatments. The flat truncheons were placed with the micro-dam surrounding the entire truncheon including the leaf material. All micro-dams were constructed at the time of planting and not maintained thereafter. The assumption with micro-dams was that more water would be focussed towards the rooting zone during precipitation events. Clumping and density were hoped to test whether any benefits (survivorship) could be derived in planting truncheons in groups and at different densities. Lastly, planting posture, i.e., vertical or flat, was tested in the hope that truncheons laid flat on the soil surface would take root and hence reduce costs considerably, relative to the upright ones for which holes must be dug.

The experimental design for the investigation of the key factors outlined in Table 4.1 followed a multi-factorial design, whereby the various interactions could be analysed using standard analysis of variance (ANOVA) statistical techniques. Individual treatments are depicted in Table 4.2.

Table 4.2: Treatments applied to *P. afra* truncheons in 25 m² plots.

Treatment no.	Stem diameter (mm)	Planting posture	Clumping no. of clumps	Density/m ²	Micro-Dam
1	20	upright	6	1.44	Yes
2	20	upright	6	1.44	No
3	20	upright	6	0.72	Yes
4	20	upright	6	0.72	No
5	20	upright	3	1.44	Yes
6	20	upright	3	1.44	No
7	20	upright	3	0.72	Yes
8	20	upright	3	0.72	No
9	20	Flat	6	1.44	Yes
10	20	Flat	6	1.44	No
11	20	Flat	6	0.72	Yes
12	20	Flat	6	0.72	No
13	20	Flat	3	1.44	Yes
14	20	Flat	3	1.44	No
15	20	Flat	3	0.72	Yes
16	20	Flat	3	0.72	No
17	40	upright	6	1.44	Yes
18	40	upright	6	1.44	No
19	40	upright	6	0.72	Yes
20	40	upright	6	0.72	No
21	40	upright	3	1.44	Yes
22	40	upright	3	1.44	No
23	40	upright	3	0.72	Yes
24	40	upright	3	0.72	No
25	40	Flat	6	1.44	Yes
26	40	Flat	6	1.44	No
27	40	Flat	6	0.72	Yes
28	40	Flat	6	0.72	No
29	40	Flat	3	1.44	Yes
30	40	Flat	3	1.44	No
31	40	Flat	3	0.72	Yes
32	40	Flat	3	0.72	No

The proportion data of surviving plants for each year was transformed using ArcSin(Sqrt) transformation, and the transformed variable used for further analysis. Results for seven plots were missing in the 2006 data, but no more than one plot in each of the 32 treatments was missing, so I employed mean value replacement to ensure a balanced design. The 2008 data are complete. R version 2.8.0 (R Core Development Team 2008) and Statistica version 8.0 (StatSoft Inc. 2008) were used to analyse the data, and significant differences were usually reported for $p < 0.05$.

4.4. Results

As a result of incessant interference from both kudu (*Tragelaphus strepsiceros*) and baboons (*Papio ursinus*) following the planting of unrooted truncheons in the open plot, all 320 replicates were abandoned. The animals uprooted the truncheons before the soil could settle and secure them in place, effectively nullifying the experimental design. All subsequent results pertain only to the enclosure plot where herbivory was prevented.

4.4.1. 2006 survivorship for the main effects

Planting posture had the greatest effect on survivorship 2006 (Table 4.3), with survivorship of flat truncheons being 33.6 ± 1.56 % compared to upright truncheons being 53.2 ± 1.65 % ($F=97.33$, $p < 0.001$). Truncheon stem diameter showed no significant influence on survivorship (~ 43 %). Clumping in 2006 played no significant role in affecting survivorship ($F=0.79$, $p=0.375$). The placement of a micro-dam at the time of planting significantly ($F=13.27$, $p=0.0025$) improved the chances of the survivorship by 8 %. Planting density showed a significant effect ($F=6.40$, $p=0.012$), although it was thought that this effect would only be noticed after subsequent years, especially with regard to episodic black frost events. Generally, the low density planting had a marginally better survivorship than the high density plants.

Table 4.3: Mean 2006 survivorship results (percentage plus standard error (SE)) for *P. afra* truncheon treatments for individual factors.

Factor	Class 1	% and SE	n	Class 2	% and SE	n	p value	F
Truncheon diameter	20 mm	43.4±1.56	160	40 mm	43.4±1.9	160	0.823	0.05
Planting posture	Upright	53.1±1.7	160	Flat	33.6±1.6	160	<0.001	97.33
Clumping	3	42.4±1.8	160	6	44.4±1.8	160	0.375	0.79
Density	0.72 m ⁻²	41.0±2.0	160	1.44 m ⁻²	45.8±1.7	160	0.012	6.40
Micro-damming	Yes	47.4±2.0	160	No	39.4±1.5	160	<0.001	13.27

Mean percentage survivorship for all treatments in 2006 was 43.2±2.8 %. When considering the survivorship in 2006 with regards to individual treatments, some results contradict the overall findings outlined above (Table 4.4). Six of the ten treatments that showed mean percentage survivorship of >50 % were all in the 40 mm stem diameter class. Clear patterns in Treatments 1 to 8 show the benefit of micro-damming (which becomes moderated in subsequent treatments).

4.4.2. 2008 survivorship for the main effects

Survivorship with regard to individual factors in 2008 followed similar patterns to that of 2006 (Tables 4.5 and 4.6). Truncheon posture appears to be the single most important factor affecting *P. afra* survivorship three years after planting (F=139.49, p=<0.001). Treatments with flat truncheons exhibited lower (24.9±1.3 % vs. 46.7±1.6 %) survivorship than upright plantings. Micro-damming again was the second most important influence on survivorship, (F=11.25, p=<0.001), improving survivorship by 7 %. Planting density in 2008 had a significant influence (F= 9.07, p=0.003) on survivorship, while clumping and truncheon stem diameter had no effect.

Table 4.4: 2006 *P. afra* mean percentage survivorship results reflected in terms of treatments applied.

Treatment number	Treatment Code	Mean % Survivorship	SE	n
23	4-U-3C-L-Y	72.8	6.1	10
17	4-U-6C-H-Y	70.0	3.1	10
3	2-U-6C-L-Y	66.1	4.9	10
1	2-U-6C-H-Y	66.0	4.5	10
22	4-U-3C-H-N	64.8	4.7	10
19	4-U-6C-L-Y	61.7	8.7	10
5	2-U-3C-H-Y	60.8	5.8	10
18	4-U-6C-H-N	57.8	4.9	10
21	4-U-3C-H-Y	57.5	4.2	10
7	2-U-3C-L-Y	51.7	6.4	10
14	2-F-3C-H-N	49.2	4.1	10
16	2-F-3C-L-N	45.0	4.5	10
4	2-U-6C-L-N	41.7	5.6	10
25	4-F-6C-H-Y	41.4	8.1	10
8	2-U-3C-L-N	41.1	4.8	10
9	2-F-6C-H-Y	40.8	6.5	10
20	4-U-6C-L-N	40.6	5.7	10
30	4-F-3C-H-N	40.6	3.2	10
10	2-F-6C-H-N	39.2	5.8	10
24	4-U-3C-L-N	37.0	3.9	10
11	2-F-6C-L-Y	35.0	6.1	10
13	2-F-3C-H-Y	32.2	6.4	10
15	2-F-3C-L-Y	32.2	6.0	10
27	4-F-6C-L-Y	32.2	7.5	10
12	2-F-6C-L-N	32.1	4.3	10
6	2-U-3C-H-N	31.4	3.4	10
2	2-U-6C-H-N	29.9	3.2	10
28	4-F-6C-L-N	28.9	5.4	10
26	4-F-6C-H-N	26.9	4.7	10
32	4-F-3C-L-N	24.4	10.1	10
29	4-F-3C-H-Y	24.1	3.8	10
31	4-F-3C-L-Y	13.3	2.2	10

2 = 20 mm stem diameter

4 = 40 mm stem diameter

U = upright

F = flat

6C = 6

3C = 3

L = low density

(0.72 truncheons per m⁻²)

H = high density

(1.44 truncheons per m⁻²)

Y = micro-dam

N = No micro-dam

The mean percentage survivorship (2008) for all 32 treatments was 35.8 ± 2.7 %. Mean survivorship ranged from 11.7 ± 5.0 % (Treatment 32) to 65 ± 3.1 % (Treatment 17). Seven treatments had lower ranges of survivorship of 0 %, while only one treatment had a lower range of more than 49 % (Treatment 17). The greatest range in percentage survivorship occurred in treatment two, with a differential between lowest percentage and highest percentage survivorship of 83 % within the 10 replicates.

Table 4.5: Mean 2008 survivorship results (percentage plus standard error (SE)) for *P. afra* truncheon treatments for individual factors.

Factor	Class 1	% and SE	n	Class 2	% and SE	n	p	F
Truncheon diameter	20 mm	36.9 ± 1.5	160	40 mm	34.6 ± 1.8	160	0.121	2.41
Planting posture	Upright	46.7 ± 1.6	160	Flat	24.9 ± 1.3	160	<0.001	139.49
Clumping	3	34.3 ± 1.6	160	6	37.3 ± 1.8	160	1.21	2.42
Density	0.72 m^{-2}	33.3 ± 1.8	160	1.44 m^{-2}	38.3 ± 1.6	160	0.003	9.07
Micro-damming	Yes	39.3 ± 1.9	160	No	32.3 ± 1.5	160	<0.001	11.25

Table 4.6: Mean % 2008 survivorship for *P. afra* with 32 treatments.

Treatment number	Treatment Code	Mean % Survivorship	SE	n	
17	4-U-6C-H-Y	65.0	3.1	10	2 = 20 mm stem diameter 4 = 40 mm stem diameter U = upright F = flat 6C = 6 3C = 3 L = low density (0.72 truncheons per m^{-2}) H = high density (1.44 truncheons per m^{-2}) Y = micro-dam N = No micro-dam
1	2-U-6C-H-Y	59.2	4.6	10	
23	4-U-3C-L-Y	58.9	7.3	10	
19	4-U-6C-L-Y	57.2	8.4	10	
3	2-U-6C-L-Y	56.1	5.7	10	
18	4-U-6C-H-Y	56.1	6.6	10	
22	4-U-3C-H-N	51.1	3.1	10	
5	2-U-3C-H-Y	50.0	5.8	10	
7	2-U-3C-L-Y	47.5	6.8	10	
21	4-U-3C-H-Y	47.2	4.4	10	
14	2-F-3C-H-N	43.9	4.6	10	
16	2-F-3C-L-N	38.9	5.0	10	

Treatment number	Treatment Code	Mean % Survivorship	SE	n
20	4-U-6C-L-N	38.3	7.1	10
8	2-U-3C-L-N	36.7	6.4	10
2	2-U-6C-H-N	35.0	8.1	10
9	2-F-6C-H-Y	33.6	4.9	10
4	2-U-6C-L-N	33.1	4.8	10
24	4-U-3C-L-N	29.4	2.9	10
25	4-F-6C-H-Y	28.6	5.1	10
10	2-F-6C-H-N	28.6	5.7	10
30	4-F-3C-H-N	28.1	4.1	10
11	2-F-6C-L-Y	27.8	5.1	10
6	2-U-3C-H-N	26.4	4.0	10
15	2-F-3C-L-Y	25.0	6.6	10
12	2-F-6C-L-N	24.4	3.9	10
13	2-F-6C-H-Y	24.2	6.2	10
26	4-F-6C-H-N	19.2	3.5	10
27	4-F-6C-L-Y	18.9	5.7	10
29	4-F-3C-H-Y	17.5	4.2	10
28	4-F-6C-L-N	16.1	3.3	10
31	4-F-3C-L-Y	12.2	3.0	10
32	4-F-3C-L-N	11.7	5.0	10

4.4.3. Survivorship trends: 2006 vs. 2008

For the total cohort of 8 622 truncheons weighed and measured in June 2005, 36.7 % were surviving in August 2008, compared to 44.1 % surviving in June 2006. The steady drop in survivorship has been uniformly distributed across the variables tested (Figure 4.1). As a whole, the 20 mm stem diameter class showed a 6.3 % drop in survivorship, whereas 40 mm stem diameter class showed the greatest drop (of all factors) at 8.6 %. All other factors showed similar drops in survivorship of between 6 and 8 %.

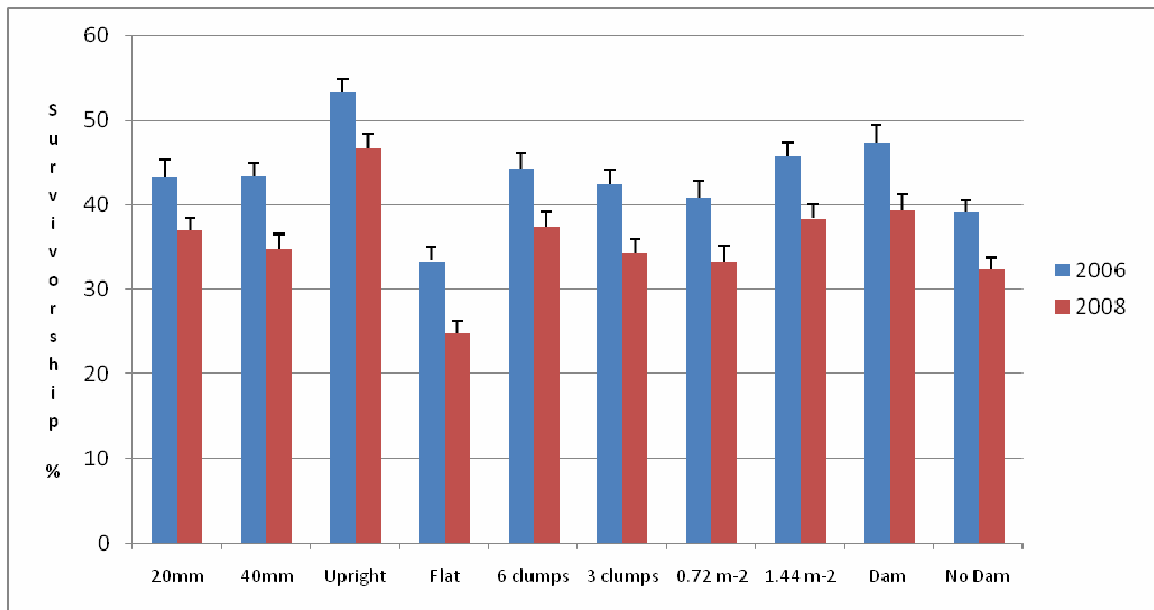


Figure 4.1: Mean *P. afra* survivorship in 2006 and 2008 for factors tested.

The drop in survivorship from 2006 to 2008 (Figure 4.2) for individual treatments followed similar patterns to those observed for individual factors – the anomaly being Treatment 2 (discussed below). On average, all factors dropped in mean percentage survivorship by 7.4 ± 0.8 %. Treatments 21, 22, 23, 25, 28, 30 and 32 showed greatest drops in mean percentage survivorship, dropping by 12–14 %.

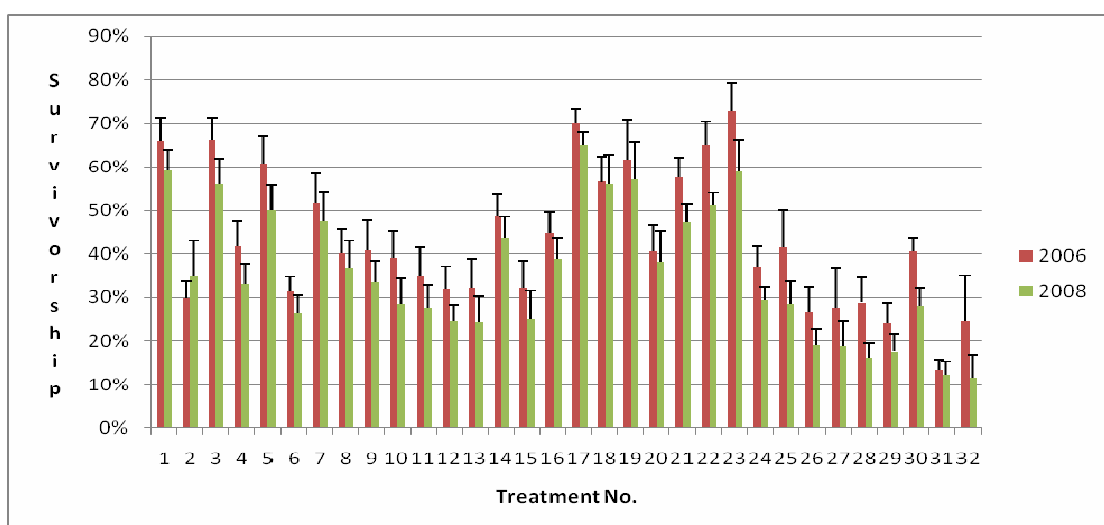


Figure 4.2: Mean *P. afra* survivorship in 2006 and 2008 for treatments tested.

4.4.4. 2006 multifactorial analysis

Assumptions are satisfied with the residuals passing the Kolmogorov–Smirnov test of normality at the five percent level of significance ($D=0.044$, $p=0.1382$) and the Anderson–Darling test of normality at the one percent level of significance ($A=0.8387$, $p=0.0304$). Homogeneity of variances is satisfied according to Levene’s test with Brown–Forsythe modification ($F=1.142$, $p=0.283$). The significant main effects on survivorship are posture, density and micro-dam, with posture and micro-dam having the greatest significance.

From Figures 4.3 and 4.4 it can be seen that survivorship is better for those planted upright in all but one case where there is a significant interaction between four of the factors (stem diameter, micro-dam, posture and density). Small truncheons planted upright with no micro-dam in the high density configuration had a lower survivorship compared to small plants planted flat in the same configuration, however this was not observed in the no micro-dam treatments, hence the significant interaction ($F=6.98$, $p=0.009$).

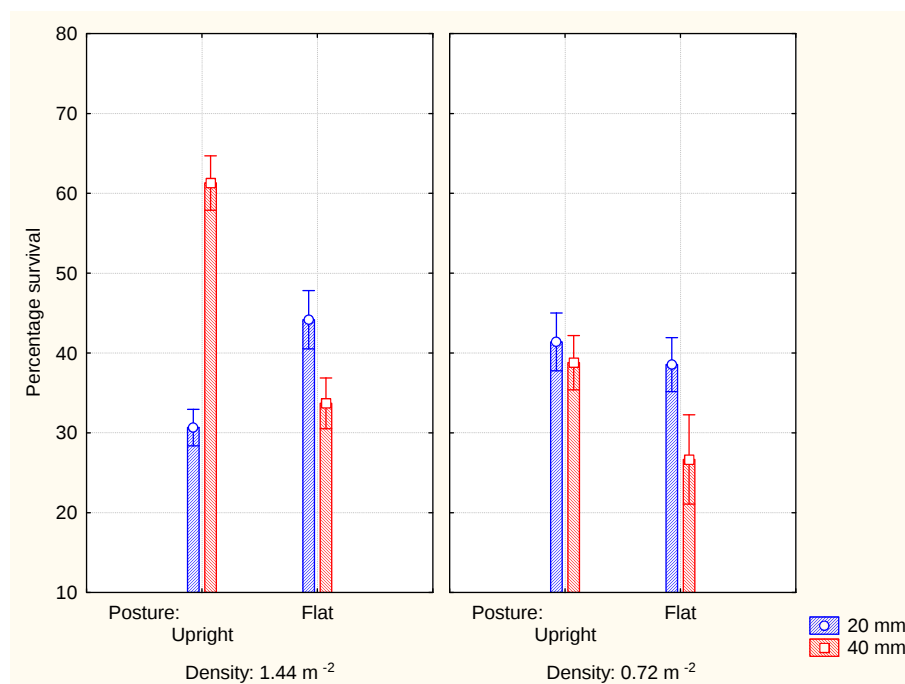


Figure 4.3: The interactive influence of micro-dam, stem diameter, density and posture on *P. afra* survivorship in 2006 (micro-dam absent) - see Table 4.7 below).

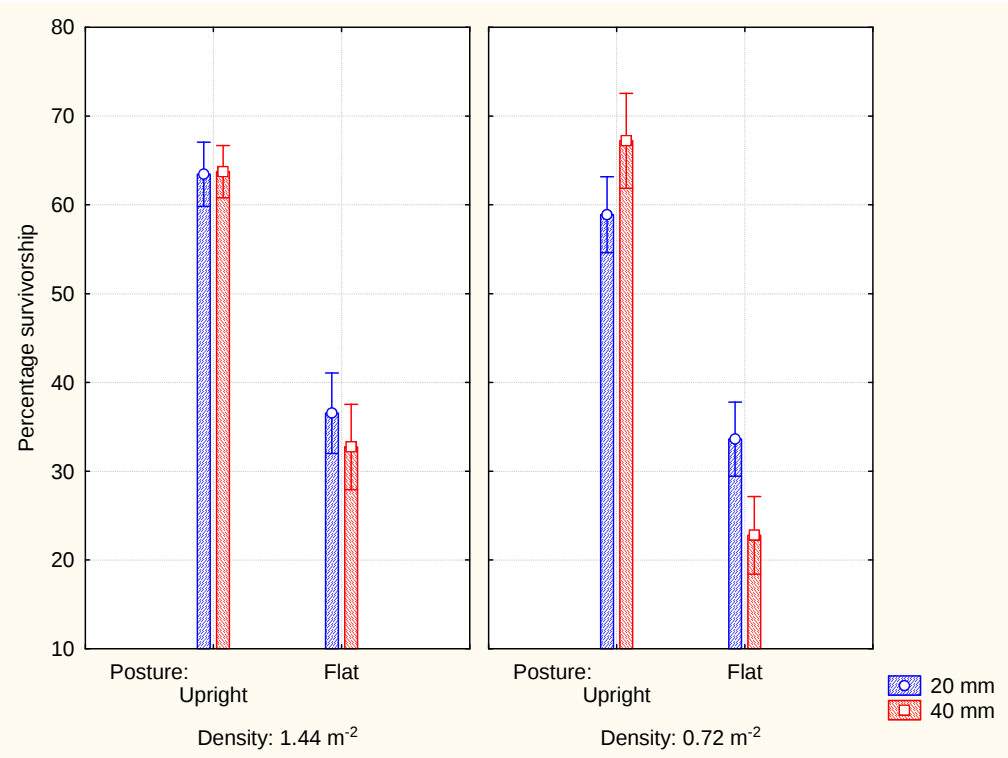


Figure 4.4: The interactive influence of micro-dam, stem diameter, density and posture on *P. afra* survivorship in 2006 (micro-dam present) - see Table 4.7 below).

Table 4.7: Factorial ANOVA results for *P. afra* survivorship (2006 and 2008).

Factors	2006					2008				
	SS	df	MS	F	p	SS	df	MS	F	p
Intercept	160.48	1	160.48	3796.68	<0.001	123.25	1	123.25	3146.81	<0.001
Stem size	0.00	1	0.00	0.05	0.823	0.09	1	0.09	2.41	0.121
Posture	4.11	1	4.11	97.33	<0.001	5.46	1	5.46	139.49	<0.001
Clumping	0.03	1	0.03	0.79	0.375	0.09	1	0.09	2.42	0.121
Density	0.27	1	0.27	6.40	0.012	0.36	1	0.36	9.07	0.003
Micro-dam	0.56	1	0.56	13.27	<0.001	0.44	1	0.44	11.25	0.001
Stem size*Posture	0.91	1	0.91	21.58	<0.001	1.13	1	1.13	28.74	<0.001
Stem size*Clumping	0.02	1	0.02	0.51	0.477	0.05	1	0.05	1.26	0.263
Posture*Clumping	0.00	1	0.00	0.02	0.885	0.10	1	0.10	2.54	0.112
Stem size*Density	0.17	1	0.17	4.06	0.045	0.17	1	0.17	4.35	0.038
Posture*Density	0.06	1	0.06	1.48	0.224	0.07	1	0.07	1.67	0.198
Clumping*Density	0.00	1	0.00	0.09	0.770	0.03	1	0.03	0.68	0.412
Stem size*Micro-dam	0.01	1	0.01	0.31	0.578	0.00	1	0.00	0.03	0.854
Posture*Micro-dam	1.44	1	1.44	34.16	<0.001	0.95	1	0.95	24.29	<0.001
Clumping*Micro-dam	0.35	1	0.35	8.33	0.004	0.21	1	0.21	5.32	0.022
Density*Micro-dam	0.03	1	0.03	0.62	0.433	0.06	1	0.06	1.50	0.222
Stem size*Posture*Clumping	0.20	1	0.20	4.66	0.032	0.05	1	0.05	1.15	0.284
Stem size*Posture*Density	0.02	1	0.02	0.54	0.463	0.00	1	0.00	0.03	0.867

Factors	2006					2008				
	SS	df	MS	F	p	SS	df	MS	F	p
Stem size*Clumping*Density	0.00	1	0.00	0.08	0.773	0.01	1	0.01	0.30	0.583
Posture*Clumping*Density	0.01	1	0.01	0.32	0.573	0.06	1	0.06	1.57	0.212
Stem size*Posture*Micro-dam	0.10	1	0.10	2.41	0.121	0.13	1	0.13	3.36	0.068
Stem size*Clumping*Micro-dam	0.00	1	0.00	0.10	0.754	0.05	1	0.05	1.37	0.243
Posture*Clumping*Micro-dam	0.10	1	0.10	2.47	0.117	0.13	1	0.13	3.22	0.074
Stem size*Density*Micro-dam	0.22	1	0.22	5.28	0.022	0.16	1	0.16	4.20	0.041
Posture*Density*Micro-dam	0.02	1	0.02	0.42	0.518	0.02	1	0.02	0.49	0.485
Clumping*Density*Micro-dam	0.12	1	0.12	2.80	0.095	0.07	1	0.07	1.88	0.172
Stem size*Posture*Clumping*Density	0.15	1	0.15	3.58	0.059	0.02	1	0.02	0.58	0.448
Stem size*Posture*Clumping*Micro-dam	0.02	1	0.02	0.56	0.454	0.01	1	0.01	0.30	0.582
Stem size*Posture*Density*Micro-dam	0.30	1	0.30	6.98	0.009	0.11	1	0.11	2.72	0.100
Stem size*Clumping*Density*Micro-dam	0.17	1	0.17	4.04	0.045	0.14	1	0.14	3.46	0.064
Posture*Clumping*Density*Micro-dam	0.00	1	0.00	0.00	0.965	0.02	1	0.02	0.61	0.434
Stem size*Posture*Clumping*Density*Micro-dam	0.02	1	0.02	0.43	0.512	0.01	1	0.01	0.25	0.616
Error	12.17	288	0.04			11.28	288	0.04		

Posture interacted highly with the presence/absence of a micro-dam at the time of planting ($F = 34.161$, $p < 0.0001$). Micro-dams enhanced survivorship when planting was upright but induced higher mortality when the plants were planted flat, as shown in the Figure 4.5 below.

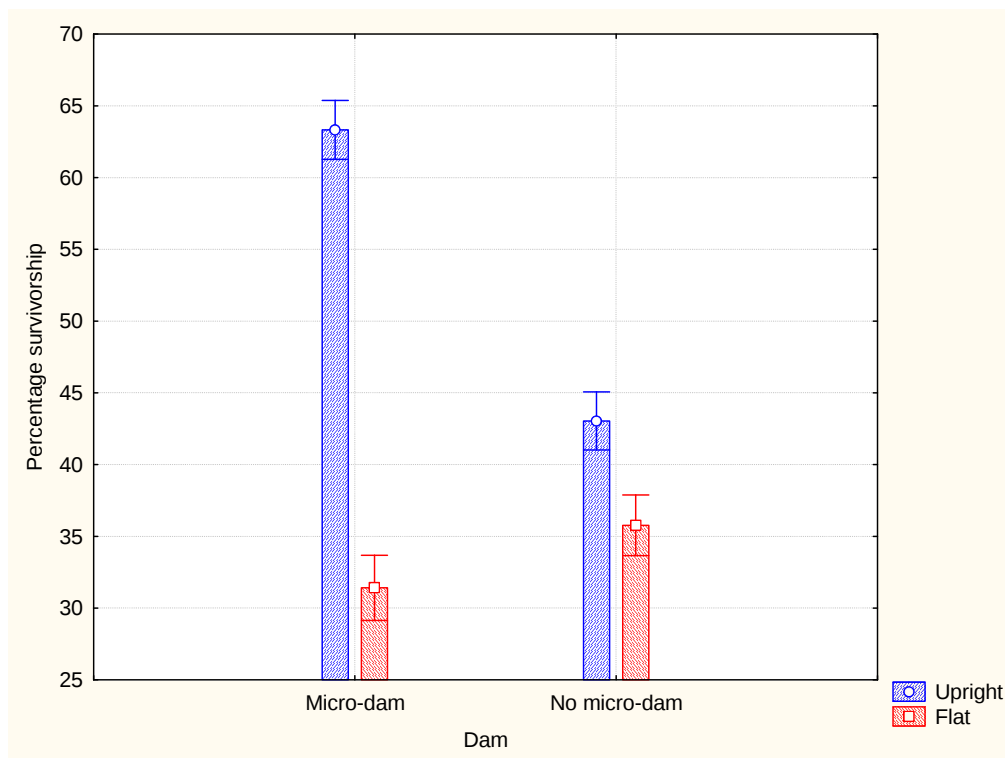


Figure 4.5: The influence of micro-dam and posture on *P. afra* survivorship in 2006.

Although truncheon stem diameter showed no significant main effect on survivorship ($F=0.050$, $p=0.823$), it does interact significantly with posture ($F=21.583$, $p < 0.0001$). Figure 4.6 shows that survivorship dropped more for the large truncheons planted flat, compared to the smaller truncheons planted flat.

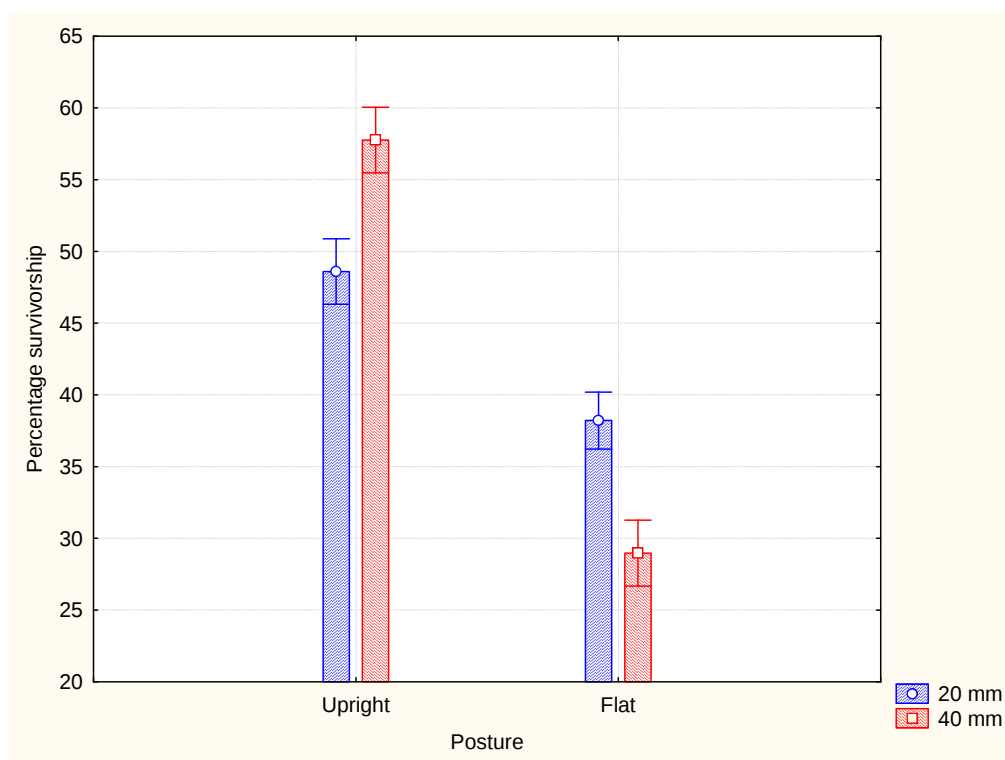


Figure 4.6: The influence of stem diameter and posture on *P. afra* survivorship in 2006.

The presence of the micro-dam also interacted significantly with clumping ($F = 8.33$, $p = 0.0042$) and is shown in Figure 4.7. Mean percentage survivorship for treatments with six clumps and a micro-dam showed survivorship in the region of 52 %, whereas mean percentage survivorship with three clumps and a micro-dam was approximately 42 %. In the reciprocal (i.e., no micro-dam), the mean percentage survivorship for three clumps dropped marginally, whereas the mean percentage survivorship for six clumps dropped massively.

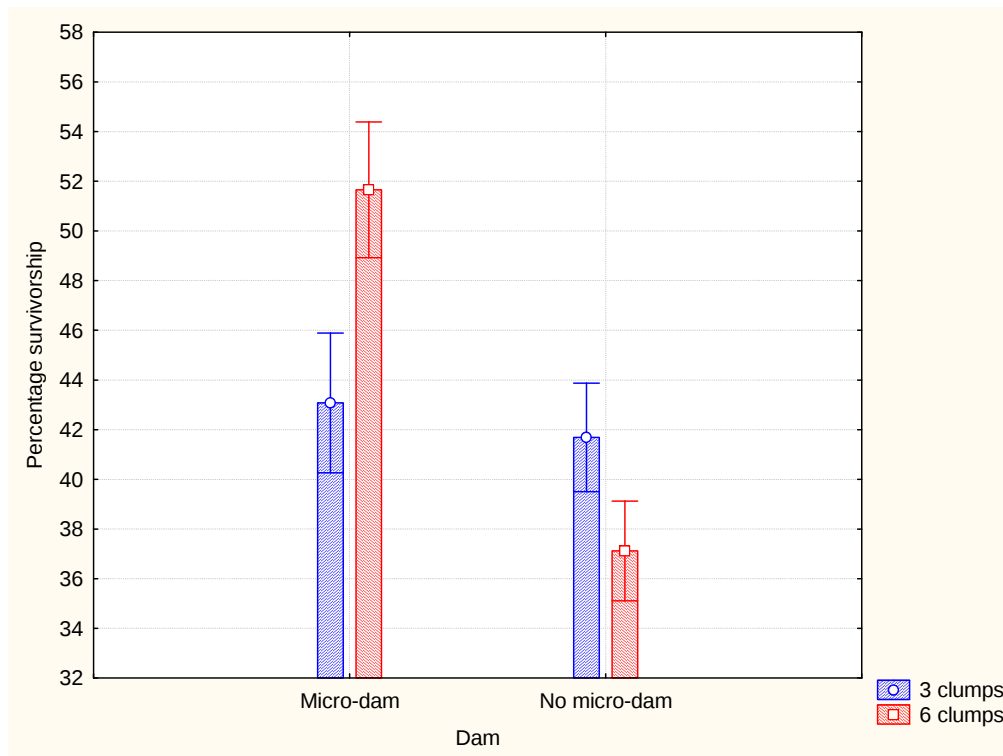


Figure 4.7: The influence of micro-dam and clumping on *P. afra* survivorship in 2006.

4.4.5. 2008 multifactorial analysis

Assumptions were again satisfied (Normality: $D=0.0386$, $p=0.2922$ and $A=0.4249$, $p=0.315$ and homogeneity of variances $F=9176$, $p=0.5968$). Interactive effects for 2008 were very similar to those of 2006 (Table 4.7). Posture was dominant through a number of permutations, with micro-dam and density playing significant but lesser roles. Four interactions that provided significant results in 2006, became not significant in 2008 (i.e., 1) stem size, posture and clumping; 2) stem size, posture, density and clumping; 3) stem size, posture, density and micro-dam; and 4) stem size, clumping, density and micro-dam).

The interaction between stem size, density, micro-dam and posture, is shown in Figures 4.8 and 4.9. With a micro-dam, the smaller truncheons show greater survivorship in the flat position relative to the larger plants at both densities, the upright truncheons experiencing greater survivorship overall. As in 2006, posture interacted significantly with micro-dam ($F=24.289$, $p<0.0001$). Importantly, with a

micro-dam present, mean percentage survivorship for both stem diameters in the upright plantings are similar and favourable (50 %+), while flat plantings are much less favourable (<30 %).

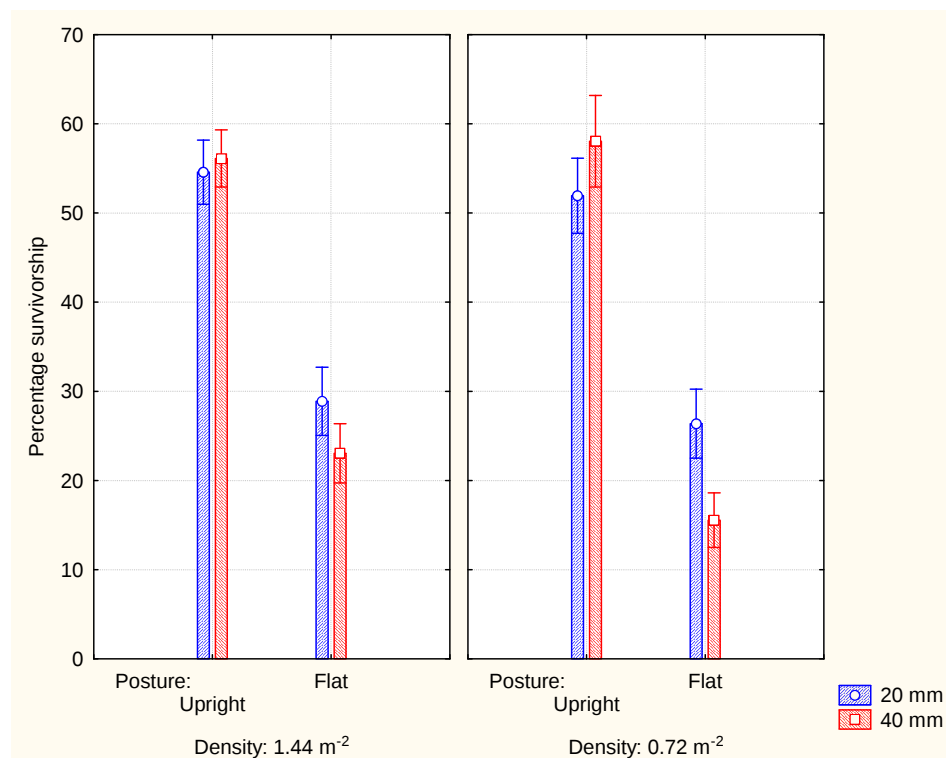


Figure 4.8: The inter-active influence of micro-dam, stem diameter, posture and density on *P. afra* survivorship in 2008 (micro-dam present) (see Table 4.7 above).

In the absence of the micro-dam, larger stem diameters showed much higher survivorship (20 %+) when planted upright and at higher densities. The same stem diameters, planted upright at low densities, provided similar survivorship results (~35 %). When planted flat, with no micro-dam, the smaller stem diameter truncheons outperformed the larger ones planted, irrespective of density – as was the case in 2006.

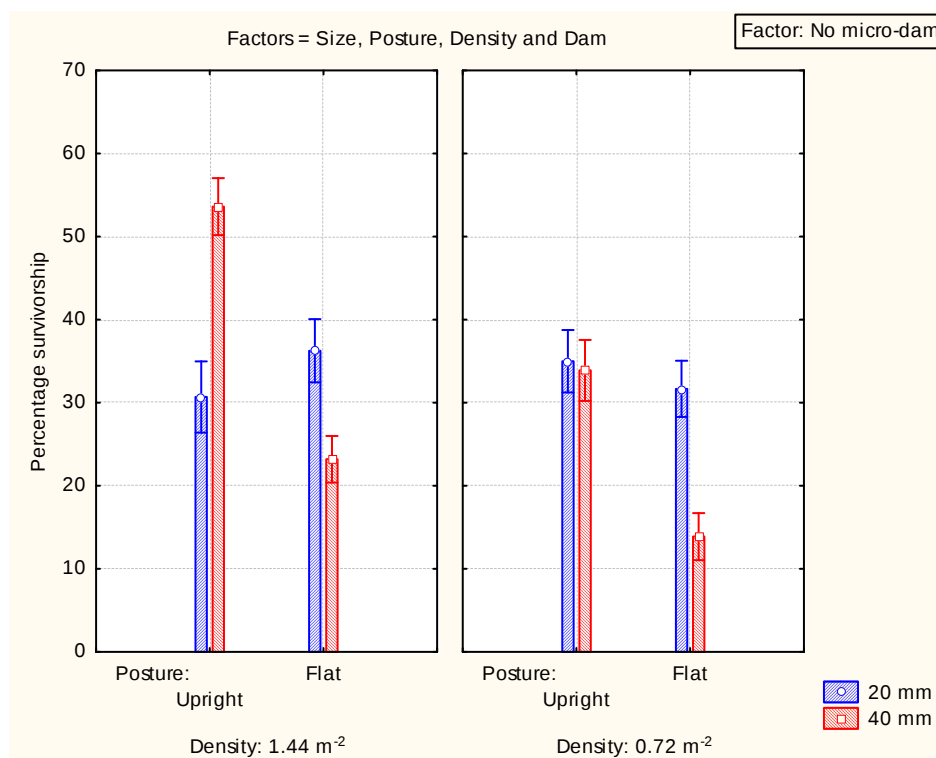


Figure 4.9: The inter-active influence of micro-dam, stem diameter, posture and density on *P. afra* survivorship in 2008 (micro-dam absent).

Planting density in 2008 had (as expected) a greater influence ($F=9.07$, $p=0.0028$) on mortality than seen in 2006, while clumping and truncheon stem diameter appeared to have no effect (in isolation) on establishment and early survivorship. Clumping did, once again, interact with micro-damming, however the significance of this was not as high as in 2006 ($F=5.32$, $p=0.0218$). Stem diameter in 2008 did interact more strongly with density ($F=4.35$, $p=0.038$) than in 2006 (Figure 4.10).

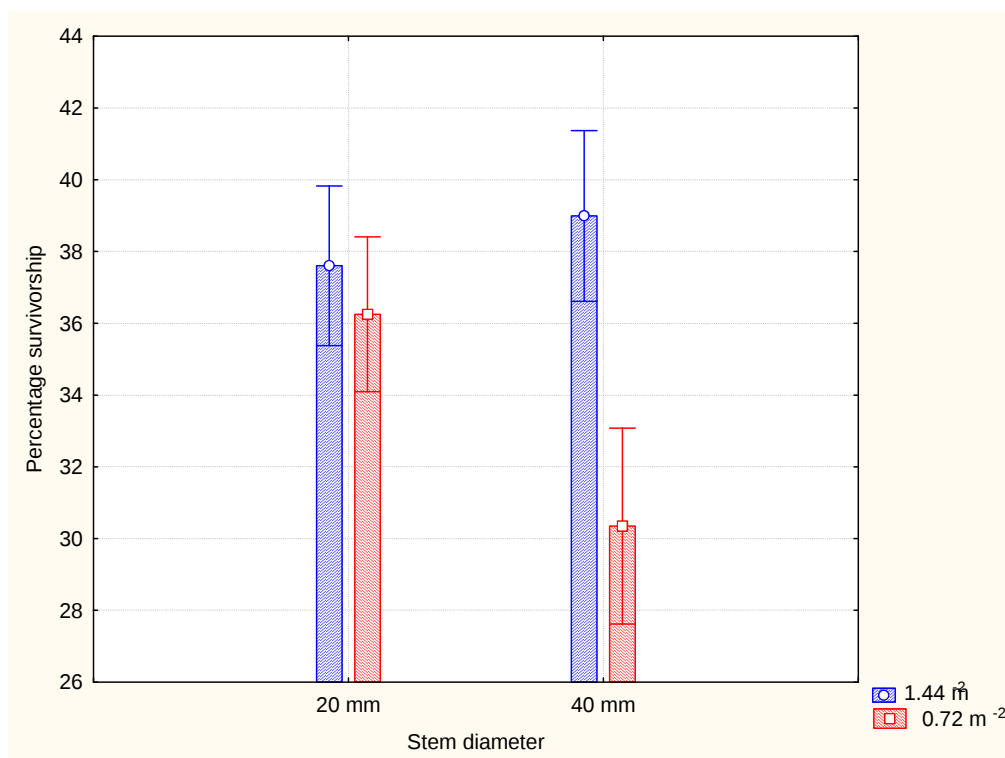


Figure 4.10: The influence of stem diameter and density on *P. afra* survivorship in 2008.

When conducting ANOVA with “year” as a factor, the same main effects of posture, density and damming were observed, with posture and damming having the most significant effect. Higher order interactions were the same as observed for the two years individually. Year definitely had a significant effect, with survivorship (Table 4.8) dropping from 2006 to 2008 from 43.4 ± 1.26 % to 35.8 ± 1.20 % on average ($F=109.71$, $p<0.001$).

Table 4.8: Repeated Measure ANOVA for *P. afra* survivorship 2006 and 2008.

Factor	SS	df	MS	F	p
Intercept	282.50	1	282.50	4021.22	0.000
Stem size	0.06	1	0.06	0.89	0.346
Posture	9.53	1	9.53	135.64	0.000
Clumping	0.12	1	0.12	1.71	0.192
Density	0.62	1	0.62	8.86	0.003
Micro-dam	1.00	1	1.00	14.20	0.000
Stem size*Posture	2.03	1	2.03	28.93	0.000
Stem size*Clumping	0.07	1	0.07	0.97	0.326
Posture*Clumping	0.04	1	0.04	0.58	0.447
Stem size*Density	0.34	1	0.34	4.87	0.028
Posture*Density	0.13	1	0.13	1.82	0.178
Clumping*Density	0.01	1	0.01	0.08	0.785
Stem size*Micro-dam	0.00	1	0.00	0.04	0.835
Posture*Micro-dam	2.37	1	2.37	33.73	0.000
Clumping*Micro-dam	0.55	1	0.55	7.84	0.005
Density*Micro-dam	0.08	1	0.08	1.16	0.282
Stem size*Posture*Clumping	0.22	1	0.22	3.06	0.081
Stem size*Posture*Density	0.02	1	0.02	0.24	0.624
Stem size*Clumping*Density	0.01	1	0.01	0.20	0.654
Posture*Clumping*Density	0.07	1	0.07	0.94	0.333
Stem size*Posture*Micro-dam	0.23	1	0.23	3.31	0.070
Stem size*Clumping*Micro-dam	0.04	1	0.04	0.63	0.430
Posture*Clumping*Micro-dam	0.23	1	0.23	3.28	0.071
Stem size*Density*Micro-dam	0.39	1	0.39	5.49	0.020
Posture*Density*Micro-dam	0.04	1	0.04	0.52	0.470
Clumping*Density*Micro-dam	0.19	1	0.19	2.69	0.102
Stem size*Posture*Clumping*Micro-dam	0.00	1	0.00	0.02	0.904
Stem size*Posture*Density*Micro-dam	0.38	1	0.38	5.38	0.021
Stem size*Clumping*Density*Micro-dam	0.31	1	0.31	4.35	0.038
Stem size*Posture*Clumping**Micro-dam	0.03	1	0.03	0.39	0.532
Error	20.23	288	0.07		
YEAR	1.23	1	1.23	109.71	0.000
YEAR*Stem size	0.03	1	0.03	3.05	0.082
YEAR*Posture	0.05	1	0.05	4.27	0.040
YEAR*Clumping	0.01	1	0.01	0.70	0.402

Factor	SS	df	MS	F	p
YEAR*Density	0.00	1	0.00	0.26	0.613
YEAR*Micro-dam	0.00	1	0.00	0.33	0.569
YEAR*Stem size*Posture	0.01	1	0.01	0.50	0.480
YEAR*Stem size*Clumping	0.00	1	0.00	0.26	0.613
YEAR*Posture*Clumping	0.06	1	0.06	5.32	0.022
YEAR*Stem size*Density	0.00	1	0.00	0.00	0.993
YEAR*Posture*Density	0.00	1	0.00	0.00	0.974
YEAR*Clumping*Density	0.02	1	0.02	2.22	0.137
YEAR*Stem size*Micro-dam	0.01	1	0.01	1.02	0.313
Factor	SS	df	MS	F	p
YEAR*Posture*Micro-dam	0.03	1	0.03	2.29	0.131
YEAR*Clumping*Micro-dam	0.01	1	0.01	0.84	0.361
YEAR*Density*Micro-dam	0.00	1	0.00	0.29	0.590
YEAR*Stem size*Posture*Clumping	0.03	1	0.03	2.39	0.123
YEAR*Stem size*Posture*Density	0.01	1	0.01	0.62	0.431
YEAR*Stem size*Clumping*Density	0.00	1	0.00	0.11	0.741
YEAR*Stem size*Posture*Micro-dam	0.00	1	0.00	0.08	0.773
YEAR*Stem size*Clumping*Micro-dam	0.01	1	0.01	1.25	0.265
YEAR*Posture*Clumping*Micro-dam	0.00	1	0.00	0.05	0.831
YEAR*Stem size*Density*Micro-dam	0.00	1	0.00	0.20	0.653
YEAR*Posture*Density*Micro-dam	0.00	1	0.00	0.00	0.973
YEAR*Clumping*Density*Micro-dam	0.00	1	0.00	0.24	0.625
YEAR*Posture*Clumping**Micro-dam	0.01	1	0.01	1.21	0.273
Error	3.22	288	0.01		

Density did not interact significantly with year, so even though the ANOVA in 2008 indicated that it had a greater effect, the difference in survivorship because of density was not significant across the years ($F=0.257$, $p=0.6125$). Planting posture, however, did interact somewhat significantly at five percent level of significance across the years ($F=4.27$, $p=0.0397$), (Figure 4.11). The only other significant interaction in terms of mean percentage survivorship which included “year”, was that of year, posture and clumping ($F=5.32$, $p=0.022$). From Figure 4.11 it is evident that 1) survivorship was better from 2006 to 2008 overall, and 2) survivorship was better for those planted upright than for those planted flat.

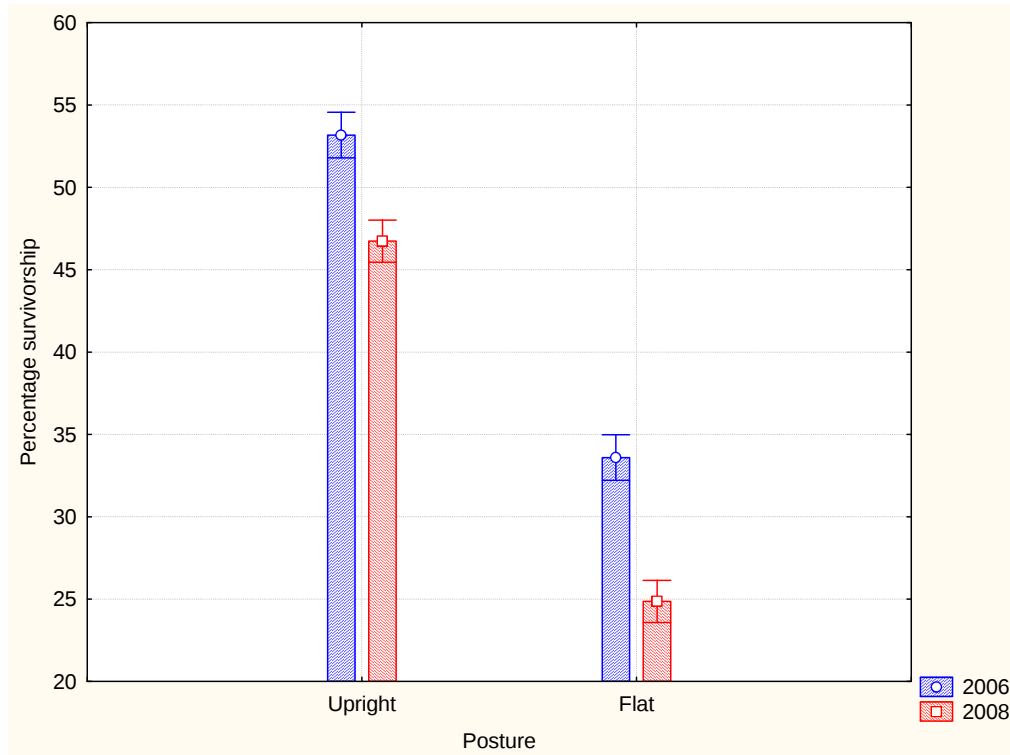


Figure 4.11: The influence of year and posture on *P. afra* survivorship.

4.5. Discussion

The overall survivorship (43 % for all treatments) recorded in 2006 in this study, two years after planting, was slightly lower than that recorded by Swart & Hobson in Kirkwood (1994), namely 56.5 %. By 2008, however, mean survivorship had dropped to 35 %. Swart & Hobson (1994) concluded that planting time or season, planting depth or water regimes made no difference with regard to mortality, but that leaf stripping negatively affected survivorship. Survivorship and growth in relation to seasonality are likely to be species-specific; the same applies to herbivory and competition. In a similar multi-species restoration trial, with herbivore exclosures, Parson *et al.* (2007) were able to demonstrate this effectively. While some species were not affected by herbivory, others were: some showed 10-fold loss in biomass accumulation due to herbivory. The authors concluded that seasonality and competition were minor influences, but that palatability and hence herbivory were major factors in designing restoration programmes.

With regard to truncheon stem diameter, Swart & Hobson (1994) found survivorship to vary significantly: 40 mm – 30.1 %, 10 mm – 56.5 % and 5 mm – 9.9 %. The mean survivorship rates (of 36.7 ± 1.8 % for the 40 mm and 36.9 ± 1.5 % for the 20 mm stem size classes respectively) reported in this study are encouraging. It is puzzling that Swart & Hobson (1994) avoided the 20 mm and 30 mm stem diameter interval.

Survivorship from planted truncheons is not yet static – mortality continues apace, despite truncheons being *in situ* for 36 months. All factors tested showed similar mortalities between the 2006 count and the 2008 count. Clumping and density appeared to be independent of increased mortality rate (2006 vs. 2008), and are only likely to become factors for carbon accrual once space between truncheons becomes minimal. The use of micro-damming following the successful establishment appears to be not significant, but key at the time of planting. This is demonstrated by the differential in 2006 survivorship (micro-dam vs. no micro-dam), which remained constant thereafter. It is possible that the micro-dam gradually diminished in size due to erosive forces, but if maintained may have improved survivorship for much longer into the trial.

Bainbridge (2007) describes a number of novel techniques to capture and harvest water for restoration in semi-arid and arid areas. Simple pitting can double water absorption rates but also serve as a seed trap for other wind dispersed seeds and is recommended for soils with capping Bainbridge (2007). Whisenant *et al.* (1995) termed the water harvesting shapes created around newly planted shrubs microcatchments, and found the survivorship improvement for *Leucaena retuse* to double relative to the controls. The study also found microcatchments to substantially improve plant biomass and to concentrate organic matter and seeds Whisenant *et al.* (1995). Although the microcatchments are considerably larger (Bainbridge 2007) than those employed in this study, the principle is a sound one.

4.5.1. Post planting establishment and stocking rates

Hobson *et al.* (1993), as cited in Stuart-Hill (1999), maintain that restored sites planted with *P. afra* truncheons require extended periods of rest before restocking. This is substantiated by my study in which the open plot had to be abandoned due to animal damage. Mortality is thus not restricted to a single event, and may be induced months or years after the initial planting. Mortality as a function of browse pressure (post-establishment of *P. afra*) at variable browse rates and chronosequences after rooting, requires investigation. Aucamp (1979) and Aucamp *et al.* (1980) demonstrated that browse pressure in the 25–50 % range of removal of the leaf material was optimal for *P. afra* shoot growth. This standard browse pressure recommendation may be imprudent in newly established truncheons, and also requires regional calibrations for rainfall and soil characteristics. Swart & Hobson (1994) recommend 2–3 year rest periods following planting. This recommendation (depending on the stocking rate) may still be insufficient for the plants to reach “maturity”, establish a skirt, form a semi-closed canopy and start replenishing the carbon pools. The use of nurse-plants (Padilla & Pugnaire 2006), especially if they are spinescent, may be a component of the solution.

4.5.2. Leaf age and palatability

Ting & Hanscomb (1977) first reported *P. afra*'s photosynthetic plasticity; Guralnick *et al.* (1984a) and Guralnick & Ting (1986) found that *P. afra*'s use of Crassulacean Acid Metabolism or CAM photosynthetic pathway was related to season and temperature. Water stress and the age of the leaf tissue also play a role, with younger leaves being less efficient at stomatal closure during prolonged drought (Guralnick *et al.* 1984b). The issue of leaf age has profound implications for the implementation of a large-scale restoration programme. Survivorship, when dealing with a slow growing species in an arid environment, is a key variable for investors. Selecting small truncheons with young leaves in dry environments could bring high mortalities, precipitating second and third phase plantings. This hypothesis is supported by Swart & Hobson (1994), who recorded highest mortalities (89 %) for

the smallest truncheons (5 mm stem diameter). Degraded soils with high clay content and soil capping (which I hypothesise induces additional aridity through rapid water runoff), would exacerbate the problem. Young truncheons may have less ability to withstand prolonged desiccation.

Leaf age also has a bearing on browse pressure and timing following successful establishment/restoration. Given Aucamp's (1979) and Aucamp *et al.*'s (1980) recommendations on rest periods following defoliation (and with the season being a key factor), it is quite likely that a critical threshold period exists before the truncheons become established and can tolerate the reintroduction of domestic herbivores. Similarly, an overabundance of indigenous herbivores (or minimal mega-herbivores) could negatively affect the growth rate of *P. afra* truncheons, arresting the growth habit into a large "bonsai" form. It also follows that browse pressure could lead to the *P. afra* truncheon perpetually producing new leaves and shoots – which have less drought resistance capabilities, and potentially could retard full *P. afra* cover indefinitely. The skirting habit (von Maltitz 1991) of *P. afra* could be prevented, which is vital for the function of the bush-clump mosaic and to ameliorate the micro-climate. Furthermore, excessive browse pressure will also impinge heavily on the *P. afra* litter fall contributions (2.5 t C ha⁻¹ yr⁻¹ Lechmere-Oertel 2003, Lechmere-Oertel *et al.* 2008) that are vital to replenishing the lost carbon stocks of degraded subtropical thickets.

The issue of the "sweet" (palatable) or "sour" (unpalatable) *P. afra* varieties is widely purported by local farmers. Palatability is related to tannin content (Ras 1996), which varies seasonally and between individual *P. afra* trees. Ras (1991) demonstrated that defoliation will increase tannin levels in *P. afra*, even two weeks after the event. Later Ras (1996) discovered that there was a fundamental difference in the ecophysiology of the palatable vs. the unpalatable species. Palatable plants were found to have higher acid content (primarily malic acid) and recovered faster. If two varieties do exist, and one can't be induced to become palatable with browsing, the sourcing of stock and the areas of planting for different varieties requires further investigation.

The tricky balancing act of using domestic herbivores to stimulate *P. afra* growth, without causing carbon stocks to leak, requires careful assessment – especially across the aridity and degradation gradients. A scientific case also needs to be made to demonstrate that low numbers of domestic herbivores are beneficial to newly restored areas and do not constitute a case of “non-permanence” as defined by the Clean Development Mechanism of the Kyoto Protocol.

4.5.3. Litter and canopy cover

Intact semi-arid subtropical thickets have a large litter layer (~10 cm) (Lechmere-Oertel *et al.* 2005b), which is lost in the degraded states due to the absence of leaf litter production and the erosive forces of rain and wind. Sigwela *et al.* (2003) hypothesise that litter is vital for seedling establishment. This loss of litter also greatly affects water infiltration rates (Scheltema as cited in Stuart-Hill 1999) and could therefore play a key role in *P. afra* truncheon survivorship with regard to restoration plantings in degraded landscapes. Litter layer or mulch addition (in conjunction with micro-damming) therefore needs to be investigated as a treatment for improving *P. afra* survivorship, but would increase costs. Bainbridge (2007) motivates that mulches provide protection from wind and water erosion, thereby improving plant establishment. A trade-off may exist where mulches may inhibit infiltration during small precipitation events, but can reduce water lost from the soil following large precipitation events. During the restoration of a landfill site where mulches were used, Athy *et al.* (2006) recorded mixed results. While some species did not require any form of mulch, the mean growth rate for all species was improved with either leaf or wood mulch, but retarded when a combination of leaf and wood mulch was applied. Leaf mulch is advantageous in that it is not a vector for fungus, but diminishes more quickly (Athy *et al.* 2006).

4.5.4. Limitations of this study

A potential complication in the experimental design of the trial was planting on high percentage clay soils with poor drainage. At the onset of the project, the issue of

soil capping, as it affects soil penetration from rainfall events and *P. afra* survivorship, was untested. Consequently, soil planting depth was not tested in the rigorous multi-factorial design and may have played a key role in some treatments – especially when correlated with pre- and post-planting rainfall events. Although Swart and Hobson (1994) argue that planting depth played no role in survivorship, I hypothesise that soil type and degradation state will, in conjunction with planting depth, influence survivorship. This is partly supported in results (see Figure 4.3) where small truncheons with no micro-dam showed significantly higher survivorship than larger truncheons. Smaller truncheons are planted closer to the soil surface, and the lack of a micro-dam – which holds water long enough to penetrate the depths needed by the larger truncheons – may have resulted in desiccation and greater mortalities. It is also not clear how the watering regimes applied by Swart & Hobson (1994) may have influenced soil depth as a factor.

A second limitation of the trial was the failure to incorporate climatic variability and seasonal influences into the multi-factorial design. Rainfall is extremely unreliable within the study area. Ambient temperature, solar radiation intensity, soil temperature and soil moisture will vary significantly between seasons, and over relatively short periods within seasons. Key seasonal windows may exist with regards to optimising *P. afra* cut truncheon survivorship.

4.5.5. From research to implementation

A central premise of these trials has been research for implementation. Survivorship data from the *P. afra* multi-factorial trials would inform the large-scale restoration planting teams, with a view to improving survivorship and improving cost-effectiveness. *P. afra* mortality in the contract plantings done by Working-for-Woodlands workers has profound implications in terms of costs per hectare – and ultimately drives the economic feasibility of the entire restoration programme, which hopes to employ the carbon economy to fund the restoration. A major confounding variable in the restoration initiative has been the abiotic variability inherent in the rugged and complex BNR. This variability has manifested itself in the variable

survivorship percentage for the *P. afra* planting contract blocks, undertaken over the last four years by the Working-for-Woodlands teams. Survivorship in the contract blocks ranges from 0–77 % (n=26, mean 34.3 ± 4.5 %) and could be attributed to a host of factors, including baboons, kudu, frost, grass competition, desiccation and management efficiency. Palmer & Pitman (1961) list cattle as being exceptionally destructive but claim that frost is not a limiting factor, provided there is protection from adjacent dense vegetation. A critical and two-fold management implication of the high mortalities reported in the multi-factorial trial and the Working-for-Woodlands planting contracts is 1) secondary planting costs, which impact heavily on the economic feasibility for carbon credits, and 2) the retardation of canopy closure, which impedes the accumulation of the vital litter layer. The leaf litter from the isolated *P. afra* clumps will be leaked from the landscape. I further hypothesise that the absence of the litter layer will affect the water-use efficiency at the landscape scale, preventing the prodigious litter production (Lechmere-Oertel 2003, Lechmere-Oertel *et al.* 2008) needed. This could have serious implications for the carbon verification, in years to come.

Given the multi-factorial nature of the experimental design and the sampling effort, the good survivorship (2008) in treatment 17 (65 ± 3.1 %) should be seriously considered as preliminary recommendations for restoration using *P. afra* cut truncheons. In other words: larger stem diameter truncheons, planted upright with a micro-dam at the base of the truncheon, and at higher densities, should yield the best results in terms of survivorship. I emphasise this may only hold true for degraded flat areas, with high clay percentage, no litter layer and soil capping. I speculate that there is, furthermore, ample opportunity for improving these survivorship figures in cases where minor water additions during critical periods are both practical and economical.

4.6. Conclusions

The multi-factorial trial conducted in the flat, degraded Gamtoos bontveld at Goedeheop (in the BNR) has revealed that some factors are critical, others not

significant, and that some measure of interplay exists. Stem diameter and clumping did not affect survivorship significantly, while planting posture, density and micro-damming did. Survivorship was highly variable between treatments and within treatments and, on average, continued to drop – even three years after planting. Mean survivorship for all treatments after three years was ~35 %.

The results presented above can only be interpreted with confidence in the absence of herbivory and disturbance agents such as baboons, herbivores, porcupines and rodents. The initial experimental design allowed for the quantification of this; however, the intensity of disturbance – principally from baboons – prevented any establishment. Newly planted plants in highly degraded environments can experience high fatalities due to herbivory (Bainbridge 2007). It would be worth investigating the influence of herbivory on planted truncheons for both survivorship and growth rate, after successful establishment. Furthermore, judicious stocking rates and timing of herbivory after planting is needed for the financial models required by carbon traders and investors. The results from this study need to be contextualised in terms of the abiotic site-specifics. These truncheons have been planted in degraded areas with high percentage clay soils, soil capping, poor drainage and no aspect. Results from well drained soils could yield significantly different results. The establishment of the trial was undertaken with a view to long-term monitoring. It remains to be seen if survivorship will stabilise and what the multi-factorial design will reveal in terms of *P. afra* growth rate.

In summary, considerably more knowledge is needed for the restoration protocols for cost-effective planting and optimal establishment of *P. afra* in the highly variable environments of the degraded spekboomveld and spekboom thickets of the Eastern and Western Cape.

Chapter Five

Concluding discussion and future research

“Providence meant to spoil our farmers in placing the spekboom on the hills of the Karroo” – anonymous in Palmer & Pitman pg 111, (1961).



5.1. Introduction

A significant passage of time has passed since the Rio Summit in 1992, which ultimately spawned the carbon economy. It has given rise to a new discipline and an army of scientists and bureaucrats. Climate change and carbon offsetting has become mainstream in the western world. South Africa has been incredibly slow to seize the opportunity for restoration funding that can be afforded through the carbon economy. Among the developing nations, Brazil, Indonesia, China and India have dominated the formal CDM carbon market over the last decade. Point Carbon (2008) report that China controls the sales of CDM market (62 %), followed by Indonesia (10 %), Brazil (8 %) and India at 5 %. South Africa is not reflected.

The recent emergence of payment for ecosystem services (PES) has also been fortuitous for restoration ecologists, planners and practitioners, carbon sequestration being an anchor ecosystem service. The fundamental key issue has been quantifying the ecosystem service and getting the goods to market. This research was specifically commissioned to close the gap with regards to the first point. Much work still needs to be done with regard to the latter. The two themes of carbon stocks and *P. afra* survivorship tackle the knowledge gap in concert, but only where *P. afra* is a dominant feature in the plant community (i.e., spekboomveld and spekboom thickets).

5.2. Allometry

The duration of the field work and financial costs in this study have largely indicated that allometric techniques have limitations in estimating woody and succulent biomass C in intact subtropical thicket vegetation, but do become more practical further down the degradation gradient. The employment of these allometric relationships as a technique for quantifying woody and succulent biomass C stocks in other subtropical thicket areas on a large scale (as required by the carbon economy) is unlikely. The cost and time factors are considerable. There is possibly scope for using these data as ground-truthing for remote sensing techniques such

as multi-temporal L band radar (see Mitchard *et al.* in review). Should the combination prove cost-effective then these relationships will become invaluable to other field workers. There will be a need to supplement with other species-specific allometric relationships, e.g., *Cussonia* spp. and species with unique vegetation architecture or carbon partitioning strategies (*Euphorbia coerulescens* or *E. bothae*). More work is needed for those species that proved problematic (e.g., *Jatropha capensis*).

It is worth noting that growth form is an important criterion when selecting a suitable guild-specific allometric equation (see Table 3.2). Although a suitably robust relationship was established for *Grewia robusta*, it was not applied to the congeneric *Grewia occidentalis*. Personal observations on the growth form indicated a more accurate estimate of standing biomass C would be derived from employing the *Ehretia rigida* guild-specific regression equation. Canopy area in *Grewia robusta*, but cumulative basal stem areas in *Ehretia rigida*, were found to be the best predictors. Similarly the *Gymnosporia buxifolia* and *Gymnosporia polyacantha* congeners did not employ the same regression equations.

5.3. Carbon stocks

Carbon stocks in the intact semi-arid subtropical thickets of the BNR are substantially lower (~50–60 % lower at similar depths) than has been reported in the literature for other South African intact semi-arid subtropical thickets (Mills *et al.* 2003, 2005a, b). This serves to increase the differential between the TCS of subtropical thickets and tropical forests. An investor is more likely to favour the 640 t C ha⁻¹ for Australian tropical forests (Mackey *et al.* 2008 as cited in Garnaut 2008) than the 117–120 t C ha⁻¹ subtropical thickets of the Baviaanskloof.

It is also crucial to keep in mind that irrespective of the TCS, the accrual rate and price of carbon are key to the financial viability (Mills *et al.* 2003). A massive carbon sequestration potential can be nullified if the carbon accrual rate makes the return on investment too long or induces more risk. Conversely a mediocre carbon

differential could attract significant interest if the accrual rate is rapid and the risk is removed completely in the later years. This needs to be clearly understood in the context of restoring degraded subtropical thickets, where the net primary productivity (NPP) is limited due to rainfall and the TCS are substantially lower than tropical forests. To make matters more difficult, tropical forest destruction has seen considerable more media and scientific exposure when compared to semi-arid thickets.

The research undertaken in this thesis was aimed at focusing intensively on the carbon pools of a number of subtropical thicket types found within the BNR, and to quantify what percentage of the TCS and various sub-pools had been lost. With this objective, relevant detail has been presented to provide the level of precision needed for the formulation of a carbon sequestration strategy for the Baviaanskloof valley and BNR.

The results of Chapter 3 show a reasonable amount of similarity among the TCS (to a soil depth of 25 cm) of the non-succulent, intact semi-arid subtropical thicket types, i.e., 70.64 ± 17.24 to 74.08 ± 15.62 t C ha⁻¹. The intact succulent forms of the subtropical thickets were estimated to have 18–24 % more TCS (to a soil depth of 25 cm) than non-succulent forms. The partitioning of the carbon is quite different too. Whereas the intact more mesic forms store 1–1.5 t C ha⁻¹ in the litter C pool, the semi-arid form has a three-fold litter layer (4.85 t C ha⁻¹) and approaches the figures reported for African tropical forests (Glenday 2006). Intact Gamtoos bontveld has nearly three-fold more carbon in the herb C pool, when compared to the other forms of intact thickets, and is reflected in the mosaic nature of isolated bushes within a mixture of karroid and grassland elements.

The TCS differentials between old lands and intact subtropical thickets in the BNR (and greater Albany Thicket biome), while being of academic interest, can generally be disregarded as largely irrelevant – in contrast to abandoned old lands in topical areas (Silver *et al.* 2000). The essence of the results from Chapter 3 (as it pertains to the restoration of subtropical thicket via the carbon economy) rests on the TCS

differentials between the degraded and intact forms of the Baviaanskloof spekboom thickets. With 65 % of the TCS being lost, one can conclude that these landscapes are highly degraded, if not desertified. The greatest net C lost was in the woody and succulent tree/shrub C pool (~86 %), followed by litter C, soil C and root C (71 %, 57 % and 27 % respectively). The herb C pooled doubled (in the degraded state) largely as a result of grass invasion, which could bring the serious risk of fire for newly planted *P. afra* truncheons.

The rate of degradation and carbon leakage for the area is unknown and beyond the scope of this study. Garnaut (2008) makes the statement that soil C stocks are variable spatially and temporally, even fluctuating seasonally and annually. It is not clear if the author is referring to agricultural soils solely or not, and the question of carbon stock leakage following degradation still remains unanswered for semi-arid subtropical thickets.

I would speculate that there have been pulses of degradation over the last 100 years, largely a function of market forces and perverse government incentives. If we assume 100 years to be a realistic estimation, it implies that C has been leaking by an average of $0.57 \text{ t C ha}^{-1} \text{ yr}^{-1}$ from the total landscape. It is plausible that the woody and succulent C was lost over a relatively shorter period of time. This would have severely impacted on the litter-fall production rates (see Lechmere-Oertel *et al.* 2008), precipitated the accelerated loss of the litter layer and ultimately started the below ground C leakage. Litter fall rates for a variety of subtropical thicket types and individual species needs to be measured. Litter fall rates and the correlation with leaf litter loss, and temperature extremes of periods of aridity need to be investigated.

With the allometric relations that have been developed through this study, it would be prudent to assess the carbon stock differential between degraded and intact Baviaanskloof thicket savannas. These degraded areas are highly visible, require biodiversity return and could qualify for the voluntary carbon market.

5.4. Carbon accrual rates

There is a dire shortage of data on growth rates for subtropical thicket species. The only known reliable study on *P. afra* growth rate is that of Mills & Cowling (2006). It is worth noting that this experimental planting mostly included a geno/phenotype of *P. afra* from another area (Steytlerville). This geno/phenotype has a markedly different growth form, being prone to upright growth and less like the classical *P. afra* umbrella-like bush with a skirt. The local *P. afra* recorded a carbon accrual rate of $2.4 \text{ t C ha}^{-1} \text{ yr}^{-1}$ vs. the $4 \text{ t C ha}^{-1} \text{ yr}^{-1}$ of the Steytleville geno/phenotype, and is closer to the Great Fish River *P. afra* geno/phenotype ($1.2 \text{ t C ha}^{-1} \text{ yr}^{-1}$) (Mills & Cowling 2006). This variability in carbon accrual rates is unfortunate for engendering confidence in the carbon investment community, and will require adopting the lowest rates less SE, (currently approximating $0.9 \text{ t C ha}^{-1} \text{ yr}^{-1}$), unless more site specific case studies can be produced. Ultimately there is a need for a spatially explicit dataset (GIS layer) of the carbon accrual rates across the Albany Thicket Biome.

Litter is probably the key carbon pool in terms of restoration and carbon accrual rates, simply because of the relationship with precipitation, water use efficiency and soil moisture. Taking Mills & Cowling's data (2006) and litter pools from this study (intact Baviaanskloof spekboom thickets) at $4.85 \pm 0.99 \text{ t C ha}^{-1}$, a simple regression (time in years vs. litter C in t C ha^{-1}), $y = 20.776x$, $R^2 = 0.91$, predicts it would take at least 100.8 years for the litter to be restored using the Steytleville geno/phenotype. Using the local phenotype is likely to reduce the time period to approximately 41 years. Most noteworthy is that the truncheons would need to be planted at a density of 1 750 per hectare, with close to zero mortality, experience exceptionally low levels of browse pressure and approach 95 % *P. afra* canopy cover. I predict the litter layer is a governing factor on the carbon accrual in the other pools.

5.5. *P. afra* genotypes

There is a lack of knowledge on the spatial distribution of *P. afra* genotypes/phenotypes (Mills & Cowling 2006). Should it become possible to quantify these patterns, correlations with TAGC as well as C accrual rates would be useful. More modelling work (similar to Robertson & Palmer 2002) is required to predict the spatial distribution, density and biomass of *P. afra* across the degraded landscapes of its former range.

In terms of restoration protocols, the advocacy of one geno/phenotype may have profound implications in the long run – especially at the landscape level. The three fold difference in litter C (3.4 t C ha^{-1} vs. 1.2 t C ha^{-1} , Mills & Cowling 2006) is directly attributable to the different genotype/phenotype used for the restoration. The implications of the lower litter C pool on the patch and landscape hydrology is unknown. The authors also reported differences in biomass C and root C between the two with the local geno/phenotype producing 43 % more root C and the Steytlerville (berg spekboom) geno/phenotype producing 44 % more biomass C.

5.6. *P. afra* survivorship

The destructive damage caused by baboons (*P. ursinus*) and kudu (*T. strepsiceros*) to the newly planted truncheons was not anticipated in the planning stages of the project. This is problematic in the sense that very few land parcels (even in the highly degraded state) will be devoid of these two species, making the prediction of survivorship in the rural farming areas risky. A short window period is needed for the soil to settle and plants to take root. A trade-off exists between the planting depth needed to limit this form of damage and water infiltration depths from precipitation. It is also highly probable that root development in *P. afra* is cued by some combination of climatic and abiotic variables. More work is needed on establishing if survivorship is enhanced during certain periods of the year (Palmer & Pitman 1961), or where soil temperature and soil moisture have reached a threshold, or a combination of both.

The stark reality of the results from the work presented in Chapter 4 centres on the implications of the mean percentage survivorship (~35 %) after three years. If we assume this to be a conservative and realistic fact, it may require seven or more years of successive planting to achieve the desired planting density needed to achieve the carbon accrual rates reported by Mills & Cowling (2006). Even at 50 % survivorship (similar to Swart & Hobson 1994), it would equate to five years of successive plantings. This is on par with anecdotal evidence from the WfW experience where successive follow-ups for the removal of alien invader plants are standard operating procedures. The fundamental difference in the case of subtropical thicket restoration is the demonstration of the *bottom line* input costs, which ultimately determine the economic feasibility. I predict that 35 % survivorship, coupled with the lower carbon stocks and accrual rates (relative to subtropical forests) will render degraded subtropical thickets unattractive to the carbon offsetting fraternity.

The issue of long-term truncheon survivorship and growth rate in the presence of browse pressure remains to be tested. For the effective restoration of the highly endemic vegetation of Amsterdam Island, it was necessary to exclude livestock completely (Micol & Jouventin 1995). In a multi-species restoration trial (Parsons *et al.* 2007) found herbivory to have a significant effect on some species (both in terms of survivorship and growth).

I hypothesise that a critical browse pressure exists whereby *P. afra* truncheons will be retarded (or “bonsaied”) from incessant pruning. This effectively prevents the skirt and canopy from forming and resulting in leaf litter being lost from the landscape and therefore not replenishing the soil carbon pool. The relationship between timing and intensity of browse pressure will not be a linear one, but more likely to require extremely light browsing after the establishment period, becoming increasingly greater with time until a peak or optimum is reached. The recommendations of Palmer & Pitman (1961) that replanting should only commence in spring with one year old cuttings should be investigated too.

Further work is needed to test for the incidence of pathogens (e.g., *Phytophthora* spp.) that may influence survivorship. There is a probability that in some soil types and during certain times of the year high risk periods exist for planting. Inoculations of the degraded soil with ectomorrhizae, mulch additions and hygroscopic gels all need testing. These data are critical in order to complete a thorough cost-benefit-analysis and various sensitivity analyses in the volatile carbon economy, especially where survivorship in spatial and temporal survivorship is low for the standard planting techniques.

Frost and fire are features that have low levels of impact for intact subtropical thickets, but become significantly enhanced in the degraded state. The resilience (and growth rate) of *P. afra* truncheons for these factors needs investigation. It has been my personal observation that, where plants/truncheons are isolated, frost has a severe and fatal effect for succulents. When the same degree of frost is evidenced on intact vegetation (at same catena position and aspect), the damage is superficial and the plants recover quickly.

Substantially more work is needed in the realm of *P. afra* eco-physiology, particularly field experimentation. Having established that seasonality and increasing aridification induce CAM for the species (Guralnick & Ting 1986), it remains to be seen what role dew and mist play in this relationship. I hypothesise that *P. afra's* phenomenal growth rate, in low rainfall and high temperature environments (Mills & Cowling 2006), is directly due to the ability to harvest dew water. The occurrence of heavy dew during certain periods of the year in the river valleys of the BNR is pronounced. There has been to date no quantification of this addition to the MAP of the semi-arid subtropical thickets, and I predict that it could double the amount of water available to the CAM dominated communities. The harvesting of this water can either be in the form of stem flow (running to the base of the stem and into the soil) or directly absorbed through the stomata in the dark periods. Carlyle-Moses (2004) found that in semi-arid Spain the volume of water supplied to the base of the plant from stem flow during precipitation events was sufficient for the plants to survive drought periods. A better understanding of the

water-partitioning behaviour of subtropical thickets (within the matrix of a changing climate and various degradation classes) may become extremely useful in the gambit of PES and watershed management. The fine-scale water-harvesting and water-shedding behaviour of *P. afra* bushes and trees, specifically with regard to water supplemented through stemflow, needs investigation.

The STRP is a unique restoration exercise in that it is largely hinged on the *en masse* vegetative propagation of a single species (for the first phase at least). There is a general belief that other biodiversity will infuse naturally into the restored sites – but this has not been proved. The mere fact that the restoration scale is in the order of 1×10^6 hectares, and dominated by CAM plants, does make it slightly extraordinary. The restoration is also ambitious given that the costs to restore the land will far outweigh the market value of the land, and most farmers will not have the financial resources to undertake the restoration. This unfortunately precludes a number of restoration techniques that have been successfully employed elsewhere (on a smaller scale and with an appropriate budget). There can be little doubt that tree shelters (del Campo *et al.* 2006) will not only improve the survivorship of *P. afra* truncheons, but also accelerate the carbon accrual rate afforded through herbivory protection. Tree shelters have the double advantage of ameliorating the micro-climate around the plant, as well as harvesting dew water. Del Campo *et al.* (2006) found that 6 % soil moisture was a threshold for active plant growth and some tree shelter designs were able to surpass this threshold. These findings are more profound when one considers that in many arid environments dew and fog are crucial to plant survival in arid and semi-arid regions (Agam & Berliner 2006). In some areas, especially where herbivore numbers cannot be controlled effectively, the employment of nurse-plants (Padilla & Pugnaire 2006, Aerts *et al.* 2007) may yield improved survivorship and ultimately enhance growth rates.

5.7. General

South Africa's semi-arid thickets have been "neglected and abused" (Hoffmann & Everard 1987), as has the same vegetation in adjacent countries (Dyer 1952).

Despite being well-described and mapped by Acocks (1975, 1988), and finally achieving biome status in 1996 (Low & Rebelo 1996), they remain neglected in terms of research, policy, monitoring and ultimately management. In the chapter “Ecology of the main grazing lands of South Africa” by Tainton (1999), subtropical thickets do not feature. In a comprehensive review of the forests and woodlands of South Africa (Lawes *et al.* 2004), the semi-arid thickets are ominously absent (see map on the frontispiece). Clearly, with total carbon stocks in the region of 100–250 t C ha⁻¹, these lands cannot be classed in the karroid group (especially with 30 fold TAGC compared to the karoo vegetation (Mills *et al.* 2005a)). The absence of fire as an ecosystem driver precludes its definition as a savanna form. With a near-closed canopy cover, a dense impenetrable tangle of spinescent woody shrubs, succulent shrubs, and TAGC of around 35 t C ha⁻¹, they should surely qualify as woodlands. The neglect of subtropical thickets from a policy perspective needs to be addressed.

Priority conservation areas in the subtropical thickets of the Albany thicket biome have received a coordinated and systematic land-use planning approach (Rouget *et al.* 2006). This is where payment for ecosystem services can be touted as a viable substitute for marginal land use practices. This is likely to be particularly true for areas that will experience water shortages and where crop irrigation no longer becomes feasible. Carbon sequestration as a potentially viable land use requires investigation, critically so for areas of irreplaceable conservation biodiversity within the subtropical thicket biome (see Knight *et al.* 2003). There is also a need for mainstreaming mechanisms for investors to accrue carbon and biodiversity credits of the same piece of land and on the same budget (Bekessy & Wintel 2008). Aronson *et al.* (2006b) contend that nature conservation, ecological restoration and economic development could and should be facilitated by the same budget.

5.8. Is carbon farming in degraded subtropical thickets viable?

Given the paucity of case studies for the growth rates of subtropical thicket species in landscape restoration experiments, we are obliged to rely heavily on the results of Mills & Cowling (2006), at least until more published records are produced. Taking

the conservative approach of using the data from the site-specific genotype would invoke the lower carbon accrual rate of 2.4 t C ha^{-1} . This raises the question as to costs vs. benefits of quantifying the carbon pool differentials. The biomass C after 27 years accounted for 33 % (@ $0.796 \text{ t C ha}^{-1} \text{ yr}^{-1}$) and would be relatively quick and cheap to quantify (see table 3.2). Litter C would require more logistics, but still is relatively simple and requires no laboratory work or laborious sieving and drying. Unfortunately this C pool accounted for merely 4.6 % of the TCS after 27 years (@ $0.111 \text{ t C ha}^{-1} \text{ yr}^{-1}$). This leaves the soil C and root C, 41.5 % and 20.9 % of the TCS after 27 years, and accrued at $1.00 \text{ t C ha}^{-1} \text{ yr}^{-1}$ and $0.504 \text{ t C ha}^{-1} \text{ yr}^{-1}$ respectively. While these are significant C additions, the variability in the landscape and the costs to conduct large-scale below ground carbon stock assessments have not been carefully quantified as yet. Also, given the complexity of the methodology to calculate soil and root C (Garnaut 2008, Mills 2008), is it financially worthwhile to calculate baselines?

The first report concerning the potential for farming carbon in South Africa within degraded subtropical thicket comes from Mills *et al.* (2003). The report includes a sensitivity analysis that shows ranges of potential income from carbon farming of $\$1\text{--}684 \text{ ha}^{-1} \text{ yr}^{-1}$ (2003 prices, TCS to a soil depth of 50 cm and approaching 250 t C ha^{-1}). Turpie *et al.* (2003) discounted the possibility of restoring semi-arid subtropical thicket via carbon credits, citing the slow growth rate of subtropical thicket plants as a major limiting factor. Similar findings for the semi-arid rangelands of Australia have been reported (Flugge & Abadi 2006). The authors concluded that the carbon price would need to be A\\$25–46 per $\text{t CO}_2\text{e}$ to make it commercially viable (2006 prices). Mills *et al.* (2007) modelled the economic feasibility of restoring subtropical thickets using a discount rate of eight percent and found the IRRs to vary from 7.9–24.0 %. Key variables included the growth rates of *P. afra*, residual standing plant biomass, stocking rates after planting and the price of C. The authors assumed restoration costs to be relatively high ($\sim \text{R}5\,000 \text{ ha}^{-1}$). It is potentially feasible to reduce planting costs in some environments to $\text{R}1\,500\text{--}2\,000 \text{ ha}^{-1}$ through mechanisation, correct workload quantification and effective management. Mills *et al.* (2007) also bargained on substantial mortalities, which

required two years of successive follow-up plantings. Survivorship of *P. afra* cut truncheons with no-after care on a large scale is highly variable. Results from Chapter 4 demonstrate that 65 % survivorship is possible after three years – even on highly capped clay soils with poor drainage. If this rate of survivorship can be repeated on a large scale (implying a cost saving through no follow-up), and if mechanisation can reduce costs to <R2 000 per hectare, one would expect incredibly favourable IRRs.

The Garnaut Report (2008) estimates that soil carbon could become the most valuable crop for Australian farmers, sequestering C @ 0.5 t C ha⁻¹ yr⁻¹ and providing an additional income to crop farmers of around US \$36 ha⁻¹ yr⁻¹ (one t CO₂e being \$20). Garnaut (2008) estimates that should carbon prices stabilise at US \$20 t⁻¹, it would provide a ten-fold increase in income for landowners in the arid and semi-arid areas of Australia. For South African semi-arid subtropical thickets, which receive similar or slightly more rainfall, but are dominated by CAM plants in the landscape, one would expect a similarly favourable forecast. With pastoralism gross current incomes of R75–150 ha⁻¹ yr⁻¹ (Kirkman 2006), carbon accrual rates of 15 t CO₂e ha⁻¹ yr⁻¹ (Mills & Cowling 2006) and carbon prices in the region of €20 t CO₂e, peaking at €33 t⁻¹ CO₂e (Point Carbon 2008), it is hypothetically possible to achieve gross incomes of R3 600 ha⁻¹ yr⁻¹. These figures exclude restoration costs, transaction fees, verification costs, and land acquisition costs. Important to keep in mind is that the C accrual rate has been averaged for the 27 year period (Mills & Cowling 2006) and accrual rates are likely to follow a sigmoidal curve and peak towards 20–25 years.

Engelbrecht *et al.* (2004) demonstrate the limitations of bio-sequestration of CO₂ in South Africa, citing that, in reality, to sequester the annual CO₂ emissions from fossil fuels would require the total land surface area of South Africa – with this solution having a limit of 30 years. The authors also allude to the fact that “enhancement” or super-saturation of C for intact subtropical thickets is not feasible. It is curious why (given the publications and Mills & others on the carbon stocks and carbon sequestration rates for ST) Engelbrecht *et al.* (2004) do not mention subtropical

thickets as a potential sink (but mention reduced fires, savanna thickening, reduced sugar cane burning, etc.) and conclude that bio-sequestration in South Africa is limited due to rainfall and net primary productivity. The window of opportunity for bio-sequestration in South Africa may further be limited should recommendations to investigate geological sequestration, oceanic sequestration and fertilisation, employing aged mines and gas fields, etc. (Engelbrecht *et al.* 2004), provide a cheap technological solution.

Considering that a mere 4.8 % of white-owned land has been successfully redistributed since 1994 (Philip & Hassen 2008) and the target is 30 % by 2014, it is unlikely that that this target will be met. More importantly the issue of sustainability for young emerging black farmers needs to be factored in. Within the areas of degraded subtropical thicket in the Eastern and Western Cape, a unique opportunity has arisen. Should land-reform proceed hastily in the highly degraded subtropical thicket areas, without a comprehensive socio-ecological-financial assessment, these emerging farmers will surely fail. The situation is akin to an inexperienced city-dweller being “given” a trading store with no stock on the shelves, and no capital to replenish the shelves either. Should a large percentage of land transfer happen to a relatively few “disadvantaged individuals” (witness the BEE in the mining sector), it will be another lost opportunity (Philip & Hassen 2008).

5.9. Enter the carbon economy.

The rate of failure for emerging farmers that have been recipients of the land redistribution and land-reform programmes is becoming more and more apparent (Du Toit 2004, Philip & Hassen 2008). The relative ease of restoring degraded subtropical thickets previously dominated with *P. afra* suits a new generation of emerging black farmers. Furthermore, the task is labour intensive and the areas are vast which could revitalise the rural economy and facilitate “keeping the people on the land in living landscapes” (Knight *et al.* 2003). To make matters slightly easier, the produce (carbon) doesn’t need to be transported to market, and the market has been created (~€40 billion annually (Point Carbon 2008)). The DWAF, through

Working-for-Woodlands (incorporating the Subtropical Thicket Restoration Programme) has been actively investigating the feasibility of employing the carbon economy as a tangible PES initiative since 2004 (Powell *et al.* 2004, 2006, Mills *et al.* 2007). Should the carbon sequestration methodology, (which is due for submission to accreditation agencies) receive accreditation, the platform will have been laid for a sustainable land-reform programme within the Eastern Cape. Additionally, all of the above emerging black farmers will have the grace of 30+ years to receive training, skills, capacity and the confidence needed to become successful and sustainable farmers – something many of the current landowners have clearly failed to do. Income from carbon farming should provide a steady income while the *P. afra* plantings grow and reach maturity. It will also provide the State with an opportunity to build the capacity and institutions needed to safeguard the restored natural capital – preventing a repeat of the tragedy of the last 120 years.

5.10. Carbon trading and biodiversity trade-offs

Although Mills & Cowling (2006) advocate *P. afra* as the solution to subtropical thicket restoration, citing *P. afra*'s vegetative propagation negating any need to construct nurseries and zoochory bringing plant biodiversity, this remains to be seen. I hypothesise that, although some measure of seed dispersal will happen, there is no guarantee that zoochory will recruit the right species, at the right densities and in good time. Furthermore, the micro-climate may be unfavourable (for a number of decades) for seedling germination, sapling establishment and ultimately canopy recruitment. A trade-off may exist between *P. afra* canopy providing the micro-climate needed for imported seeds and the *P. afra* canopy excluding the same species (light and water harvesting). Subtropical thicket restoration may yet prove that we indeed do need the ecological knowledge to reach the "tipping point" for natural succession to proceed. Furthermore should the economic incentives from obtaining carbon credits become highly attractive (see Caparrós & Jacquemont 2003), it is possible that "natural capital speculators" will seek to produce vast stands of *P. afra* in habitats where it was previously absent or

at low densities. This could have dire consequences for the succulent endemic flora of the region and requires urgent policy interventions.

As biodiversity decline continues across the globe, more cases will be made for the payment of biodiversity credits. Ferraro & Kiss (2002) make compelling arguments for direct payments for conserving biodiversity. Although restoration is expensive (Woodworth 2006b, Bainbridge 2007) the fusion of economics and conservation under the banner of “restoring natural capital” (RNC) may yet provide the answers. PES as a discipline being the timely delivery vehicle. In the context of a South Africa needing to be moving towards a low-carbon economy, production costs are likely to increase, making restoration work all the more expensive. There is a real danger in over-reliance on the income from carbon credits to finance the restoration of degraded subtropical thickets. In a study of three tree species (*Pinus resinosa*, *Picea abies* & *Juglans nigra*) used for commercial afforestation in southern Ontario, Canada, the income from carbon credits increased the IRRs fractionally Yemshanov *et al.* (2007). There is a need inside the STRP to quantify the potential gains made through the same restoration intervention (i.e., same budget), but with another beneficiary. Water security, water quality, NPP, and biodiversity should be bundled as a composite PES package, invoking the option for cross-subsidisation, and with a view to effecting catchment-level sustainable land management. The piecemeal sale of carbon credits will undermine the chances of ever achieving this goal.

The DWAF has extended the lifespan of the project and increased the geographic range. For the next three years, further trials and assessments will be undertaken in the three megareserves in the Eastern Cape: Great Fish River Reserve, Addo Elephant National Park and the Baviaanskloof Nature Reserve. Furthermore, trials have been commissioned on private land across the entire Albany Thicket Biome to establish a standardised experiment for the survivorship of *P. afra* truncheons. These trials will focus on the degraded spekboomveld and spekboom thicket areas with a view to covering a wide range of abiotic variables. This will further strengthen the robustness of the restoration protocols needed for the private farmers and communal rangeland managers to enter the carbon economy with the least risk.

5.11. Conclusion

Whisenant *et al.* (2005) came to the conclusion that despite excessive degradation it is possible to initiate a process to get to a point for natural succession to start operating in semi-arid environments. The secret lies in concentrating the scarce resources, selecting keystone species and changing the microclimate (Whisenant *et al.* 2005). The work undertaken in this thesis made some progress with regards to the above. The trials undertaken on *P. afra* cut truncheon survivorship have provided a first draft for restoration protocols in the *P. afra* deficient rangelands.

A significant investment has been made in the accurate quantification of the carbon stocks within the study area. Our knowledge base has been increased with regard to the total carbon pools and how the carbon is portioned (sub-pools) in the various subtropical thicket types. The total carbon stocks for intact subtropical thickets in the BNR are substantially lower than reported in the literature for intact subtropical thickets in the same area. However, the carbon differentials between the intact and degraded state are substantial and may prove sufficient to warrant a revision of unsustainable land-use practices and the restoration of the lost natural capital. Much will depend on the price of carbon and the cost of the restoration, of which the latter will largely be a function of *P. afra* survivorship across the Albany Thicket Biome.

While considerable progress has been made on the biological front with regard to attaining knowledge on the restoration of degraded subtropical thicket, much still needs to be done in the socio-political and institutional realms (see Corbera *et al.* 2009) to ensure the ecosystem goods are well marketed and dividends flow to rural communities.

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