# Carbon sequestration on the subtropical dunes of South Africa: a comparison between native regenerating ecosystems and exotic plantations

by

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# Carbon sequestration on the subtropical dunes of South Africa: a comparison between native regenerating ecosystems and exotic plantations

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## Abstract

Rehabilitation and revegetation of mined coastal sand dunes on the east coast of South Africa makes sense. It recovers ecosystem services such as carbon sequestration. The outcome of rehabilitation, which covers a third of the mined area, is a secondary coastal dune forest similar to the forests in the region. The aim of revegetation, on the other hand, is to establish *Casuarina equisetifolia* J.R.Forst. & G.Forst. plantations on the remaining two thirds of the mined area, for the small-scale production of charcoal. The ratio of these two post-mining land use options has consequences for the carbon sequestration potential of the mined area.

As growth rate could be a reflection of carbon sequestration rate, this study compared the growth rate of *Acacia kosiensis* P.P.Swartz, the species that dominates rehabilitated stands, to that of *C. equisetifolia* in response to rainfall. The carbon sequestration potential of the two post-mining land use options was subsequently evaluated by measuring carbon storage in wood, the herb layer, litter and soil in different-aged stands. I also compared the financial potential of the two land use options.

Tree ring analysis could not be applied to the two species. No correlation between growth and rainfall could be found either. On average, *A. kosiensis* grew twice as fast as *C. equisetifolia*. Carbon storage in the wood, herb layer, litter and soil in rehabilitated stands of known age (7, 11, 17 and 21 years old) differed from the revegetated stands (8, 12, 16 and 19 years old). More carbon was stored in the revegetated stands than the rehabilitated stands. I attribute this primarily to the relatively larger wood and litter components of the former. *C. equisetifolia*, however, is harvested for charcoal production after about sixteen years of age, thus releasing most of the carbon stored in wood. The present ratio of rehabilitation to revegetation (1:2) is not optimal for long-term carbon sequestration.

Rehabilitation costs more, but the income potential thereof, as determined in this study, is less than that of revegetation. This, however, does not reflect the true financial potential of the two land use options. The financial analysis performed in this study only considered income from the sale of timber, charcoal and carbon credits. It excluded other potential

benefits of the rehabilitation of coastal dune forests. These may include the contribution to biological conservation and ecological services such as dune stabilisation and water purification. Coastal dune forests also provide habitat for a variety of organisms adapted to live in them. All these have a value. Their inclusion in a detailed cost-benefit analysis could render rehabilitation as the more financially efficient option.

## Acknowledgements

My supervisor, Rudi van Aarde and my co-supervisor, Theo Wassenaar tactfully steered me towards this exciting study. Little did I know how difficult it would be. Without their help this study would have been impossible. Prof always knew just when to push me and how hard; always encouraging me to beat my own best (or what I thought was my best. Oh, ye of little self-belief!). Many times when I felt like giving up, Theo would re-assure me that I could do it. For their unwavering belief in my ability to “kill it”, I give my sincerest thanks.

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The girls at CERU, my friends and, most of all, my partner would be my wings when my legs had trouble remembering how to walk. When consumed by self-doubt and had an overwhelming urge to drop out (yeah, I was often tempted to), they would encourage me to go on. My family were there with me every step of the way. Always believing in me and never complaining about the length of time I took to complete this study. I know their prayers carried me through. I thus dedicate this work to my father, my mother, my very many sisters and of course, to God, the Almighty Father who makes all things possible.

## Disclaimer

The work contained in this thesis is the original work of Phumza Ntshotsho (except where stated), done under the supervision of Professor Rudi van Aarde and co-supervised by Dr Theo Wassenaar. No part of this work has been previously submitted for a degree or for examination.

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# Chapter 1

## General introduction

### Why be bothered about carbon?

Climate change is driven by man-made changes in atmospheric conditions (UNFCCC, 1992; Karl & Trenberth, 1999; Middleton, 2000). A reduction in atmospheric carbon dioxide ($CO_2$) concentration through stimulating the build-up of biomass makes environmental sense. The Kyoto Protocol validates carbon sequestration in vegetation as a means to mitigate the increase in atmospheric $CO_2$ concentration (UNFCCC, 1997; Schlamadinger & Marland, 2000; IPCC, 2000). Although carbon sequestration in terrestrial stores may not be permanent, it does allow time for the development and implementation of other mitigation strategies. Terrestrial carbon sequestration thus provides opportunities for large $CO_2$ emitters like Richards Bay Minerals (RBM) to take responsibility and reduce the impact of their activities by recapturing their emissions.

### Mining and carbon release

RBM mines coastal sand dunes north of Richards Bay for various heavy minerals (Camp, 1990). The mining process (dredge mining) destroys coastal dune vegetation and disturbs the soil. This conceivably releases all stored carbon into the atmosphere as $CO_2$.

### Restoration and carbon recovery

Revegetation and rehabilitation follow mining (Scott *et al.*, 1994; van Aarde *et al.*, 1996a). The Tisand Lease contract, signed between RBM (the leaseholder) and the KwaZulu government (the landowner), dictates that RBM revegetates two thirds of the lease area with

*Casuarina equisetifolia* J.R.Forst. & G.Forst. and rehabilitates the remaining third with indigenous vegetation (Camp, 1990; van Aarde *et al.*, 1996a). This ratio approximates the state of the area before mining (~60% *Eucalyptus*, ~20% grassland and ~20% coastal dune forest; Camp, 1990). Post-mining rehabilitation and revegetation provide for the recovery of ecosystem services like carbon sequestration. Their ratio may have consequences for the amount of carbon that can be sequestered and stored in the lease area.

It is not known how much carbon is being put back in response to revegetation and rehabilitation. The rate at which this carbon recovery process is occurring is also unknown. There is a need to evaluate the carbon sequestration potential of the two alternatives (exotic revegetation and indigenous rehabilitation) so as to aid the optimisation of post-mining land use. Through this evaluation, the leaseholder may be advised whether to alter the ratio of exotics to indigenous vegetation in order to improve the total carbon sequestration potential of the lease area.

## What influences carbon sequestration?

The amount of carbon that a plant can sequester is limited by the rate at which photosynthesis can bind carbon into plant tissue (Catovsky *et al.*, 2002). This rate is, in turn, constrained by the availability of resources such as water, nutrients and sunlight (Hopkins, 1995; Lambers *et al.*, 1998). The availability of resources therefore determines the amount of biomass that accumulates, and hence the amount of carbon sequestered, per unit area and unit time. As part of this study I examined the effect of variability in rainfall on the growth rate, and presumably the carbon sequestration rate, of trees dominating the two post-mining land use options.

In addition to resource availability, the rate at which biomass accumulates in an ecosystem can be influenced by the plant species present. Increasing plant diversity has a positive asymptotic effect on ecosystem productivity (Fig. 1.1). Two of the hypotheses that have been proposed for this positive interaction are resource use efficiency and the sampling effect (Nijs & Impens, 2000). The resource use efficiency hypothesis proposes that, in diverse ecosystems, species could be complementary in resource uptake, either in time or space. This is sometimes referred to as niche complementarity (Tilman *et al.*, 2001). It allows such species assemblages to acquire more of a limiting resource (Loreau, 1998). The sampling effect hypothesis states that highly diverse ecosystems may be highly productive, simply because they have a higher probability of containing highly productive species (Aarssen, 1997; Tilman *et al.*, 1997).

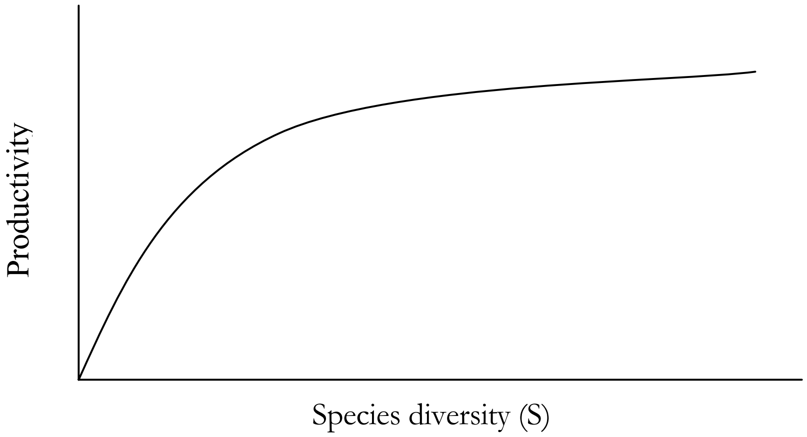

**Fig. 1.1.** Ecosystem productivity is a positive saturating function of plant species diversity. Adopted and modified from Tilman *et al.* (1997) and Loreau (1998).

Rehabilitation and revegetation give rise to vegetation types with different levels of biodiversity. Because the rehabilitated stands are more biologically diverse, I expect them to be more productive, and thus sequester more carbon than the exotic stands. Altering the ratio in favour of rehabilitation could thus enhance the carbon sequestration potential of the area, while also promoting biodiversity conservation.

## The economic consequences of rehabilitation and revegetation

Economics is usually an important factor in land use, and is a major influence in competition between potential land uses (Mather, 1986). Motivation to increase the portion of the land that is rehabilitated, in order to increase the carbon sequestration potential of the mined area, can be provided in terms of the financial benefits that can be accrued from this activity.

The Clean Development Mechanism (CDM, Article 12 of the UN Framework Convention on Climate Change) refers to, among others, land use projects that create carbon sinks (IPCC, 2001). CDM permits trade in carbon credits between industrial countries (sources) and developing countries (sinks of carbon) (Scholes, 2004). International trade in carbon credits has already begun (Haar & Haar, 2005; Klaassen *et al.*, 2005; www.pointcarbon.com), and South Africa can participate fully through its CDM office. The generation of profits through carbon trading could conceivably serve as an incentive for the enhancement of the carbon sequestration potential of the land.

In addition to the money potentially accruable from the sale of carbon credits, there are other potential income streams. These may be the sale of timber and non-timber forest products (medicinal plants, edible plants, etc.), as well as ecosystem services supplied by indigenous forests (e.g. water purification, reduction of soil erosion, etc.). All these can serve as additional incentives to increase the ratio of indigenous to exotic stands in the mining lease area. Profit generation by the charcoal industry, however, can serve as a perverse incentive for this alteration of ratios. A comparison of the financial costs and benefits of the two post-mining land use options might aid the leaseholder in deciding whether to alter the present ratio of rehabilitation to revegetation for maximum cost efficiency.

### Objectives, hypotheses and key questions

The general objectives of the present study were:

- To relate the growth rate of exotic *C. equisetifolia* and indigenous *A. kosiensis* to annual rainfall
- To compare the carbon sequestration potential of rehabilitation and revegetation
- To compare the financial potential of rehabilitation and revegetation

The above was achieved by testing the following hypotheses:

H1: The growth rate of *C. equisetifolia* is higher than that of *A. kosiensis* in response to rainfall

H2: The more biologically diverse rehabilitating stands have a higher carbon sequestration and storage potential than the *C. equisetifolia* stands

H3: The financial potential of the two land use options is not different

To test the hypotheses, I addressed these key questions:

- What is the growth rate of *A. kosiensis* and *C. equisetifolia* in response to rainfall?
- How much carbon is stored in the four carbon stores (wood, herb layer, litter and soil) in rehabilitated and revegetated stands?
- What is the cost of rehabilitating compared to that of revegetating and how does this compare to the income generating potential of either land use option?

In Chapters 3 and 4 I address the first question. Chapters 5 and 6 address the second and third questions, respectively.

## Chapter 2

# Study area and study species

### Study area

The study was conducted on a stretch of coastal sand dunes north of Richard's Bay (28°43′S, 32°12′E) and south of the Mapelane Nature Reserve (28°24′S 32°26′E). The area falls within the Maputaland Centre of Endemism (van Wyk, 1994). Van Aarde *et al.* (1996b) and Smith (2000) give detailed descriptions of the area and its climate. Richards Bay Minerals (RBM) holds a lease to mine the area for heavy minerals (Camp, 1990). A narrow strip of coastal dune forest directly adjacent to the sea is left unmined. The rehabilitation and revegetation programme that follows the mining activities results in a sequence of different-aged stands of indigenous vegetation on one third of the area, and beefwood (*Casuarina equisetifolia*) plantations on the remaining two thirds (Camp, 1990). The layout of the areas that have been rehabilitated and revegetated up to 2004, together with the unmined forest is shown in Fig. 2.1.

The rehabilitation/revegetation chronosequence, together with the unmined coastal dune forest represent varying degrees of biodiversity. The commercial stands (the beefwood plantations) are the least diverse, the indigenous stands having intermediate biodiversity and the unmined forest being the most diverse. The rehabilitated dune forests of Richards Bay are considered secondary forests (Camp, 1990) because of the lack of floral diversity and multiple strata characteristic of a climax forest (Smith, 2000). The plant diversity of the developing dune forest is relatively low, with the dominant tree species being *A. kosiensis*, interspersed with a few broadleaved forest tree species and herbaceous species. A description of the dominant tree species will be provided.

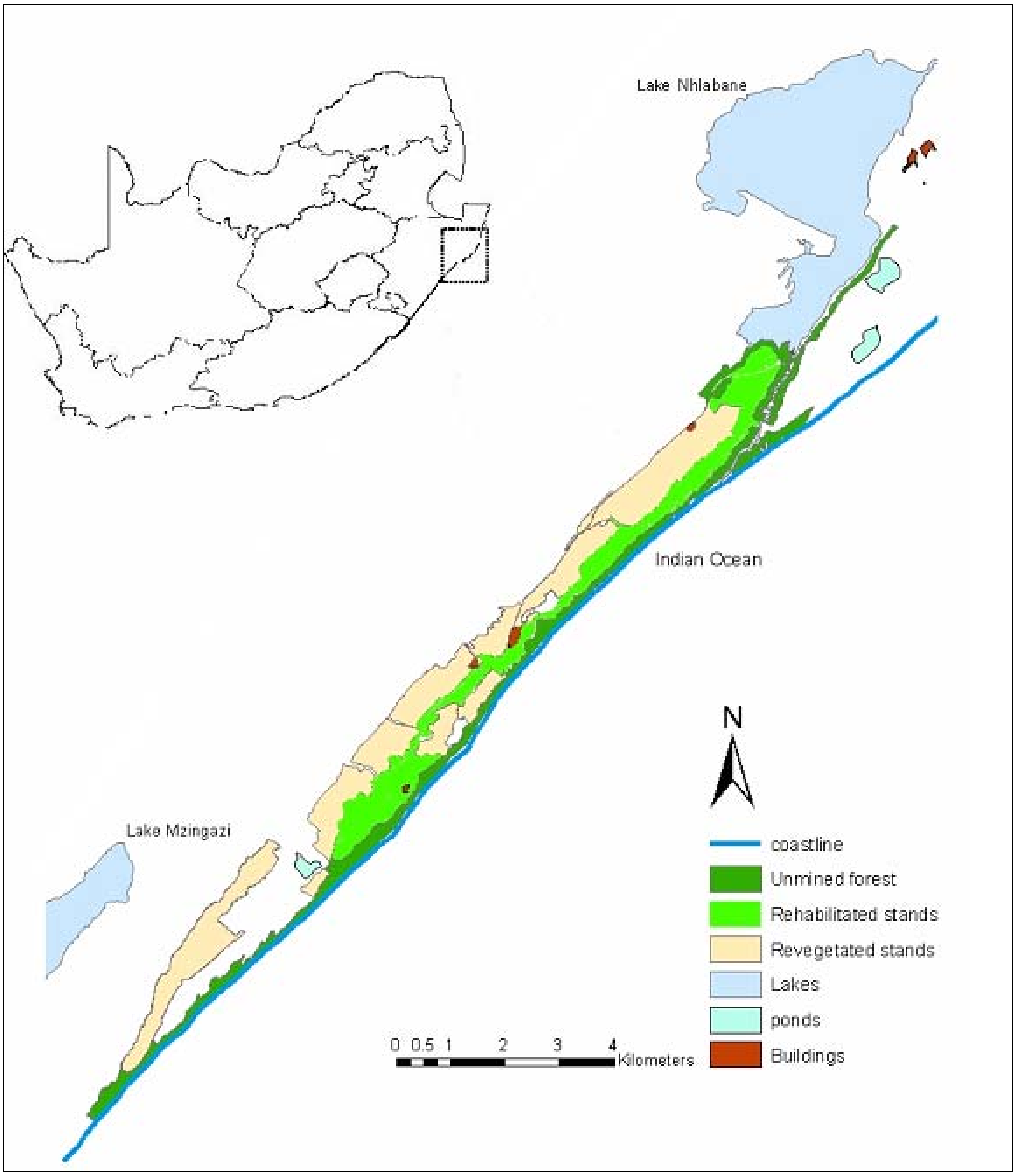

**Fig. 2.1.** A map denoting the location of the study area, the layout of rehabilitated and revegetated areas in relation to each other as well as the unmined forest. The map is based on information collected in 2004.

### Dominant tree species

Barnes *et al.* (1996) gave a description of a "dune form" of *A. karroo*, which has subsequently been named a species, *Acacia kosiensis* P.P.Swartz, in its own right (Coates Palgrave, 2002). *A. kosiensis* is a dune forest tree species belonging to the family Leguminosae, sub-family Mimosoideae. Because of the recent split from the more well-known *A. karroo*, little information exists in literature about *A. kosiensis*. This species is found along the east coast from the Tugela Mouth northwards into Mozambique. It is confined to a narrow belt along the coast on the coastal plain, among the coastal dunes and in the mouths of river estuaries (Barnes *et al.*, 1996). It usually forms dense pure stands and dominates in loose sand dunes and regenerating coastal forest. It is a large tree with long, slender stems, reaching as high as 17m in height (Coates Palgrave, 2002). The bark is greyish-white or light brown and smooth. *A. kosiensis*, as a leguminous species, has great potential to rehabilitate land and improve soil fertility (Gourlay & Barnes, 1996).

**Fig. 2.2.** A picture of a 19 year-old *A. kosiensis* tree.

*Casuarina equisetifolia* J.R.Forst. & G.Forst. is a native of Australia, Melanasia, Micronesia and Polynesia (Ndiaye *et al.*, 1993). It belongs to the family Casuarinaceae and is commonly known as beefwood, horsetail tree, she oak, Australian pine. It is a rough-barked, fast-growing tree with nearly erect or semi-spreading main branches and slim branchlets, as well as evergreen, needle-like foliage and woody cones (Elfers, 1988).

**Fig. 2.3.** A picture denoting the rough-barked, erect stem of a 16 year-old *C. equisetifolia* tree, in the foreground, and trees of similar age with evergreen foliage in the background.

*Casuarina equisetifolia* is also a nitrogen-fixing tree of considerable social, economic and environmental importance in tropical/subtropical littoral zones of Asia, the Pacific and Africa (Srivastava, 1995). It is commonly used in agroforestry systems, for soil stabilisation and reclamation work and in coastal protection and rehabilitation. It has a wide natural range and is one of the most extensively introduced tree species outside its natural range. It has been introduced into India, East and West Africa, the United States of America, the Caribbean, southern China, Vietnam, and some Middle East countries (Pinyopusarerk & Williams, 2000).

# Chapter 3

# Relating the growth rates of *Acacia kosiensis* and *Casuarina equisetifolia* to rainfall

## Introduction

The mining of coastal sand dunes north of Richards Bay is followed by forest rehabilitation and revegetation as different land use options (see Chapter 1). The sweet-thorn (*Acacia kosiensis*) and beefwood (*Casuarina equisetifolia*) dominate the areas that have been earmarked for the respective land use options. These species may grow at different rates. Therefore, their potential to sequester carbon may differ, even when exposed to similar conditions such as rainfall. It is thus worthwhile to evaluate the growth performance of the two species in relation to rainfall.

Water is an important resource for plant growth (Ting, 1982; Taiz & Zeiger, 1991). It limits physiological processes such as photosynthesis (Hopkins, 1995; Lambers *et al.*, 1998), which in turn determine growth rates. More accurately, water limits plant growth through its influence on effective soil moisture. Effective soil moisture content is the amount of available subsurface water coming from all sources, corrected for losses through evaporation and runoff (Stokes & Smiley, 1968). Plants respond to decreasing effective soil moisture by continuing to grow roots but decreasing the growth of shoots (Kramer & Boyer, 1995). Shoots grow by increasing in height (apical growth) and breadth (radial growth). It is this radial growth that is the basis of dendrochronology (the use of growth rings to determine the age of wood and to obtain information on tree growth responses to variations in precipitation; Lilly, 1977; Schweingruber, 1996). It follows that variation in precipitation, and subsequently in effective soil moisture, should manifest as variation in plant growth.

In temperate regions where there is a distinct growing season, growth is strongly periodic (Lilly, 1977; Gourlay, 1995). In comparison, not all trees from tropical and subtropical regions form distinct growth rings due to the apparent lack of a precipitation pattern (Stokes & Smiley, 1968). Consequently, dendrochronological studies in tropical and subtropical regions are not as common as those in temperate regions.

A growth ring is the layer of wood added to a tree during a single growing season (Steenkamp, 2000). The annual ring is divided into two parts, earlywood and latewood (Lilly, 1977). It is the sharp contrast between the last-formed latewood cells of one growing season and the first-formed earlywood cells of the following growing season that delineates the boundary of an annual ring (Stokes & Smiley, 1968). By counting the number of growth rings, one can theoretically determine the age of a tree. Furthermore, by measuring the width of the rings, one can determine the amount (and hence rate) of growth in one season.

As part of my study I tested the hypothesis that the growth of *A. kosiensis* and *C. equisetifolia* is determined by rainfall. Additionally, I proposed that the two species respond differently to the limitation posed by rainfall and this would be reflected in growth ring widths. That is, a species that is able to maximise growth despite the limiting factor would have wider growth rings than one that suffered stunted growth because of the same limiting factor. The differences in growth rate would then be a reflection of differences in the species' potential to sequester carbon. The aim of this study was to relate the growth rate of known-aged *A. kosiensis* and *C. equisetifolia* to past rainfall events in the study area using dendrochronological techniques.

## Materials and methods

I collected stem disks between January and February 2004 from four stands of *A. kosiensis* (aged 7, 11, 17 and 21 years) and four stands of *C. equisetifolia* (aged 8, 12, 16 and 19 years). I cut a disk (~ 5-10 cm thick) of the stem at ankle height (~ 10 cm above ground level) from four randomly selected trees in each of the stands. I dried the disks to constant weight in an oven set at 70°C and then used a hand-held belt sander to smoothen the surfaces, starting with course grit (60 grain), followed by medium grit (80 grain). I used an industrial belt sander (using 80, 100 and 120 grain, successively) to further smoothen the surfaces and an orbital sander to polish them (gradually increasing grain size from 120, 340, 600, 800 and ultimately to 1200). I removed excess dust from the disks before viewing them using a dissecting microscope at 2.4X, 4X, 6X and 40X magnification. The disks were also examined microscopically by an experienced wood anatomist[1]. Images of the disks were taken with a Fujifilm Fine Pix S2Pro digital camera (Fuji Photo Film, Inc., USA). I viewed the digital images using "Imaging for Windows" (Eastman Software, Inc., USA).

I measured the diameter of each of the disks (3 measurements on each) and calculated a weighted average diameter for each age class according to the equation $\overline{Y}_w = (\overset{n}{\Sigma} w_i Y_i)/(\overset{n}{\Sigma} w_i)$ (Sokal & Rohlf, 1995); where $n$ is the number of disks in each age class, $w_i$ is the number of measurements on each disk and $Y_i$ is the average diameter of each disk. I then calculated the radius for each age class and delineated these on the disks and their digital images to indicate the extent of radial growth during periods that correspond to the ages of trees. For example, on a 21-year old *Acacia* disk I would have four points along one radius, each one corresponding to either 7, 11, 17 or 21 years of growth (Fig. 3.1). I then counted the number

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[1] Ms C. Steenkamp (Senior lecturer, Department of Land Management, Polytechnic of Namibia, Windhoek, Namibia) is experienced in dating *Acacia* species using growth ring analysis.

of rings within each growth period along four radii on each disk. I did linear regression analysis to relate the growth ring counts to chronological age in years.

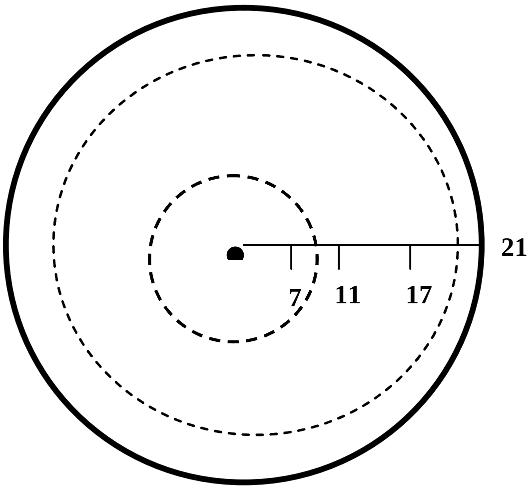


**Fig. 3.1.** A schematic representation of the method used to count growth rings (shown as broken lines on the diagram) within known growth periods (shown as numbers denoting age in years). The pith is shown as the dark dot in the centre and the bark is shown as a dark solid line. To prevent clutter, only two hypothetical growth rings are shown.

I obtained rainfall data from the mining, planning and rehabilitation office of RBM. To relate growth rate to rainfall history I calculated the cumulative deficit and surplus rainfall between 1983 and 2001. I measured diameter at breast height (DBH) of 80 trees from each of known-aged stands (7, 8, 10, 11, 12, 17 and 21 years for *A. kosiensis* and 4, 6, 8, 10, 12, 14, 16 and 19 years for *C. equisetifolia*). I then calculated the growth rate of each tree using the equation:

growth rate (cm.year$^{-1}$) = diameter at breast height (cm) / age (years)

I plotted the growth rate against cumulative deficit/surplus rainfall values for the years corresponding to the years in which the trees were established. I expected that trees established during years of below average rainfall would grow slower than those established during years with surplus rainfall. The trend in tree growth rates therefore would follow that of cumulative deficit and surplus rainfall.

**Results**

Apparent incremental growth lines were evident on cross-sections of all the stems of *A. kosiensis* and *C. equisetifolia* that I prepared. For both species it was difficult to distinguish anatomical features to which a chronological age could be linked. A digital image of a section of a polished stem disk of *A. kosiensis* (Fig. 3.2) shows the numerous white lines (black arrow on figure points to one such line) that denote aliform parenchyma. The black markings denote the location of marginal parenchyma and were taken to represent “growth rings. Some of these lines are discontinuous, as illustrated by the unequal number of markings on the opposite sides of the disk.

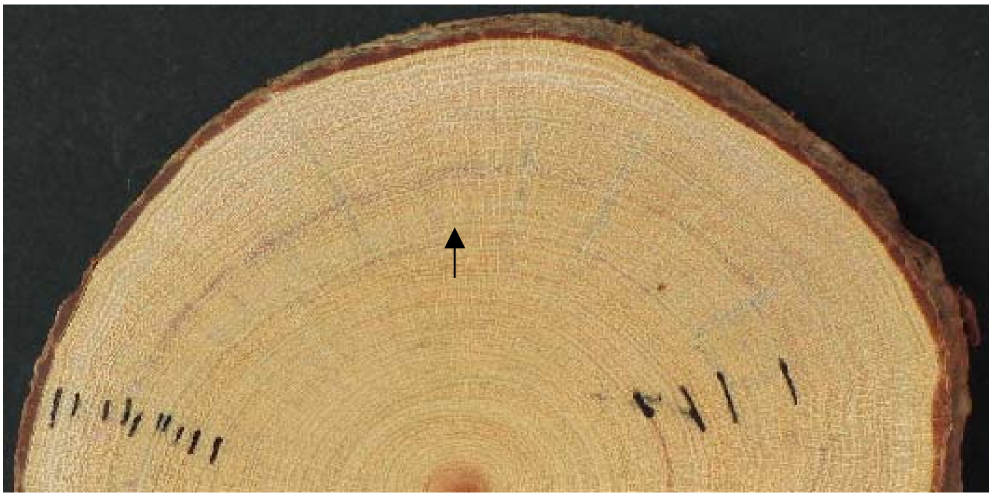


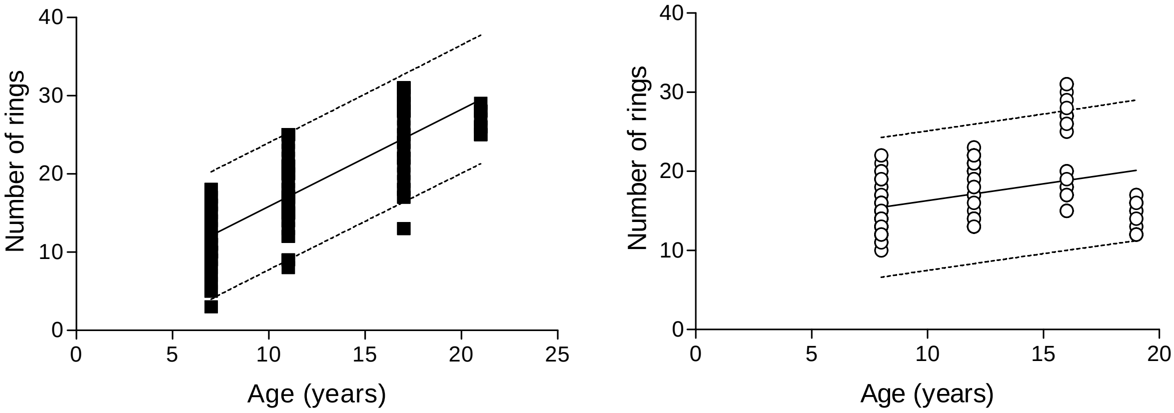
**Fig. 3.2.** A digital image of a section (pith to bark) of a polished 12 year-old *A. kosiensis* stem disk. The black lines denote the location of marginal parenchyma identified under a dissecting microscope. The arrow points to a line that denotes aliform parenchyma.

There was no evidence of marginal parenchyma on the cross-sections of beefwood stem disks. Some darker “lines” (black arrows on Fig. 3.3), which could be latewood, were visible, but these were discontinuous. Moreover, some of the lines were too close to each other and could therefore not indicate annual rings.

**Fig. 3.3.** A digital image of a section (pith to bark) of a polished 8 year-old *C. equisetifolia* stem disk. The black arrows point to two of many dark lines that could denote latewood. All such lines may denote the end of consecutive growth seasons.

The number of apparent incremental growth rings varied from 3 to 18 in the 7 year-old *A. kosiensis* samples and this trend of high variability in the number of counted rings was similar for all wood samples <21 years old (Fig. 3.4a). There was less variability in the number of counted rings on the 21 year-old wood samples and the counts were consistently higher than the known age. For *C. equisetifolia*, the general trend was that of more lines counted than the known chronological age for samples from trees <16 years of age and fewer lines than age for trees >16 years of age (Fig. 3.4b). On disks from 16-year old trees I counted either fewer or more lines than expected from their age. The regression equations for *A. kosiensis* and *C. equisetifolia* were: $y = 1.20x + 3.4$ and $y = 0.43x + 12$; respectively.

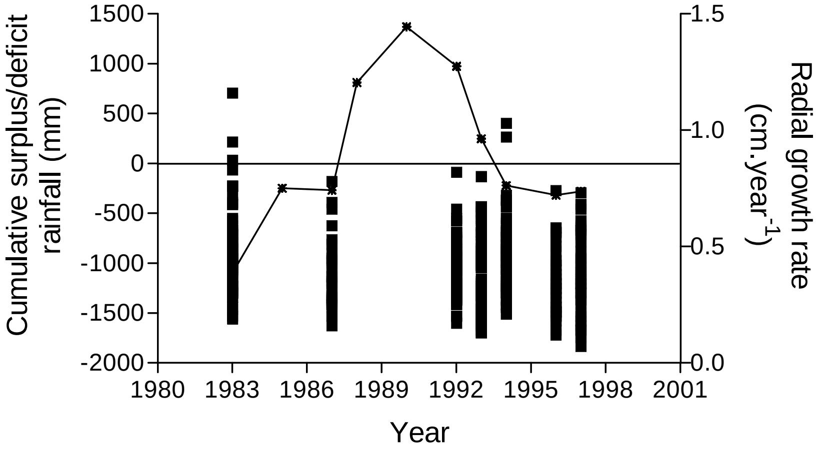
The deviation of the slopes of the regression lines from 1 suggests that the rings identified in this study are not annual. Age accounted for 64% and 13% of the variability in the number of apparent growth rings on *A. kosiensis* and *C. equisetifolia* stem disks, respectively.



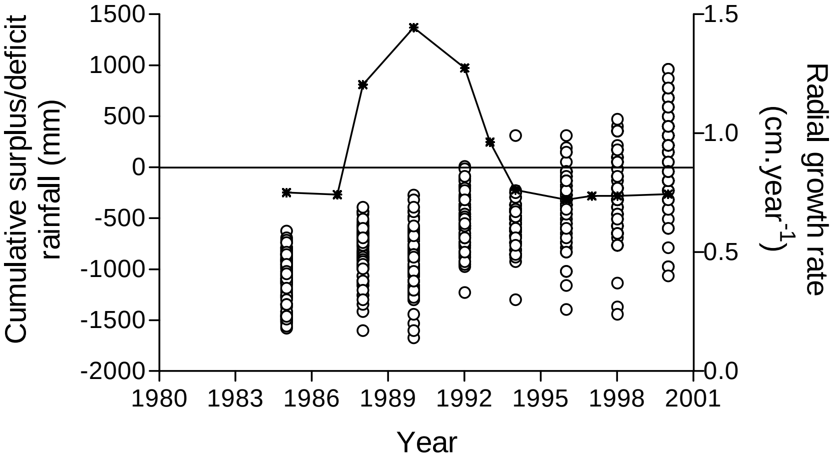
(a) (b)

**Fig. 3.4.** The number of apparent growth rings counted on the cross-sections of (a) *A. kosiensis* and (b) *C. equisetifolia* as a function of age. The linear regression lines, shown with 95% prediction intervals are $y = 1.20x + 3.4$ and $y = 0.43x + 12$, with $r^2 = 0.64$ and 0.13 for *A. kosiensis* and *C. equisetifolia*, respectively.

The growth rate of trees established within the same year varied for both species (Figs 3.5a & b). The growth rate of *A. kosiensis* decreased slightly from 0.48 ± 0.02 cm.year$^{-1}$ for the trees established in 1983, to 0.38 ± 0.02 cm.year$^{-1}$ for those established in 1997 (Fig. 3.5a). The growth rate of *C. equisetifolia* increased progressively from 2.4 ± 0.08 cm.year$^{-1}$ for the oldest trees to 5.83 ± 0.14 cm.year$^{-1}$ for the youngest trees (Fig. 3.5b). Rainfall was below average prior to 1987 and after 1993. This was interrupted by a period of surplus rainfall. The trend in radial growth rate of both species did not reflect this unimodal trend in rainfall.



(a)



(b)

**Fig. 3.5.** The relationship between radial growth rate (plotted on the right y-axis) of (a) *A. kosiensis* (n = 560) and (b) *C. equisetifolia* (n = 640) and cumulative rainfall surplus/deficit (asterisks and line plotted on left y-axis).

## Discussion

The lack of distinct annual rings on the sweet-thorn and beefwood samples examined in this study suggests that these species from Richards Bay may not be suitable for dendrochronological studies. Tree-ring analysis is a problem for southern African tree species (Lilly, 1977; Steenkamp, 2000). The difficulty results from a lack of distinct ring boundaries that correspond to seasonal climatic extremes. Thus, even though *C. equisetifolia* is not a southern African species, the trees I sampled may not be suitable for dendrochronology, probably because they were exposed to a subtropical climate.

For the successful application of dendrochronological techniques, four conditions must be fulfilled (Stokes & Smiley, 1968), viz.,

(1) Trees used for dating purposes must add only one ring for each growing season,

(2) Although the total seasonal growth is the result of many interacting factors, such as genetics and environment, only one environmental factor must dominate in limiting the growth,

(3) This growth-limiting factor must vary in intensity from season to season and the resulting growth rings must faithfully reflect such variation in their width,

(4) The variable growth-limiting factor must be uniformly effective over a large geographical area.

My results showed that the first three conditions were not fulfilled in this study. I could not ascertain the seasonal nature of the rings. Neither could I demonstrate that rainfall is indeed a growth-limiting factor. Mean annual rainfall in the region averages 1575 ± 107mm (van Dyk, 1996), which may be sufficient to maintain effective soil moisture. Stokes & Smiley (1968) emphasised that if effective soil moisture is sufficient in most years for a tree to produce optimum growth, the ring structure becomes indistinct. That is, there is insufficient variation in ring widths to produce any recognizable sequence. Dendrochronological techniques clearly cannot be applied reliably to trees growing under such conditions.

Gourlay & Barnes (1994) and Gourlay (1995) correlated growth zones on *A. karroo* samples from Richards Bay to peaks in rainfall using X-ray scanning electron microscopy, in addition to visual assessment. Martin & Moss (1997), Eshete & Stahl (1999) and Steenkamp (2000) have also successfully applied dendrochronology to other *Acacia* species. All of them agreed on the difficulty of distinguishing growth zones in *Acacia* species.

The lack of a relationship between growth rate of the two species and rainfall suggests that rainfall does not limit the growth of either of the two species on the coastal dunes of Richards Bay. Thus, the hypothesis stated earlier is rejected. If there is any limitation on growth, it may be a result of constraints posed by other environmental factors. These may include competition between neighbouring plants, temperature, light and nutrient availability. Moreover, the growth rates I calculated here may not reflect the true growth rates of the species. The method I used to estimate tree growth rate is crude. It assumes a linear relationship between tree size and age. Growth rate may vary at different stages of growth under the influence of various environmental variables such as aspect, slope, crowding, etc. Consequently, non-linear growth models that incorporate such environmental variables have been developed for various species (e.g. Beck, 1974; Ndiaye *et al.*, 1993; Monserud & Sterba, 1996; Zhang *et al.*, 1996; Lee *et al.*, 2004).

If the rate of carbon sequestration by *A. kosiensis* and *C. equisetifolia* is to be deduced from their respective growth rates, then a more accurate estimate of growth rates is needed. This requires the development of a growth model that includes variables that may potentially affect growth. In Chapter 4 I develop such growth models for the two species.

## Chapter 4

# A comparison of the radial growth of *Acacia kosiensis* and *Casuarina equisetifolia*

### Introduction

As a tree grows, its trunk and branches do not only get longer, but also thicker (McMahon, 1975). Thus, tree growth can be viewed as lateral growth (radial increment), vertical growth (height increment) or a combination of the two. Increase in diameter at breast height (DBH) is often used to measure tree growth (Lee *et al*., 2004). DBH is related to tree size (volume or biomass) through a logarithmic function (see van Dyk, 1996). DBH, and hence circumference, can be used as a direct index of size or biomass. However, there is some uncertainty about how to deal with multi-stemmed trees. Trees with more than one stem can be seen as representing several trees of different circumferences. On the other hand, multiple stems could represent one large tree with a circumference equal to the circumference of a stem with a total surface area at breast height that is the sum of all separate stems. This subject receives remarkably little attention in the literature. Even basic textbooks on tree growth do not explicitly deal with the measurement of growth in multi-stemmed trees. In this study I work with the circumference of only the largest stem, under the assumption that the growth rate of that particular stem reflects the maximum growth rate of the tree.

The aim of this part of my study was to compare the radial growth rates of *A. kosiensis* and *C. equisetifolia* in order to compare their carbon sequestration potential. I tested the hypothesis that the radial growth rate of *A. kosiensis* is less than that of *C. equisetifolia*. I based this hypothesis on the expectation that *C. equisetifolia* maximizes growth as it does not compete with other species for resources.

As a first step, I developed growth models based on age. Because tree growth is not only a function of tree age, I considered other confounding factors (multi-stemmedness, total tree size and environmental variables) to improve the abilities of the models to describe growth in these species.

## Materials and Methods

I measured 560 *A. kosiensis* trees from different known-aged stands (7, 8, 10, 11, 12, 17 and 21 years; 80 trees in each stand) and 640 *C. equisetifolia* trees (aged 4, 6, 8, 10, 12, 14, 16 and 19; 80 individuals in each age class). Sampling took place at five random points using the Point-Centred Quarter method (Cottam & Curtis, 1956) along four transects of ~200m, about 50m apart in each of the stands. For each tree I recorded the total number of live stems and measured stem circumferences at breast height (1.2 m above ground). I then computed a composite stem circumference for each multi-stemmed tree (Appendix 1). I recorded a GPS reading at each point in the *A. kosiensis* stands and only at the beginning of the first transect in the *C. equisetifolia* stands. These data were superimposed in ArcView GIS Version 3.3 (Environmental Systems Research Institute, Inc. USA) on a Digital Terrain Model of the study area to calculate elevation, slope and distance from the nearest edge for each sampling point in the *A. kosiensis* stands. (The Digital Terrain Model did not include the areas planted with *C. equisetifolia*, consequently the elevation, slope and distance from edge could not be calculated for sample points in these stands).

To compare the growth rate of the two species I regressed log-transformed circumference on age using Graphpad Prism Version 3 (GraphPad Software, Inc. USA). To further improve the growth models, I used stepwise multiple regression in STATISTICA Version 6 (StatSoft, Inc. USA) to relate circumference growth to age, number of live stems, composite circumference, elevation, distance from edge and slope, where applicable.

I also used some circumference data for *A. kosiensis* from the CERU database. These data (1376 values taken from six stands) were collected in 1999 in a similar manner to mine, and could thus be analysed for comparison between the two datasets.

## Results

### *A. kosiensis growth*

The untransformed data did not satisfy the assumptions of normality and homoscedasticity. Transformation of the data, however, did not improve the fit of the model ($R^2 = 0.54$ for log-transformed data versus $R^2 = 0.59$ for untransformed data). Hence I report the results of the untransformed data. Up to the age of 21 years, *A. kosiensis* growth follows an exponential curve (Fig. 4.1a), described by the equation $y = 9.66e^{0.089x}$, where $y$ is stem circumference and $x$ is age. This species grows slowly during the first eight years, after which the growth rate progressively increases. The data collected in 1999 fit the exponential growth model better and predicted slightly slower growth for the species, as described by the equation $y = 12.88e^{0.078x}$ (Fig. 4.1b, $R^2 = 0.65$).

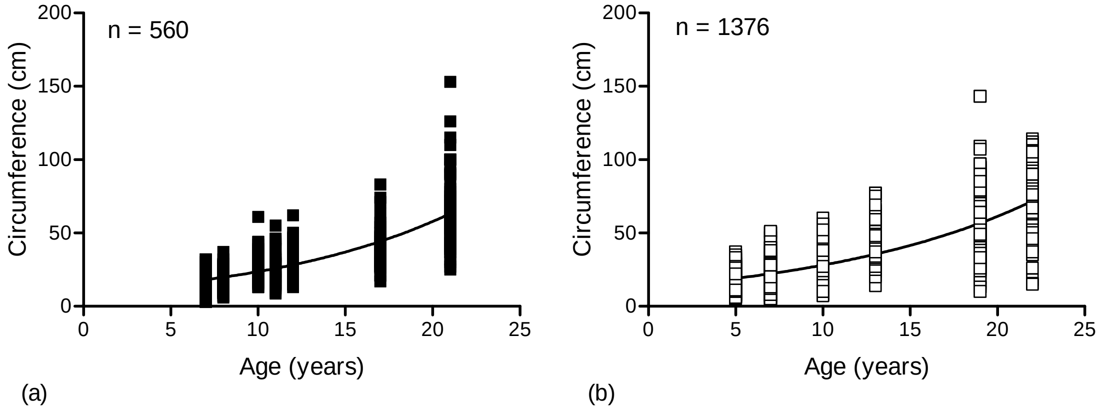


**Fig. 4.1.** Exponential growth curve for *A. kosiensis* based on (a) the 2004 and (b) the 1999 datasets. The curves, which show the change in circumference (y-axis) with increasing age (x-axis), are described by the equations $y = 9.66e^{0.089x}$ and $y = 12.88e^{0.078x}$.

### *C. equisetifolia growth*

The best-fit model for *C. equisetifolia* was a Boltzman sigmoidal growth curve (Fig. 4.2), described by the function $y = 2.98 + (3.81 - 2.98)/(1 + e^{(6.84 - x)/1.80})$. Log transformation of the data to satisfy the assumption of homoscedasticity marginally improved the fit of data to the model ($R^2 = 0.48$ for untransformed versus $R^2 = 0.49$ for log-transformed data). The runs test also supported this observation ($P = 0.045$ and 0.18, untransformed and transformed data respectively). Based on this model early growth is rapid (Fig. 4.2). DBH in this species continues to increase until about the age of 13 years.

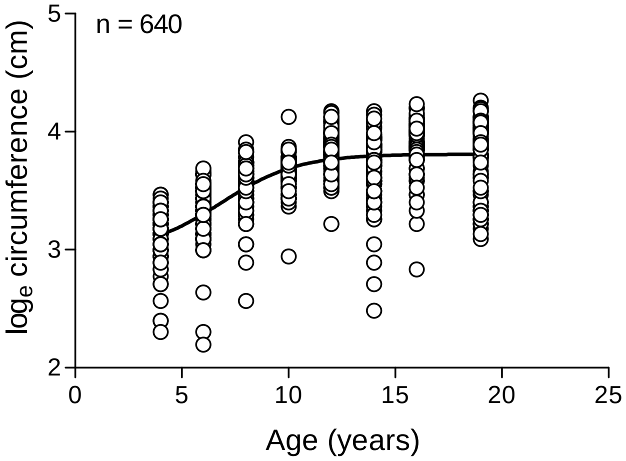


**Fig. 4.2.** Sigmoidal growth curve for *C. equisetifolia* showing the change in circumference (y-axis) with age (x-axis). The curve is described by the equation $y = 2.98 + (3.81 - 2.98)/(1 + e^{(6.84 - x)/1.80})$.

### *A comparison of the growth rate of A. kosiensis and C. equisetifolia*

For comparison the growth of the two species could be described in terms of the linear regression equations:

$y = 2.13 + 0.096x$; for *A. kosiensis* and

$y = 3.10 + 0.045x$; for *C. equisetifolia*

The corresponding $r^2$ values are 0.55 and 0.38, respectively.

Despite the weakness of the curve fits, the following could be deduced: On average, the growth rate in DBH of *A. kosiensis* was more than twice as high as that of *C. equisetifolia* (Fig. 4.3). The difference in growth rates was significant at $P < 0.0001$

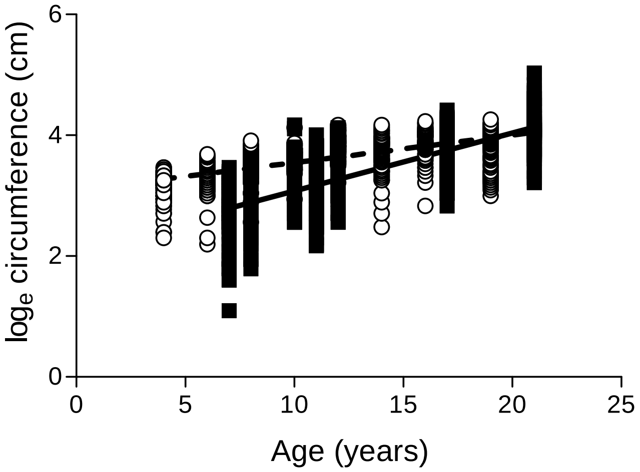


**Fig. 4.3.** Linearized growth curves to compare the growth of *A. kosiensis* (solid squares and solid line, n = 560) and *C. equisetifolia* (open circles and broken line, n = 640). Least squares linear regression analysis was used to determine growth rates. The regression lines are described by the functions: $y = 2.13 + 0.096x$; for *A. kosiensis* and $y = 3.10 + 0.045x$; for *C. equisetifolia*.

*Determinants of growth in A. kosiensis and C. equisetifolia*

At first glance, multi-stemmedness was not common in either *A. kosiensis* or *C. equisetifolia* (Fig. 4.4). More than 70% of the sampled *Acacia* trees and more than 90% of the *Casuarinas* were single stemmed. However, the multi-stemmed trees accounted for approximately 63% and 12% of the total biomass in *A. kosiensis* and *C. equisetifolia* stands, respectively (Appendix 2).

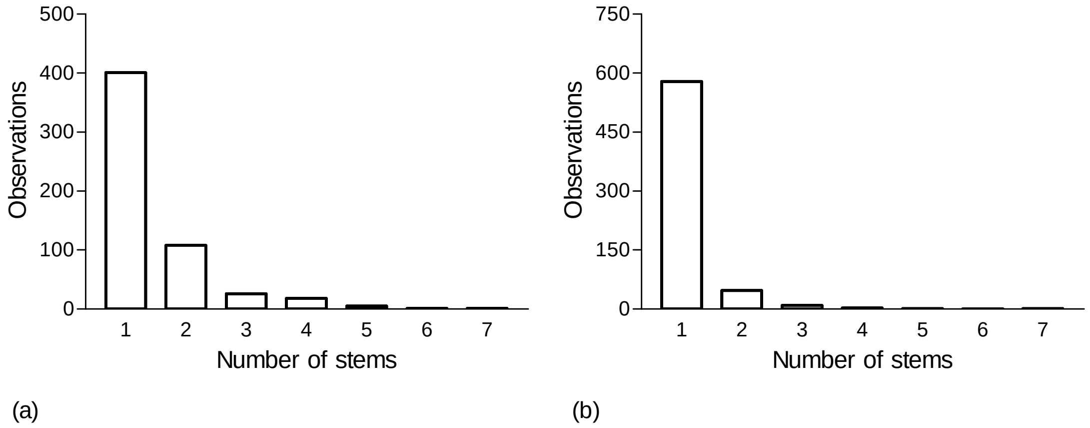


**Fig. 4.4.** The number of trees with different numbers of live stems on (a) *A. kosiensis* (n = 560) and (b) *C. equisetifolia* (n = 640) trees. The majority of the trees are single-stemmed in both instances.

A t-test showed that for *C. equisetifolia,* there was no difference between the circumference of the single-stemmed trees and the circumference of the largest stems of multi-stemmed trees in six out of eight stands (6, 8, 10, 14, 16 and 19 year-old stands). In the remaining two stands (4 and 12 year-old stands), the largest stems of the multi-stemmed trees were significantly smaller than the single stems of the single-stemmed trees (Table 4.1). A t-test for *A. kosiensis* showed that multi-stemmed trees had larger circumference values for the biggest stems than the single-stemmed trees in all but two of the sampled age classes (10 and 17 year-old stands). In the 10 and 17 year-old stands the largest stems of the multi-stemmed trees were not significantly bigger than the single-stemmed trees (Table 4.1).

**Table 4.1.** Results of t-tests comparing the circumference of the largest stem of multi-stemmed trees to that of single-stemmed trees in *A. kosiensis* and *C. equisetifolia* stands of various ages. The mean and standard deviation are shown for single-stemmed and multi-stemmed trees in each age class. Sample sizes are shown in parentheses.

| ***Acacia kosiensis*** | | | | ***Casuarina equisetifolia*** | | | |
|---|---|---|---|---|---|---|---|
| Stand age | Single-stemmed $\bar{x}$ ± SD (n) | Multi-stemmed $\bar{x}$ ± SD (n) | *P*-value | Stand age | Single-stemmed $\bar{x}$ ± SD (n) | Multi-stemmed $\bar{x}$ ± SD (n) | *P*-value |
| 7 | 13.51 ± 6.90 (44) | 20.32 ± 6.02 (36) | < 0.01 | 4 | 23.66 ± 4.61 (74) | 19.33 ± 6.59 (6) | 0.035 |
| 8 | 15.33 ± 5.79 (51) | 20.17 ± 7.29 (29) | <0.01 | 6 | 27.77 ± 6.11 (77) | 30.33 ± 4.51 (3) | NS |
| 10 | 28.34 ± 8.82 (74) | 33.33 ± 16.86 (6) | NS | 8 | 35.26 ± 6.62 (74) | 34.00 ± 5.44 (6) | NS |
| 11 | 25.60 ± 10.66 (65) | 32.85 ± 9.26 (15) | 0.02 | 10 | 38.53 ± 5.53 (75) | 40.60 ± 6.47 (5) | NS |
| 12 | 27.12 ± 7.84 (52) | 34.96 ± 8.43 (28) | <0.01 | 12 | 48.39 ± 8.23 (62) | 44.11 ± 6.74 (15) | 0.047 |
| 17 | 41.79 ± 14.30 (61) | 46.84 ± 14.49 (19) | NS | 14 | 42.68 ± 10.64 (78) | 28.00 ± 2.83 (2) | NS |
| 21 | 56.02 ± 16.92 (54) | 78.23 ± 26.88 (26) | <0.01 | 16 | 49.83 ± 10.46 (76) | 49.50 ± 6.45 (4) | NS |
| | | | | 19 | 44.89 ± 13.94 (63) | 48.41 ± 10.65 (17) | NS |

Based on the information in Fig. 4.4 and Table 4.1 above, the number and total mass of extra stems may influence the growth in circumference of the largest stem of *A. kosiensis* substantially, while exerting little or no influence on the growth of *C. equisetifolia*.

The addition of the two factors (number of stems per tree and composite circumference) in forward stepwise multiple regression analyses improved the fit of the model (from $r^2 = 0.69$ to multiple $R^2 = 0.73$ for *A. kosiensis*). Thus, 73% of the variability in *A. kosiensis* stem circumference was attributable to the combined effects of age, number of stems and stem size. Unsurprisingly, the final step of the regression model for *C. equisetifolia* excluded the effect of the number of stems and composite circumference, and only included the effect of age. The final multiple $R^2$ of 0.39 was a slight improvement on the $r^2$ of 0.38 obtained for the simple linear regression model. The equation for *A. kosiensis* was:

$\hat{Y} = \mathrm{a} + \mathrm{b}\,X_1 + \mathrm{b}\,X_2 + \mathrm{b}\,X_3$

$= 24.23 + 0.14\,X_1 + 0.56\,X_2 - 0.0063\,X_3$

with $R^2 = 0.73$, where a = Y-intercept; $X_1$ = Age, with ß = 0.62; $X_2$ = Composite circumference, with ß = 0.51; $X_3$ = Number of stems, with ß = -0.14

The equation for *C. equisetifolia* was

$\hat{Y} = \mathrm{a} + \mathrm{b}X_1$;

$= -91.46 + 8.54X_1$

with $R^2 = 0.39$, where $X_1$ = Age; ß = 0.62

For both species, the strongest predictor of tree size was age (disproportionately large ß values). For *A. kosiensis*, composite stem circumference had the second largest influence on tree size, followed by the number of stems, which had a negative ß value, suggesting that the presence of additional stems on a single tree hampers growth.

*Additional environmental variables*

The further inclusion of environmental predictor variables (distance from an edge, elevation, aspect and slope) did not significantly improve the fit of the *A. kosiensis* regression model (final multiple $R^2 = 0.75$). The final regression model excluded elevation, aspect and slope, ending up with four predictor variables for circumference. The final equation for *A. kosiensis* was thus:

$$\hat{Y} = a + b X_1 + b X_2 + b X_3 + b X_4$$

$$= 1.06 + 0.16 X_1 + 0.55 X_2 - 0.0055 X_3 + 0.25 X_4$$

with $R^2 = 0.75$

where $X_1$ = Age, with ß = 0.68; $X_2$ = Composite circumference, with ß = 0.49; $X_3$ = Number of stems, with ß = -0.13; $X_4$ = Distance from an edge, with ß = 0.10

The final equation shows that the additional environmental variable (distance from an edge) has a weak positive effect on the radial growth of *A. kosiensis*. It is apparent even in this final model that age and composite stem circumference have the strongest predictive power, whereas multi-stemmedness influences growth negatively.

## Discussion

Age accounted for about half the variation in the circumference of *A. kosiensis* and *C. equisetifolia* trees. My data suggest that *A. kosiensis* grows exponentially and continues to grow even at 21 years of age. Analysis of data collected previously at the same site (1999 dataset) also yielded an exponential growth model. Though I have not yet come across a study that describes the growth function of *C. equisetifolia*, Rockwood *et al.* (1983) found that this species grows rapidly initially, with height increments exceeding 1.5 m per year found commonly. My data suggest that growth in girth of *C. equisetifolia* is also rapid in the first few years of growth, levelling off at about 13 years. This is slightly lower than the age of

16 years quoted in literature (Midgley *et al.*, 1983) as the age at which *C. equisetifolia* stops growing. Overall, age seems to be a good predictor of radial growth for both species.

The species conform to different growth models. Comparison of their growth rates through linear regression suggests that *A. kosiensis* grows faster than *C. equisetifolia*, under the same environmental conditions. Differences in tree density may be partly responsible for these differences in growth rate. Studies in young stands have shown that initial spacing has significant influence on tree growth (Zhang, *et al.*, 1996). Neighbouring plants generally compete for limiting resources (Damgaard, 2004). Competition is a reciprocal negative interaction between two organisms, which may arise either through direct interference or indirect exploitation of shared resources (Connell, 1990). Planting density should thus affect the intensity of competition. For example, in dense stands growth may be hampered as a result of intense competition. Notice the slow growth of *A. kosiensis* (Fig. 4.1) during the early years of growth when the trees are crowded, compared to the faster growth beyond 15 years when the canopy is opening up (see Chapter 5 for tree densities). This indeed seems to serve as further support for the idea that crowding inhibits growth. It would be an interesting exercise to compare tree growth in permanent sampling plots of different tree densities.

A tree may be pre-disposed to growing faster than another one simply as a consequence of its genetic makeup. For example, *C. equisetifolia* is a commercial species and may possibly have undergone generations of selection for maximal growth in the shortest time possible. But because there is an upper limit to the size to which a tree can grow (Ryan & Yoder, 1997), *C. equisetifolia* stops growing at a relatively young age, when it reaches this upper size limit. *A. kosiensis*, on the other hand, has not been selected for optimal yield. It may thus grow for a longer period, at an ever-increasing rate, and only reach the upper size limit at an older age.

Plant productivity is limited by water and phosphorus availability in most terrestrial ecosystems (Ho *et al.*, 2004). The root system (tap root versus fibrous root) may confer a growth advantage through the increased efficiency in resource absorption. In general, fibrous root systems are relatively shallow, whereas tap root systems may penetrate the soil to considerable depths (Robbins *et al.*, 1959), thereby gaining access to water and minerals that may be inaccessible to species with a fibrous root system. Depending on site conditions (shallow versus deep water table), *C. equisetifolia* can either have a deep taproot or long horizontal roots (Yadav, 1983). In sandy areas where the water table tends to fluctuate widely (as is likely the case in Richards Bay), the plant develops a deep taproot. *A. karroo* (a species of which *A. kosiensis* is a form) has a deep taproot (Barnes *et al.*, 1996). Thus, similarity of the root systems excludes rooting behaviour as a possible contributor to the difference in growth rates.

The growth form or architecture of trees is of high importance as it affects height gain, light interception, defence and reproduction (Archibald & Bond, 2003; Kuppers, 1989). These in turn affect the competitive advantage and fitness of trees (Farnsworth & Niklas, 1995). A highly ramified shoot has a larger photosynthetic area than an unbranched shoot (Archibald & Bond, 2003). Competition for light may therefore be a selective force for multi-stemmedness in a forest environment. In this study I found that the presence of many stems on a single tree has a slightly negative effect on the growth of *A. kosiensis* (as measured by the increase in circumference of the largest stem). It would thus seem that promoting multi-stemmedness has trade-offs. The highly ramified shoot optimises light interception, which in turn promotes growth. But this is negated by competition between sibling stems, which grow sub-optimally, thus negatively affecting the growth of the entire tree. Multi-stemmedness therefore has two components, the number of extra stems and the size of those extra stems, which act in opposite directions to affect tree growth.

Distance from an edge had a slightly positive effect on the growth of *A. kosiensis*. In ecological terms an edge is defined as a zone where two plant communities meet, or where successional stages within plant communities come together (Noss, 1983 in Weiermans, 2000). William-Linera (1990) and Jose *et al.* (1996) define forest edges as sharp transitions from forest type to a more open type of vegetation, characterized by high soil temperature, low soil moisture, high solar radiation and low relative humidity. Increased light penetration close to an edge may thus favour tree growth because trees growing closer to an edge experience less competition for light. This means that increased competition for light further away from the edge should hamper growth. I suggest that other abiotic factors that are characteristic of edges (e.g. low humidity and soil temperature) may have a negative effect on growth, hence the positive relationship between growth and distance from an edge.

Topographic factors (elevation, slope and aspect) did not explain any of the variability in tree size. Monserud & Sterba (1996) also found that these variables explained only 3% of the variation in basal area increment of Austrian forest tree species. Lee *et al.* (2004), on the other hand, found elevation and slope to have a significant effect on DBH growth of Japanese red pine (*Pinus densiflora*) but only elevation was significant for Oriental oak (*Quercus variabilis*). It seems that the significance of the effect of topographic variables on the growth of species varies according to site and species.

*Acacia kosiensis* grows exponentially, at least for the first 21 years of age. *C. equisetifolia* grows sigmoidally and stops growing at around 13 years of age. *A. kosiensis* grows more than twice as fast as *C. equisetifolia* under the same site conditions. Tree age is the best predictor for growth in girth for both species. For *A. kosiensis*, the presence and the size of additional stems on a tree significantly influence growth. Topographical variables have no influence on the growth of *A. kosiensis* but distance from the edge of the stand has a positive effect, albeit a weak one.

The hypothesis that *C. equisetifolia* grows faster than *A. kosiensis* is rejected. Considering the higher growth rate of *A. kosiensis*, the question arises: does this species store twice as much carbon as *C. equisetifolia*? In Chapter 5 I compare carbon sequestration by the two species as well as carbon storage in the other three stores in areas set aside for rehabilitation and revegetation.

## Chapter 5

# Carbon sequestration and storage in rehabilitated and revegetated stands

### Introduction

The destruction of coastal dune vegetation prior to mining by Richards Bay Minerals (RBM) conceivably releases most of the carbon stored in biomass and in the soil. This destruction of vegetation also sees the end of the sequestration of carbon by forest plants through the process of photosynthesis. Post-mining dune rehabilitation and revegetation, however, present an opportunity for the recovery of this storage function. Because growing plants sequester carbon, plant growth rate can be used as a surrogate for the rate of carbon sequestration.

In Chapter 4 I showed that the two tree species dominating the post-mining land-use options had different radial growth rates. This finding raised a question of whether they also differ in carbon sequestration rate. Even though wood is a major component of aboveground carbon storage (Ilic *et al.*, 2000), there are other stores of carbon and it would be misleading to ignore these.

In an ecosystem, carbon is stored in four pools: aboveground live biomass (primary store), belowground live tissue, necromass (which consists mainly of plant litter usually on the soil surface or in the soil, but some may take the form of standing or attached dead material) and in the soil (Smith, 2000; Pregitzer & Euskirchen, 2004). Litterfall and decomposition drive the transfer of carbon between these stores (Attiwill & Adams, 1993; Rajendran & Devaraj, 2004). In this study I divided aboveground live biomass into woody biomass and herbaceous biomass. I thus measured carbon storage in woody biomass, herb layer[2], litter and soil for comparison between rehabilitated and revegetated stands. I did not address carbon storage in belowground live tissue.

The revegetated stands differ from the rehabilitated stands in that they are monocultures comprising evenly spaced trees of similar age and height. The floor is covered with thick layers of needle-like leaves. Occurrence of a herb layer is rare. The rehabilitated stands, on the other hand, have a more complex vegetation structure. The dominant trees, which form the canopy, are *A. kosiensis*. The under-story comprises emerging broadleaved tree species, climbers, forbs and grasses. The intensity of competition for resources with co-existing tree species possibly reduces the growth potential, and subsequently the carbon accumulation potential, of *A. kosiensis*. The presence of a dense and diverse under-story, however, probably enhances biomass accumulation in the rehabilitated stands. The litter layer comprises mainly dead plant material at various stages of decay. Because of such structural and functional differences between the two vegetation types, I expected carbon storage in the different compartments to differ. Overall, because the rehabilitated stands are more biologically diverse, I expected them to store more carbon than the revegetated stands.

I thus tested the following hypotheses:

$H_1$: Carbon storage in the wood of *A. kosiensis* is less than in the wood of *C. equisetifolia*

$H_2$: The amount of carbon stored in the herb layer of rehabilitated stands is more than that stored in the herb layer of revegetated stands

$H_3$: There is more carbon stored in the litter of the revegetated stands than in that of the rehabilitated stands

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[2] For the purposes of this study herb layer refers to all non-woody vegetation below 1m in height.

$H_4$: There is more carbon stored in the soil of the rehabilitated stands than in that of the revegetated stands

$H_5$: The total carbon pool of the rehabilitated stands is more than that of the revegetated stands

## Materials and Methods

Wood, herb layer, litter and soil samples were collected between January and February 2004 from four stands of *A. kosiensis* (aged 7, 11, 17 and 21 years) and four *C. equisetifolia* stands (aged 8, 12, 16 and 19 years). I sampled along two parallel, 10 meter wide transects, ~20 meters apart, running through each stand. Two trees were cut down along each transect to obtain wood samples. I took four herb layer and litter samples, as well as five soil samples randomly along each transect. I obtained herb and litter samples using a custom made 1 $m^2$ sampling frame. I used clippers to remove the herb layer, whilst litter samples were simply picked up from the forest/plantation floor within a 0.25 x 0.25 $m^2$ quadrat of the sampling frame. I extracted soil cores (0-15 cm) using a soil auger of ~600 ml (Thiart Augers, Potchefstroom). Van Aarde *et al.* (1998) suggested that most changes in soil attributes during succession take place in the upper 10 cm, hence my sampling of the upper 15 cm. Ten soil samples were also collected from Sokhulu, an unmined forest contiguous with Mapelane Nature Reserve (28°24'S 32°26'E). Data from this site, together with data collected earlier in the same forest as well as Zulti (south of St. Lucia lighthouse), were used to determine the average benchmark value for soil carbon storage. These data are stored in the Conservation Ecology Research Unit (CERU) database.

All samples and cores were put in separate brown paper bags, weighed to determine wet mass, oven dried at 70 °C, weighed again to determine dry mass and then stored until analysis for carbon concentration. The Walkley-Black method, as suggested by The Non-Affiliated Soil Analysis Working Committee (1990), was used to determine the organic carbon concentration in the soil samples. The Kjeldahl method of digestion by sulphuric acid was used to determine the carbon concentration in the wood, herb and litter samples. Carbon concentration is expressed as a percentage of the dry mass of a sample.

A major component of carbon storage is in the woody stem (Ilic *et al.*, 2000). An estimate of stand biomass is required to obtain a measure of carbon storage in wood. Mean stand biomass can be calculated as:

$$bm = d \times m,$$

where $bm$ = biomass (kg.ha$^{-1}$), $d$ = density (trees.ha$^{-1}$) and $m$ = the mean mass of individuals in a specific stand (kg) (van Dyk, 1996). I measured tree density (based on the Point-Centred Quarter method, Cottam & Curtis, 1956) at 20 randomly selected points in each of the four *A. kosiensis* stands. Density within each stand was expressed as the average of those 20 values. An estimate of tree density in the *C. equisetifolia* stands was obtained from Mr. R. Kok (*pers. comm.)*[3]. Tree mass can be determined directly by weighing whole trees or by extrapolating from pre-existing regression lines. I measured the height and circumference at breast height of four trees at 20 randomly selected points in each stand. An infrared rangefinder (Impulse model, Laser Technology Inc., Englewood CO, USA) was used to measure individual tree height and distances between trees, whereas a tape measure was used to measure circumference, which was subsequently converted to DBH (diameter at breast height, 1.2m) using the equation DBH = $C/\pi$, where C is circumference

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[3] Mr. Rynhard Kok. Rehabilitation superintendent, Richards Bay Minerals, Richards Bay, South Africa.

as measured and $\pi$ is a constant (3.14). I then used DBH and height to calculate tree mass for *A. kosiensis* using the regression equation developed by van Dyk (1996):

$y = 0.864x - 1.445$; where $y$ is $\log_e$mass and $x$ is $\log_e(\mathrm{DBH}^2 \times \mathrm{height})$.

I weighed four randomly chosen *C. equisetifolia* trees in each stand using a cattle scale, accurate to within a kilogram. I subsequently multiplied stand biomass ($\mathrm{kg.ha}^{-1}$) by carbon concentration in the wood (%C) to obtain values of carbon storage in wood ($\mathrm{kgC.ha}^{-1}$).

Alternatively, tree mass can be computed from density and total volume of the wood (Fearnside, 1997; Ilic *et al*., 2000). Multiplying wood density and volume yields an estimate of tree mass. Wood density or basic specific gravity (Fearnside, 1997) is expressed as the ratio of the mass of the oven dry sample to its wet volume ($\mathrm{kg.m}^{-3}$). I determined wood density using the water-immersion method as set out in Ilic *et al*. (2000). However, I used wood disks instead of wood cores. I calculated wood volume, based on the assumption that most carbon storage in wood is in stem wood, and that the shape of a stem is either conical, cylindrical or paraboloid (Appendix 3).

### Data analysis

I used a *t*-test in STATISTICA Version 6 (StatSoft, Inc. USA) to test for differences between the wood densities of the two species. For both land use options, total carbon storage in each stand was determined by summing random combinations of carbon storage in all four stores using Microsoft Excel 2000 (Microsoft Corporation, UK). There were 16, 8, 8 and 10 values for the wood, herb layer, litter and soil stores for each stand. Consequently, the random combinations of sums thereof were in excess of 10000 values. A subset of 1000 values was used in the least squares linear regression analyses. I used GraphPad QuickCalcs (GraphPad Software, Inc. USA) to test for outliers, using stepwise exclusion. Graphpad Prism

Version 3 (GraphPad Software, Inc. USA) was used to draw scatter-plots for wood density, tree density, tree dry mass, dry mass and percentage carbon of the herb layer and litter, as well as the amount of carbon in the wood, herb layer, litter, soil and total carbon pool of the different stands. Linear regression analysis (Zar, 1996) was used to assess rates of carbon accumulation in the various carbon stores.

## Results

### *Wood density*

The wood of *C. equisetifolia* was significantly denser than that of *A. kosiensis* (*t*-value = 8.80, $p < 0.0001$) and ranged from ~750 kg.m$^{-3}$ to just below 900 kg.m$^{-3}$ (mean ± S.E. = 829.2 ± 8.21, n = 16) (Fig. 5.1b). The wood of *A. kosiensis* ranged between 600 and 775 kg.m$^{-3}$ (mean ± S.E. = 712.8 ± 10.49, n = 15) (Fig. 5.1a). Wood density did not change significantly with increasing age for either species ($P$ = 0.69 and 0.38 for *A. kosiensis* and *C. equisetifolia* respectively).

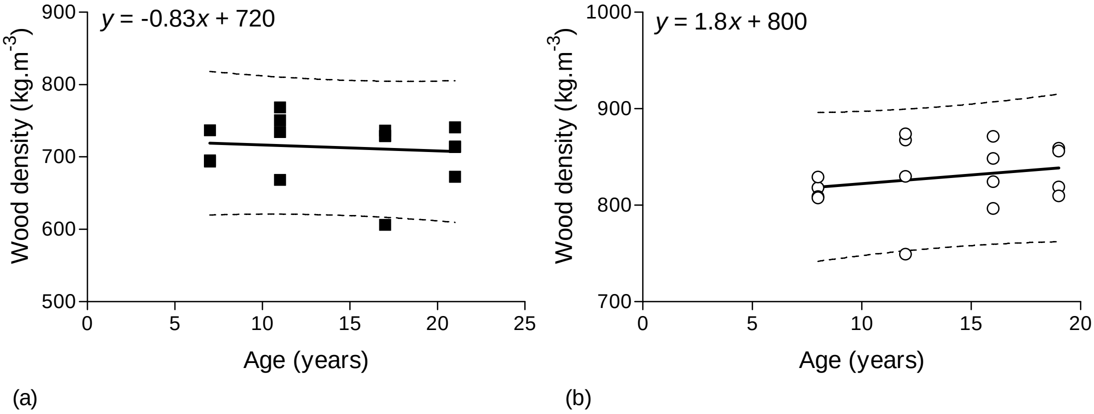


**Fig. 5.1.** Wood density as a function of age of (a) *A. kosiensis* (n = 15) and (b) *C. equisetifolia* (n = 16). Least squares linear regression lines, shown with 95% prediction intervals, illustrate a lack of significant change in wood density with age. The functions defining the regression lines are shown on the figures.

*Tree density*

Tree density in *A. kosiensis* stands decreased from a maximum of 10851 trees/ha (mean ± S.E. = 4917 ± 604, n = 20) in the 7-year-old stand to a maximum of 125 trees.ha$^{-1}$ (mean ± S.E. = 616 ± 94, n = 20) in the 21-year-old stand (Fig. 5.2). The density remained at ~1736 trees.ha$^{-1}$ across all ages in the revegetated stands.

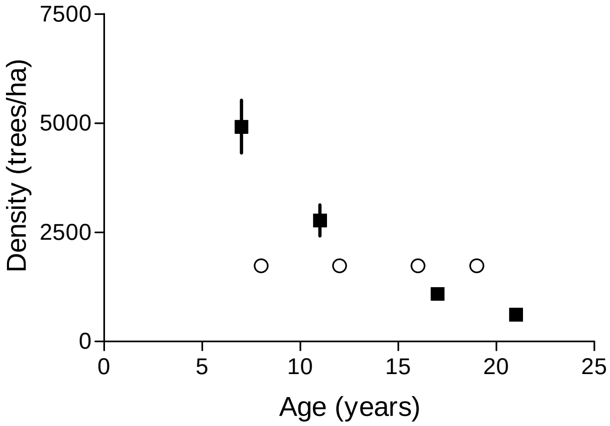


**Fig. 5.2.** Tree density as a function of age in rehabilitated (solid squares, n = 80) and revegetated (open circles, n = 4) stands. The values for the rehabilitated stands are expressed as mean ± SE. The Point Centred Quarter method was used to determine the density of *A. kosiensis*, whereas a known fixed number was quoted for the density *C. equisetifolia* (see text).

*Tree mass*

*Acacia kosiensis* increased in dry mass at a rate of 13 ± 0.91 kg.year$^{-1}$ ($P < 0.0001$) across the sampled range (Fig. 5.3a). The dry mass of *C. equisetifolia* trees, on the other hand, increased to a maximum at 16 years of age (Fig. 5.3b). The change in dry mass with age was not significant ($P = 0.07$). Surprisingly, the oldest *Casuarina* trees (19 years old) were lighter on average (157.4 ± 25.46 kg) than the 12 and 16 year-old trees (175.8 ± 43.5 kg and 195.4 ± 21.31 kg, respectively). Generally, *C. equisetifolia* trees were heavier than *A. kosiensis* trees in stands ≤19 years.

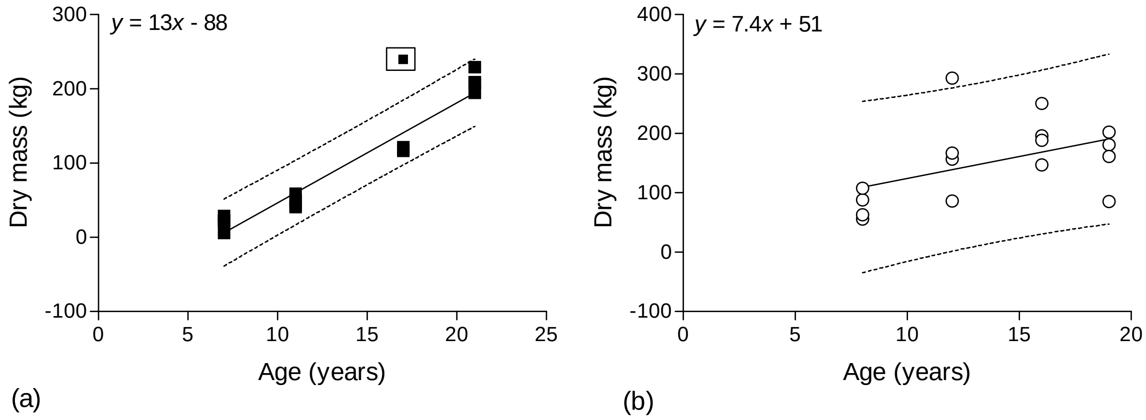


**Fig. 5.3.** Dry mass of individual (a) *A. kosiensis* (n = 16) and (b) *C. equisetifolia* (n = 16) trees ranging from 6 to 21 years of age. The boxed data point (a) is an outlier (Grubb's test $P < 0.05$). Least squares linear regression was used to show rates of change, equations are shown on the figures. *C. equisetifolia* trees were weighed and the mass of *A. kosiensis* was derived from a regression equation developed by van Dyk (1996) (see text for details).

*Carbon storage in wood*

Figures 5.4a and b show carbon storage in the wood of *A. kosiensis* and *C. equisetifolia* respectively, as a function of the age of trees. There was more carbon stored in *C. equisetifolia* wood (mean across ages ± S.E. = 121.64 ± 6.42 tC.ha$^{-1}$, n = 80) than in *A. kosiensis* wood (mean across ages ± S.E. = 74.00 ± 5.53 tC.ha$^{-1}$, n = 79). Carbon storage increased at a rate of 1.00 ± 0.33 and 13.0 ± 2.2 t.ha$^{-1}$.year$^{-1}$ ($P < 0.01$) in the wood of *A. kosiensis* and *C. equisetifolia*, respectively. The high variability in the carbon values is attributable to the wide range of tree density and tree mass values (Figs 5.2 and 5.3).

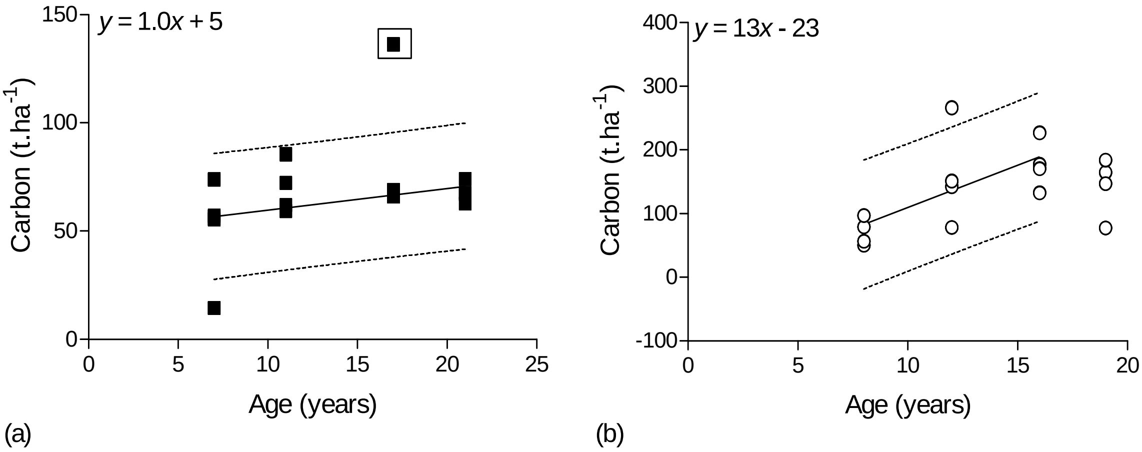


**Fig. 5.4.** The amount of carbon (t.ha$^{-1}$) in the wood of (a) *Acacia kosiensis* (n = 16) and (b) *C. equisetifolia* (n = 16) as a function of stand age. Linear regression lines (± 95% prediction interval), based on least squares regression analyses, were used to determine accumulation rates. Data for *C. equisetifolia* were analysed only for the period of active growth (up to 16 years of age). The linear regression equations are shown on the figures. The boxed data point is an outlier (Grubb's test $P < 0.05$).

*Carbon storage in the herb layer*

The herb layer in revegetated stands was sparse. Only five herb layer samples were obtained in all the revegetated stands sampled in this study. Because of this, a meaningful statistical analysis could not be carried out for this land use option. The carbon content of the herb layer of rehabilitated stands was 46 ± 0.43% and did not change significantly with age ($P = 0.15$, Fig. 5.5a). Herb layer biomass increased at 20 ± 3.8 g.m$^{-2}$.year$^{-1}$ ($P < 0.0001$), from 123± 31.2 g.m$^{-2}$ in the seven year-old rehabilitated stand to 400.5 ± 46.1 g.m$^{-2}$ in the 21 year-old stand (Fig. 5.5b).

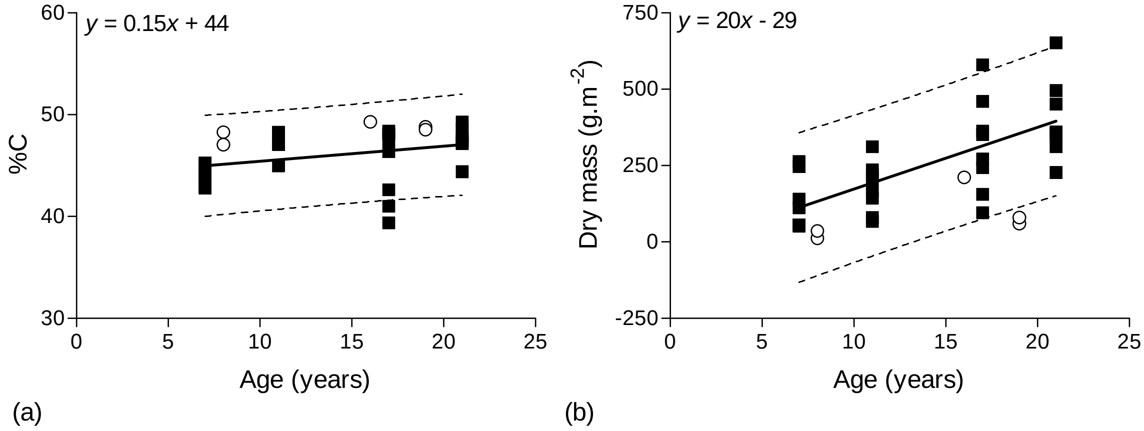


**Fig. 5.5.** Carbon concentration (a) and dry mass (b) of herb layer samples, as a function of age of rehabilitated (solid squares) and revegetated (open circles) stands. Linear regression lines, shown with 95% prediction intervals, show rates of change for *A. kosiensis* data. The functions describing the lines are shown on the figures. *C. equisetifolia* data were not analysed because of small sample sizes.

Combining the carbon concentration and dry mass data (Figs 5.5a and b) gave values of carbon storage in the herb layer (Fig. 5.6). The amount of carbon in the herb layer in the rehabilitated stands increased with age at a rate of 0.10 ± 0.02 tC.ha$^{-1}$.year$^{-1}$, from123 ± 31.2 tC.ha$^{-1}$ in the youngest stand to 401 ± 46.1 tC.ha$^{-1}$ in the oldest stand. Again, the data for the revegetated stands could not be analysed because of the small sample size.

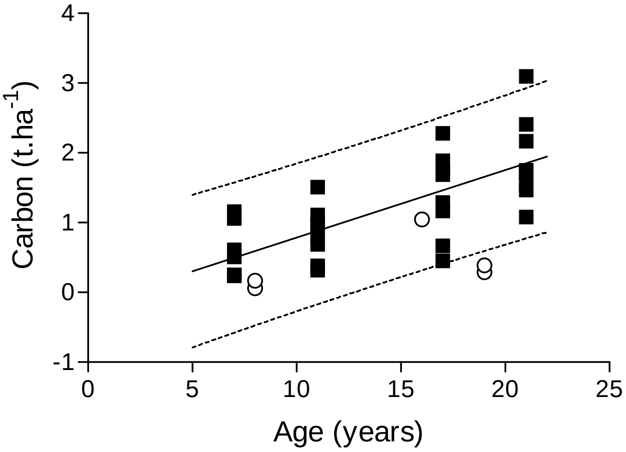


**Fig. 5.6.** Carbon (t.ha$^{-1}$) in the herb layer of rehabilitated (solid squares, n = 32) and revegetated (open circles, n = 5) as a function of age. The linear regression line (shown with 95% prediction interval) defined by the function $y = 0.10x - 0.18$ was used to determine the rate of increase in carbon storage in the rehabilitated stands. Linear regression analysis was not done for *C. equisetifolia* because of a small sample size.

*Carbon storage in the litter*

The carbon content of litter in revegetated stands did not change significantly with age ($P = 0.49$, Fig. 5.7b). It ranged from 40.8 ± 2.99% (n = 8) in the youngest stand to 39.5 ± 3.11% (n = 8) in the oldest stand. On average, it was higher than the carbon content of litter in the rehabilitated stands, which increased significantly from 19.5 ± 1.97% (n = 8) in the youngest stand to 35.1 ± 1.42% (n = 8) in the oldest stand (Fig. 5.7a).

There was more litter in the revegetated stands (mean dry mass across ages ± S.E. = 1.01 ± 0.06 kg.m$^{-2}$, n = 28) than in the rehabilitated stands (mean dry mass across ages ± S.E. = 0.55 ± 0.04 kg.m$^{-2}$, n = 32) (Fig. 5.7c & d). The rate of increase in the amount of litter in the revegetated stands was 0.14 ± 0.05 kg.m$^{-2}$.year$^{-1}$ ($P < 0.05$) but there was no significant increase in the rehabilitated stands ($P = 0.09$).

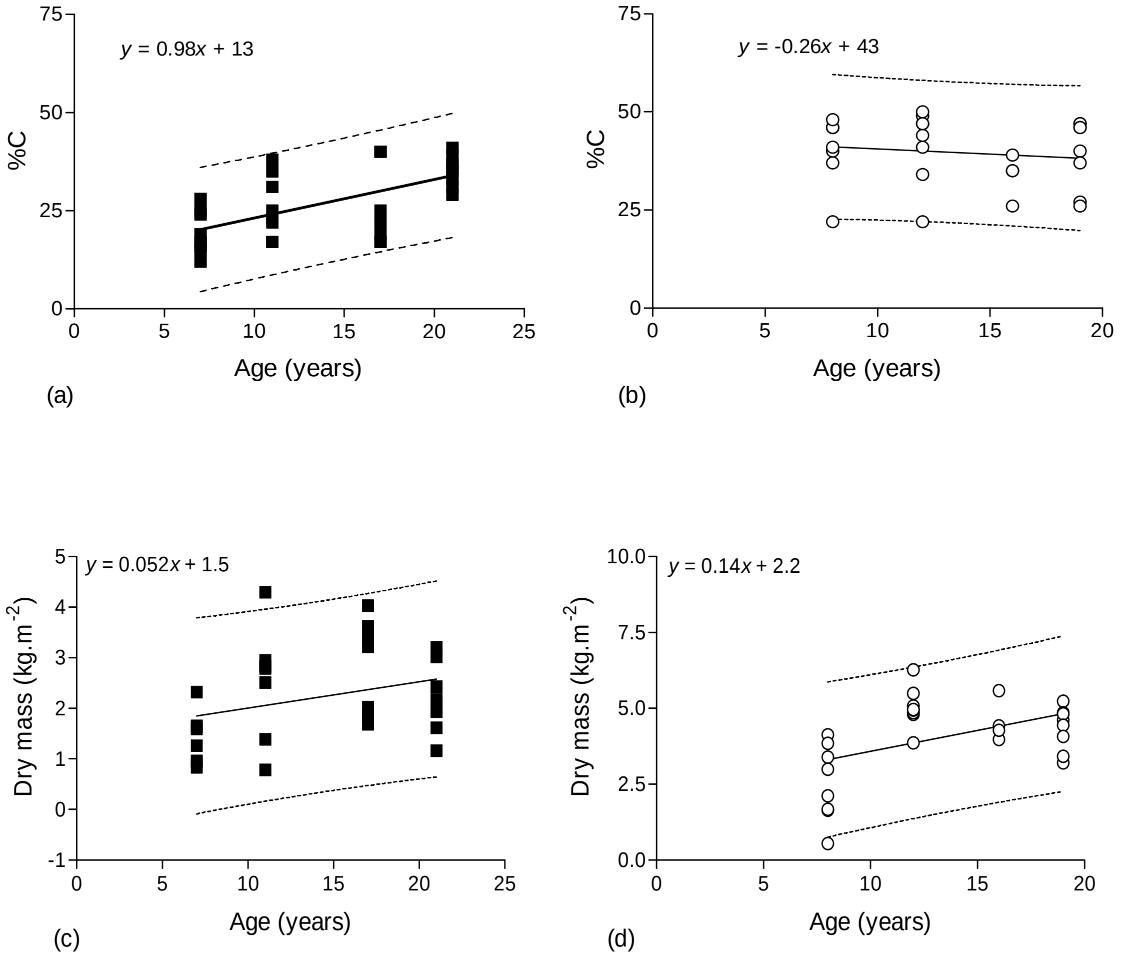


**Fig. 5.7.** Carbon concentration (a & b) and dry mass (c & d) of litter samples as a function of age of rehabilitated (solid squares, n = 32) and revegetated (open circles, n = 32) stands. Linear regression lines, based on least squares regression analysis, were used to determine rates of change. The corresponding equations are shown on the figures.

The combination of percentage carbon and mass yielded an estimate of carbon storage in the litter. Values ranged from 2.77 ± 0.49 $tC.ha^{-1}$ in the youngest rehabilitated stand to 7.82 ± 1.08 $tC.ha^{-1}$ in the oldest (Fig. 5.8a). There was more carbon in the revegetated stands, ranging from 10.47 ± 2.16 $tC.ha^{-1}$ in the youngest stand to 17.20 ± 1.81 $tC.ha^{-1}$ in the oldest (Fig. 5.8b). The rate of increase in litter carbon in the rehabilitated stands was 0.32 ± 0.08 $tC.ha^{-1}.year^{-1}$ ($P = 0.0002$). The increase in carbon in the litter of revegetated stands was not significant ($P = 0.13$).

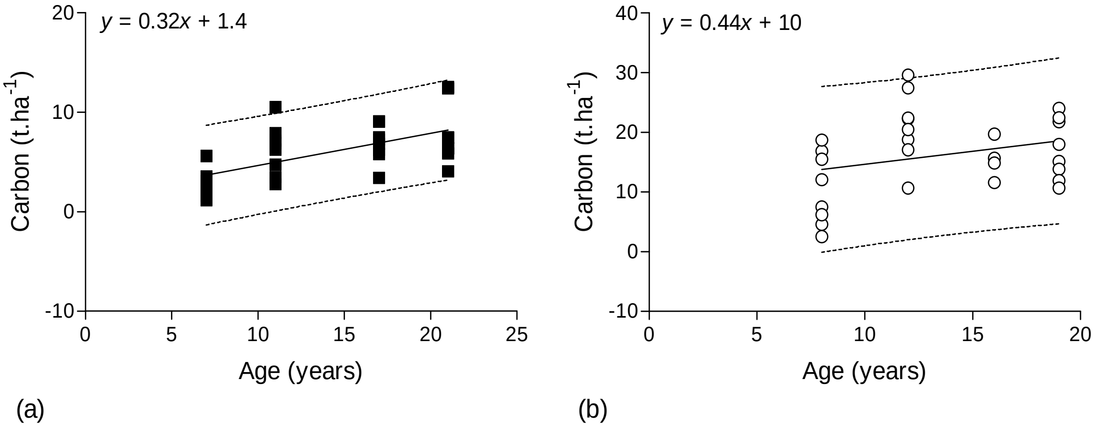


**Fig. 5.8.** Carbon (t.ha$^{-1}$) in the litter of (a) rehabilitated stands (n = 32) and (b) revegetated stands (n = 32) expressed as a function of age. Least squares linear regression lines (shown with 95% prediction intervals) were used to determine rates of increase. The lines are defined by the functions shown on the figures.

*Carbon storage in the soil*

I recorded a decrease of -0.66 ± 0.24 t.ha$^{-1}$.year$^{-1}$ in the amount of carbon stored in the soil of rehabilitated stands with age (Fig. 5.9). This decrease was significant ($P = 0.009$). These results were directly opposite to the findings of Smith (2000) and Wassenaar (2004) who had worked at the same site and documented an increase in the amount of carbon between the ages of one and 24 years.

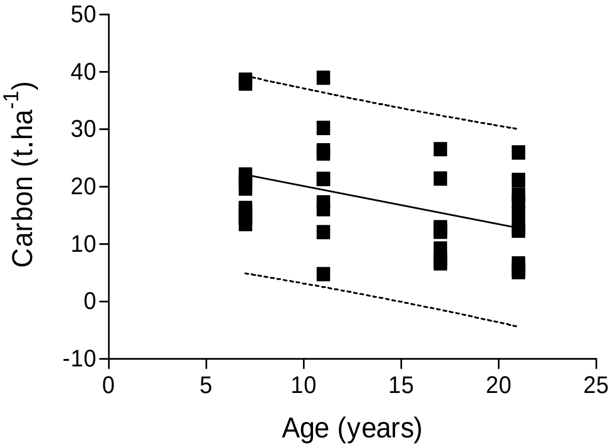
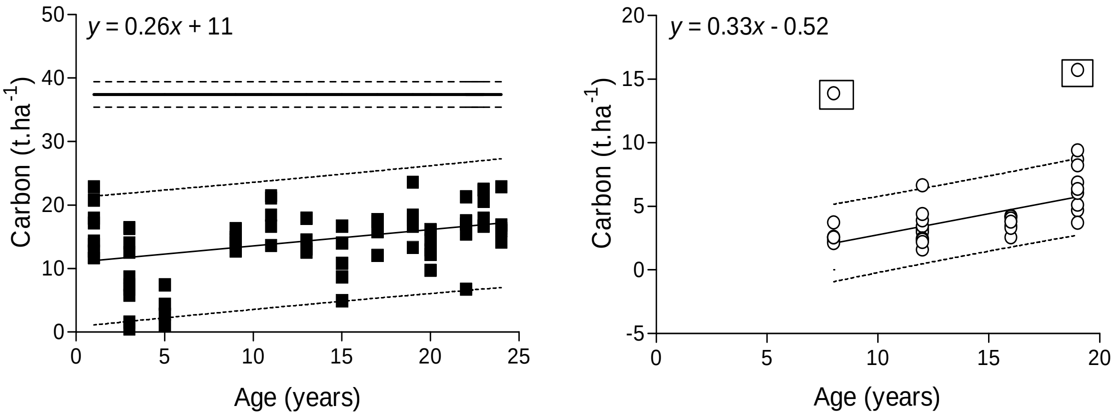


**Fig. 5.9.** Carbon (t.ha$^{-1}$) in the soil of rehabilitated stands of different ages (n = 32). Linear regression line (shown with 95% prediction interval) was fitted using least squares regression to illustrate the decrease ($P = 0.07$). The regression line is defined by the function $y = -0.66x + 27$.

Subsequently, I used soil data that were collected in other studies (Smith, 2000 and Wassenaar, 2004) in stands aged 1, 3, 5, 9, 11, 13, 15, 17, 19, 20, 22, 23 and 24 years old. These data form part of the Conservation Ecology Research Unit (CERU) database.

Carbon in the soil of rehabilitated stands varied within stands. It ranged between 11.73 and 22.94 tC.ha$^{-1}$ (n = 9) in the youngest stand and between 14.23 and 22.93 tC.ha$^{-1}$ (n = 5) in the oldest stand (Fig. 5.10a). All rehabilitated stands stored less carbon than the benchmark sites. Carbon storage in the soil of revegetated stands ranged between 2.11 and 3.74 tC.ha$^{-1}$ (n = 7) in the youngest stand, and between 3.73 and 9.41 tC.ha$^{-1}$ (n = 9) in the oldest stand (Fig. 5.10b). Generally, there was about three times more carbon in the soil of rehabilitated stands than revegetated stands (e.g. a 19 year-old rehabilitated stand stored, on average, 17.99 ± 1.67 tC.ha$^{-1}$ while a revegetated stand of similar age stored 6.58 ± 0.64 tC.ha$^{-1}$). The amount of carbon stored in the soil in the rehabilitated stands increased by 0.26 ± 0.07 tC.ha$^{-1}$.year$^{-1}$ ($P = 0.0008$) while that in the revegetated stands increased by 0.33 ± 0.06 tC.ha$^{-1}$.year$^{-1}$ ($P < 0.0001$).



(a) (b)

**Fig. 5.10.** Carbon (t.ha$^{-1}$) in the soil of (a) rehabilitated stands (n = 73, data from Smith, 2000 & Wassenaar, 2004) and (b) *Casuarina* plantations (n = 40) of different ages. Linear regression lines (shown with 95% prediction intervals) were used to determine accumulation rates. Functions describing the lines are shown on the figures. The horizontal line in (a) represents the average benchmark value ± SE (37.43 ± 1.74, n = 19). The boxed data points are outliers (Grubb's test $P < 0.05$).

*Total carbon storage*

The total carbon pool of rehabilitated and revegetated stands of known ages is shown in Figs 5.11a & b respectively. The total pool in rehabilitated stands increased from an average of 72.14 ± 2.44 tC.ha$^{-1}$ (n = 1000) in the 7 year-old stand to 136.47 ± 6.63 tC.ha$^{-1}$ (n = 1000) in the 17 year-old stand and then dropped to 89.93 ± 2.24 tC.ha$^{-1}$ (n = 1000) in the 21 year-old stand (Fig. 5.11a). The total pool of revegetated stands was larger than that of rehabilitated stands and increased from 80.97 ± 2.01 tC.ha$^{-1}$ (n = 1000) in the 8 year-old stand to 211.32 ± 3.73 tC.ha$^{-1}$ (n = 1000) in the 16 year-old stand, then dropped to 158.17 ± 4.44 tC.ha$^{-1}$ (n = 1000) in the 19 year-old stand (Fig. 5.11b). This trend of an increase in carbon storage up to a certain age, followed by a drop in the oldest stand is similar to the trend exhibited by carbon storage in wood of the two species dominating the rehabilitated and revegetated stands (Fig. 5.3). Based on the linear regression lines (Fig. 5.11 a & b), carbon accumulated at a rate of 1.3 ± 0.07 and 7.9 ± 0.21 $t.ha^{-1}.year^{-1}$ in the rehabilitated and revegetated stands, respectively.

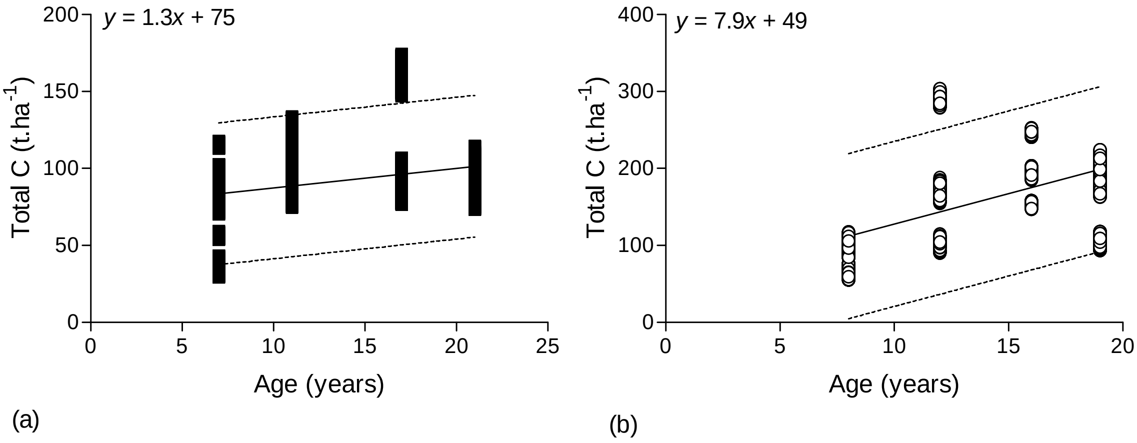


**Fig. 5.11.** The total carbon pool ($t.ha^{-1}$) of (a) rehabilitated (n = 4000) and (b) revegetated (n = 4000) stands as a function of age. Linear regression lines (shown with 95% prediction intervals) were used to calculate accumulation rates. Functions describing the lines are shown on the figures.

The contribution to total stand carbon by each of the stores measured in this study is shown in Fig. 5.12. Most carbon was stored in the wood of the tree species dominating the two land use options. The second largest contribution to the total carbon pool came from the soil and from the litter in the rehabilitated and revegetated stands, respectively. In both cases the herb layer contributed the least to the total carbon pool.

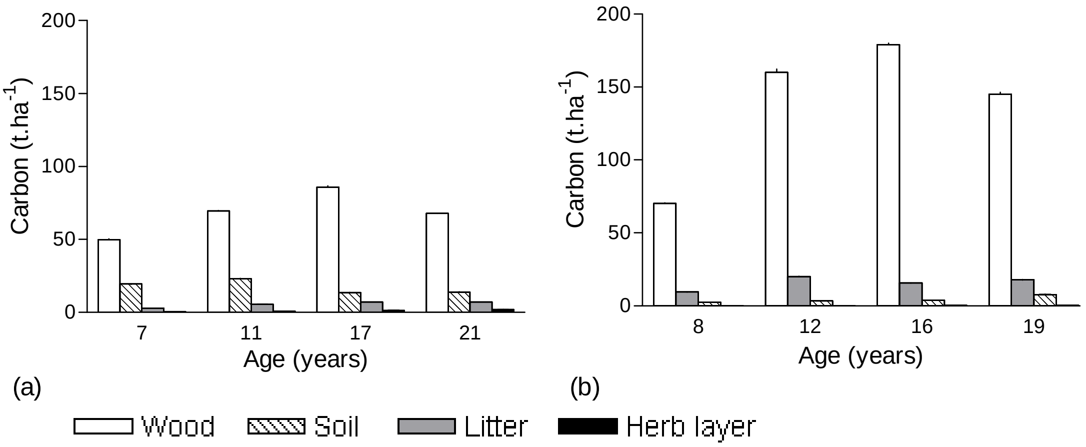


**Fig. 5.12.** Actual contribution of the various stores (wood, herb layer, litter and soil) to the total carbon pool in (a) rehabilitated and (b) revegetated stands of known ages.

## Discussion

The wood density values I obtained in this study are less than published values for the two species (see Barnes *et al.*, 1996; Doran & Hall, 1983). These differences are probably a consequence of geographic differences in growing conditions (see Ilic *et al.*, 2000). In the end, I did not use wood density to determine carbon storage in wood, as such an exercise requires prior knowledge of stem volume. Estimates of stem volume are influenced by shape. For example, a conical stem occupies less volume than a paraboloid stem of equal height and basal area, which in turn occupies less volume than a cylindrical stem (Appendix 3). Thus, stem shape influences the estimates of carbon storage in wood, as it determines which mathematical equation should be used to obtain stem volume (Appendix 4).

The results support the hypothesis that more carbon is stored in the wood of *C. equisetifolia* than in the wood of *A. kosiensis*. This finding is not surprising, considering that the former has much denser wood than the latter (Appendix 3). The higher wood density of *C. equisetifolia* means that it puts on more wood, and hence more carbon, per unit volume. Furthermore, the low rate of increase in carbon stored in wood in the rehabilitated stands is a result of the opposing effect of decreasing tree density on any increase in tree mass. *A. kosiensis*, being a pioneer species, decreases in density with age. But, as the biomass of this species decreases, secondary tree species become more common (Mentis & Ellery, 1994; van Aarde *et al*., 1996c & d; van Dyk, 1996; Wassenaar, 2004). In this way, the total carbon pool in wood in the rehabilitated stands may be increasing at a rate higher than what was recorded in this study, but this increase cannot be picked up through the single-species approach adopted here. A better approach would be to include the secondary tree species in this assessment.

The significant increase in the amount of carbon stored in *C. equisetifolia* with increasing age serves to show the important role stand age plays in determining carbon storage in woody biomass. The value of 74.00 ± 5.53 $tC.ha^{-1}$ for carbon storage in the wood of *A. kosiensis* falls within the range published for living biomass in tropical forests (20 to 227.94 (mean = 87.18 $tC.ha^{-1}$, Soepadmo, 1993; Pregitzer and Euskirchen, 2004). The values reported by Smith (2000) ranging between 37.38 and 31.06 $tC.ha^{-1}$ for rehabilitated stands ranging in age from 17 to 21 years, are much lower than the values I obtained, despite having worked at the same site. This I ascribe to his use of an indirect measure of biomass (the so-called Rutherford's equation), which could have resulted in a gross underestimate of actual biomass. Woody biomass is nearly always the largest carbon pool in carbon storage forest-type projects (MacDicken, 1997). This was true for both the rehabilitated and revegetated stands, where I found that woody biomass contributed >60% of the total carbon in all the stands.

Carbon storage in the herb layer increased with age in the rehabilitated stands and this is purely a consequence of an increase in herb layer biomass rather than an increase in the carbon concentration (%C) in the vegetation itself. Lubke *et al*. (1992), Mentis & Ellery (1994) and Wassenaar (2004) documented an increase in the species richness of herbaceous species with increasing age. However, this is not synonymous with an increase in herb layer biomass, which would contribute to an increase in carbon storage with increasing age. Smith (2000) reported values of ~80 and ~60 $g.m^{-2}$ for the 17 and 21 year-old stands, respectively. These are almost three times less than what I measured in stands of similar age (316 ± 56 $g.m^{-2}$ and 400.50 ± 46.10 $g.m^{-2}$). Moreover, Smith (2000) did not observe a significant increase in biomass between the ages of 17 and 21 years, whereas I observed an increase of ~85 $g.m^{-2}$ between the same ages.

*Casuarina* species restrict under-storey growth (Duke, 1983) through physical and chemical mechanisms. Accumulation of thick carpets of litter may hinder the emergence of seedlings of other species, while chemical compounds exuded by the litter itself may also add to the inhibition of seedling establishment (Barritt & Facelli, 2001). The inability of other plant species to co-exist with *C. equisetifolia* has negative consequences for ecosystem development in that ecosystem structure complexity is not encouraged, which may in turn hamper nutrient cycling. Because there was a limited herb layer in the revegetated stands, it follows that total carbon storage in the herb layer was less than in the rehabilitated stands. I thus accept the hypothesis that carbon storage in the herb layer of rehabilitated stands is more than that of revegetated stands.

The third hypothesis, proposing that carbon storage in the litter of rehabilitated stands is less than that of revegetated stands, is also accepted. More carbon was stored in the litter of *C. equisetifolia* than that of rehabilitated stands. This results from the carbon concentration (%C) of the former being ~1.5 times higher than that of the latter. I suggest that this difference is a consequence of the chemical and physical composition of the litter constituents of the litter. It is likely that the litter components of the rehabilitated stands are at different stages of mechanical and chemical degradation. This would then mean that the dead plant material has already released some of the carbon stored therein into the soil.

In addition to there being a higher concentration of carbon in the litter, more litter accumulated in the *C. equisetifolia* stands than in the rehabilitated stands. Litter layer amounts depend on a number of factors such as litterfall, trampling by vertebrate herbivores and decomposition (Ford & Grace, 1998; Smith, 2000). Smith (2000) quantified litterfall in the rehabilitated stands and found that litter accumulation rates range between 65 and 150 g.m$^{-2}$.month$^{-1}$. Because no similar study has been carried out for the revegetating stands, it is possible that differences in litterfall may be responsible for the difference in litter layer amounts. Cattle that frequent the rehabilitated stands (Wassenaar & van Aarde, 2001) may also contribute to the mechanical breakdown and incorporation of litter into the soil trough trampling, thus facilitating further chemical breakdown by microbial agents (Naeth *et al*., 1991; Hammel, 1997). Scholes (2004) stated that material such as grass decays more rapidly if it is trampled. This would then reduce the volume of litter.

Overall, values for the amount of carbon in the litter in rehabilitated stands obtained in this study fall in the lower end of the range reported for tropical forests (3 tC.ha$^{-1}$ to 104 tC.ha$^{-1}$, Schlesinger, 1977; Brown & Lugo, 1992). Also, the amount of carbon stored in the oldest *Casuarina* stand (19 years) was less than half of what was measured in a 37 year-old stand in coastal Senegal (42.05 tC.ha$^{-1}$, Gourbiere & Debouzie, 1995). These differences may be a result of differences in age, climate and litter quality, which directly affect litter accumulation and decomposition rates.

Soils are important and large stores of carbon, (Gupta & Rao, 1994; MacDicken, 1997). The soil carbon pool of tropical forests generally ranges between 40 and 228 tC.ha$^{-1}$ (Smithwick *et al*., 2002; Pregitzer & Euskirchen, 2004) while that of the Miombo woodlands is ~83 tC.ha$^{-1}$ (Walker & Desanker, 2004). Mills & Fey (2004a & b) reported values ranging from 40 to 71 tC.ha$^{-1}$ for carbon in the upper 10 cm of soils in the thicket biome. The values recorded in the present study (e.g. 17 tC.ha$^{-1}$ for the oldest sampled stand) fall outside the lower limits of these published ranges. This is probably because the soils need more time to accrue carbon after the mining process. The carbon found in the topsoil that is spread over the dunes at the start of rehabilitation (van Aarde & Wassenaar, 1999) is subsequently used up by the growing plants in the first few years after rehabilitation (notice the decrease in the first five years after rehabilitation, Fig. 5.10a). It thus takes time for the soil properties to revert to what they used to be prior to mining. Moreover, the sandy nature of the soil on these dunes makes it susceptible to leaching (Smith, 2000). However, I recorded an increase with age in the amount of carbon in the top 15 cm of soil in both the rehabilitated and revegetated stands. Indeed, the trajectory of the regression of soil carbon values of the rehabilitated stands seems to suggest that carbon storage is increasing towards that of unmined forests.

Most of the carbon in the soil comes from the vegetation, with smaller amounts coming from animal excreta, dead animal matter and the atmosphere (Hamblin, 1989 in Smith, 2000). Carbon input from the vegetation occurs via the decomposition of plant litter. Thus, the difference between litter production and its decomposition controls the size of the carbon store within the soil (Kurz *et al*., 2000). Three interacting groups of factors regulate the rate at which litter decomposes. These are the environmental conditions, the quality of the resource and the decomposer community (Gallardo & Merino, 1993; Heal *et al*., 1997). Because both the rehabilitated and revegetated stands are exposed to the same climate, I suggest that the "environmental conditions" factor does not contribute to the differences in carbon storage in soil. Instead, I propose that the differences in litter quality (e.g. structural and chemical properties) and biological activity are responsible for the observed differences.

Nitrogen (N) content or C:N ratio, as well as cutin and lignin content are of critical importance in the decay of litter (Gallardo & Merino, 1993; Prescott, 1995). Smith (2000) gave N and C:N values for the rehabilitated stands, but there are no such values for the revegetated stands to carry out a comparison. As a sequel to this study, it would be worthwhile to do a comparative structural and chemical analysis of the litter in both rehabilitated and revegetating stands in order to determine whether the quality differs to such an extent that it affects decay.

Perhaps the most important variable driving decomposition is the presence of decomposers, which may take the form of microbial agents like fungal hyphae (Gallardo & Merino, 1993), bacteria (Smith, 2000), millipedes (Smit & van Aarde, 2001) and/or nematodes and protozoa (Wardle & Lavelle, 1997). Fragmentation by animals (e.g. herbivorous invertebrates) significantly accelerates the degradation rate of tough types of litter such as tree leaves (Hammel, 1997). This, in turn, facilitates the release of elements into the soil. Smit & van Aarde (2001), however, found that millipede biomass and species richness had no effect on the rate of change in carbon concentration. They then suggested that individual millipede species affect the rate at which elements accumulate in the soil differently. Van Aarde *et al*. (1996) reported high densities of millipedes in rehabilitated stands, whereas sightings of millipedes in the *Casuarina* stands were few and far between (*pers. obs.*). It is therefore possible that the millipede community in the revegetated stands excludes species that are good at mobilizing carbon, which results in the slow rate of carbon accumulation in the soil. I suggest that biological activity in the *Casuarina* stands may be too limited or insufficient to transfer carbon from the litter into the soil at a rate that balances carbon input into the litter from litterfall. Because carbon storage in the soil differed between rehabilitated and revegetated stands, I thus accept the hypothesis that carbon storage in the soil of the rehabilitated stands is more than that of the revegetated stands.

Taking all four stores into consideration, more carbon was stored in the revegetated stands, mainly due to the large woody store, than in the rehabilitated stands. I thus reject the hypothesis that the total carbon sequestration potential of the rehabilitated stands is more than that of the revegetated stands. If the goal of the landowner is to maximize the carbon sequestration potential of the area, an alteration of the ratio of revegetation: rehabilitation to 1:0 would achieve this. However, the aim of establishing the *Casuarina* plantations in the first place is to produce charcoal. The short-term nature of this carbon storage potential thus makes this option less attractive.

Published estimates of the carbon pool of tropical forests are variable – values from 16.9 $tC.ha^{-1}$, for subtropical dry forests, to 250 $tC.ha^{-1}$, for tropical moist forest in China and West Africa have been reported, with an average of 137.2 ± 70.5 $tC.ha^{-1}$ ($n = 44$ independent studies; see Brown & Lugo, 1984 and references therein). The value of total carbon recorded in this study for the oldest rehabilitated stand (91.01 $tC.ha^{-1}$) is less than the normal range for mature forests (100 to 200 $tC.ha^{-1}$; Pregitzer & Euskirchen, 2004). But, the rate of accrual (1.3 $tC.ha^{-1}.year^{-1}$) compares well with the range for tropical forests (1.1 to 2.2 $tC.ha^{-1}.year^{-1}$; Malhi & Grace, 2000). Considering this rate of increase, it seems likely that in time, carbon storage in the rehabilitated stands will tend towards that of mature forests. Moreover, the values I recorded for the total pool of rehabilitated stands are probably underestimates as they do not include carbon storage in other forest tree species that co-exist with *A. kosiensis* in older stands. Thus, the true value may be closer to the values for mature forests.

Scholes (2004) estimated the total amount of carbon in arid fertile *Acacia* woodland, covering an area of $1 \times 10^{12}$ $m^2$, at $2108 \times 10^{12}$ g. This equates to 21.08 $tC.ha^{-1}$, which is almost four times less than the average obtained in this study for the rehabilitated stands (92.50). The discrepancy between values reported in different studies may be a result of ambiguity in the definition of a forest. For example, some authors clump forests and woodlands under the "forest" vegetation type while others treat these two as separate vegetation types. Discrepancies between studies may also result from differences in site-specific conditions as well as data collecting methods.

Historically there has been a bias in research on the RBM lease area, with most research done in the rehabilitated stands and little or no attention paid to the adjacent *Casuarina* plantations. This has led to a lack of published data pertaining to the revegetated stands, which made it difficult to assess how realistic are the values obtained in this study.

# Chapter 6

# The financial potential of rehabilitation and revegetation

## Introduction

Richards Bay Minerals (RBM) holds a lease to extract heavy minerals from the dunes northeast of the town of Richards Bay (van Aarde *et al.*, 1996c & d). The ownership of the land will revert to the government once the lease expires. Government may then appoint a local agency (e.g. the local municipality) to manage the use of land. What will happen to the area then? One option is that of the development of a financially viable and sustainable land use.

Sustainable development is not just about the protection of the environment. It refers to a balanced use of resources that will ensure their availability to future generations (Newton & Freyfogle, 2005). It also includes the use of natural resources to eliminate poverty and thereby improve social welfare (Morvaridi, 1994). Sustainable development is high on South Africa's agenda (e.g. Green, 2005). The owners of RBM (BHP Billiton and Rio Tinto) are committed to sustainable development (BHP Billiton HSEC Report, 2002; Johnson, 2003). So is RBM (RBM Sustainable Development Report, 2003), who accepts the Brundtland definition of sustainable development. This definition considers sustainable development as part of the socio-economic process (Morvaridi, 1994). To this end, the rehabilitation of land that has been disturbed by mining is recognised as part of sustainable development. Additionally, rehabilitation is an indication of good environmental performance on the part of RBM. Environmental performance measures how successful an organisation is in reducing and minimizing its impact on the environment (Klassen & McLaughlin, 1996).

Two thirds of the Tisand lease area are revegetated with exotic *Casuarina equisetifolia* plantations. On the remaining third, RBM establishes an indigenous coastal dune forest through ecological succession. We refer to this as “rehabilitation”. The Tisand Lease agreement dictates this ratio (van Aarde & Wassenaar, 1999). This ratio has implications for the efficacy of both the rehabilitation and the revegetation programmes since the measurable products derived from these options differ. In the case of rehabilitation, the company aims to establish a secondary dune forest and restore the conservation potential of the area (Camp, 1990; Wassenaar & van Aarde, 2005). On the other hand, the revegetation programme aims at providing for the development of informal charcoal production plants driven by local people.

The rehabilitation and revegetation policy is evidence of compliance with environmental regulations. This can be costly and might hurt an organisation's bottom line (Cohen *et al.*, 1997). Thus, we expect some measurable benefit to balance the cost. Presently, RBM is spending money on rehabilitation and revegetation. We do not know what the benefits of this expenditure are.

The benefits derived from the direct and consumptive use of natural resources include wood for construction, timber and energy, medicinal products, edible fruit, herbs and vegetables, and the hunting of game. These benefits may not be readily accessible to the community presently, but they may be after mine closure. Indirect benefits include watershed protection, erosion protection, micro-climatic regulation, carbon storage and sequestration, water filtration and soil stabilisation. Where there is no existing market, such as for most of the indirect benefits listed above, environmental economists try to set up imaginary artificial markets through contingency valuation surveys. These are surveys of people's willingness to pay for environmental benefits (Bedder, 1994). This is a time-consuming exercise and beyond the scope of my study.

Ecosystems can also have option values (e.g. recreation and tourism) and non-use values (e.g. existence values and bequest values) (Blignaut & Lumby, 2004). All these have an economic value and an in-depth economic analysis, requiring an economist's expertise, would conceivably address all of them. This is a less economically oriented study and will only address financial benefits potentially accruable from the sale of timber, charcoal and certified emission reduction units (CERs). A CER is equal to one metric ton of $CO_2$-equivalent emissions reduced or sequestered through a Clean Development Mechanism project.

Most studies previously carried out on the site have primarily evaluated the ability of communities to recover in response to post-mining rehabilitation (van Aarde *et al.*, 1996 a, b & c; Ferreira & van Aarde, 1997; Kritzinger & van Aarde, 1998; Davis, *et al.*, 2002; Davis *et al.*, 2003; van Aarde *et al.,* 2004; Redi *et al.*, 2005; Wassenaar *et al.*, 2005). A few studies have evaluated the recovery of soil attributes (van Aarde *et al.*, 1998; Smith, 2000; Smit & van Aarde, 2001; Kumssa *et al.*, 2004). One study has alluded to the cost of post-mining rehabilitation by RBM (Lubke & Avis, 1998). No study, however, has contrasted the cost of rehabilitation to the potential benefits afforded by the recovered variables. Of interest, as well, is the lack of studies focusing on the revegetation programme.

In this Chapter I test the hypothesis that the financial costs and benefits of rehabilitation and revegetation are not different. I focus on the costs of establishment as well as costs of production of the marketable product, where applicable. I also address the potential financial benefits of the two post-mining land-use options.

## Methods

I gathered information on the environmental performance of RBM from published literature and the company's internal reports. I used published information and information provided through personal communication[4] to estimate the costs of rehabilitation, revegetation and the production of charcoal. I based labour wages on the minimum labour wages for farm workers (~R4.hour$^{-1}$) as set out by the Department of Labour for the year 2004. I used this hourly rate for labour associated with the rearing of the *C. equisetifolia* seedlings, livestock guarding, charcoal production and the operation of vehicles used to transport timber and charcoal. Appendix 5 shows the detailed calculations of all costs. I used labour costs for other steps in the rehabilitation and revegetation programme as they were provided (R.ha$^{-1}$). I based estimates of the financial benefits potentially accruable from the sale of charcoal and certified emission reduction units (CERs) on current (2005) market prices (www.pointcarbon.com). I also did expenditure and income analysis of the rehabilitation and revegetation programmes. This is based on the difference between the input costs and income generated from the products and by-products of a programme. In the case of rehabilitation, this is the difference between the costs of the rehabilitation process itself and the income potentially accruable from the sale of the programme's by-product (CERs). For revegetation, there are various streams of expenditure and income. Thus, I based the analysis on the difference between the sum of all expenses and the sum of all income streams (see Appendix 5). The different expense and income values were in different units (e.g. rehabilitation and revegetation costs as well as timber and CER income values were in R.ha$^{-1}$. The charcoal production costs and charcoal sale income values, on the other hand, were in R.t$^{-1}$). Consequently, all values were standardised and expressed in R.year$^{-1}$.

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[4] Information on the charcoal industry was provided by the owner of the charcoal-making company operating on the lease area. Information on rehabilitation and revegetation was supplied by the Mining and Planning Superintendent at RBM.

## Results

### *Costs*

#### *(1) The cost of rehabilitation and revegetation*

Based on Camp (1990) and van Aarde *et al.* (2004), the first step in revegetation and rehabilitation is the reshaping of dunes. I assume that the cost of reshaping the dunes for revegetation and rehabilitation is the same. Differences in the total cost of rehabilitation and revegetation result from differences in the cost of subsequent steps (Table 6.1).

**Table 6.1.** Approximate costs (R.ha$^{-1}$) of the steps involved in the rehabilitation and revegetation of sand dunes by Richards Bay Minerals based on figures for 2004. Values with asterisks next to them were obtained from the rehabilitation office of the RBM, the rest were calculated (as shown in Appendix 5).

| **Activity** | **Rehabilitation (R.ha$^{-1}$)** | **Revegetation (R.ha$^{-1}$)** |
|---|---|---|
| Purchase of seeds/seedlings | 2000* | Not applicable |
| Rearing seedlings | Not applicable | 351 |
| Topsoil replacement | 90 000* | Not applicable |
| Erection of windbreaks[5] | 18 000* | 25 000* |
| Maintenance of windbreaks | 3000* | 10 000* |
| Planting of seeds/seedlings | Not applicable | 1 900* |
| Alien species control | 600* | Not applicable |
| Livestock guarding | 0.37 | Not applicable |
| Re-planting bare patches | Not applicable | 500 if needed* |
| Total | 113 600 | 37 751 |

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[5] Wind protection is only required for about 6 months for rehabilitation, while it needs to be in place for sometimes up to three years for *Casuarina*. A lot of the nets are re-used in the rehabilitation, while new nets have to be bought almost every time for Casuarinas. The windbreaks in the *Casuarina* plantations also need much more maintenance because of the long time that they are up, hence the difference in costs.

The total yearly cost of rehabilitation was R6 250 00, based on the 55 hectares of dunes rehabilitated in 2004. The cost of revegetating 128 hectares during the same year amounted to about R4 810 000. The cost of rehabilitating a hectare of mined dunes (R113 600) exceeded the cost of revegetating (R37 751) a similar area. This difference primarily results from the cost of spreading topsoil across areas earmarked for rehabilitation (see Table 6.1).

*(2) The cost of making charcoal*

The charcoal making plant is located on the RBM lease area. It is operated by local people and the product is transported to the nearby town of Richards Bay. The most costly step in the manufacture of charcoal is the harvesting of timber, followed by the carbonisation process. This is because these steps are the most labour intensive in the whole operation. The total cost of producing a ton of charcoal is less than R 1 000 (Table 6.2).

**Table 6.2.** Steps and approximate costs involved in making charcoal. Costs are in R.t$^{-1}$ of charcoal produced. See Appendix 5 for detailed calculations.

| **Activity** | **Cost** |
|---|---|
| Purchase of timber | 90 |
| Timber harvesting | 500 |
| Timber transportation | 71 |
| Carbonisation | 216 |
| Charcoal transportation | 49 |
| Total | 926 |

Based on the 2 080 tons that are produced annually, the cost is R1 926 000 per year.

*Benefits*

*(1) Positive market valuation for RBM*

In 2003, RBM received the "Excellence in Mining Environmental Management (EMEM)" award for the third year in a row. This was in recognition of their environmental management practices (RBM Sustainable Development Report, 2003). The Department of Minerals and Energy (DME) developed the EMEM award system to reward mines that show environmental responsibility and strive towards excellence in environmental management (DME, 2000). The rehabilitation and revegetation programme that follows mining by RBM is demonstration of environmental responsibility.

*(2) Income from the sale of timber*

*C. equisetifolia* is harvested for charcoal production at the age of 16 years. At this age, biomass of utilizable timber[6] is ~390 t.ha$^{-1}$. Thus, if the timber is sold for R15 per ton, the standing value (defined as the tangible value of marketable timber that is present in a stand at the age when the value is required; Uys & Daugherty, 2000) is ~R5 900 per hectare. To meet the annual demand of 2 080 tons of charcoal, the landowner sells 12 480 tons of timber. This requires the felling of 32 hectares. Thus, the annual income, based on the standing value of one hectare is ~R190 000.year$^{-1}$.

*(3) Income from the sale of charcoal*

The charcoal produced sells for R2 per kilogram, equivalent to R2000 per ton. The charcoal producer makes >R1000 in profit per ton of charcoal, considering that production

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[5] This is stem biomass and excludes the leaf and branch components.

cost is just under R1000 (Table 6.2). The annual income (R4 160 000.year$^{-1}$) is the product of the price of charcoal and the annual production thereof (2 080 t).

*(4) Income from the sale of CERs*

In the case of the rehabilitated stands, potentially tradable carbon (carbon that may be put up for sale as CERs) comprises the carbon accumulating in all four stores (wood, understorey, litter and soil). Accumulation rate of tradable carbon is 7.69 tC.ha$^{-1}$.year$^{-1}$. This value is the weighted average of the accumulation rates depicted in Fig. 6.1. The weighted average was calculated using the equation: $\overline{Y}_w = (\overset{n}{\Sigma} w_i\ Y_i)/(\overset{n}{\Sigma}\ w_i)$ (Sokal & Rohlf, 1995); where $n$ is the number of stands or age classes, $w_i$ is the number of observations and $Y_i$ is average accumulation rate within each age class.

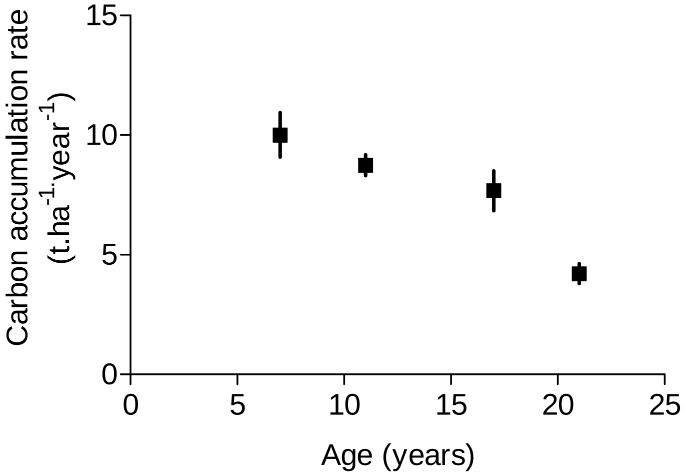

**Fig. 6.1.** The rate of accumulation of total tradable carbon (t.ha$^{-1}$year$^{-1}$) in the rehabilitated stands as a function of age.

Carbon currently sells for ~€20.tC$^{-1}$ (Carbon Market Daily, October 2005). Based on present exchange rates (1€ = R8), the landowner stands to make R1 230 per hectare from the rehabilitated stands, annually. The annual potential income from the sale of CERs, based on the 785 hectares that have been rehabilitated so far, is thus R965 550 per year.

*Expenditure and income analysis of the rehabilitation and revegetation programmes*

The annual expenditure on rehabilitation exceeds the potential annual income from the sale of CERs by about 5.3 million rands. Revegetation, on the other hand, is less costly as the cost thereof only exceeds the benefit by some 2.4 million rands (see Appendix 5).

## Discussion

Lubke & Avis (1998) estimated the cost of rehabilitating one hectare of mined dunes at between R25 000 and R30 000. This is about four times less than what I estimated in this study (R113 600). This discrepancy is presumably a result of changes in the economy since their study. In my study rehabilitation costs more than revegetation. Why should RBM comply with the environmental regulations imposed by the Tisand Lease if they are so costly? The only reason we would expect compliance is if the expected cost of non-compliance exceeded the cost of compliance (Cohen *et al.*, 1997). The benefits of compliance would include increased market valuation (McGuire *et al.*, 1988; Dowell *et al.*, 2000; King & Lenox, 2001; Adams & Zutshi, 2004). Environmental awards that recognise strong environmental performance result in a significant, positive change in market valuation (Klassen & McLaughlin, 1996). Moreover, there is strong evidence of a positive correlation between good environmental performance and financial gain (King & Lenox, 2003). Based on the above, good environmental performance, as demonstrated by RBM, can only be good for business.

The income generated through the sale of the products and by-products of the rehabilitation and revegetation programmes presumably benefit the local society. The landowner sells the *C. equisetifolia* trees to a contractor who makes charcoal on site. The timber owner earns revenue equal to standing value (~R190 000.year$^{-1}$). This accrues to the landowner and, through the relevant governance structures, should benefit the local society. The financial benefits of the charcoal industry (R4 160 000.year$^{-1}$) also accrue to the society because local people run the charcoal making plants.

A Clean Development Mechanism (CDM) office opened in Pretoria on 1 December 2004. The purpose of the CDM is to assist non-Annex I countries (such as South Africa) in achieving sustainable development and in contributing to the ultimate objective of the United Nations Framework Convention on Climate Change (UNFCCC), and to assist Parties included in Annex I in achieving their emission reduction commitments (UNIDO, 2003). Together with the Kyoto Protocol, which came into effect on 16 February 2005, this makes international carbon trading a reality. Money generated through the sale of CERs would presumably accrue to the landowner and subsequently to the local society. In order for carbon sequestered in any project to qualify as CERs, such a project must fulfil the additionality criterion. Additionality is the enhancement of greenhouse gas removals by Land Use, Land Use Change and Forestry activities that is additional to any that would occur in the absence of such a project (IPCC, 2000). This requires a demonstration that the carbon sequestered in the project is additional to that of a baseline. A baseline is the reference scenario against which a change in greenhouse gas removals is measured (IPCC, 2000). Usually, this is the business-as-usual scenario, the scenario before the start of the project. For my study area, this would be the situation before mining. Thus, for a more appropriate approximation of the potential income from the sale of CERs from the Tisand lease area, an assessment of the baseline would be required. That is, carbon sequestration would have to be quantified in an area similar to the rehabilitated portion of the lease area before it was mined. Subsequent to the demonstration that the present land use option sequesters more carbon than the baseline, the excess carbon sequestered would qualify as CERs. It seems unlikely that the rehabilitation programme could qualify for CERs. At most, the rehabilitated stands can grow back to a state where they store as much carbon as the intact dune forest and thus hopefully become carbon neutral.

The Kyoto Protocol recognises “forestry” as an activity that can be used to meet emission reduction commitments (IPCC, 2000). However, there is ambiguity regarding what constitutes “a forest”. Can I regard the commercial *C. equisetifolia* plantations in my study area as forests? The primary goal of establishing the plantations is to produce charcoal. Assuming the exotic plantations that existed before mining were not commercial and were therefore not harvested intensely, then they presumably stored more carbon in the long term than is the case presently. Thus, the revegetation programme does not fulfil the additionality criterion (IPCC, 2000). Consequently, I do not regard the carbon stored through the revegetation programme as CERs. If, however, the trees could either remain standing or be harvested to make more long-lived wood products, the carbon stored therein could be included in the carbon accounting system (MacDicken, 1997). Harvested wood materials are recognised as relevant carbon pools (Bateman & Lovett, 2000; IPCC, 2000). The use of renewable energy sources (e.g. biofuel) is also recognised as a valid means of curbing human-induced climate change (IPCC, 2001). Thus, using the charcoal to fuel the mining operation (as a biofuel substitute for fossil fuels) could also warrant its inclusion in the carbon accounting system.

In a given land area and time period, a full carbon accounting system would consist of a complete accounting for changes in carbon stocks across all carbon pools (IPCC, 2000). Applying full carbon accounting to an activity should, in principle, yield the net carbon exchange between terrestrial ecosystems and the atmosphere. Thus, in the case of RBM, the whole mining operation should be examined as a unit, instead of separating the post-mining activities into revegetation and rehabilitation.

It seems that revegetation makes more sense in financial terms, as it presents two income streams that, in combination, overshadow the moneymaking potential of the rehabilitation option, as assessed in this study. Then why does RBM not revegetate the entire mined area? The answer to this question lies in the stipulations of the Tisand lease and other considerations such as the direct and other indirect benefits that I did not quantify in this study. For instance, Wassenaar *et al.* (2005) demonstrated that the species diversity in the rehabilitated stands tends towards that of undisturbed forests. In this way, rehabilitation provides for the conservation of biological diversity. This land use option also provides for the extraction of a wide range of products (e.g. fruit, medicinal and edible plants) by the local people.

A cost-benefit analysis carried out for the rehabilitation of indigenous forests in Transkei (DWAF, 2001) estimated the net benefit at between R2040 and R18 790 per ha. This analysis included ten possible benefit streams. Had this analysis only included carbon sequestration benefits, the cost would have exceeded the net benefit by between R1 680 and R4920 per ha. Twine *et al.* (2003) and Shackleton & Shackleton (2004a; 2004b; 2005) quantified the extent and value of resource extraction in some rural areas of South Africa. Annually, this amounts to between R497 and R697 per person and about R1800 per household. Mander (1998) estimated trade in medicinal plant material alone to the value of R500 million per year. Lawes *et al.* (2004) give a comprehensive list of studies that quantify the use and value of resources from indigenous forests and woodlands. Were such values to be incorporated in the financial analysis carried out in this study, rehabilitation would probably make more financial sense than revegetation.

Considering that the input costs of rehabilitation are higher than those of revegetation, with the topsoil replacement step being the most expensive, perhaps the rehabilitated area could be expanded after the first *C. equisetifolia* rotation. The same seed mixture of pioneer plant species used to kick-start rehabilitation could be used. Because the presence of *C. equisetifolia* during the first rotation enriches the soil (Mailly & Margolis, 1992), there would probably be no need for the spreading of the topsoil. Ultimately, the cost of establishing a coastal dune forest through rehabilitation would be comparable to or less than that of re-establishing a C. equisetifolia plantation.

RBM's activities show their commitment to the concept of sustainable development. These activities have market competitiveness implications, because environmentally responsible companies are rated higher than their environmentally irresponsible counterparts. On the other hand, environmental responsibility is costly. I have given an account of this cost. I have also assessed some of the tangible benefits of the present rehabilitation/revegetation policy. The hypothesis that the financial costs and benefits of the two post-mining land use options are not different is rejected. A major shortcoming of this study was a lack of a baseline for a more appropriate "carbon market" assessment. This presents opportunities for further research.

# Chapter 7

# Synthesis

The genesis of this thesis was in a desktop study directed at evaluating the carbon sequestration consequences of the destruction of vegetation during mining and the subsequent rehabilitation and revegetation of mined dunes in Richards Bay, South Africa (van Aarde *et al.*, 2003). The study highlighted the need for a detailed assessment of carbon sequestration in the areas set aside for rehabilitation and revegetation as post-mining land use options. This assessment was needed to assist Richards Bay Minerals to evaluate the efficacy of the present 2:1 ratio of revegetation to rehabilitation, hence this present study. The mandate was to perform a detailed evaluation of carbon sequestration through the two post-mining land use options.

I undertook to relate the growth rate of *Acacia kosiensis* and *Casuarina equisetifolia*, the species dominating the two land use options, to rainfall. I did this in order to deduce the species' carbon sequestration potential. Initially, I assumed that tree-ring analysis (dendrochronology) would provide the answer. This assumption was based on the premise that as trees grow they add growth rings and the width of these rings reflects the trees' response to rainfall. Conceivably, a tree that puts on wider rings grows faster than one that does not. Subsequently, this faster-growing tree would sequester more carbon.

I found no relationship between apparent growth ring counts and rainfall. The samples of *A. kosiensis* and *C. equisetifolia* collected in Richards Bay were not suitable for dendrochronological analysis (Chapter 3). This is probably a result of a lack of extreme climatic conditions and distinct growing seasons. Trees growing in such conditions do not produce a ring structure that can be easily examined.

When dendrochronology did not work, I used statistical methods to model the growth of the species and to relate that to topographical variables. From the modelling exercise I was able to work out the growth rates of the aforementioned species. I then measured the actual carbon content of the woody tissue. I also measured carbon storage in three other recognised carbon stores (understorey, litter and soil).

*A. kosiensis* grew exponentially during the first 21 years while *C. equisetifolia* followed a sigmoidal growth curve and stopped growing around 13 years of age (Chapter 4). Tree age was the strongest predictor of growth in both species. The number and size of additional stems (multi-stemmedness) had a significant influence on *A. kosiensis* growth but had none on the growth of *C. equisetifolia*. Of the four topographical variables examined (slope, aspect, elevation and distance from an edge), only the last one had a weakly positive effect on the growth of *A. kosiensis*.

I found that the revegetated stands stored more carbon than the rehabilitated stands (Chapter 5). This contradicts the proposition that more biologically diverse ecosystems are more productive (Chapter 1). It is likely, though, that total carbon storage in the rehabilitated stands was underestimated, as it excluded carbon storage in secondary forest tree species. The herb layer and soil stores of the rehabilitated stands were bigger than those of the revegetated stands. This was a result of the higher herb layer biomass and supposedly a higher litter decomposition rate in the rehabilitated stands.

The higher carbon sequestration potential of the revegetated stands was largely a result of the bigger wood and litter components thereof (Chapter 5). If, however, we consider that most of the carbon stored in the wood, which constitutes ~90% of the total carbon pool, is released back into the atmosphere as charcoal is burnt, it becomes clear that the revegetated stands are not effective as long term carbon stores. Moreover, as shown in Chapter 4, active growth in *C. equisetifolia* ceases after 13 years of age. Thus, after this age, the species does not sink any additional carbon. *A. kosiensis*, on the other hand, continues to grow exponentially and sink carbon, at least throughout the age range covered in this study (5 to 22 years). Fig. 7.1 illustrates the different life history traits of the two species.

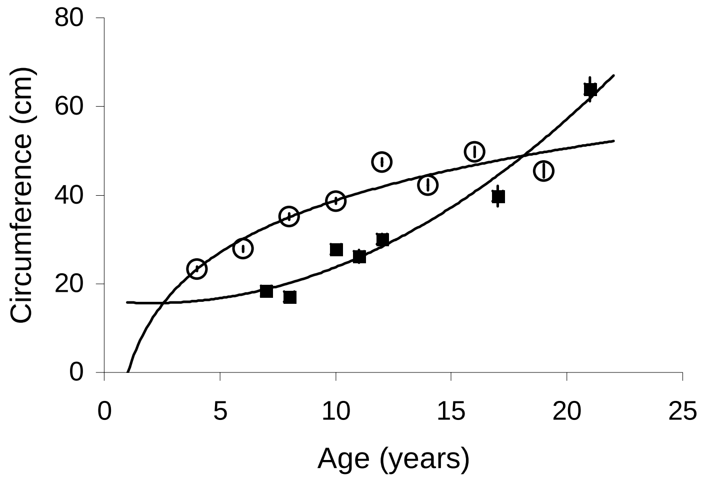

**Fig. 7.1.** Circumference, expressed as mean ± SE, of *A. kosiensis* (dark squares, n = 540) and *C. equisetifolia* (open circles, n = 640) as a function of age. Trend lines illustrate the different life history traits of the species.

The amount of carbon sunk and stored in trees is not only a function of their size, but a combination of size and mass. Thus, even though *C. equisetifolia* does not capture any additional carbon after 13 years of age, the carbon stored therein is still more than that stored in *A. kosiensis* because generally the former weighs more than the latter (Chapter 5). But, *A. kosiensis* continues to attain mass as it grows in size. Thus, beyond 19 years, the age at which the mass accumulation curves intersect (Fig. 7.2), *A. kosiensis* becomes heavier than the heaviest (16 year-old) *C. equisetifolia*.

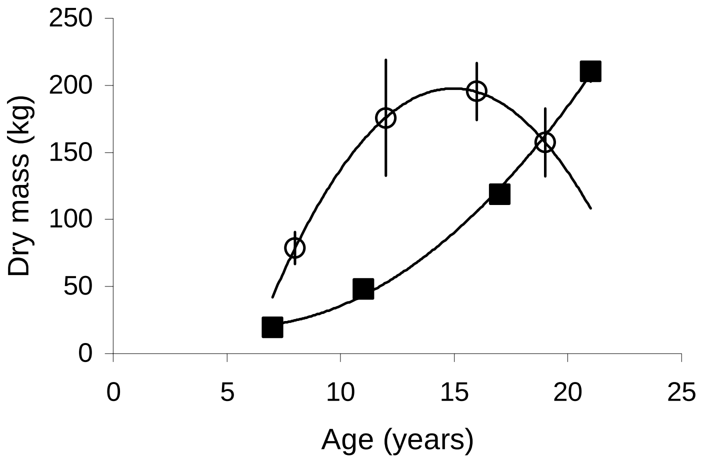

**Fig. 7.2.** Dry mass, expressed as mean ± SE, of *A. kosiensis* (dark squares, n = 16) and *C. equisetifolia* (open circles, n = 16) as a function of stand age. Trend lines were fitted for illustration.

The combination of the two factors, increasing size and mass with age, renders *A. kosiensis* a better species for carbon sequestration in the long term than *C. equisetifolia*. Therefore, maximum long-term carbon sequestration in the lease area can, in the future, be attained by focusing only on rehabilitation. This may not be a feasible, or even readily acceptable option, as the charcoal industry is an important source of income for the local people. Income generation through the sale of carbon credits, however, may serve as an incentive for this alteration of ratios. Moreover, coastal dune forests as an ecotype within the Maputaland centre of endemism (MCE) have a special conservation value (van Wyk, 1994). Thus, rehabilitating the entire RBM lease area would contribute to biodiversity conservation. Woolley (2003) concluded that coastal dune forest is very rare on a landscape level, making up 0.83% of the total land surface area of the MCE. Indeed, in KwaZulu-Natal this ecotype is constrained to narrow strips along the eastern shore. The narrowness of these strips of forest may render them vulnerable to edge effects.

If the coastal dune forest ecotype is to be protected from future threats, one approach would be to increase the size of the existing strips. This can be done by rehabilitating a bigger portion of the mined area. If perhaps the whole lease area could be rehabilitated to a continuous stretch of coastal dune forest, the threat of edge effects could be minimised. Conceivably, this would compromise the income-generation potential of the lease area, as the charcoal industry is more lucrative than the potential trade in carbon credits (Chapter 6).

In Chapter 6 I demonstrated that rehabilitation is more costly than revegetation. Also, in terms of directly measurable financial benefits, the former land-use option is less appealing. If, however, other direct benefits that are derivable from the consumptive use of natural resources (e.g. fruit, fuel, etc.) and intangible benefits (e.g. watershed protection, soil stabilisation, micro-climate regulation, etc.) are considered, rehabilitation may be the better option. There is, therefore, a need for a detailed cost-benefit analysis that will incorporate all potential sources of income in order to determine the true financial potential of the two post-mining land use options.

The question remains: how much carbon is released during the clearing of vegetation in front of the mine? The answer to this question will determine the baseline for carbon accounting purposes. In turn, this will determine whether RBM will be in carbon balance, deficit or surplus at the end of its operations. Thus, there is a need for a detailed evaluation of carbon storage in an undisturbed coastal dune forest, the equivalent of a baseline. Knowing the total area that will have been mined by the end of the lease period, one can work out how much carbon will have been released in total. This can then be compared to the amount that will have been recaptured through the rehabilitation and revegetation programme. Also, total biomass accrual by emergent forest species in the rehabilitated stands is unknown. This may be substantial and may mean that the value reported in this study for total carbon storage in the rehabilitated stands is an underestimate.

The main objective of this study, which was to compare the carbon sequestration potential of a native regenerating coastal dune forest and exotic plantations, was achieved. However, this was done for only the first few years (21 and 19 respectively). Because indeterminate changes can occur beyond the ages covered here, the future carbon storage potential of these ecosystems is uncertain. Therefore, a few pertinent questions still remain, most important of which is the question of how much carbon an indigenous coastal dune forest stores. Because of the structural complexity of this ecosystem, it would be a task beyond the scope of this study to answer such a question. To address it, and a few other questions arising from this study, further studies are required.

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## Summary

Rehabilitation and revegetation are the two post-mining land use options that follow after mining by Richards Bay Minerals (RBM) along the coastal sand dunes of Richards Bay. The outcome of rehabilitation is potentially a secondary coastal dune forest. Revegetation, on the other hand, is aimed at establishing beefwood plantations for the development of a local charcoal industry. Both post-mining land-use options have the potential to sequester carbon.

Carbon sequestration in rehabilitated and revegetated stands was quantified. The soil and herb layer stores of the rehabilitated stands were larger, and the litter and wood stores smaller, than those of the revegetated stands. Overall, the revegetated stands stored more carbon than the rehabilitated stands. Revegetated stands may be more efficient at carbon sequestration, but the aim of this land use option in the first place is to cut the trees down for charcoal-thus releasing all stored carbon. In terms of long-term carbon storage, therefore, rehabilitation is more efficient. Total carbon storage in rehabilitated stands may have been underestimated, as storage in emergent forest tree species was not determined. For long-term carbon sequestration, the present ratio of revegetation to rehabilitation is not optimal.

Rehabilitation costs more than revegetation. The financial benefits of revegetation, through the sale of timber and charcoal, are more than the potential financial benefits of rehabilitation (at least the ones addressed in this study). The true financial potential of the two land-use options can be determined through a detailed cost-benefit analysis.

## Appendix 1: The calculation of composite stem circumference, based on the assumption that a stem is circular in cross section

1. Circumference (C) at breast height was measured in the field
2. The circumference was converted to radius (r) using the equation: $C = 2\pi r$
3. The radius values were converted to area (A) using the equation: $A = \pi r^2$
4. A sum of the areas ($\sum A$) of all the stems, excluding the largest stem, of a single tree was used to calculate diameter (D) using the equation: $D = 2(\sqrt{(A/\pi)})$
5. Composite circumference (C*) was calculated using the equation: $C^* = \pi D$

Note: Composite circumference (C*) for single-stemmed trees is equal to zero because the sum of areas of additional stems ($\sum A$) is zero.

## Appendix 2: The calculation of biomass contributed by multi-stemmed trees to total biomass

1. Circumference (C) at breast height was measured in the field
2. The circumference was converted to radius (r) using the equation: $r = C/2\pi$
3. The radius values were converted to basal area (A) using the equation: $A = \pi r^2$
4. Total basal area ($\sum A$) for each species was calculated by summing the areas of all the stems of that particular species.
5. The proportion of biomass (bm) contributed by multi-stemmed trees was calculated by dividing the sum of the areas of the stems of multi-stemmed trees ($\sum A_m$) by the total basal area ($\sum A$).

## Appendix 3: The volume of stems with different shapes is different

For illustration I use a hypothetical stem with basal area (BA) = 0.027 $m^2$, and height (H) = 15.8 m. The calculated volumes are:

1. For a conical stem

   V = (BA x H)/3

   = (0.027 $m^2$ x 15.8 m)/3

   = 0.14 $m^3$

2. For a paraboloid stem

   V = (BA x H)/2

   = (0.027 $m^2$ x 15.8 m)/2

   = 0.21 $m^3$

3. For a cylindrical stem

   V = BA x H

   = 0.027 $m^2$ x 15.8 m

   = 0.42 $m^3$

Values of carbon storage in stems with different shapes are calculated in detail in Appendix 4.

## Appendix 4: Stems of different shape yield different values of carbon storage in wood

Given:

- 3 trees with conical, paraboloid and cylindrical stems, respectively, each with a volume (V) of 0.14, 0.21 and 0.42 $m^3$ (as calculated in Appendix 3)
- % carbon of the wood = 52.29
- wood density (D) = 818.80 kg.m-$^3$

Task:

Calculate the amount of carbon (kgC) stored in the wood of each of the trees

Method:

1. For the tree with a conical stem

Mass (M) = V x D

= 0.14 $m^3$ x 818.80 kg.m-$^3$

= 114.63 kg

Carbon in wood = 114.63 kg x 0.52

= 59.94 kgC

2. For the tree with the paraboloid stem

Mass (M) = V x D

= 0.21 $m^3$ x 818.80 kg.m-$^3$

= 171.95 kg

Carbon in wood = 171.95 kg x 0.52

= 89.91 kgC

3. For the tree with the cylindrical stem

Mass (M) = V x D

= 0.42 $m^3$ x 818.80 kg.m-$^3$

= 343.89 kg

Carbon in wood = 343.89 kg x 0.52

= 179.81 kgC

## Appendix 5: Costs involved in rehabilitation, revegetation and charcoal production (approximate values have been used)

### *1. Cost of rearing C. equisetifolia seedlings*

Four full-time workers work in the nursery. They work six days a week for nine hours a day at an hourly rate of R4 per worker. Assuming that they work 52 weeks a year and given that 128 hectares were revegetated in 2004 (this supposes that in a particular year, the nursery only rears seedlings that will be enough to use in revegetation for that particular year), I worked out the per hectare cost of running the nursery as follows:

4 workers x R4.hour$^{-1}$.worker$^{-1}$ = R16.hour$^{-1}$
R16.hour$^{-1}$ x 9 hours.day$^{-1}$ = R144.day$^{-1}$
R144.day$^{-1}$ x 6 days.week$^{-1}$ = R864.week$^{-1}$
R864.week$^{-1}$ x 52 weeks.year$^{-1}$= R44 928.year$^{-1}$
R44 928.year$^{-1}$ / 128 ha.year$^{-1}$ = R351.ha$^{-1}$

### *2. Cost of cattle guarding*

Ten and three workers during the week and on weekends, respectively, patrol the rehabilitated stands to keep livestock out. In total, the rehabilitated stands cover 785 hectares. Optimally, all 785 hectares should be patrolled on any particular day. Thus, during the week the cost of guarding is:

9 hours.day$^{-1}$ x R4.hour$^{-1}$.worker$^{-1}$ = R36.day$^{-1}$.worker$^{-1}$
R36.day$^{-1}$.worker$^{-1}$ x 10 workers = R360.day$^{-1}$
R360.day$^{-1}$ / 785 ha.day$^{-1}$ = R0.46.ha$^{-1}$

On weekends, the cost is:

9 hours.day$^{-1}$ x R4.hour$^{-1}$.worker$^{-1}$ = R36.day$^{-1}$.worker$^{-1}$
R36.day$^{-1}$.worker$^{-1}$ x 3 workers = R108.day$^{-1}$
R108.day$^{-1}$ / 785 ha.day$^{-1}$ = R0.14.ha$^{-1}$

If R0.46.ha$^{-1}$ is paid out for cattle guarding 71% of the time (5 days out of seven) and R0.14.ha$^{-1}$ is paid out the remaining 29%, then weighted cost of guarding is:

(R0.46.ha$^{-1}$ x 0.71) + (R0.14.ha$^{-1}$ x 0.29)
= R0.33.ha$^{-1}$ + R0.04.ha$^{-1}$
= R0.37.ha$^{-1}$

*3. Annual cost of rehabilitation and revegetation*

I calculated the annual cost of rehabilitation and revegetation for the year 2004 as the total area rehabilitated or revegetated in 2004 multiplied by the total cost per hectare.

= 55 $\text{ha.year}^{-1}$ x R113 600.$\text{ha}^{-1}$

= 6 248 000 ≈ R6 250 000.$\text{year}^{-1}$

In comparison, the annual cost of revegetation is:

= 128 $\text{ha.year}^{-1}$ x R37 751.$\text{ha}^{-1}$

= 4 832 128 ≈ R4 830 000.$\text{year}^{-1}$

*4. Cost of producing charcoal*

Approximately 40 tons of charcoal are produced weekly. The following steps outline the costs of meeting this demand.

*(a) Timber purchase*

The landowner sells timber for R15 per ton. Six tons of timber are needed to produce one ton of charcoal. Thus, the cost of timber for the production of one ton of charcoal is:R15.$\text{t}^{-1}$ x 6 = R90.$\text{t}^{-1}$

*(b) Timber harvesting*

The charcoal producer hires a chainsaw operator and his assistant for R1000 per day. He can cut six trees per hour (*pers. obs.).* The stem of a 16 year-old tree (only stems are used to make charcoal), on average, weighs 220 kg. Thus, if the chainsaw operator works for nine hours a day, he harvests approximately 12 tons of timber to make charcoal. However, charcoal production efficiency is 1:6 (see point 3a above). Thus, the chainsaw operator harvests the equivalent of 2 tons of charcoal a day. I calculated the cost as follows:

6 $\text{trees.hour}^{-1}$ x 9 $\text{hours.day}^{-1}$ = 54 $\text{trees.day}^{-1}$

54 $\text{trees.day}^{-1}$ x 220 $\text{kg.tree}^{-1}$ = 11880 $\text{kg.day}^{-1}$ ≈ 12 $\text{t.day}^{-1}$ (timber)

12 $\text{t.day}^{-1}$ (timber) = 2 $\text{t.day}^{-1}$ (charcoal)

Therefore, cost is:

R1000.$\text{day}^{-1}$ / 2 $\text{t.day}^{-1}$ = R500.$\text{t}^{-1}$

*(c) Timber transportation*

(i) To meet the demand of 40 tons of charcoal produced weekly, 240 tons of timber are transported to the kilns per week. There are two tractors used for transportation (Mr Vickus, *pers. comm.*)[7]. I made the following assumptions regarding the tractors:

- they are Leyland 154 farm tractors with an engine capacity of 1500cc
- the tractors run on diesel, priced at R3.49.litre$^{-1}$ (2004 price, http://www.aa.co.za)
- they each pull a single axle trailer with a load capacity of two tons
- each one transports 120 tons of timber a week
- thus, each tractor makes 60 trips of approximately 20 km each per week

Using the above parameters, the average running cost (includes fuel and maintenance costs) for each of the tractors was calculated by the AA online rates calculator (http://www.aa.co.za/live/ratescalc.php) as 65 cents.km$^{-1}$.

Total distance travelled by both trucks in one week is:

20 km.trip$^{-1}$ x 60 trips.week$^{-1}$ x 2 = 2400 km.week$^{-1}$

The total running cost per week is:

2400 km.week$^{-1}$ x 65 cents.km$^{-1}$ = 156 000 cents.week$^{-1}$

= R1 560.week$^{-1}$

Total running cost per ton of charcoal produced is:

R1 560.week$^{-1}$ / 40 t.week$^{-1}$

= R39.ton$^{-1}$

(ii) I made the following assumptions regarding the labour costs associated with the transportation of timber:

- there's one driver for each of the two tractors
- there are two assistants to help with the loading and offloading of the timber
- the six people each work for nine hours a day for six days a week

Thus, the labour costs are:

6 workers x R4.hour$^{-1}$.worker$^{-1}$ = R24.hour$^{-1}$

R24.hour$^{-1}$ x 9 hours.day$^{-1}$ = R216.day$^{-1}$

R216.day$^{-1}$ x 6 days.week$^{-1}$ = R1 296.week$^{-1}$

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[7] Mr Vickus is the owner of the charcoal making plant on the RBM lease area.

R1 296.week$^{-1}$ / 40 t.week$^{-1}$ = R32.4.t$^{-1}$

The total cost (R.t$^{-1}$) of transporting timber is the sum of vehicle running cost and labour cost:

39 + 32.4 = 71.4

*(d) Carbonisation and packaging*

The charcoal plant produces approximately 8000 bags (5 kg each) a week. Forty people work in the operation. They work for six days a week and for nine hours each day at a rate of four rands per hour per worker. I calculated this cost as:

40 workers x R4.hour$^{-1}$.worker$^{-1}$ = R160.hour$^{-1}$

R160.hour$^{-1}$ x 9 hours.day $^{-1}$ = R1 440.day$^{-1}$

R1 440.day $^{-1}$ x 6 days.week$^{-1}$ = R8 640.week$^{-1}$

R8 640.week$^{-1}$ / 40 t.week$^{-1}$ = R216.t$^{-1}$

*(e) Transportation of charcoal to point of sale*

(i) I made the following assumptions concerning vehicle running costs associated with the transportation of charcoal from the kilns to the point of sale in Richards Bay:

- charcoal is transported by means of a Mitsubishi Canter SE5-106 1.5 ton mini-truck with an engine capacity of 2 835 cc
- the vehicle runs on diesel priced at R3.49.litre$^{-1}$ (http://www.aa.co.za)

Using these parameters, I calculated the average running cost for the mini-truck using the AA online rates calculator (http://www.aa.co.za/live/ratescalc.php) as 90 cents.km$^{-1}$.

The mini-truck makes, on average, 27 trips of 54 km each per week to deliver 40 tons of charcoal. The weekly distance travelled is thus 1458 km.

Running cost, corrected for the amount of charcoal transported is:

(1 458 km x 90 cents.km$^{-1}$) / 40 t = 3 280.5 cents.t$^{-1}$

= R32.81.t$^{-1}$ ≈ R33.t$^{-1}$

(ii) The labour costs associated with the transportation of charcoal are based on the assumption that one driver operates the vehicle with two assistants. Thus, the cost is:

3 workers x R4.hour$^{-1}$.worker$^{-1}$ = R12.hour$^{-1}$

R12.hour$^{-1}$ x 9 hours.day $^{-1}$ = R108.day$^{-1}$

R108.day $^{-1}$ x 6 days.week$^{-1}$ = R648.week$^{-1}$

$R648.week^{-1} / 40\ t.week^{-1} = R16.2.t^{-1}$

The total cost ($R.t^{-1}$) of transporting charcoal is the sum of the vehicle running cost and labour cost: $33 + 16.2 = 49.2 \approx 49$

*5. Expenditure and income analysis of the rehabilitation and revegetation programmes.*

(i) For rehabilitation, the difference between expenditure and income is:

annual rehabilitation cost – annual income from the sale of CERs

= 6 250 000 – 965 550

= 5 284 450 ≈ $R5\ 300\ 000.year^{-1}$

(ii) For revegetation, there are two streams of expenditure and income. The difference between cost and income is thus:

(annual revegetation cost + annual charcoal production costs) – (annual timber sale income + annual charcoal sale income)

= (4 810 000 + 1 920 000) – (190 000 + 4 160 000)

= 6 730 000 – 4 350 000

= $R2\ 380\ 000.year^{-1}$