

**Resource Quantification, Use and Sustainable
Management of Coastal Forests in the
Eastern Cape Province.**

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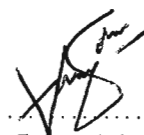
PREFACE

The work presented in this thesis is the original work of the author and has not been submitted in any form for a degree to any other institution. Where use has been made of the work of others it has been duly acknowledged in the text.

The study was undertaken between June 1998 and November 2001 through the Forest Biodiversity Programme, School of Botany and Zoology of the University of Natal, under the supervision of Prof. Michael J. Lawes.

This thesis has been compiled as a collection of papers (apart from the introduction – chapter I and the conclusion – chapter VII). Each chapter is written in accordance with the required format for the particular journal it has been published in or submitted to (except that tables and figures are in the text and not on separate sheets at the end of each chapter). The chapters also depart from the format used in the journals by having one acknowledgements section. Chapters will be referred to in the text by their roman numerals (i.e. I to VII).

Signed.....



John A.F. Obiri

ABSTRACT

Indigenous forests of South Africa are few, small in size and highly fragmented, yet they face intense exploitation particularly in the rural areas. Management of these forests is challenging. High rural dependency on forests and the need to ensure the maintenance of the forest ecological processes that maintain biodiversity and ecosystem integrity are at odds with one another. Rural needs from forests are mainly short-term and interfere with the longer cycle of ecosystem maintenance. In Umzimvubu District of Eastern Cape Province forest management through sustainable use is hampered by a lack of information about the forests' ecology, resource availability and exploitation patterns. Thus it is difficult to set quotas or sustainable harvesting levels. This study addresses these challenges by tackling crosscutting themes of (1) forest policy and use, (2) forest resource availability and exploitation and, (3) the ecological processes of forest regeneration - all vital components for sustainable forest management.

Using questionnaires it was found that all forest stakeholders (i.e. forest resource users and managers) were opposed to a ban on forest resource use but agreed to regulated harvesting. Although the new forest policy advocates the devolution of forest management from the state to communities, resource users preferred a greater role for the state in forest management than expected. Given the choice stakeholders selected state forest management over community forest management. However, the combination of roles of the state and communities in forest management, as exemplified by the new policy of participatory forest management (PFM) is probably the most applicable management practice, although it is not without its problems.

Tree species are the focus of this study. Trees were largely used for fuelwood, medicinal purposes, craftwork, fencing posts and building poles and involved twenty species. Poles and posts were indiscriminately harvested from the medium (10-20cm dbh) tree size-class. Fuelwood harvesting was selective and only certain species were used. Fuelwood harvesting is unsustainable because the average amount of deadwood produced by the forest marginally balanced that removed from forest as fuelwood. Similarly medicinal tree harvesting (largely through stem debarking) was unsustainable and at least 28% of the debarked trees died. Only one species (*Macaranga capensis*) could withstand the current stem bark harvesting pressure.

Species suitable for pole and post harvesting were determined by a graphic method, based on linear-programming approach that examined the spatial scale or grain of regeneration of a species. The grain of a species is established by comparing the density of stems from a species at the forest canopy and sub-canopy levels and sanctions harvesting only if a species was adequately represented at both levels i.e., fine-grained. Only one species (*Englerophytum natalense*) met these requirements in all

forests and could sustain high levels of pole and post exploitation. *Harpephyllum caffrum* and *Heywoodia lucens* are among the most coarse-grained species and their use is discouraged.

A relatively high percentage of the forest is under gaps (7.8%) created via natural disturbances of windthrow (50%), breaking tree branches (20%) and snags (13%). Another 17% result from selective tree cutting activities. The gap-phase dynamics paradigm appears to play a minor role in forest tree regeneration, as gap-size niche-differentiation is weak and there is no gap-filling guild of pioneer species. A lottery paradigm best explains tree regeneration in gaps in the forests of Umzimvubu District. Although selective tree harvesting creates gaps, in moderation gap creation is unlikely to change the forests' species composition since there is no gap-filling guild that is favoured by gaps and recruitment is a chance event.

Resource use in the forests of Umzimvubu District is unsustainable and PFM offers a viable option for managing these forests. Sustainable use of forest cannot be achieved without an integration of the multifaceted social and ecological issues of forest management and more importantly without prioritising and understanding the ecology of forests.

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This study would probably not have been possible in the first place had it not been for the encouragement from my parents (Malova Obiri and Terry Mbone) and one elder of the Avamuruga people – Hezron Janira. I greatly thank them. I also thank three families in Pietermaritzburg (The Calders, Nsahlais and Shands) who enriched my stay in South Africa.

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CHAPTER I

INTRODUCTION

CHAPTER I**INTRODUCTION:****FOREST MANAGEMENT IN A NEW ERA IN SOUTH AFRICA**

The indigenous forests of South Africa are small, highly fragmented and face high levels of human exploitation. In Umzimvubu District of South Africa, where this study was undertaken, local communities are poor and highly dependent on forest products, which range from medicines, fuelwood, posts, craftwork raw materials and construction poles. Attempts have been made by the state to use these forests judiciously by combining their conservation with community development or community based natural resource management. Though of good intent, these attempts have focussed more on addressing the short-term socio-economic needs of people (i.e. social sustainability) and less on the long-term survival of the forests. While it is necessary to address peoples' needs, meaningful sustainable use of forest has first to consider the ecological processes that drive the forests' biodiversity and ecosystem integrity, i.e. ecological sustainability, since it is through these processes that forests are established and maintained. Furthermore, the availability of resources and their appropriate off-take levels with respect to forest integrity need to be quantified. This study emphasises an integrated ecological and socio-economic approach to forest management. It seeks to investigate how natural forests of Umzimvubu District can be sustainably managed to address the ecological and social sustainability of the forests and its people, respectively.

Sustainability in Forestry

The sustainability of forest resources and forest ecosystems has been debated almost from the beginning of modern forest management. The issues have undergone numerous transformations. Ecologists and environmental scientists stress the maintenance of forest ecosystems in the face of different types of human interventions,

such as the harvesting of forest products, land conversion, and air and water pollution. The view of socio-economists, politicians, and commercial forest managers of forests is much more utilitarian and emphasises the need for forests to provide human societies with highly valued goods and services, even though often forests may not have the capacity to sustain this use (see Clark 1995). In many respects the scientific community has helped to perpetuate an illusion of sustainable development through scientific and technological progress, resulting in a misguided view among political leaders of the dynamics of resource exploitation (Ludwig *et al.* 1993). In most instances resource problems are not really environmental problems but are human problems created repeatedly under a variety of political, social and economic systems. For successful forest management to occur the concerns of people and the environment must be harmonised. This requires an understanding of the functions and interactions of various forest components and systems at different temporal and spatial scales, as well as an appreciation of the role of local people in conservation. This partnership between professional scientists and local communities is widely regarded as the way forward for conservation (Getz *et al.* 1999), although the efficacy of community-based conservation projects continues to receive mixed reviews (e.g. Clark 1995, Dickinson *et al.* 1996, Getz *et al.* 1999, Attwell and Cotterill 2000).

Although the concept of sustainability has been a guiding principle in forestry since the 18th century when it was explained as the principle of sustained yield (Rubner 1992), the precise meaning and operational definition of the concept remains surprisingly elusive. Difficulties with the concept of sustainability arise from the diversity of its meanings to different stakeholders and, in the case of forests, the complexity of the biological system being managed (Ludwig *et al.* 1993, Becker and Ostrom 1995). The application of the concept of sustainability to forest management depends on whether attention is focussed on the conservation of forest ecosystems, maintenance of human systems that are forestry dependent, the maintenance of a dominant product or product mix or the sustenance of production capacity (Gale and Cordray 1991). The latter have been grouped into the ecological and social dimensions of forest sustainability (Table 1). In this study I am especially interested in the conservation and maintenance of forest processes and the influence of the use of wood products on these.

Throughout the 19th century and up to the middle of the 20th century, forest management focussed on regulating yield or sustained yield management (Parry *et al.* 1983). Forests were managed to maximize timber supply with little regard to the soil or other important factors involving the integrity of forest ecosystems. However, by the middle of the 20th century it was apparent that the supply-oriented concept was no longer appropriate because forests could not sustain long-term timber production. It also became apparent that replacement of forests with intensive plantation forestry removed some elements of forest structure, composition and function that were vital for the long-term productivity and survival of indigenous forests (Parry *et al.* 1983, Perry *et al.* 1989). Thus, the principle of sustained yield changed such that it became interpreted as not only incorporating the maintenance of a certain output, but also the maintenance of production capacity in forest lands and / or the maintenance of the natural regenerative capacity of forests (Maini 1992). This interpretation was later broadened to include the concept of multiple-use. Although of good intent, the sustained yield principle was based on a linear economic assumption that was at odds with ecological reality (Maser 1994). The assumption being that the essential elements required by the forest (e.g. soil and its nutrients, water, air and sunlight, climate) remained constant while the forest output and extraction of products was maximized. This linear thinking ignored the cyclic nature of ecological reality (e.g. the fluctuation of essential elements) while attempting to maximize short-term gains from a system that was fundamentally controlled by ecological laws. Since the 1970s the multiple-use sustained yield concept was gradually replaced by the more inclusive concept of sustainable forest use. This concept advocates the management of forest ecological systems to sustain their capacity to remain reproductive in perpetuity, with resource extraction levels being set within this context. The difficulty of applying this concept in modern society is that the context is often ignored and the demand exceeds supply leading to ecological collapse followed by economic and social instability in even marginally forest dependent communities.

In developing countries, especially Africa, much attention has been given to the enhancement of the social dimension of the concept of sustainability (Alcorn 1996,

Lebbie and Freudenberg 1996, Peters 1996, Mortimore 1998). Features of sustainable forest management have been identified in the tropics and subtropical regions that have received little attention in the temperate regions. For instance, in tropical and subtropical regions the involvement of communities living close to forests in the mainstream of forest management is regarded as imperative for sustainable forest use. Consequently, the concept of social or community forestry has gradually been preferred (Arnold 1991, Poffenberger and McGean 1998).

Table 1: Ecological and socio-economic factors implicit in the concept of sustainable forestry (after Wiersum 1995)

1.	Maintenance of forest ecological factors ^a
	<ul style="list-style-type: none"> • Maintenance of biodiversity and natural forest ecological processes • Maintenance of production capacity of forest soils • Maintenance of vegetative renewal capacity
2.	Maintenance of yields of vital forest goods and services for human benefits
	<ul style="list-style-type: none"> • Maintenance of production of dominant commercial good • Maintenance of ecological benefits in relation to non-forest areas • Maintenance of production mix of diverse products and services for human benefits • Maintenance of production of goods for those population who depend on forest for basic needs
3.	Sustenance of human institutions that are forest-dependent
	<ul style="list-style-type: none"> • Maintenance of cultural integrity of tribal communities • Maintenance of labour- and income-generating benefits derived from forests
4.	Sustenance of human institutions that ensure forests are protected against negative external institution
	<ul style="list-style-type: none"> • Maintenance of involvement of local forest users in forest management • Maintenance of proper socio-economic conditions for communities living near forests. • Maintenance of effective legal and organisational frameworks for forest protection

^aThe understanding of the meaning of the ecological factors depends on the temporal and spatial scales considered (Fresco and Kroonenberg 1992, Cox and Moore 2000).

Community forestry management involves the active protection of a forest area and regulation of its use by a community. In many cases, a community's primary goal in protecting a forest area is to generate products for subsistence use, but in some cases, particularly in India (Conroy *et al.* 2000), income generation is the main objective. In the latter case, income from the sale of forest products is placed in a common fund and used to finance community activities such as cultural and religious festivals or a school construction project. Where subsistence needs are required, for example, one or more households may need poles to repair a home, a village committee comprised of elders (i.e. *panyachat* in India, Poffenberger and McGean 1998) decides whose needs are

greatest and how much wood can be removed. This community management among other types of management is discussed later in chapter II.

The preference for community forestry is not only associated with the socio-economic development of rural communities but is also linked to the notion, held mostly by conservationists, that forests resources are best maintained if their management involves local community participation (Getz *et al.* 1999). The relationship between community management and other management options is dealt with in the next two sections.

Ecological limits, social dynamics and forest sustainability

The norms of sustainable forest management are based on ecological characteristics, social characteristics and the reciprocal relationships between these two categories (Falconer 1995). In South Africa the extent or limit to which social and/or ecological characteristics affect forest sustainability is dependent on whether the concerned forests are under state or community authority (Cawe 1992, Shackleton 1993, DWAF 1997, Castley and Kerley 1996, Obiri *et al.* in press; see Chapter III). Intensities of resource exploitation and thus forest sustainability differ in state and community forests (see Chapters III and IV). A fundamental difficulty in understanding forest sustainability in such interacting systems is that ecological and social dimensions of sustainability are subject to different time scales of management (Sutherland and Reynolds 1998). While long-term processes dominate forest ecosystems, social systems are predominately driven by short-term factors that fluctuate with the changing needs of man. Inevitably, the two systems come into conflict since societal demands structured around short-term gains interfere with the longer cycle of ecosystem maintenance. There are no indications of how these challenges can be resolved in a scientifically rational way (Kenan *et al.* 1999, Lackey 1999). Instead, it is becoming increasingly clear that values determining an individual or a society's concept of sustainability are diverse and largely depend on the society's fundamental needs. In South Africa the government, through the revised Forest Act No. 84 (National Forest Act 1998) and the National Forest Action Programme (DWAF 1997), has established new policies that stress the participation of rural communities in forest management. This is seen as the best way

of achieving sustainable development and conservation or conservation based community development. Under these policies, the achievement of sustainable forest management is envisaged to ultimately depend on reconciling socio-economic and conservation needs. The question is, given the significant socio-economic demands on indigenous forests, will community management ensure sustainable forestry? Stated another way, can the local communities meet their needs without compromising those of forests (i.e. basic ecological needs)? In the past this would have been easily answered since community needs were largely at the subsistence level (Johnson 1983), but today forests are viewed as major sources of income and indeed they are (e.g. in South Africa an annual US\$61 million worth of traditional medicines are extracted from forests, woodlands and grassland; Mander 1998). The problem is that these economic returns are promoted in isolation of any consideration of whether forests can sustain these economic yields – these figures may be a flash in the pan and create short-term incentives for detrimental harvests from forest environment. A further question is, how does community management compare to other forms of forest management? Chapter II further explores these issues and consider peoples' attitudes toward, and knowledge of, forest management systems and resource use in Umzimvubu District of the Eastern Cape Province of South Africa.

The three faces of sustainable forest management in South Africa.

The debate over which policies and management practices are ideal for implementing sustainable forestry has been confusing (Yade 1997, Grundy 2000, Neil 2000), in part because people truly mean different things when they use the term sustainable management. These different meanings reflect the different goals, interests, knowledge and values of different people (Van der Zel 2000). Fundamentally, there are three views of sustainable forest management in South Africa; community, state and participatory forest management.

Community Forest Management (CFM)

This view considers community management as the key to sustainable forests. This view, advocated mostly by resource users and politicians, takes an anthropocentric perspective and seeks to foster multiple human uses of forest resources. In CFM the decision-making is solely by the community that develops by-laws regulating the use and protection of resources with limited or no participation of the state (DWAF 1997). CFM has largely arisen over the last three decades following the disappointing results of natural resource conservation under state-centred policies (Mark 1984, Ostrom 1990, Poffenberger 1990, Blockhus *et al.* 1992, Ascher 1995, Sarin 1998). These authors offer differing lists of conditions considered necessary for successful functioning of CFM however most analysis include three fundamental requirements. First, a community must highly value a natural resource if it has to sustainably manage the resource. Second, in order to benefit from managing a resource, property rights must be devolved to individuals who use the resource. Third, the individuals or communities using the resource must have the ability to establish micro-institutions that regulate the resource use.

Successes of CFM have been reported in several different studies. For instance in southern Africa Wily *et al.* (2000) report on village forest reserves (VFR) that have been established throughout Tanzania. These VFR are run by village governments that pass by-laws on the regulation and use of resources within the VFR and mobilise the local people to police forests. In Bolivia, South America, Becker and Leon (2000) suggest that a significant amount of the expansive Yuracare forest and its biological diversity still benefit from traditional rules for human activities in the forest. However, some studies have doubted the effectiveness of CFM and particularly the notion that forests can be managed and conserved through indigenous knowledge systems (see Attwell and Cotterill 2000) or through indigenous institutions (Gibson and Baker 2000). But again some communities who have shown long-standing awareness of the value of ecosystems and habitats in their socio-cultural systems have reflected useful resource management initiatives. Examples are communities with sacred groves such as the Kaya forests of the Digo people in coastal Kenya (Nyamweru 1996) and Thathe

forest of the Venda in Limpopo Province of South Africa (Sikhitha 1999). However, a significant problem with CFM systems is that they may operate with little consideration or understanding of how the standing forest stock is regenerated. This management process is often predicated upon a vague and spurious understanding of, and the value of indigenous knowledge systems – mainly that these are superior to contemporary understanding of system responses and management processes (see Attwell and Cotterill 2000).

State Forest Management (SFM)

SFM in contrast to CFM has a biocentric approach which advocates prevention of excessive use and the protection of forest resources by the state authority. SFM is characterised by a centralised, authoritarian structure with a top-down approach to management and decision-making that often excludes local people (Horn 2002). This approach emphasises the protection of ecological processes and biodiversity *per se*, as necessary for forest establishment and growth and ultimately controlled harvesting of resources. These include processes of tree regeneration and species assembly in forest, which are covered under chapters V and VI. In the past most indigenous forest were entirely under state management and locals were only allowed to purchase permits to harvest limited quantities of certain products as long as they did not interfere with overall timber production and conservation objectives of forest managers (Grundy 2000). A major concern of this management policy is the capacity of the state to police forests and to protect them in a meaningful way from illegal exploiters (Shackleton 1993, Castley and Kerley 1996).

Participatory Forest Management (PFM)

PFM recognises the authority of different parties (e.g. the communities and the state) and advocates the sharing and decision making about, and the reward from, natural resource management for the benefit of a wide group of stakeholders (Grundy 2000, Wily 2000, Bass 2001, Horn 2002). PFM is also referred to as collaborative

management, shared forest management or Joint Forest Management (JFM) (National Forest Act 1998, Grundy 2000, Neil 2000). PFM has a two-sided approach to management; it uses the power of the state to create incentives for community participation and uses local action to drive the process of conservation. The PFM concept has arisen from the growing view that traditional forest methods in the past have failed to halt both the area and quality of national forests and woodlands (ODA 1996). A critical factor in PFM is how the element of participation or joint management is interpreted. Participation has often been interpreted differently by different people and may mean a range of things that extent from state manipulation or coercion, state decentralisation without devolution of control, partial devolution, to the community controlling and owing the rights over the resources and the land. In South Africa it is not clear where forest department lies on this spectrum of participation in part because PFM is a relatively new initiative (Horn 2000) and PFM may have to be applied on a case-by-case basis precluding a set position along the “continuum”.

An advantage of PFM over SFM is that local people can be mobilised into initiatives such as policing large forest blocks, which would be unmanageable under the state control (Poffenberger and McGeen 1998). Even though advantages of partnership seem obvious, there are good reasons why some communities may not collaborate with the state. For instance, collaboration may open communities to the danger of manipulation by the administrative structures of the state. Campbell (1991) reports such a case in Ethiopia where forests were initially jointly managed under a participatory programme but gradually fell under complete government control with locals losing their land-use options. Issues on PFM are further discussed under chapter II.

Overall, there is a continuum of forest management options from the socio-economically inclined resource-use community management at one end to the more ecosystem-maintenance and restricted resource use and complete protection of forests, at the other (Figure 1). Understanding the three views of indigenous forest management in South Africa as different points along this continuum of management paradigms may assist in clarifying the different visions of forest management. It is worth noting that there is no particular management practice that is ideal for all forests. Rather the task is

to determine which management approach along the continuum is best suited for a particular forest. For instance, what management options ought to be applied to coastal Eastern Cape Province (ECP) forests that were (1) originally controlled by local chiefs, and (2) those that have always been under state authority. Solutions require fundamental information on the state of the resource base, an understanding of the ecological processes that underpin resource dynamics, and knowledge of the exploitation patterns in these forests within a framework of socio-economic and ecosystem considerations.

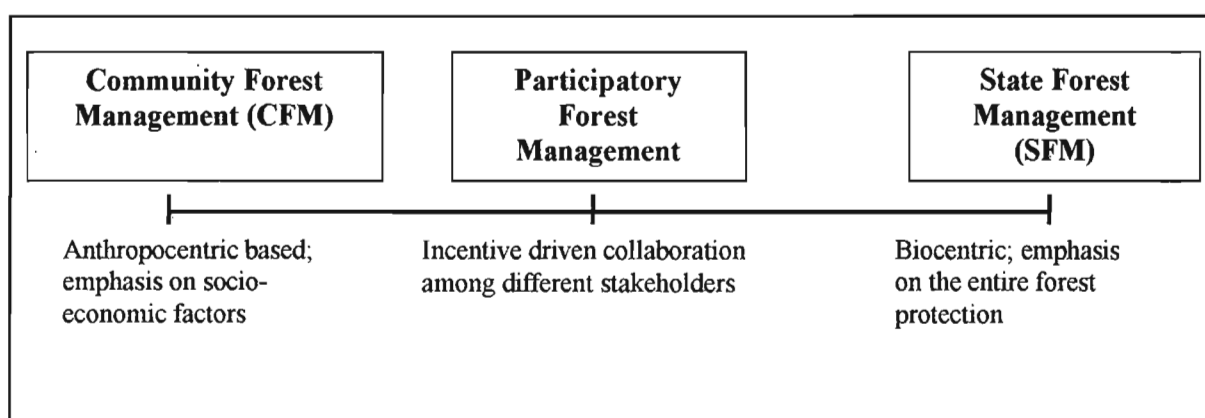


Figure 1: The community / state control spectrum of forest management (from Castley and Kerley 1996, Grundy 2000, Neil 2000, Van der Zel 2000, Obiri *et al.* in press; see Chapter III)

Indigenous forests of South Africa

Indigenous closed-canopy forests of South Africa are small and highly fragmented (Eeley *et al.* 1999). Earlier studies estimated forest cover at 0.1-0.2% of the total land surface of South Africa (Huntley 1984), recent work show this to be 0.5% (Low and Rebelo 1996). Of these forests, about 58% are in protected areas under state authority while the rest are in other legally defined areas (DWAF 1997). Despite being relatively well protected under legal instruments of the state, local administration and conservation bodies (Geldenhuis and MacDevette 1989, National Forests Act 1998), considerable legal and illegal exploitation still occurs in indigenous forests, seriously

impacting on their biodiversity. Given the relatively small area of forest these remain a major challenge to forest conservation in South Africa (Castley and Kerley 1996).

Challenges to conservation of indigenous forest in Eastern Cape Province

The Eastern Cape Province (ECP) has approximately 3702 km² of indigenous forests of which 907 km² are coastal forest (Low and Rebelo 1996). These forests fall under two types of ownership or tenurial system, state management and community control (Cooper and Swart 1992). Currently community forests make up 30% of the total forests in ECP while the rest is state forest. This tenurial set-up dates back to the frontier wars of the early 19th century when Xhosa tribes, defeated by the colonial government, had their forest land annexed as crown land, which subsequently became state forests (King 1941, Johnson 1983). However, chiefs who accepted colonial rule were given marginal control of some annexed forests, which later came to be known as Headmen (*Nkosi*) or community forests. The first signs of forest management in the ECP began under the colonial government in 1913 when the Native Trust Land Act was enacted (King 1941). This followed the 1888 Cape Forest Act 28 (Sim 1907) which had conferred all forest management in the Cape Colony (i.e. the current Western Cape Province) to forest officers. Thus, state forests in ECP region have been under state control for close to 90 years, although under different governments.

Although the sustainable use paradigm and community-based conservation (CBC) have been freely advocated by many conservationists worldwide (e.g. Western *et al.* 1994, Nhira *et al.* 1998, Poffenberger and McGean 1998) as solutions to forest encroachment these approaches face two main challenges in the coastal forest of ECP:

- the forests are too small and fragmented to sustain commercial levels of production; and
- there is a high demand for forest products by very poor communities. The economic choices available for such poor communities, often leads to actions that are not compatible with forest conservation (e.g. Noss 1997, Sutherland and Reynolds 1998, Gibson and Becker 2000).

Furthermore, managing these forests under the often recommended models of sustainability (i.e. harvesting or removing individuals, in a population, within the limits

that can be accommodated by density-dependent responses of fecundity and mortality; see Sutherland and Reynolds 1998, Appanah 2001) is currently not possible because:

- there is scant information on the availability, distribution and abundance of the harvestable tree species. This information outlines the available resource base from which sustainable harvesting quotas can be determined;
- although some studies have identified the exploited trees (Johnson 1982, Cawe and Mckenzie 1989, Van Eck *et al.* 1997, Obiri *et al.*; see Chapter III), most previous work emphasised floristic and structural classification (Shone 1985, Cawe and Mckenzie 1989, Cawe 1992, Cooper and Swart 1992). Yet it is the impact of harvest off-take and the harvesting pattern on the various forest tree species that is most important in addressing sustainability;
- the harvested forest products, particularly the non-timber forest products (NTFP's), are largely unaccounted for in the national economy (i.e. the gross domestic product). Thus, their markets are uncontrolled and this could lead to over-exploitation and local extinction of species; and
- there is no information on vital forest ecological processes such as seedling recruitment, succession patterns or growth characteristics of harvested species. Yet this information is a prerequisite for deciding harvesting quotas and reforestation or enrichment planting programmes.

In essence, it is fair to state that under the current situation, the sustainable use model, though widely advocated, faces many challenges in coastal ECP.

Given the central role of forests in the socio-economic livelihoods of rural communities, it has been suggested that the long-term viability of forests is inevitably linked to integrating communities into mainstream management and conservation practices (Falconer 1995, Nhira *et al.* 1998, Getz *et al.* 1999, Banana and Gombya-Ssembajjwe 2000, Weis 2000). With the majority of coastal ECP forests under state control, critical questions have arisen about whether or not more forests should be (1) community managed, (2) jointly managed through participatory initiatives or, (3) remain under state authority.

Aims and Objectives

Although specific hypothesis and aims are dealt with in the chapters, the overall objectives of this study are to:

- Develop an understanding of the local peoples knowledge and attitude towards forest resource use, and participation in conservation and management of the forests.
- Identify key forest tree resources used by the local communities, quantify their availability, establish existing harvesting structures and their efficacy under the different tenurial systems and, determine which tree species are exploitable.
- Establish the nature of forest regeneration or tree replacement pattern in forests (i.e. essential ecological processes) and the consequences for these process and forest conservation of tree harvesting patterns.

In addressing these issues, this thesis compares the resource base, exploitation trends and regeneration patterns between community and state forests within a conceptual framework that involves both forest ecology and socio-economic factors.

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CHAPTER II

CHALLENGES FACING NEW FOREST POLICIES IN SOUTH AFRICA: ATTITUDES OF FOREST USERS TOWARD MANAGEMENT OF THE COASTAL FORESTS OF THE EASTERN CAPE PROVINCE

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SUMMARY

Here we question the efficacy of new forestry policies that govern the sustainable use and conservation of forests in South Africa. These policies advocate the devolution of forest management from the state to local communities. The attitudes of forest stakeholders (forest resource users and managers) toward forest management policies and resource use were investigated using questionnaires. Results showed a concordance in the attitudes of resource users and managers. There was a general lack of support for Community Forest Management (CFM), particularly among older resource users, but strong support for State Forest Management (SFM). Forest stakeholders were positively disposed toward Participatory Forest Management (PFM) and generally favoured joint community/state forest management. Socio-economic correlates of stakeholders influenced attitudes toward the use and management of forest. Socio-economic factors must be considered when designing and implementing forest management. In this regard, younger users, low-income earners and educated resource users, were all positively inclined toward PFM. This enthusiasm should be used to drive conservation initiatives in these rural areas. No stakeholders supported a total ban on forest resource use, but all supported controlled use. These latter attitudes were independent of the preferred management system. This study adds to the growing literature that suggests that CFM has been overemphasised. Rather than narrowing forest management around communities we recommend an all-inclusive approach that provides for some management input by the state.

KEYWORDS: forestry policy, community attitudes, community forest management, participatory forest management, state forests, sustainable development

INTRODUCTION

A first step in formulating effective policies for conservation is to understand the knowledge and attitudes of people towards the use and management of the resource base (Jacobson & Marynowski 1997). We investigate the latter from a background of the conservation and sustainable management of coastal Pondoland forests of Eastern Cape Province in South Africa.

Recent trends in forest management in southern Africa have focused on the devolution of forest tenure and management from state authorities to local communities (DWAF 1997; Neil 2000; Olsen & Helles 2000; Wily 2000). People living around forests in rural areas are now being provided with opportunities to take direct control of the forests they use. This approach has arisen out a global acceptance that forest management under state protection-orientated management has failed to halt, let alone turn around, over-exploitation, fragmentation and loss of forests (Vandergeest 1996; Nhira *et al.* 1998; Wily 2000). Furthermore, there is an increasing popular awareness that sustainable use is unachievable without public participation (Mukherjee & Gangopadhyay 1997; Sarin 1998; Wily 2000; Bass 2001). Limited financial resources and lack of trained personnel have been previously identified as the key factors contributing to poor management by state authorities (Persson 2000). Many forest authorities have sought to solve this by seeking substantial donor funding (Bruenig 2000). In spite of funding support poor management has continued and new resolutions demand a thorough review of indigenous forest management policies (Kaimowitz 2000).

Newly introduced policies seek to reduce (but not eliminate) state control by establishing Participatory Forest Management (PFM) and Community Forest Management (CFM) processes (Neil 2000; Wily 2000). PFM, also referred to as Joint Forest Management (JFM), shared forest management or collaborative management, is the key process chosen to drive the management of South African indigenous forests (National Forest Act 1998). PFM aims to integrate planning, research, and decision making into a comprehensive system with participation of the state and the various other stakeholders particularly in the local communities (Slocombe 1993; Neil 2000;

Wily 2000; Bass 2001). In CFM, the community develops by-laws governing the use and protection of resources with the state playing a minimum or no role in the decision-making process (DWAF 1997; National Forest Act 1998). Similar new initiatives based upon PFM structures are acclaimed to be successful in Asian countries like India, Nepal, Pakistan (Mukherjee & Gangopadhyay 1997; Pandey & Gangopadhyay 1997; Pratima & Jattan 1999), and some parts of Africa, for example in Tanzania (Wily *et al.* 2000). Their success in South Africa has yet to be demonstrated (Kepe 1997; Leach *et al.* 1999).

There are no studies reported on the awareness, perception or attitudes that forest stakeholders in South Africa have towards the management practices promoted by the new policies. Yet these are the key policies upon which the future of forest management in South Africa depends (DWAF 1997; National Forest Act 1998). Although the policies are well articulated to forest resource managers (Mjwara *et al.* 2000; Neil 2000) the same cannot be said for resource users, and thus the implementation of policy remains problematic in rural areas. Effective policy instruments are required for implementation. Jacobson and Marynowski (1997) suggest these can be best developed by understanding stakeholders' knowledge and attitudes toward the use and management of forests. Besides examining attitudes towards management, we further scrutinise the attitudes and knowledge of different socio-economic categories of resource users. Our aims are to (1) investigate the perceptions and attitudes of various forest stakeholders towards forest resource use and management, and (2) identify the socio-economic groups within the community, with positive attitudes, that are critical to the process of forest conservation in rural areas.

METHODS

Study Area and the Stakeholders

Forest stakeholders were interviewed in Umzimvubu District of the Eastern Cape Province of South Africa. This is the main district of the Pondoland coast where forests rich in biodiversity are entwined with communities that place numerous demands on forest resources. Stakeholders were defined in accordance with Borrini-Feyerabend (1996) as social actors who (1) have a direct, significant and specific interest in an

area's natural resources, (2) are aware of their own interest in management of the resources, (3) possess specific capacity (skills, knowledge) and comparative advantages (proximity, mandate) for such management, and (4) are usually willing to invest specific resources (e.g. money, time, authority) toward some form of management. In this study 53 stakeholders were interviewed. They were divided into resource users (e.g. villagers, traditional leaders, woodcarvers, local craft-work traders, medicinal plant collectors, fuelwood gatherers) and resource managers (e.g. staff from the departments of forestry, agricultural, environment and tourism, nature conservationists, forest based researchers and non-governmental organisations). All resource managers besides the forest-based researchers were largely people with tertiary training at diploma level and with a long living experience in the Eastern Cape Province. Of the 53 stakeholders 33 were resource users randomly selected from three different categories of age, income and education levels of different households (N = 200 households) from two communities neighbouring on the forest. The other 20 interviewees were resource managers randomly sampled from a population of 80 forestry-based resource managers, who were listed by the forester in charge of Umzimvubu District, as working or undertaking research in the forests of Umzimvubu District. This represented a sample size of 17% and 25% of the resource users and resource managers respectively in the study area.

Questionnaire and Survey Design

The questionnaire survey was based upon two main issues: forest resource use and the system of forest management. These issues and subsequent questions were designed around: (1) problem statements identified in current forest policy documents of South Africa (e.g. the National Forestry Action Plan - DWAF 1997; the National Forest Act 1998); (2) previous studies on forest resource use in the study area (Cawe 1992; Obiri 1997; Van Eck *et al.* 1997; Mukolwe 1999; Obiri *et al.* in press; Chapter III); and (3) views of independent social and natural scientists.

The level of support from resource users and resource managers for the issues raised in the survey was assessed using attitude questions that were formulated around a symmetric 5-point Likert scale (1 - strongly agree to 5 - strongly disagree) with well-

defined separate alternatives and a central neutral point (Borg & Gall 1989). The awareness or familiarity of resource users and resource managers to the issues in the survey was assessed using knowledge questions designed on a true-false format (Jacobson & Marynowski 1997).

The questionnaire consisted of three main sections with attitude, knowledge and socio-economic questions. The attitude section had seven questions placed into two groups concerning (1) forest resource use (i.e. questions on the control and banning of forest resource use); and (2) forest management systems (i.e. questions seeking views on different management practices; see Table 1). Attitude questions about forest resource use were phrased around the use of poles, posts and medicinal trees because these were considered the main resources under contention in these forests (Cooper & Swart 1992; Obiri 1997; Palmer *et al.* 1997; Van Eck *et al.* 1997; Mukolwe 1999; Obiri & Lawes 2000; Timmermans 2000; Obiri *et al.* in press; Chapter III).

Table 1: Attitudes questions on forest resource use and management. Questions were phrased as statements to which respondents express their opinion by selecting a choice from the Likert scale (strongly agree =1 to strongly disagree = 5).

Attitude statement of the survey
<p>Forest use</p> <ol style="list-style-type: none"> 1. Harvesting of poles and posts from forests should be controlled. 2. Harvesting of medicinal trees from forests should be controlled 3. Poles and posts harvesting should be banned. 4. The harvesting of medicinal trees should be banned. <p>Forest Management</p> <ol style="list-style-type: none"> 1. Forests can be better conserved if managed by state authorities (SFM) rather than local communities (CFM). 2. Forests can be best conserved if managed by the community (CFM) rather than the state (SFM). 3. Forests should be jointly run under a participatory forum (PFM) to ensure their conservation.

In order to test how attitudes towards forest management and resource use varied with the socio-economic characteristics of the communities, the questionnaire data was analysed across the three different categories of age, income and education among the resource users. The age categories comprised of youths (16-35 yrs), middle age users (36-56 yrs) and the elderly users (>56 yrs). Education level was categorised as primary, secondary (high school) and tertiary (diploma colleges, university), while monthly

income was categorised as low (<US\$125), middle (US\$125-375) and high (>US\$375) (US \$1 = 8 Rand at 2001 exchange rate).

Correspondence Analysis

For exploratory purposes, the responses of the 53 stakeholders to the attitude questions (see Table 1) were analysed in a correspondence analysis using CANOCO (Ter Braak & Smilauer 1998). This analysis was used to determine whether or not there was concordance between the attitude of resource users and managers with regard to the resource use and management practices in the forests. In the case of forest management practices we compared the attitudes toward state and community forest management. Participatory forest management was not considered in the comparison because it is a new forestry initiative in the region and relatively unknown to forest users and the local communities. Thus, stakeholder experience constrained meaningful comparison to between SFM and CFM. Nevertheless, stakeholders were asked whether PFM (broadly explained to interviewees as forests jointly run by the state and community) should be implemented in their area as a strategy for forest conservation.

Analysis of Attitudes and Knowledge on forest use and management

The attitude of resource users and resource managers to each question on forest use and management were determined using the most commonly selected response (i.e. the modal class) from a frequency distribution of responses on the Likert scale. Knowledge scores are reported as the percent correct out of a possible 100% (Table 2; see also Jacobson and Marynowski 1997). A response was considered as correct if it was similar or agreed with the forest or tree use and management ideals reported from several studies in the study region (e.g. Johnson 1983, Butchart 1989, Cawe and Mckenzie 1989, Cawe 1992, Cooper and Swart 1992, Pooley 1993, Van Eck *et al.* 1997, Bokwe and Bhat 1998). Knowledge questions were interpreted in two groups, i.e. from the point of view of forest use and the forest management system (see later Table 2). Knowledge questions on the use of forest sought to determine if stakeholders were aware of (1) the commonly exploited forest resources, (2) change in forest area, and (3) the scarcity of the highly desired multi-purpose tree - *Ptaeroxylon obliquum*. Knowledge questions on forest management were based on two issues that affected

resource users; first, forest harvesting permits, and second, the ability of users to identify state forests from community forests. The latter was important to both resource users and managers since trespassers found harvesting in state forests are fined. Differences in the support for forest use and management practices between resource users and resource managers are reported using (1) correspondence analysis, (2) modal classes from frequency distributions of the responses, and (3) knowledge scores.

Chi-square analyses were conducted on the count data derived from the questionnaires. Collation of responses sometimes resulted in low counts for some expected frequencies in the chi-square analyses. In these instances we combined the data from some classes of response and/or applied Yate's correction to 2x2 contingency table analyses.

RESULTS

Correspondence Analysis (CA): Are Attitudes of Resource Users and Managers the Same?

In the CA ordination the first axis accounted for 33% of the variation in the attitudes of stakeholders (eigenvalue = 0.077) and the second for 28% (eigenvalue = 0.063). Most of the variation associated with Axis-1 was attributed to attitudes involving the banning of post, pole and medicinal tree harvesting (Figure 1), and biplot analysis shows the support for SFM is negatively correlated with CFM. This is expected since respondents who favoured SFM generally rejected CFM as an option and *visa versa*. Attitudes toward banning of the use of pole, post and medicinal tree species were positively correlated and independent of the management system preferred by the interviewee. In other words, whatever the preferred forest management system (i.e. either CFM or SFM), forest stakeholders held similar and supportive views toward the banning and/or control of forest resource use.

There was considerable overlap in the responses (minimum convex polygons about the groups) of resource users and resource managers (Figure 1) to attitude questions. In fact, responses of managers are largely nested within the variation displayed by resource users. This suggests concordance in the views of the two groups of stakeholders.

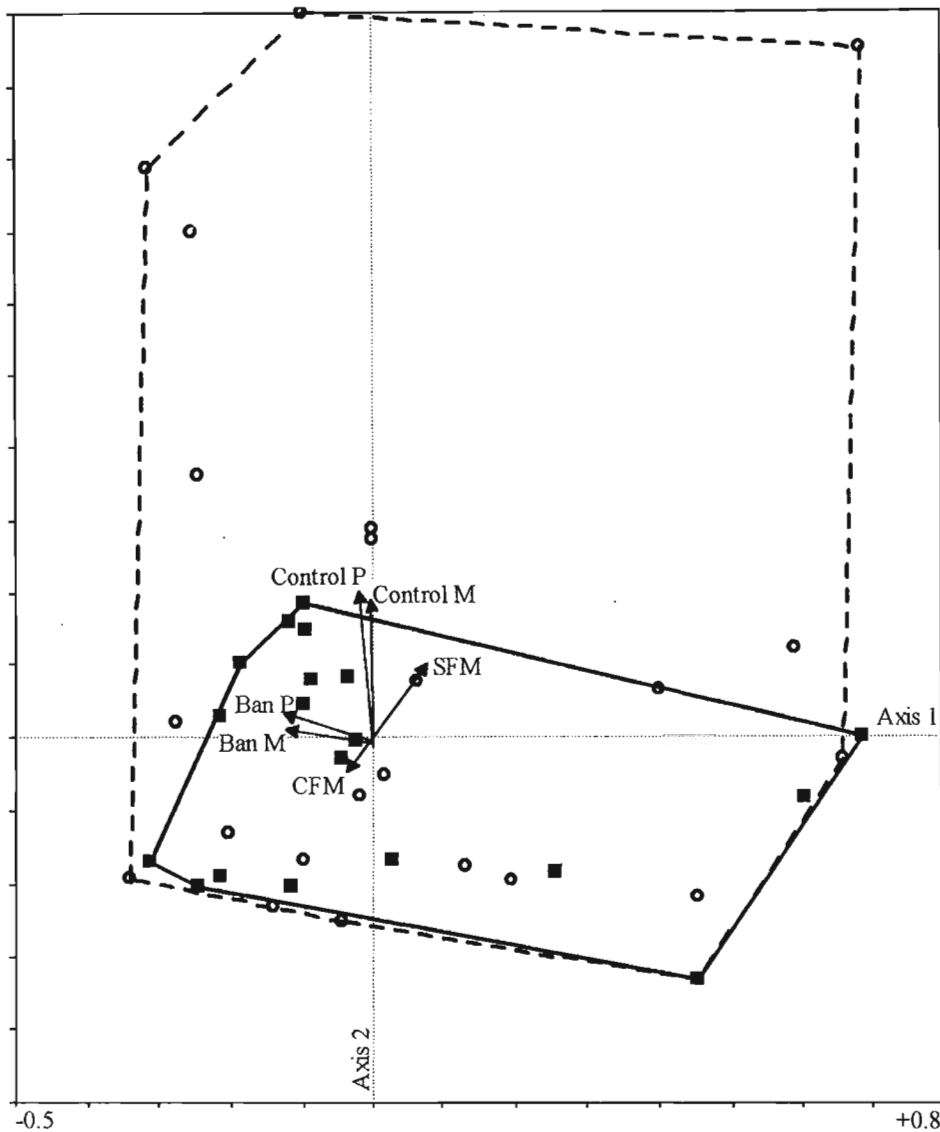


Fig. 1. Ordination diagram of a correspondence analysis of interviewee responses to questionnaires, with six explanatory variables (arrows) relating to the control of forest use and management policies (see Table 1).

Fifty-three forest stakeholders were interviewed. Data are derived from counts of responses to questions ranked along a Likert scale. The diagram is symmetrically scaled. Square symbols represent resource managers (n=20) and open circles are resource users (n=33). Abbreviations: Ban M – ban the harvesting of medicinal trees, Ban P - ban the harvest of posts and poles, Control M – control medicinal tree harvest, Control P – control posts and poles harvest. CFM and SFM represent community and state forest management respectively.

Attitudes Toward Forest Resource Use

a. Control of forest resources

The responses towards the control or restriction of poles and posts harvesting did not differ between resource users and resource managers ($\chi^2 = 4.78$, $P = 0.09$, D.F. = 2). Eighty-five percent ($n = 28$) of resource users (modal class, $n = 28$) and 85% ($n = 17$) of resource managers (modal class, $n = 17$) strongly agreed with the controlled use of posts and poles (Figure 1a). Combining the responses of resource managers and users (i.e. all stakeholders) confirmed significant support among stakeholders for the control of the use of poles and posts (modal class = strongly agree, $n = 45$, $\chi^2 = 47.4$, $P < 0.001$, D.F. = 4).

Sixty-seven percent ($n = 22$) of the resource users and 75% ($n = 15$) of resource managers strongly agreed with the control of the use of medicinal tree species (Figure 1b). The responses of these users (modal class = strongly agree, $n = 22$) and resource managers (modal class = strongly agree, $n = 15$) toward the control of the use of medicinal trees did not differ significantly ($\chi^2 = 3.43$, $P = 0.18$, D.F. = 2). Both resource users and managers supported the regulation of harvesting of medicinal trees (modal class = strongly agree, $n = 37$, $\chi^2 = 37.2$, $P < 0.001$, D.F. = 4). These results confirm the general concordance observed in the correspondence analysis in attitudes between users and managers toward the control of the use of forest resources.

All resource users (100%, $n = 33$) had incomes $< \text{US\$}375$ (i.e. no high income earners) and 97% ($n = 32$) had either a primary or secondary education (Figure 2). Thus, attitudes of users with regards to their socio-economic characteristics were tested across two levels for both education (primary and secondary) and income (low, $< \text{US\$}125$; middle, $\text{US\$}125\text{-}375$).

Support for the controlled use of medicinal trees did not differ among resource users categorised by educational standard ($\chi^2 = 0.75$, $P = 0.39$, D.F. = 1), income level ($\chi^2 = 1.05$, $P = 0.31$, D.F. = 1), or age ($\chi^2 = 0.47$, $P = 0.79$, D.F. = 2). Likewise, support for the control of post and pole harvesting did not differ across educational levels among resource users ($\chi^2 = 0.11$, $P = 0.74$, D.F. = 1) or age ($\chi^2 = 2.2$, $P = 0.33$, D.F. = 2), but

did differ across income levels ($\chi^2 = 5.1$, $P = 0.02$, D.F. = 1). Twenty-three percent of low-income resource users ($N = 13$, <US\$125) did not support control measures, however, all the middle income earning users ($N = 20$, US\$125-375) agreed with the regulation of post and pole harvesting.

b. Banning forest resource use

The responses of all stakeholders (i.e. resource users and managers) toward a ban on post and pole harvesting differed significantly across the 5-point Likert scale ($\chi^2 = 9.8$, $P = 0.04$, D.F. = 4). Thirty-eight percent ($n = 20$) of stakeholders strongly disagreed with a ban on resource use. Resource managers and resource users differed in their responses to a ban on posts and poles ($\chi^2 = 16.0$, $P = 0.003$, D.F. = 4). Fifty percent ($n = 10$) of the managers disagreed with a ban on harvesting posts and poles while 52% ($n = 17$) of the users strongly disagreed (modal class, $n = 17$) with the ban (Figure 3a).

Responses to a ban on harvesting medicinal trees did not differ between users and managers ($\chi^2 = 3.4$, $P = 0.34$, D.F. = 3). Thirty three percent ($n=11$) of the users and 50% ($n = 10$) of managers strongly disagreed i.e. did not support the ban (Figure 3b). Lack of support for a ban on medicinal tree use did not differ among resource users of different ages ($\chi^2 = 3.67$, $P = 0.16$, D.F. = 2), income levels ($\chi^2 = 0.01$, $P = 0.93$, D.F. = 1) or educational level ($\chi^2 = 0.13$, $P = 0.72$, D.F. = 1).

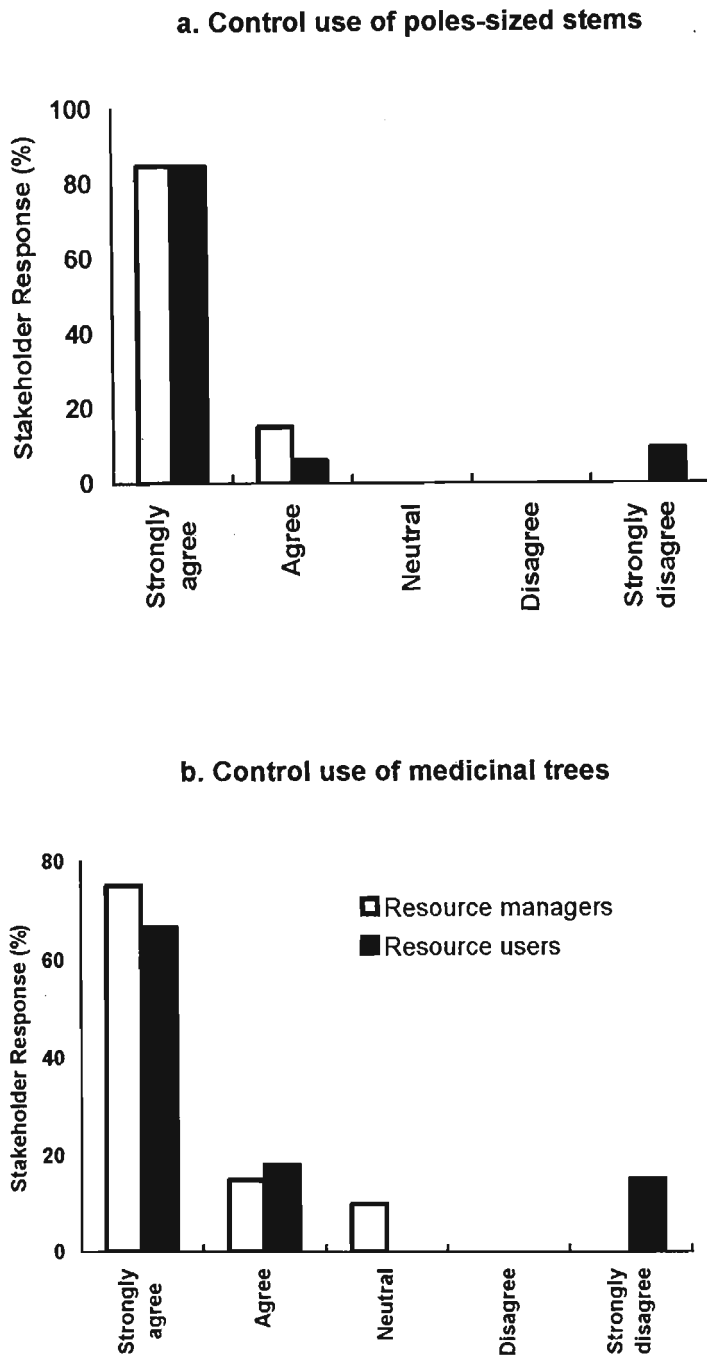


Fig. 1: Attitudes of stakeholders (resource managers and resource users) toward the control of harvesting (a) medicinal tree species and (b) pole sized-trees.

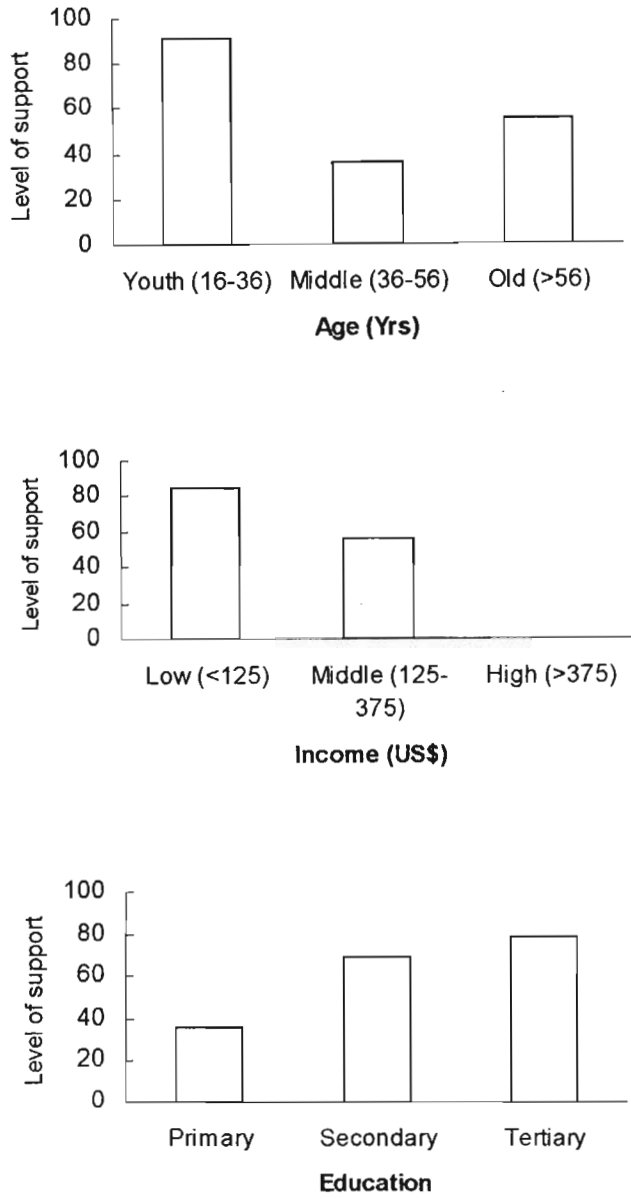


Fig. 2. Level of support (%) for participatory forest management among resource users of different socio-economic status.

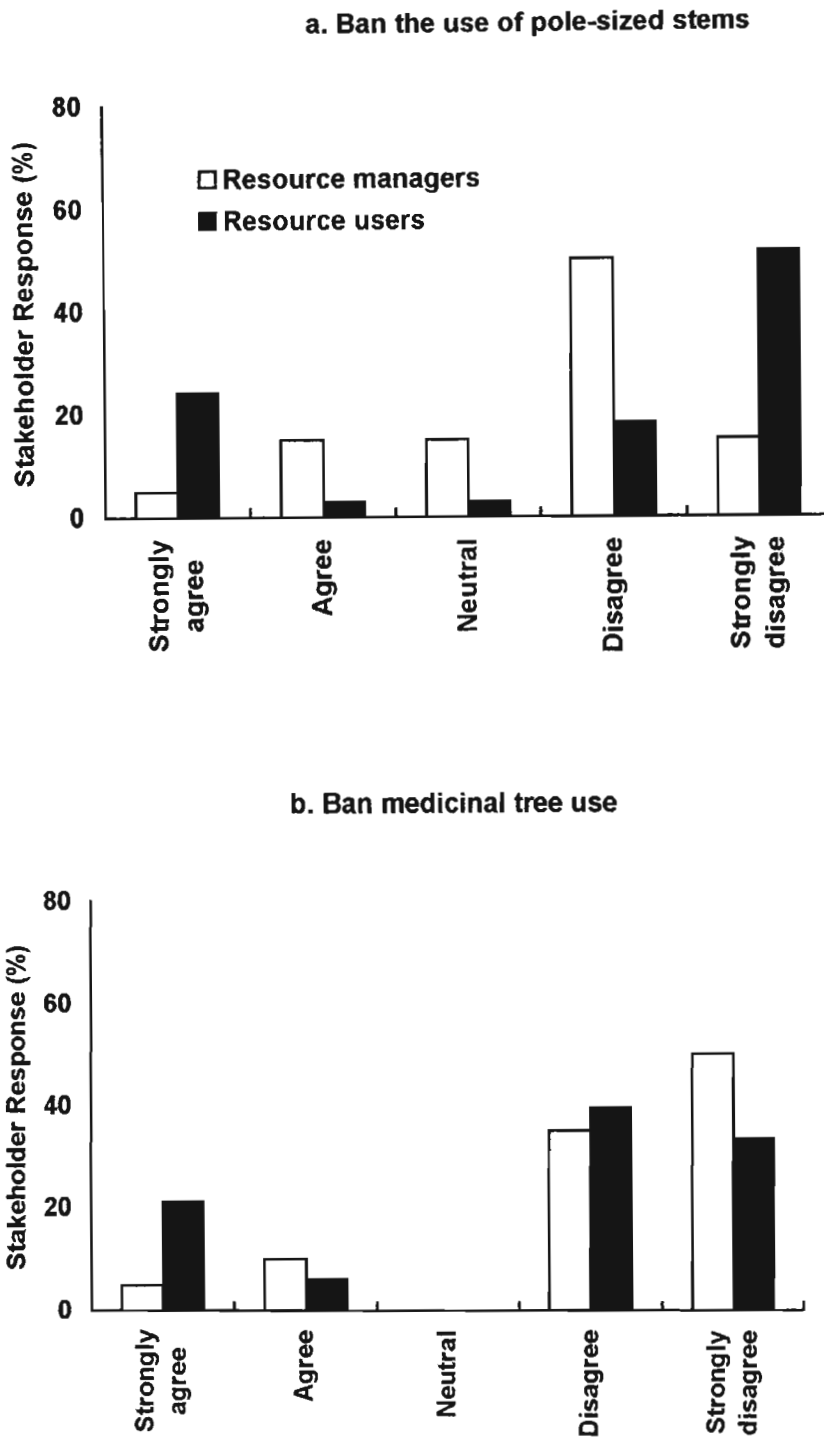


Fig. 3: Attitude of stakeholders (i.e. resource managers and users) toward the banning of (a) pole size trees and (b) medicinal tree species.

Attitudes Toward the Type of Forest Management System

a. State Forest Management (SFM)

Forest stakeholders (i.e. users and managers combined) gave significantly different responses toward the adoption of state forest management ($\chi^2 = 0.003$, $P = 16.4$, D.F. = 4), with strong agreement to SFM being the modal response ($n = 24$). Resource users and managers differed in the strength of their desire for state management ($\chi^2 = 18.9$, $P = 0.001$, D.F. = 3). Approximately half of the resource users strongly agreed (modal class, $n = 16$, 49%) with state management, while resource managers were less enthusiastic and only 40% ($n = 8$) responded in the same way (Figure 4).

Support for the state management of forests did not differ among resource users of different age ($\chi^2 = 3.3$, $P = 0.19$, D.F. = 2) and education level ($\chi^2 = 0.14$, $P = 0.71$, D.F. = 1) but differed across income levels ($\chi^2 = 4.41$, $P = 0.04$, D.F. = 1). The majority of the low-income earners agreed with (75%, $N=20$) state forest management but most of the middle income earners (62%, $N=13$) disagreed. All categories of users except the middle income earning users support state management.

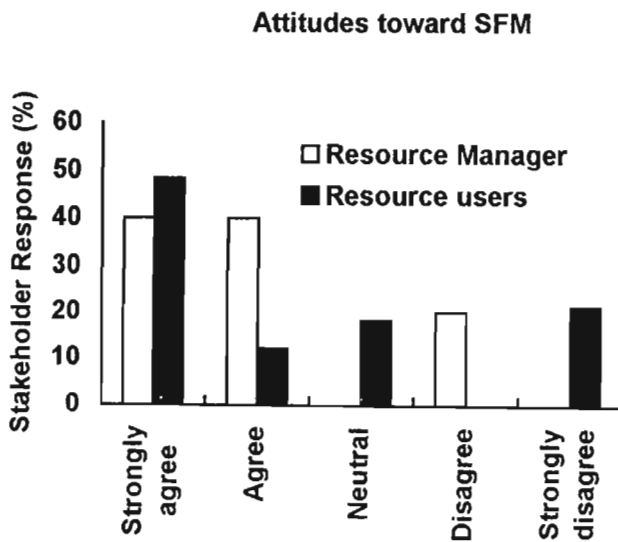


Fig. 4: Attitudes of stakeholders toward state forest management

b. Community Forest Management (CFM)

The responses of all forest stakeholders (i.e. resource users and managers) to CFM differed significantly ($\chi^2 = 18.2$, $P = 0.001$, D.F. = 4) and the modal response category of the stakeholders was that of strong disagreement to CFM (modal class, $n = 23$). Forty percent ($n = 8$) of resource managers disagreed with CFM (modal class, $n = 8$) while 52% ($n = 17$) of users strongly disagreed (modal class, $n = 17$) to the same (Figure 5). The two stakeholder groups did not differ in their response to CFM ($\chi^2 = 2.54$, $P = 0.47$, D.F. = 3).

The lack of support or negative attitude toward CFM did not differ across income levels ($\chi^2 = 0.5$, $P = 0.48$, D.F. = 1) or educational levels ($\chi^2 = 0.25$, $P = 0.62$, D.F. = 1) of resource users. However, it differed with resource users' age ($\chi^2 = 9.24$, $P = 0.01$, D.F. = 2) with all old users disagreeing to CFM and only the young showing considerable agreement. There appears to be a lack of confidence among resource users in community forest management. The results are in agreement with the correspondence analysis and confirm the negative correlation between SFM and CFM.

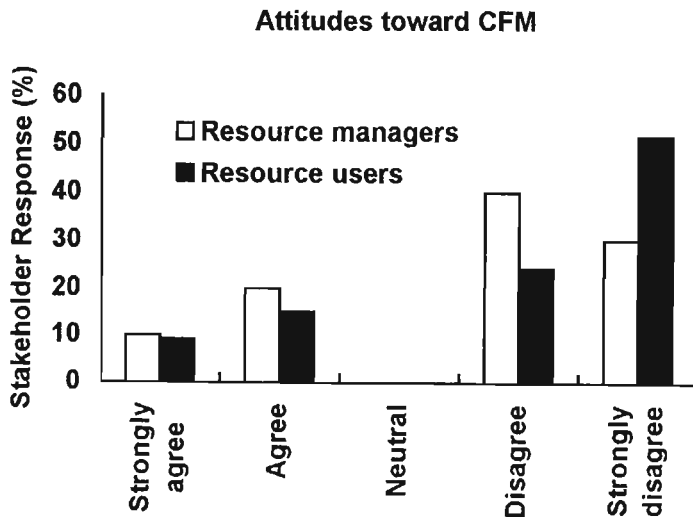


Fig. 5: Attitudes of stakeholders toward community forest management.

c. Participatory Forest Management (PFM)

Stakeholders (resource users and managers) favoured the PFM initiative and strongly agreed (modal class, $n = 21$, 40%) to the view that the state and the communities should jointly manage forests. Forty-five percent ($n = 15$) of resource users strongly agreed with PFM. Thirty percent ($n = 6$) of managers also strongly agreed (modal class, $n = 6$) with PFM (Figure 6). This support for PFM did not differ between resource users and resource managers ($\chi^2 = 2.9$, $p = 0.4$, D.F. = 3). However, the support for PFM differed across resource users' age ($\chi^2 = 7.1$, $P = 0.029$, D.F. = 2), income ($\chi^2 = 5.18$, $P = 0.02$, D.F. = 1), and education level ($\chi^2 = 6.3$, $P = 0.01$, D.F. = 1). The largest proportion of PFM supporters (with regard to age, education and income level of users) were found among the youth (91%, $N=11$), the lowest-income earners (85%, $N=13$) and the most educated users (79%, $N=19$) (Figure 2). These results clearly indicate that socio-economic factors have to be considered when designing and implementing forest management programmes.

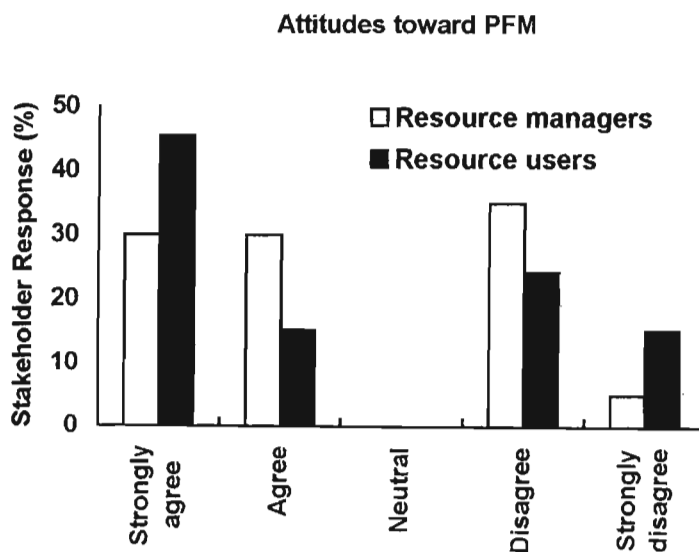


Fig. 6: Attitudes of stakeholders (resource users and managers) toward participatory forest management.

Knowledge of the Forest Resource Base

a. Indigenous tree use

Overall, forest users and managers were aware of the commonly used forest resources and correctly identified trees such as *Englerophytum natalense* (96%, n = 51), *Ptaeroxylon obliquum* (64%, n = 34), *Millettia grandis* (92%, n = 49) and *Millettia sutherlandii* (58%, n = 31). Of the two groups, resource users were more knowledgeable about the uses of the forest tree species ($\chi^2 = 18.06$, $P = 0.006$, D.F. = 6; see Forest use in Table 2). However, the responses of both users and managers, to forest use and management (Table 2), differed, in many cases, with that reported on tree use and forest management in the area (see Johnson 1983, Butchart 1989, Cawe and Mckenzie 1989, Cawe 1992, Cooper and Swart 1992, Van Eck *et al.* 1997, Bokwe and Bhat 1998). This suggests a gap in their knowledge of the forest use and management, which must be addressed in order to develop and implement management policies.

b. Change in forest area

While 80% (n = 16) of resource managers knew that forest had become more fragmented and decreased in area over the last 25 years, only half of resource users (54%, n = 18) professed to be aware of this. In spite of the knowledge that forest had fragmented and declined, few resource managers could point to any species (e.g. *P. obliquum*) that had been largely removed. Only 30% (n = 6) of the resource managers were aware that *P. obliquum*, previously common in these forests, is now very scarce (Mukolwe 1999; Obiri *et al.* in press; Chapter III). However, most resource users (80%, n = 26) were aware of the scarcity of *P. obliquum*. Awareness of the change in forest area over the last 25 years did not differ across resource users' educational levels ($\chi^2 = 2.4$, $P = 0.12$, D.F. = 1) or income levels ($\chi^2 = 0.25$, $P = 0.62$, D.F. = 1), but differed with users' ages ($\chi^2 = 6.16$, $P = 0.05$, D.F. = 2). The older resource users were the most aware (82%, N=11) of the changes in forest area while, as expected, the youngest users were least informed (33%, N=12), and mistakenly thought that forests had increased in area. Knowledge of the rarity of *P. obliquum* only differed with age of resource users ($\chi^2 = 6.2$, $P = 0.045$, D.F. = 2) but not with their income or education level. In matters

of environmental change (e.g. changes in area and important species' abundance) older users are the most informed and management planning should make provision for their participation in forest conservation.

Knowledge of Forest Management Practices

Overall, resource users and managers did not differ in their knowledge of forest management practices ($\chi^2 = 4.05$, $P = 0.26$, D.F. = 3; see Forest management in Table 2). However, while all resource users (100%, $n = 33$) knew that local chiefs had control over some of the indigenous forests in the rural areas (community or *Nkosi* forests), only 45% ($n = 9$) of resource managers were aware of this and most resource managers mistakenly thought all forests in the rural areas were under state management. The current practice of issuing harvesting permits in state managed forests was well known to all resource users and managers. On the other hand 80% ($n=16$) of resource managers and 82% ($n=27$) of resource users were aware of the permit requirement in community forests and that the permits were issued by the local chiefs (Table 2). Despite knowing the latter, none of resource users actually paid for permits to harvest in the community forests. Most users claimed that nobody stopped or prevented them from harvesting in community forests. Clearly the permit system in community forests is a non-operational practice and exists only on paper. Most resource users (79%, $n=26$) and resource managers (85%, $n=17$) correctly identified the white painted boulders on forest edges as beacons used by the forest department to mark boundaries of state and community forests. Thus, it is unlikely that users harvesting forest resources unknowingly stray into state forests.

Table 2: Knowledge scores of resource users and resource managers to issues related to forest use and forest management. For each statement the knowledge score for resource users and managers as a percentage of respondents who were correct.

Statement from the survey	Knowledge scores (%) and observations (n)			
	Resource Users	n	Resource Managers	n
<u>Forest Use</u>				
Woodcrafts in local markets are mostly from the <i>Millettia grandis</i> tree.	94	31	90	18
The Natal Mahogany tree (<i>Trichilia dregeana</i>) is not a medicinal tree.	67	22	80	16
The Giant Umzibeet (<i>M. sutherlandii</i>) is a useful timber & craft tree.	85	28	15	3
Fruits of the Natal Milkplum (<i>Englerophytum natalense</i>) are inedible.	76	25	50	10
The popularly planted hedge, <i>Cestrum laevigatum</i> is an indigenous tree	54	18	70	14
Within the last 25 years indigenous forests have increased in area.	54	18	80	16
The Sneezewood tree (<i>Ptaeroxylon obliquum</i>) is a rare among the Coastal forests.	85	28	30	6
<u>Forest Management</u>				
Harvesting forest resources in state forests requires permits.	100	33	100	20
Permits are not required for harvesting resources in community forests.	82	27	80	16
None of the forests in rural areas are under community control.	100	33	45	9
Boulders painted in white demarcate state from community forests.	79	26	85	17

DISCUSSION

Controlling Resource Use

With the exception of the low-income category of resource users, who disagree with controlling post and pole harvesting, all other forest stakeholders supported the controlled use of forest products. However, no stakeholders supported a total harvesting ban on these resources. In these forests a ban already exists on harvesting tree species such as *P. obliquum*, *M. grandis* and *Strychnos henningsii* (Cooper & Swart 1992) but these continue to be poached (Obiri 1997; Mukolwe 1999; Obiri *et al.* in press; Chapter III). Banning forest resource use often has little impact on the overall conservation of resources because the costs of policing a ban are prohibitive (Shackleton 1993a). In communities neighbouring on forest the impetus for forest protection is often strongest among the higher income earners since they are less dependent on forests. Low-income users are often reluctant to participate in protection for fear of losing access to resources and the obvious economic consequences of this (WRI 2000). Our results support this trend. If low-income earners supported restrictions on harvesting they would, in fact, be subsidising forest conservation since they would have to bear the brunt of the costs of lost opportunity through reduced access to resources (Field 2001). The low-income earners also have the most impact on forest conditions and conservation. For both the latter reasons low-income earners (and those factors relating to their livelihoods) are the most important target stakeholder group to include in developing rural conservation programmes (Weis 2000).

Perceptions of the Forest Management Systems

Although a large body of literature supports community management processes (Mukherjee & Gangopadhyay 1997; Nhira *et al.* 1998; Pratima & Jattan 1999, Wily *et al.* 2000) our study adds to the growing evidence (e.g. Mosse 1997; Hackel 1998; Leach *et al.* 1999) that point to the weakening support for community management systems. We take support for this view from two issues. First, in contrast to our expectations, resource users have negative attitudes toward, and hence low support for, community forest management i.e., they are not confident that their own community management structures led by local chiefs can sustainably manage forests. Second, the

resource users support the protection-orientated state forest management system. That communities support state involvement in forests is further confirmed by their positive perception of joint state / community forest management as outlined under the new initiative of PFM. Sikhitha (1999) reports similar attitudes from the Limpopo Province of South Africa where 89% of individuals interviewed from communities around Thathe forest were of the opinion that the government is better placed to manage the forests than themselves and 43% supported PFM initiatives.

Why is Community Forest Management Poorly Supported?

The efficacy of community forest management in Umzimvubu District has declined. This may be linked to the introduction of Transitional Rural Councils, where the local chiefs' powers have been drastically reduced (Pycroft 1999; South African National NGO Coalition 2001) and so has their authority over community forests. Nomtshongwana (1999) reports a similar decline in chiefs' powers over community forests in the southern Drakensberg region of South Africa. He attributes this to recent changes in the governing structures, where villagers are more politicised and less inclined to follow traditional rules. Castro (1990) suggests that breakdown of traditional systems can have adverse effects on the future of a community since traditions are taught and conserved through generation-sets, which once broken cannot be regained. Thus, the previous authority of chiefs and their strong community management structures that once ensured potential sustainable use in community forests, are unlikely to be achieved in future – even under a state advocated community management.

Although the lower support for CFM among resource users is an unexpected outcome, it is not surprising that this lack of support derives largely from the older users. The old users are most aware of the negative impacts on forest (e.g. declining forest areas, increased scarcity of key tree species and the disintegration of previous community management structures) that have occurred over the last 25 years, mostly in community forests. Obiri *et al.* (in press; see Chapter III) have shown that the community controlled forests are heavily overexploited and have significantly lower tree density and fewer trees species than state forests. In the past villagers obtained permits from local chiefs (*amakosi*) before harvesting trees in community (*Nkosi*) forests (Obiri

1997). Furthermore, village committees of selected elders met regularly to discuss forest use issues (e.g. monitoring, sanctions, conflict resolution) and would mete out fines to forest transgressors (Obiri 1997). These practices reinforced local leadership in community forests but are currently not being implemented, to the extent that some resource users are not aware of their existence.

Disintegration of community management structures and the consequent degradation of forests has also been reported from other community-controlled forests in southern Africa (Shackleton 1993b; Vermeulen 1996) and South America (e.g. Yuacare Forest, Ecuador, Gibson & Becker 2000). However, studies from Tanzania (Wily *et al.* 2000), India and some South-east Asian countries (Poffenberger 1998; Pratima & Jattan 1999; Pokharel 2000) point to successes in community management. For example, in the Chingra forests of West Bengal, India, Poffenberger (1998) reports that the community leadership (*panyachat*) have, without outside funding, established extensive forest areas on their own by managing natural regenerations and effectively mobilising community commitment for forest protection. However, in all of the latter successful cases the authority of traditional leaders is highly respected – this must be addressed for the successful implementation of community participation in the management of forest resources.

Because users of our coastal forests indicated that state forest management is preferable to community management, critical questions emerge for forest policy makers. Gibson *et al.* (2000) suggest three such questions on community management: (1) what factors make certain communities succeed in their design and implementation of self-governing rules, (2) why do certain communities self-govern in the first place, and (3) why do some communities neglect introducing organisational changes when their external environments change. Although answers vary with geography, Poffenberger (1998) and Sarin (1998) suggest that certain general characteristics of a community determine its success in communal management. These include (1) a high dependency on forests, (2) a perception of forest scarcity or outside threat, (3) geographical proximity to forest, (4) presence of indigenous resource management institutions, (5) sacred forests and traditional religious forest issues, and (6) strong local leadership.

While some of these characteristics are common to communities neighbouring our forests, the lack of a strong local leadership and strong management institutions stands out as the key issue resulting in poor community management and the negative attitudes shown by resource users.

How Viable is Participatory Forest Management?

While the new forestry policies are geared towards devolving power from state authorities to local communities (National Forest Act 1998; Neil 2000; Wily 2000) our results show that communities favour increased involvement by the state in the management of forest. The success of PFM depends on a variety of factors that are linked to the participating stakeholders. In South Africa Mjwara *et al.* (2000) suggest that the important factors are: (1) recognising various interest groups, (2) reconciling the views and needs of the different groups, (3) raising awareness of participatory initiatives, (4) providing viable and sustainable alternatives, and (5) that there is sufficient political will and support for the process. In Umzimvubu District, PFM initiatives are a compromise between complete devolution of power to communities and the state's control of forests. Clearly the state does not have the resources to police forests without the help of local communities and PFM appears to be a useful means of increasing the control over resources. Some conservationists recommend PFM over community or state forest management because PFM enhances collaboration between forest communities and state authorities (e.g. Murphee 1993; Becker & Leon 2000; Pokharel 2000). However, Poffenberger and Singh (1998) and Campbell *et al.* (2001) warn that implementation of PFM can be difficult, particularly where securing representation on joint management committees and reaching consensus on issues such as distribution of benefits to communities are concerned. Grumbine (1994) and Jacobson (1995) suggest that these issues can partly be overcome if resource users and managers are aware of the forest management ideals and practices and have positive attitudes towards conservation. In this study the largest proportion of PFM supporters were among the younger users, the educated and the low-income earners. These three categories of users are the desired target audience for educational programmes aimed at training leaders to spread the PFM initiatives to the rest of the local communities.

Although resource users in this study support PFM, some studies have questioned the capacity of participatory initiatives to sustain natural resources (Campbell 1991; Western *et al.* 1994; Sarin 1995; Hackel 1998). This question is raised largely because of the tendency (despite rhetoric to the contrary) of the “intended beneficiaries” in PFM programmes to be sidelined as passive participants. For example, in Ethiopia, community forests under PFM have been taken over and are entirely run by government policies with a resultant loss of land-use options for rural people (Campbell 1991). Certainly PFM is more easily advocated than implemented. Indeed, Hackel (1998) has cautioned that because PFM is often supported without appreciating the difficulties involved in reconciling conservation and community needs, it is a concept that is losing coherence as people define it for their own purposes.

Nevertheless, participatory initiatives have proved to be more realistic and successful in situations where the natural resources are plentiful and attract economic benefits. For example, the Zimbabwean CAMPFIRE programme and Zambia’s Administrative Design for Game Management have managed to raise sufficient funds from their hunting and ecotourism ventures (Western *et al.* 1994), and have succeeded in working with local communities largely because they offer sound economic incentives. It is unlikely that this success could be duplicated in the forestry situation in Umzimvubu District for one important reason - the economic incentives necessary for attracting strong local participation are lacking due to the limited revenue-generating potential of local forests. Even if forests had sufficient resources, continued local participation can never be guaranteed because communities are inclined to change their views of participation should the incentives they receive be considered insufficient, or better alternatives arise. Norton-Griffiths and Southey (1995) reports such a case among the Masai of Kenya who have converted their wooded grassland from traditional livestock management and tourism to full-time cereal crop agriculture. Thus, the implementation of PFM programmes are bolstered and paradoxically most threatened by the economic incentives offered to communities and may fail if sufficient revenue is not realised from what is to be conserved. There is a very real danger of social and ecological disaster where PFM is badly managed or implemented.

The Role of Protectionism

Because conservation through PFM is strongly linked to resource users protecting a resource, by foregoing some benefits from it, in return for other incentives, protectionism is still at the core of conservation. The level of protectionism needs careful consideration in all conservation programs. For example, to what level can forest protection be assured under PFM initiatives? Can participation of communities be assured if very high levels of protection are set? Furthermore, Hackel (1998) points out a weakness of PFM in that it assumes that communities will accept whatever level is set for protection because they receive incentives. Even if this assumption is correct can the incentives be sustained? Details of many issues are lacking from the PFM literature and conservationists should use PFM with caution. In particular, the link between economic incentive and resource protection poses a serious practical and conceptual dilemma for PFM.

It is difficult to have conservation without some form of protectionism. Disregarding state forest management on the basis of its protective role may not be the complete solution for forest management. State forest management needs reconsideration since the benefits that accrue from it are more widely spread through the society, unlike CFM and PFM, which largely address only the interests of communities immediate adjacent to forests. PFM should be seen as one process that fosters better relations between communities and resource managers (Pagiola *et al.* 1998) - just like other conservation based processes such as the compensation of communities for crop damage from wildlife (Naughton-Treves 1998) or environmental education initiatives (Cambray 2000). Neither PFM nor community management can solve the problems of forest conservation on their own, but a comprehensive system that also considers protection of the resource base by the state has a better chance of achieving this goal.

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CHAPTER III

THE DYNAMICS AND SUSTAINABLE USE OF HIGH-VALUE TREE SPECIES OF THE COASTAL PONDOLAND FORESTS OF THE EASTERN CAPE PROVINCE, SOUTH AFRICA

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Abstract

We investigate the ecological status of the standing crop of high-value tree species in coastal scarp forest of the Eastern Cape Province of South Africa. We compare resource availability, distribution and use in several forests that are under State and Community management. Species suitable for harvesting were identified using a simple linear programming model based on the frequency and scale of regeneration (grain) of the tree species.

Twenty high-value tree species were widely used. Nine species were apparently over-exploited and absent from Community forests. Tree densities were higher in State forests than Community forest. Size-class distribution profiles for all species, except *Harpephyllum caffrum* and *Heywoodia lucens*, displayed the characteristic inverse J-shaped distribution that generally indicates stable populations. Tree species varied in their spatial scale of recruitment, but were mostly intermediate- to coarse-grained. Two fine-grained species in State forest (*Englerophytum natalense* and *Millettia grandis*) and only one species in Community forest (*E. natalense*) had densities that could potentially sustain current levels of exploitation.

The prevailing tenurial system (i.e., State or Community forest), the tree species and category of stem size (dbh) used, all affected tree harvesting offtake levels. Trees in the 10-20 cm diameter size class (poles) were most harvested. Harvesting levels of pole-sized stems were more influenced by tree size than species, were significantly greater in Community than State forests, and could be explained by the closer proximity of households to Community ($\bar{x}=1.5$ km) than State forests ($\bar{x}=3.2$ km). Given the willingness of communities distant from forests to plant multipurpose indigenous trees, there is the potential for strengthening community forestry, and thus rural development, via forestry interventions such as agroforestry. However, the survival of these forests depends on the establishment of suitable integrated community-based institutions and management practices.

Keywords: ecological grain, harvesting, community forestry, forest products, poles, size class distributions, sustainable use, subtropical forest.

Introduction

In this study we examine the use, by rural communities, of high value tree species from the coastal scarp forests of Umzimvubu district in the Eastern Cape Province of South Africa. Our aims are to: (1) quantify resource availability and distribution in State and community managed forests; (2) document the current patterns of tree species use; (3) establish local attitudes toward forest conservation; and (4) provide harvesting and conservation guidelines for their future use.

An optimal harvesting system should consider the availability of resources, their rate of use, and their regeneration potential (McGregor, 1994; Hartshorn, 1995). Management systems that marginally alter the resource availability, and whose off-take patterns do not exceed resource regeneration, should be promoted. An understanding of the dynamics of the resource base is fundamental to developing sound management systems for harvesting resources (Seydack, 1991; Everard et al., 1995). Unfortunately there are few longitudinal studies from which the population dynamics of important species can be estimated. Forest dynamics are most often inferred from single survey, size-class frequency descriptions of population structure (Swaine et al., 1990; Shackleton, 1993a; Poorter et al., 1996; Sano 1997). Although these data have the relative merits of being easily collected and are often available from forestry records, they are static representations of population composition at a certain moment in time, and confound attempts to accurately estimate rates of change. Nevertheless, in the absence of long-term data on population dynamics, as much information as possible must be gleaned from static population descriptions. We combine a traditional analysis of population size-class frequency distributions (SCDs; Condit et al., 1998; Lykke, 1998) with a further analysis of the apparent spatial grain of population regeneration (e.g., Everard et al., 1995) from these same static data. This approach allows us to estimate what tree species can potentially sustain moderate harvesting levels, and further identify those species that should not be used. While these data do not permit an estimate of absolute optimal harvesting limits, used judiciously they do provide a useful framework upon which to base operational harvesting rates.

The success of resource management under two different tenurial systems has been widely debated: management under park (State) authorities as opposed to community management (Hartshorn, 1995; Lewis and Alpert, 1997; Nhira et al., 1998).

Sustainable use of natural resources under community tenure can improve the living standards of rural people and decrease conflicts with park authorities (Song et al., 1997; Wiersum, 1997; Deboer and Baquete, 1998; Banana and Gombya-Ssembajjwe, 2000). But there are many cases where community control has resulted in resource degradation (Paulson, 1994; Laurance, 1999) and a conflict of interests between conservationists and resource users (Noss, 1997; Sutherland and Reynolds, 1998; Gibson and Becker, 2000). On the other hand, while resource management under park authorities may initially increase the resource base, it too can lead to conflict with local communities (Vandergeest, 1996).

The division between State and Community forests in South Africa can be traced back to the early nineteenth century when tribal land was annexed (Johnson, 1983). Chiefs who accepted colonial rule were granted some authority to oversee resource use in the annexed forest reserves. These forests subsequently became Community forests (Headman's or *Nkosi* forest) while other annexed reserves became State forests under government control. Today 70% of the indigenous forest in the Eastern Cape Province (370 km², Low and Rebelo, 1997) is under State management and the remainder is under community control (Cooper and Swart, 1992; DWAF, 1997). These Community forests developed into local resource management systems, supplying the community with timber and non-timber forests products (NTFP) (van Eck et al., 1997; Mander, 1989; Mckenzie, 1989). However, in recent years a decline in the control of forest resources has been noted in both State and Community forests (DWAF, 1996), resulting in resurgence in the exploitation of forest products. We compare and contrast the use of resources from State forest and Community forest and comment on the consequences of relatively uncontrolled exploitation of Community forests. Our approach integrates our understanding of the ecological processes underpinning the population dynamics of the high-value tree species with the needs and social attitudes of local peoples towards conserving these trees in State and Community forests.

Methods

Study site

The forests in this study lie in the Umzimvubu District ($31^{\circ} 30' - 31^{\circ} 45'S$ and $29^{\circ} 15' - 29^{\circ} 30'E$) of the Eastern Cape Province of South Africa (formerly called the Transkei Republic; Fig. 1). This coastal area has an undulating terrain with altitude ranging from 0-350 metres a.s.l. Two features dominate the landscape; the numerous fragmented forests, and open grasslands with densely populated hilltop settlements. Grasslands are burned annually to regenerate new growth but not the forests. Major land uses include state conservation areas (e.g., indigenous State forests), communal grazing land, indigenous community forests, commercial plantation forestry, subsistence farming and horticultural farms that are restricted to narrow irrigable belts along rivers. Economic return from communal land is low, and the unemployment rate surpasses 80% (Ridsdale and Kallman, 1996). Household incomes range between 32-160 US\$ per month (Kieplei and Quinlan, 1996). Income is mainly from remittances from migrant workers and pensions. Natural resources are highly exploited to subsidise these meager incomes.

Resource distribution and availability

Resource distribution, availability and use was investigated during 1999 and 2000, in seven Community forests and ten State forests (Table 1) neighbouring on four local communities (Caguba, Vukandlule, Sicambeni and Tombo villages). Since local people readily identify all tree species by common name, it was possible to identify the most popular indigenous tree species used by the communities from questionnaire responses. Of these, the population densities and size-class distributions of 20 commonly used indigenous tree species were estimated using the modified Whittaker plot sampling method (Stohlgren et al., 1995). Stohlgren (1994) showed that the shape of the plot influences the number of species encountered in a sample, with more species detected in long-thin rectangular plots. Each sampling plot was 1000 m^2 (50 m x 20 m).

Ten proportionately long thin 1 m² subplots and two 10 m² subplots were arranged independently around the perimeter of the 50 m x 20 m sample plot (Fig. 2). In each plot the diameter at breast height (dbh) of each stem was measured. The number of seedlings and saplings (dbh < 5 cm) of each species were counted in the 1 m² plots. The number of sub-canopy trees in the dbh size class of 5-10 cm were counted for each species in the 10 m² plots. Subcanopy trees (dbh = 10-20 cm) were counted over the whole 50 m x 20 m plot, rather than from a smaller, centrally located 20 m x 5 m subplot as in Stohlgren's design. Similarly, canopy trees (>20 cm dbh) were counted over the whole 1000 m² plot too. Our modified design maximized the number of target species encountered in each plot, permitted independent estimates of size-class densities, and provided considerable economy in sampling effort.

Table 1: Selected community and state forests, their size and distance to the nearest community.

Forest Name	Management System	Area (ha)	Distance to nearest community (km)
Nomyezo	Community Forest	56	<0.5
Mkolwane	Community Forest	44	<0.5
Kobemnyango	Community Forest	21	<0.5
Telelo	Community Forest	15	<0.5
Umgazi	Community Forest	10	1.5
Magobiyani	Community Forest	6	2
Ludume	Community Forest	4	2
Nstwenstwe	State Forest	210	10
Nstonga	State Forest	184	2
Sonkwe	State Forest	160	3
Bulolo	State Forest	150	2
Iskilingidi	State Forest	120	9
Nyando	State Forest	101	5
Pungane	State Forest	80	2
Mbiza	State Forest	49	3
Nositemu	State Forest	10	4
Pepeni	State Forest	8	12

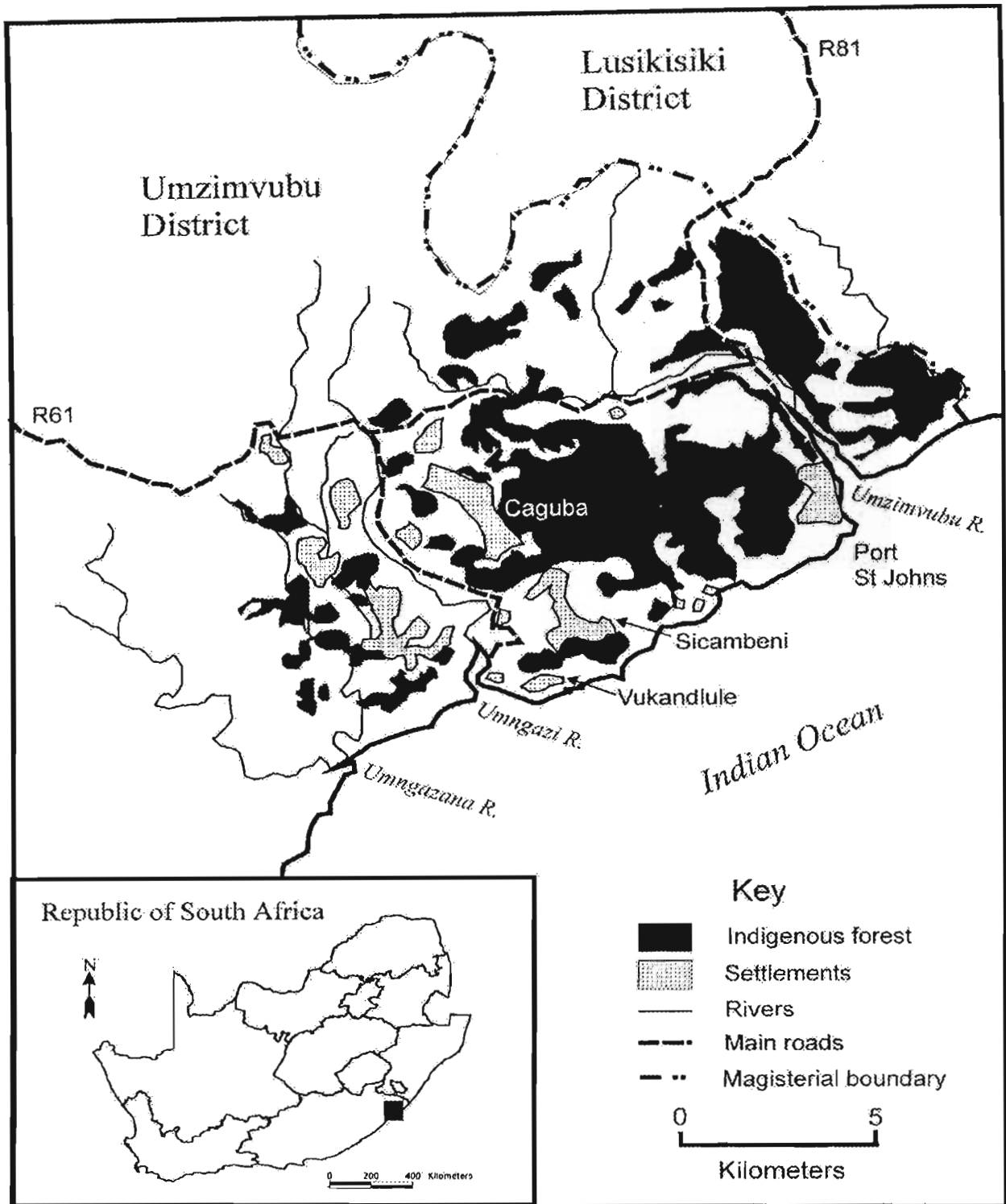


Fig. 1. Map of study area and forest included in the study.

Unlike other species *Ptaeroxylon obliquum* was most frequently observed at forest margins. In order to sample this highly desirable species we used ten 100 m x 5 m plots extending from the forest edge 100 m into the forest interior. In this way we established whether or not *P. obliquum* was a forest margin species. The numbers of seedlings/saplings of *P. obliquum* were counted in fifteen subplots (1m x 1m) placed five meters apart along the 100 m transect. Larger size classes were counted in contiguous 5 m x 5 m subplots over the 100 m length of the plot.

Seventy modified Whittaker plots were randomly sited within two broad strata - the forest margin and the forest interior. The forest interior was defined as that region of the forest greater than 60 m from the edge. The 60-m criterion was used because most edge species, for example *Millettia grandis*, do not occur more than 50 m from the forest margin (Butchart, 1989; Obiri, 1997). Fifty plots were sited in State forests and 20 in Community forests. Fewer sample plots were sited in Community forests because these forests were few in number and smaller in size compared to State forests.

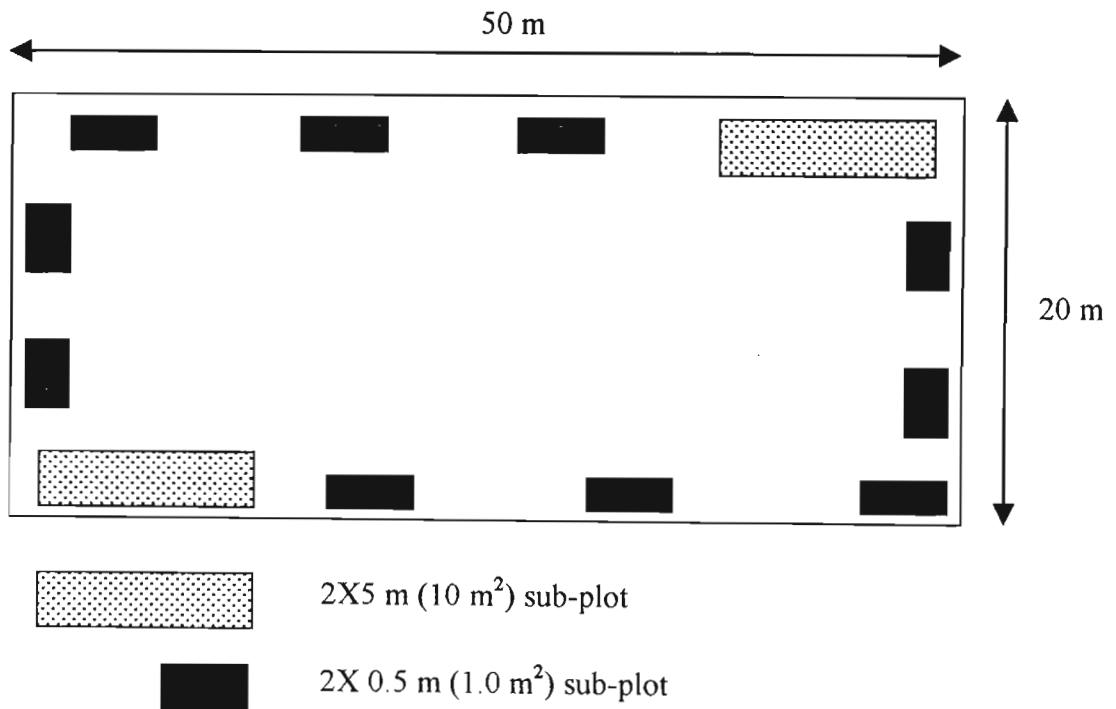


Fig. 2. Modified Whittaker plot design, adapted from Stohlgren et al. (1995).

All stumps, representing harvested trees, within the 1000 m² plots, were identified and their diameters were measured. Coppicing stumps were noted and the number of coppiced stems (diameter ≥ 1.0 cm) were counted for each stump.

Size class distributions (SCDs) were analysed using the method proposed by Condit et al. (1998) and Lykke (1998). Twenty dbh size-classes were used between 0 cm – 100cm. Each size class had an interval of 5cm (i.e., 0-5cm, 6-10 cm, 11-15 cm...96-100 cm). All stems with a dbh < 5 cm were considered newly derived stems, indicating the regenerative capacity of a species through seed germination and establishment (Shackleton, 1993a). For each of the 20 species a least-squares linear regression was calculated with size-class midpoint as the independent variable and the average number of individuals in that class (N_i) as the dependent variable. The size classes are a linear increment of dbh and were not transformed (see Condit et al. 1998), however in order to derive straight-line plots of the size-class distribution the average number of individuals (N_i) in each size class was transformed by $\ln(N_i+1)$ because some classes had zero individuals. The slopes of these regressions are referred to as SCD slopes in this study. SCD slopes were used as an indicator of population structure (Lykke, 1998). The slope values were used to summarise, in a single number, the shape of the size distribution for a species. The interpretation of the SCD slopes was based on the four types of SCD described in Everard et al. (1994). Slopes are usually negative, since larger size classes have fewer individuals, and indicate recruitment. Flat distributions with a slope of zero indicate equal numbers of regenerating trees and mature individuals. Positive slopes are sometimes referred to as unimodal since they are typically characterised by relatively many canopy individuals but no regeneration (Shackleton, 1993a; Everard et al., 1994).

The grain (sensu Everard et al., 1994) or spatial scale of forest dynamics, in particular sapling and understory tree regeneration, was investigated by interpretation of the size class distributions of species. Grain is a form of spatial analysis that compares the number of individuals of a species at the canopy level with the number of individuals in the sub-canopy / understory class (dbh=5 – 20 cm). In fine-grained species we would expect to find that canopy and sub-canopy individuals are well

represented over a small area and their SCD shows an inverse J-shaped distribution (Midgley et al., 1990; Everard et al., 1995).

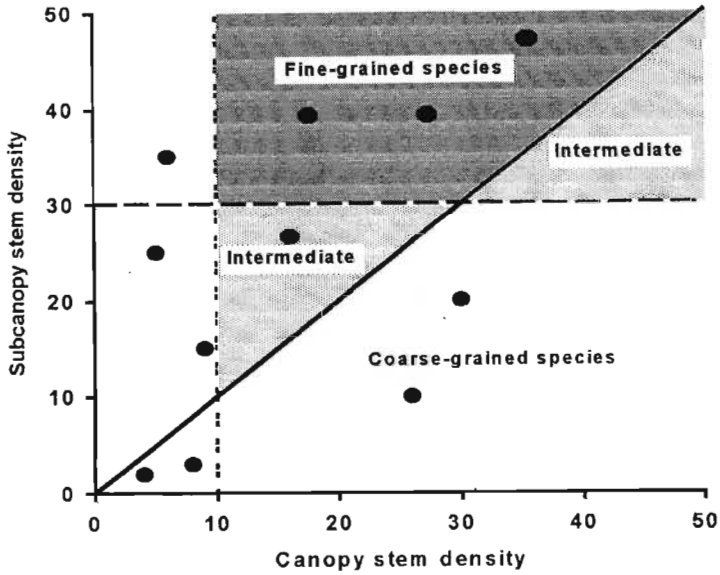


Fig. 3. Theoretical representation of the spatial scale or grain of regeneration of tree species. Stem density (individual ha^{-1}) of subcanopy/understorey stem (dbh = 5-20 cm) are plotted against the canopy stem density. In a fine-grained species subcanopy and canopy individuals are well represented over a small area. Dashed lines represent lower harvesting limits for subcanopy and canopy tree densities (see text).

These species recruit from advanced regeneration and are generally shade tolerant (Everard et al., 1995). If the scale of variation from plot to plot is small over a large number of species then a forest could be considered fine-grained (e.g., Midgley et al., 1990). In coarse-grained species there are few individuals of a species in the subcanopy relative to the canopy stratum and their SCD slope is typically flat or slightly positive. These species regenerate over a large area at relatively low densities. Because forest grain incorporates the scale of regeneration it gives forest managers a method of

predicting the influence of harvesting intensity on species survival. Since coarse-grained species regenerate over large areas at low densities they cannot withstand intense harvesting pressure. Fine-grained species, on the other hand, may tolerate moderate harvesting levels and sustained use.

In a further analysis of the grain of the harvestable fraction, we compared the mean frequency of sub-canopy stems in a sample plot, derived from the size-classes most frequently harvested (i.e. in the dbh range 5-20 cm, see later) with the number of canopy stems (reproductive age-class) of each species, in a scatter-plot. The graphical solution of a linear-programming model describing the theoretical determination of the grain of a species is illustrated in Fig. 3. The model seeks to identify those fine-grain species that can be harvested. Thus the model emphasizes the determination of grain by comparison between the stem densities of the largely vegetative and harvested (subcanopy) size-class and the reproductive "canopy" size-class. Species are of intermediate grain if their average subcanopy and canopy stems densities are similar. In general, species points that lie above this threshold are fine-grained, and those positioned below the threshold are coarse-grained. The model incorporates two limits to these generalities: (1) that stems from the harvestable size-classes should not be harvested if canopy tree densities (i.e., the density of the reproductive class) for that species are very low; and similarly (2) species with few subcanopy stems should not be harvested. These limits describe the lower harvesting limits for canopy stem density and subcanopy stem density, for fine-grained species, respectively (see Fig. 3). In our model these limits have been set from experience, and would differ from forest to forest. Plotted on the graph the potential set of harvestable fine-grained species can be identified. Furthermore, to account for the vagaries of distribution associated with ecological specialisation (e.g. gap colonisers, forest margin species or shade tolerant understory tree species) we have further refined the model by incorporating the relative abundance of the species in the habitat (i.e., species incidence, plotted on the z-axis). We assume that fine-grained species in the previously defined set that occur in more than 50% of sample plots (incidence=0.5) could be harvested.

Resource use

Open-ended questionnaires were used to compare the use of tree species by rural people between State and Community forests. The questionnaires were structured to obtain information about (1) what tree species were harvested and their uses, (2) the influence of household-to-forest distances on exploitation level, (3) local perceptions of the availability of tree species, and (4) the willingness of the local community to plant indigenous trees on their homesteads. Indigenous knowledge of the forest trees is substantial and we are confident that species were correctly identified and referred to in the questionnaires. Questionnaires were administered by one of the authors (JO) to the household heads of 60 households, with households being chosen randomly from the four villages; Caguba, Vukandlule, Sicambeni and Tombo. This represented 15% of the total number of households.

Stem densities of selected size classes representing the harvestable fraction of the population were directly compared to stump densities (harvested fraction) in the same size classes. Stump SCD's of selected species were compared between forest types (i.e., State or Community forests) to determine (1) the most exploited forest patches, (2) the most harvested tree species, and (3) the tree size-classes that were most harvested. Stumps were grouped into three diameter classes based on their local uses: (1) < 10 cm (roofing poles or rafters), (2) 10-20cm (pole/post class), and (3) > 20 cm (large tree class).

The influence of tree species and stem size on the likelihood that an individual of a species would be harvested was estimated using a harvesting index calculated as the proportion of harvested stems (stumps ha^{-1}) to the total harvestable stems (stumps plus standing trees ha^{-1}). This index was determined for the 11 most used tree species (defined as those species for which more than 15% of households claimed their use for either poles, construction, or craft work) over three size classes (dbh: <10 cm, 10-20 cm and >20cm). The harvest index ranged from zero to one, with zero occurring in cases of no harvesting and one indicating that all stems had been harvested in that size class.

Results

Resource distribution and availability

All tree species except *Harpephyllum caffrum* and *Heywoodia lucens* showed inverse J-shaped size class distributions in both State and Community Forests (Fig. 4). *H. caffrum* showed a marked absence of stems in the 40-50cm dbh size-class, and saplings were rare. *H. lucens* saplings were rare also. For both these species the SCD slopes did not differ significantly from zero (Table 2). Another species that had an unusual size-class distribution was *P. obliquum*. Individuals were absent, mostly harvested, from the dbh range of 15-20 cm, 25-40 cm and 45-70 cm (Fig. 4). *P. obliquum* saplings were abundant at forest margins but sapling density and height decreased significantly with increasing distance from the forest margin (Density [saplings.m²] = 6.27 – 0.19*distance [m], $F_{1,15} = 5.83$, $r^2 = 0.28$, $p = 0.03$; height [m] = 2.08 – 0.053*distance [m], $F_{1,15} = 15.17$, $r^2 = 0.5$, $p = 0.001$). No *P. obliquum* saplings were found more than 30 m from the forest edge, except in open gaps in the forest. Large *P. obliquum* trees (dbh>35 cm) were rare and restricted to gaps and the margins of State forests.

Based on SCD slope values the 20 analysed tree species were classified into four groups (Table 2):

Group 1 comprised *H. lucens* and *H. caffrum* and had flat SCDs (see above) whose slopes approached zero. They were characterised by poor seedling establishment, an inability to produce coppice stems and very large sized mature trees (> 70-cm dbh). They were mainly used for medicinal purposes and were regarded as declining in numbers by local people.

Group 2 displayed inverse J-shaped distributions with SCD slopes between 0.04-0.1 (Fig.3; Table 2). Saplings and seedlings occurred at low densities, but at higher densities than Group 1. Although four species showed stem coppicing, local users regarded all seven species in this group as declining populations. Apart from *P. obliquum* and *Combretum kraussii*, their uses were limited to one or two functions,

mainly medicinal (Table 3). The limited use of most species in this group (except for *P. obliquum* and *C. kraussii*) is related to the fact that these species are softwoods and decay rapidly.

Group 3 had SCDs that were clearly inverse J-shaped and had SCD slopes between 0.1-0.2. All six species in this group coppiced (Table 2). Although these tree species were intensively exploited for multiple purposes (Table 3), local users claimed that population numbers had increased over the last 20 years for all six species.

Group 4 species were characterised by abundant seedling and sapling recruitment. They had conspicuously steep inverse J-shaped SCD slopes (>0.2) (Table 2). The five species in this group readily coppiced and, with the exception of *Cassipourea gerrardii*, were regarded as abundant and as increasing populations by local users. *C. gerrardii* was used very intensively, mainly for its bark, and was regarded as scarce. Besides *C. gerrardii*, these species were used for multiple functions but, unlike the other groups, had no medicinal significance. Three species in this group (*Englerophytum natalense*, *Vepris lanceolata* and *Tricalysia lanceolata*) are particularly noteworthy for their abundance and moderate popularity among users.

The high SCD slope values of Group 4 species are a consequence of high seedling, sapling and small understory tree densities suggesting that these species are fine-grained. Conversely, the low SCD slopes and low seedling and sapling densities in Group 1 species suggest that these are coarse-grained species.

The shapes of SCD curves for a species in State and Community forests did not differ significantly in 8 of the 11 species that occurred in both forest types (Kolmogorov-Smirnov two-sample goodness of fit test, $p > 0.4$; Table 2). Those species whose SCDs differed in shape (*H. caffrum*, $KS = 0.5$, $p = 0.038$; *Celtis africana*, $KS = 0.56$, $p = 0.004$ and *Zanthoxylum davyi*, $KS = 0.73$, $p = 0.03$) were from Groups 1 and 2. These differences were largely due to there being fewer seedlings and middle- to large-sized trees in Community forests (Fig. 4).

Analysis of resource availability, using stem densities, (Table 4) showed a significantly greater density of the 10-20 cm size class trees (i.e., poles) in State forests compared to Community forests (paired t-test, $t = 2.39$, d.f. = 19, $P = 0.03$). However,

The dynamics and sustainable use of high-value tree species

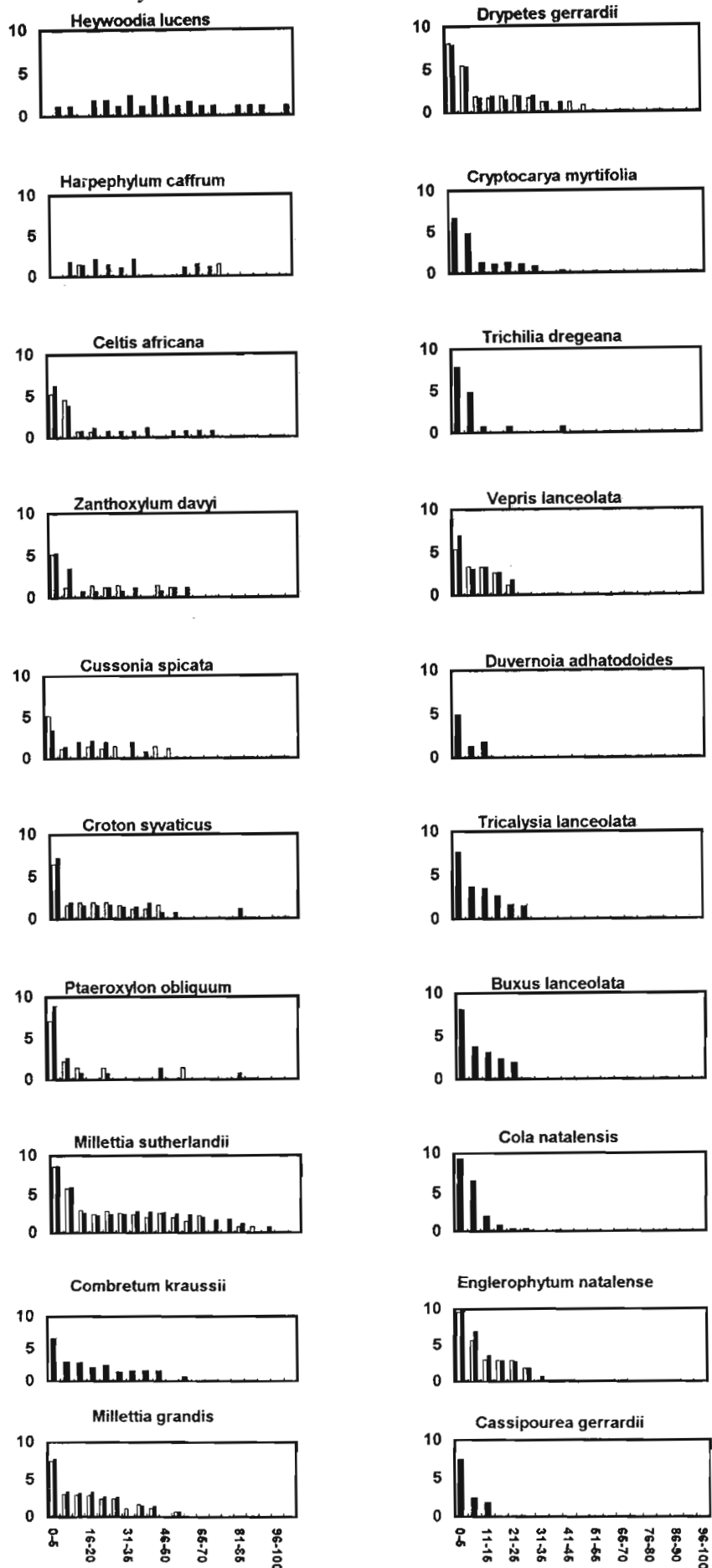


Fig.4. Diameter size-class distributions of 20 tree species arranged in order of increasing SCD slope values (see text and Table 2). Community forests are represented by open bars and state forests by filled bars. The y-axis represents $\ln(\text{individuals ha}^{-1} + 1)$ and the x-axis is the diameter size-class in 5cm intervals from 0 to 100 cm.

stem densities between the two forest types did not differ for the 5-10 cm size class (paired t-test, $t = 1.82$, d.f. = 19, $P = 0.08$) or the >20 cm size class (paired t-test, $t = 0.06$, d.f. = 19, $P = 0.96$). All of the 20 analysed and high-value species were present in State forests, however nine of these species were absent from Community forests (Table 2). The species absent from Community forests were also the most used species (Table 3). They were also mostly the fine-grained species as judged from SCD slope analysis (Table 2). This is contrary to expectation since theory suggests fine-grained species should be able to withstand moderate levels of use. Clearly considerable harvesting pressure on these fine-grained species in Community forest has led to their extirpation, and they now survive in State forest only.

Table 2: Regression slope values of size-class distributions of 20 commonly harvested species in state and community forests^a.

Species	State Forests			Community Forests			Group	Mean coppice Stems/ tree and S.E	KS value	P value
	Slope	<i>t</i> value	<i>r</i> ² (%)	Slope	<i>t</i> value	<i>r</i> ² (%)				
<i>H. lucens</i>	-0.004 ^b	1.14	6	*	*	*	1	nc	*	*
<i>H. caffrum</i>	-0.012 ^b	1.11	9	0.002 ^b	0.40	1.0	1	nc	0.50	0.038
<i>C. africana</i>	-0.040	3.61	45	-0.036 ^b	3.07	37	2	nc	0.56	0.004
<i>Z. davayi</i>	-0.050	2.16	34	-0.046	2.24	36	2	nc	0.73	0.003
<i>C. spicata</i>	-0.052	4.70	69	-0.040	2.13	31	2	nc	0.36	0.421
<i>C. syvaticus</i>	-0.053	3.43	48	-0.076 ^b	1.3	12	2	0.33 (0.30)	0.18	0.986
<i>P. obliquum</i>	-0.055	2.28	29	-0.045 ^b	2.74	35	2	1.00 (0.31)	0.06	1.0
<i>M. sutherlandii</i>	-0.059	5.06	62	-0.067	5.93	69	2	0.50 (0.19)	0.18	0.93
<i>C. kraussii</i>	-0.082	5.01	72	*	*	*	2	0.82 (0.18)	*	*
<i>M. grandis</i>	-0.102	5.02	72	-0.094	5.23	73	3	2.08 (0.16)	0.17	0.985
<i>D. gerrardii</i>	-0.105	4.46	67	-0.105	4.09	63	3	0.95 (0.24)	0.08	1.0
<i>C. myrtifolia</i>	-0.106	4.42	66	*	*	*	3	2.25 (0.62)	*	*
<i>T. dregeana</i>	-0.107 ^b	2.88	48	*	*	*	3	0.33 (0.30)	*	*
<i>V. lanceolata</i>	-0.152	5.44	81	-0.133	7.14	88	3	0.80 (0.37)	0.22	0.970
<i>D. adhatodoides</i>	-0.170 ^b	3.12	71	*	*	*	3	1.00 (0.29)	*	*
<i>T. lanceolata</i>	-0.207 ^b	5.36	85	*	*	*	4	2.30 (0.33)	*	*
<i>B. natalensis</i>	-0.235 ^b	5.27	85	*	*	*	4	2.60 (0.25)	*	*
<i>C. natalensis</i>	-0.243	3.94	72	*	*	*	4	0.60 (0.24)	*	*
<i>E. natalense</i>	-0.251	6.19	87	-0.233	5.48	83	4	2.09 (0.24)	0.11	1.0
<i>C. gerrardii</i>	-0.350	3.56	81	*	*	*	4	0.50 (0.28)	*	*

^aMean coppice stems per tree species and their S.E are given. Comparison of overall diameter size class distributions of each species in State and Community Forest (KS; Kolmogorov-Smirnov Goodness of fit 2 sample tests) and their significance levels are shown. See text for an explanation of Groups 1 to 4; and caption to Fig. 5a for full latin binomials for each species.

^bBetween 9-15 plots analysed; nc: non-coppicing tree.

* Species absent in Community Forest.

Table 3: Use of tree species for timber and NTFP by local communities. Data expressed as percentages of households (n=60) using a particular species.

Species	Construction	Medicinal	Fuelwood	Poles	Fruit	Craft	Cattle yokes	Sled	Grinding mortar	Total No. of uses
<i>D. gerrardii</i>	11	0	2	11	2	2	2	0	0	6
<i>P. obliquum</i>	23	20	16	55	0	5	0	0	7	6
<i>E. natalense</i>	20	0	2	25	34	0	5	0	0	5
<i>D. adhatodoides</i>	11	0	5	18	0	2	0	0	0	4
<i>M. grandis</i>	34	0	2	23	0	59	0	0	0	4
<i>V. lanceolata</i>	25	9	0	11	0	9	0	0	0	4
<i>C. kraussii</i>	10	0	0	12	0	10	0	0	0	3
<i>H. caffrum</i>	0	58	2	0	30	0	0	0	0	3
<i>B. natalensis</i>	5	0	2	25	0	0	0	0	0	3
<i>T. lanceolata</i>	8	0	5	25	0	0	0	0	0	3
<i>D. obovata</i>	0	0	16	0	0	0	0	2	0	2
<i>C. africana</i>	0	0	0	0	0	0	2	9	0	2
<i>T. dregeana</i>	0	62	0	0	2	0	0	0	0	2
<i>C. laevigatum*</i>	2	0	0	23	0	0	0	0	0	2
<i>H. lucens</i>	2	0	0	0	0	0	0	3	0	2
<i>Z. davyi</i>	0	20	0	0	0	0	2	0	0	2
<i>C. natalensis</i>	0	0	0	15	0	8	0	0	0	2
<i>C. myrtifolia</i>	0	55	0	0	0	0	0	0	0	1
<i>C. sylvaticus</i>	0	50	0	0	0	0	0	0	0	1
<i>M. sutherlandii</i>	0	0	0	0	0	0	0	7	0	1
<i>M. capensis</i>	0	56	0	0	0	0	0	0	0	1
<i>A. karroo</i>	0	0	59	0	0	0	0	0	0	1
<i>C. gerrardii</i>	0	80	0	0	0	0	0	0	0	1
<i>C. spicata</i>	0	20	0	0	0	0	0	0	0	1

*exotic species; see the caption to Fig. 5a for full latin binomial for each species.

Table 4: Tree densities (stems.hectare⁻¹) in State and Community forests for three commonly exploited size-classes from 20 species.

Species	Community Forests			State Forests		
	5-10 cm	10-20 cm	>20 cm	5-10 cm	10-20 cm	>20 cm
<i>E. natalense</i>	275	32	72	962	47	6
<i>M. sutherlandii</i>	316	27	0	288	20	89
<i>Z. davyi</i>	78	0	20	10	2	11
<i>D. gerrardii</i>	225	9	0	198	1	1
<i>C. africana</i>	222	2	10	44	3	11
<i>C. spicata</i>	2	3	3	3	15	7
<i>H. caffrum</i>	0	3	2	5	10	19
<i>V. lanceolata</i>	25	35	10	19	36	2
<i>P. obliquum</i>	8	3	18	15	1	9
<i>C. sylvaticus</i>	4	12	29	80	2	3
<i>M. grandis</i>	19	32	85	25	40	23
<i>H. lucens</i>	0	3	0	2	2	52
<i>B. natalensis</i>	0	0	0	40	31	1
<i>C. kraussii</i>	0	0	0	18	22	16
<i>T. lanceolata</i>	0	0	0	39	43	3
<i>T. dregeana</i>	0	0	0	119	1	1
<i>C. gerrardii</i>	0	0	0	320	2	0
<i>C. myrtifolia</i>	0	0	0	120	5	3
<i>C. natalensis</i>	0	0	0	680	7	0
<i>D. adhatodooides</i>	0	0	0	5	10	0

See the caption to Fig. 5a for full Latin binomial for each species.

The results of the linear-programming model confirmed the trends described in the SCD grain analysis above (Fig. 5a). In other words the species are broadly grouped into the Groups 1 to 4 (coarse- to fine-grained) as before. However, the final model shows that only two species (*E. natalense* and *M. grandis*) from Groups 3 and 4 are unequivocally harvestable (Fig. 5b). Although *T. lanceolata*, *V. lanceolata*, and *B. natalensis*, are all fine-grained, and stems in the harvestable size-class are abundant, their density of canopy stems is very low, reducing their potential in the model as harvestable species. However, bearing in mind that the lower canopy stem density limit

was set arbitrarily at 10 stems.ha⁻¹, and these 3 species have canopy stem densities > 6 stems.ha⁻¹, their controlled use could be contemplated. *M. sutherlandii* is unusual among all the species in that there are very high densities of canopy individuals and relatively few understory trees (Fig. 5a).

Resource use

Of the 20 commonly used species, eleven were used for garden / homestead fencing poles and for construction purposes, and seven for wood carving (Table 3). Three stem size-classes of extractive use were identified: (1) rafters and fence-droppers (cut from saplings and small trees <10 cm), (2) poles and posts (cut from mostly understory trees of 10-20 cm), and (3) sawn timber (from trees >20 cm). Distribution of stumps in these three classes differed significantly between State and Community forests ($\chi^2 = 21.8$, d.f. = 2, $P < 0.001$; Fig. 6) with Community forests having a greater density of stumps. Among the three most exploited size classes (<10 cm, 10-20 cm and >20 cm) the pole-sized trees (10-20 cm), across all species, had the highest harvest indices (0.04-0.71) (Table 5). Even in the few cases where an unfavourable species, such as the soft-wooded *Millettia sutherlandii*, were harvested, it was the 10-20 cm class that was most often cut. The trends in levels of use indicated by the harvest indices were verified by the responses to the questionnaires. The species regarded by villagers as important for fencing and construction (e.g., *Drypetes gerrardii*, *Duvernoia adhatodoides*, *E. natalense*, *P. obliquum* and *Millettia grandis*) had the highest harvest indices (>0.2), and those seldom used in construction/fencing uses (e.g. *M. sutherlandii* and *C. kraussii*) had low harvest indices (< 0.2).

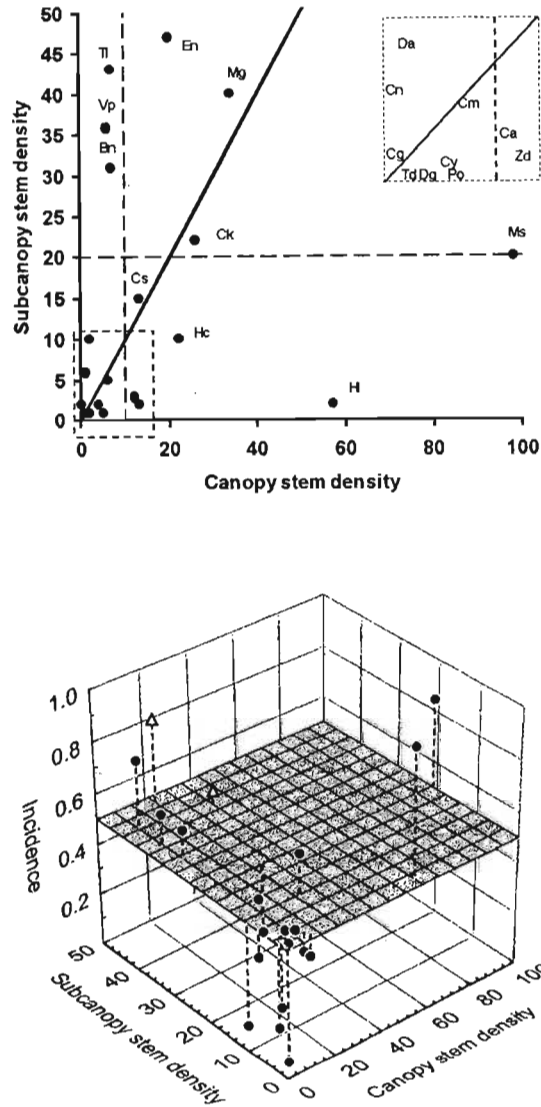


Fig. 5. The spatial scale of regeneration or grain of the 20 harvested tree species and their harvesting potential. (a) Scatter-plot of the density of harvestable subcanopy stems against canopy stem density for each species. Refer to Fig. 3. and the text for an explanation of the trends illustrated here. The inset graph is an enlargement of the area enclosed in a dashed line. Axes units are individuals ha^{-1} . Note that *E. natalense* and *M. grandis* are the only fine-grained species that should be harvested. (Bn-*Buxus natalensis*, Ca-*Celtis africana*, Cg-*Cola natalensis*, Ck-*Combretum kraussii*, Cm-*Cryptocarya myrtifolia*, Cn-*Cola natalensis*, Cs-*Cussonia spicata*, Cy-*Croton syvaticus*, Da-*Duvernoia adhatodoides*, Dg-*Drypetes gerrardii*, En-*Englerophytum natalense*, Hc-

Harpephyllum caffrum, Hl-*Heywoodia lucens*, Mg-*Millettia grandis*, Ms-*Millettia sutherlandii*, Po-*Ptaeroxylon obliquum*, Td-*Trichilia dregeana*, Tl-*Tricalysia lanceolata*, Vl-*Vepris lanceolata*, Zd-*Zanthoylum dayvi*). (b) The incidence of each species (relative abundance) is included in the above model (Fig. 5a). In the resulting 3-dimensional graphic, any fine-grained species that lies above the incidence surface (i.e., species occur in >50% of plots) are potentially harvestable. This graphic serves to confirm that *E. natalense* and *M. grandis* (Δ) are the only fine-grained species that should be harvested.

Conservation perceptions

Seventy-eight percent (n= 47) of households claimed that the indigenous trees they often used had increased in numbers over the past twenty years. For instance, *M. grandis* and *P. obliquum* were perceived as having increased in abundance, and the abundant seedling regeneration along forest margins was stated as evidence in support of this perception. Almost all homesteads (91%, n=55) agreed there was need to control harvesting of poles and posts, but, many (67%, n = 40) disagreed with a state imposed ban on harvesting. Nevertheless, most homesteads (61%, n = 37) believed that the state, through the Department of Water Affairs and Forestry, were better placed to manage harvesting in the forests than current community institutions.

Even though forests and forest products were relatively accessible, on average 2.1 km from households, 53% (n=32) of households were willing to plant their own indigenous trees. Households farthest from forests (≥ 2 km) were more willing to plant indigenous trees than those closer to forests ($\chi^2 = 9.43$, d.f. = 2, $P < 0.01$). Demand for indigenous seedlings was mostly for *P. obliquum* (50%, n=32) and *M. grandis* (33%, n=20). Also in demand were seedlings of the popular exotic species *Cestrum laevigatum*. Although considered a weed in most parts of South Africa (Henderson, 1994), this species was planted as a hedgerow in 80% (n=48) of homesteads.

Table 5. The harvesting indices for three frequently exploited size classes of 11 tree species used for poles, construction or craftwork by more than 15% of households. The index is expressed as a ratio of cut stems (B) to accumulated harvestable stems (stumps plus standing stems=A+B). Data are combined for state and community forests.

Species	Diameter size class (cm)	Stems.Ha ⁻¹ (A)	Stumps.Ha ⁻¹ (B)	Index (C) (Proportion of harvested trees =B/A+B)
<i>Drypetes gerrardii</i>	5-10	208	5	0.02
	10-20	2	5	0.71
	>20	2	1	0.33
<i>Duvernoia adhatodoides</i>	5-10	200	0	0.0
	10-20	4	4	0.5
	>20	0	2	0.5
<i>Ptaeroxylon obliquum</i>	5-10	15	1	0.06
	10-20	3	2	0.40
	>20	6	1	0.14
<i>Cola natalensis</i>	5-10	680	3	0.0
	10-20	7	4	0.36
	>20	1	0	0.0
<i>Englerophytum natalense</i>	5-10	739	3	0.0
	10-20	41	14	0.25
	>20	21	2	0.09
<i>Millettia grandis</i>	5-10	24	5	0.17
	10-20	47	15	0.23
	>20	36	10	0.22
<i>Vepris lanceolata</i>	5-10	20	3	0.13
	10-20	36	10	0.12
	>20	4	0	0.0
<i>Tricalysia lanceolata</i>	5-10	39	1	0.03
	10-20	43	8	0.16
	>20	7	0	0.0
<i>Buxus natalensis</i>	5-10	40	1	0.02
	10-20	31	6	0.16
	>20	6	1	0.14
<i>Combretum kraussii</i>	5-10	19	0	0.0
	10-20	26	5	0.16
	>20	33	0	0.0
<i>Millettia sutherlandii</i>	5-10	296	0	0.0
	10-20	23	1	0.04
	>20	91	0	0.0

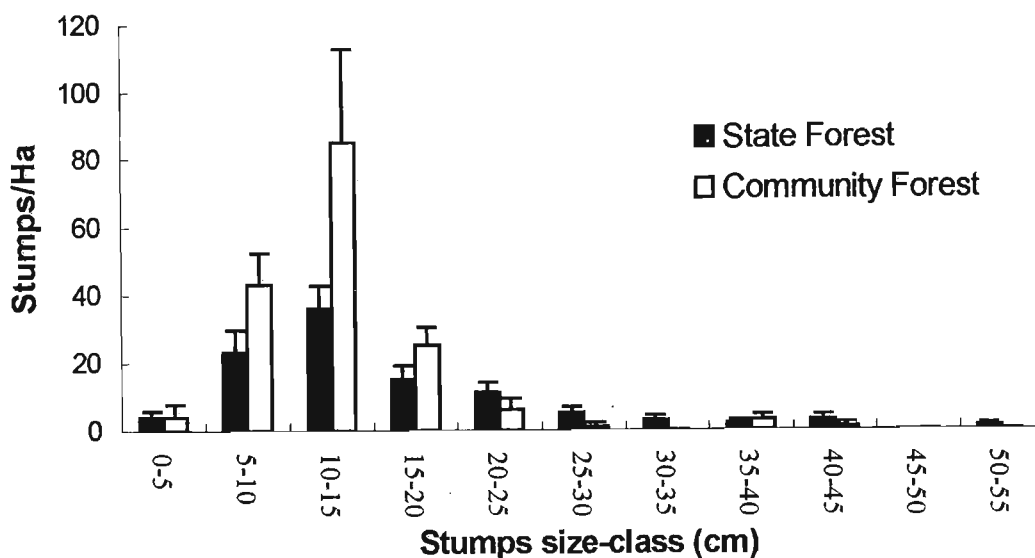


Fig. 6. Size-class distribution of stumps (i.e. harvested stems in state and community forests)

Discussion

Resource distribution and availability

Coastal scarp forests of Eastern Cape Province provide a wide array of timber and non-timber forest products (Cooper and Swart, 1992; Obiri, 1997; van Eck et al., 1997, Bokwe and Bhat, 1998; Cloete, 1998). Management of these resources requires knowledge of target species abundance and distribution, as well as regeneration and harvesting rates. In this study we apply our knowledge of the spatial scale of regeneration (i.e. forest grain) to managing these coastal scarp forests.

Forest grain has two important implications for the management of forest communities. First, it enables forest managers to decide which species and what stem size classes can be sustainably harvested. Fine-grained species like *E. natalense*, for example, produce a large supply of understory pole-sized trees whose exploitation

would have less effect on forest structure and composition than would the exploitation of other more coarse-grained species. Nonetheless, harvesting fine-grained species would still require close regulation to ensure that they are not over exploited to local extinction levels as some species have in Community forests. The harvesting of coarse-grained species such as *H. lucens* and *H. caffrum* requires careful management and should be limited. Second, forest grain enables one to determine whether forests have evolved under conditions of large-scale disturbance or fine-scale disturbance (Everard et al., 1994). Consequently, this enables foresters to use disturbance regimes as a forest management tool. For example, in forests with abundant coarse-grain species, disturbances such as fire are often necessary to maintain the natural species composition (Pinard and Huffman, 1997; Lykke, 1998). This is because coarse-grain species, like shade intolerant species, generally require gap openings or cleared forest margins to develop (Grubb, 1996; Poorter et al., 1996). If disturbances are removed it is likely that shade tolerant, fine-grain species would increase at the cost of shade intolerants or coarse-grain species.

Since over 80% of the economically exploited species in coastal ECP are intermediate- to coarse-grained, for purposes of harvesting planning, consideration of the effects of disturbance in their management is required. Disturbance is partly provided by the strong winds that occasionally blow down trees, creating gaps in these coastal scarp forests (Everard et al., 1995) and facilitating regeneration. An intriguing possibility is the putative role that harvesting plays in creating gaps and perpetuating “natural” forest dynamics (Lawes and Obiri, unpublished data). One species in particular, *M. sutherlandii*, appears to have flourished under these conditions of selective harvesting and canopy individuals dominate these forests. This is a relatively unpopular species among users (Table 5) because the timber does not work well and decays quickly. The effect of *M. sutherlandii* on the dynamics and diversity of tree communities in these forests is of concern. Thinning out this species and/or creating gaps by cutting down large trees is a management option that requires further research.

Our analysis of the grain of the harvestable fraction of the size classes of these forest tree species shows that, although five species are fine-grained and potentially harvestable, only two species should be used (*E. natalense* and *M. grandis*). We advocate the use of our conservative model of the spatial scale of regeneration for

identifying those species that can be harvested under current conditions. We have lessened the effect of combining all species and life-histories in one analysis by emphasizing that the harvestable fraction should derive from a size-class interval that is largely unproductive. Fortunately, poles are harvested mostly from the largely unproductive 10-20 cm dbh size-class. Further adjusting the harvestable size-class intervals according to the size of the mature tree is necessary to avoid recommending the use of particularly those naturally small tree species (e.g., *B. natalensis*, *T. lanceolata*) that are currently harvested for poles from the reproductive size class. Application of this model by experienced foresters should secure the long-term survival of a species in a forest by ensuring that reproductive individuals are protected, and successional collapse is avoided by limiting the numbers of vegetative understory individuals that are removed. The model does not make specific recommendations about harvesting rates, but this is not necessary if the lower limit on the extraction of subcanopy individuals of a species is not exceeded.

Patterns of use of tree species

Extra-ordinary features of the forest structure were a lack of seedling regeneration in species represented by large tree individuals, such as *H. caffrum* and *H. lucens*, and an absence of intermediate sized trees in *H. caffrum* and *P. obliquum*. This lack of seedlings is a common phenomenon of trees in South African forests (Midgley et al., 1995) and is possibly caused by high levels of seedling herbivory, although the numbers of large mammalian herbivores in these forests has declined drastically over the last century (King, 1941; Cooper and Swart, 1992). Seedling scarcity could also be attributed to the lack of suitable habitats for seedlings establishment following an increased forest encroachment and opening up of the canopy. A marked absence of intermediate sized individuals is usually an indication of a degraded population under some form of disturbance (Peters, 1996) and it is worth noting that *H. caffrum* and *P. obliquum* were intensively harvested in the past (King, 1941). In *H. caffrum* the absence of intermediate sized trees is not entirely caused by past exploitation, but is also due to its rising popularity and current use as a medicinal tree. Differences in SCD

profiles for *H. caffrum*, *C. africana* and *Z. davyi* between State and Community Forests are related to higher human disturbance in Community Forests, where these coarse-grained species are particularly vulnerable. This appears to be linked to the inability of these species to coppice, resulting in low individual survival rates in Community forests. Coppicing and resprouting facilitates the persistence of a plant in the absence of seedling regeneration, and is an advantage to plants in unpredictable environments (Basnet, 1993; Bellingham et al., 1994; Zimmerman et al., 1994; Bond and Midgley, 2001). The ability to persist through vegetative regeneration is an attractive trait in species that are targeted for harvesting (Daniels et al., 1995). Those species that cannot resprout (i.e., Group 1) should probably not be heavily used.

We expected tree selection to be influenced more by tree species than size, as this is the case elsewhere in Africa (e.g., Abbot and Homewood, 1999; Kirubi et al., 2000). However, tree stems were harvested from forest closest to homes and mainly on the basis of tree size (mostly 10-20 cm dbh) and to a lesser extent by species. The fairly indiscriminate selection of tree species for poles may be a consequence of the difficulties of carrying heavy poles back to distant homesteads, and selective harvesting for certain size classes in Community forests has also been reported for fuelwood (Shackleton, 1993b). A more plausible explanation is that the most desired species, particularly *P. obliquum*, have been over-used and thus pole cutting has diversified to previously less used species (Table 3), provided they are within the favoured 10-20 cm size range. A positive consequence of this size selection in terms of conservation, is that uncontrolled tree-cutting pressure is distributed among many species, albeit that very few species can withstand current harvesting levels, as indicated by the absence of many harvested species from Community forest. A further benefit of size-selective harvesting for management is that the abundance of stems in a class, in this case the 10-20 cm class, can be used as an indicator of the state of the standing stock in a forest, and monitoring and policing efforts can be adjusted accordingly.

The higher density of stumps in Community forests compared to State forests is due to the closer proximity of households to the former (\bar{x} = 1.5 km) than the latter (\bar{x} = 3.2 km). This is a common trend (i.e., exploitation is greatest in forest nearer to homes; Zimbabwe – Vermeulen, 1996; South Africa – Shackleton, 1993b; Kenya – Kirubi et al., 2000). In East Africa non-timber forest products use is highest among

communities living within 1.5 km of the forest edge and this use declines sharply at distances greater than 5 km (Wass, 1995). Agroforestry initiatives such as hedgerow intercropping (Huxley, 1999) and the establishment of community woodlots, as a substitute resource base, may help reduce the harvesting pressure on forests. However, these practices should not only be focussed on communities neighbouring State forests, as is the current practice (Shone, 1985), but also extended to more distant communities (i.e., those over 5km distance) (cf. Dodoma, Tanzania; Allen, 1985). This is supported by the willingness of householders farthest from forests to plant their own trees.

The high demand for *P. obliquum* and *M. grandis* seedlings for planting on homesteads, was contrary to our expectation since these two popular species were regarded by users as abundant in all forests and presumably readily available. However, it is possible that local perceptions of abundance were based on the visible presence of these species growing at the forest margin, although they are uncommon in the forest interior. It is equally likely that users have deliberately over-estimated the abundance of these species because of their popularity. Unfortunately these over-estimates are used to justify the current high harvesting rates that could result in local extinction of some species. Future forestry management will have to reconcile the difference in perception of tree abundance with the actual abundance of a species, since no harvesting limit on a species will be accepted if it is perceived to be abundant.

Conservation recommendations

Although we recommend the use of some fine-grained, Group 4, species in State forest these same species have been virtually extirpated in Community forest. Clearly, the uncontrolled use of even the most abundant and resilient resources can result in severe resource depletion and possibly irreversible degradation (e.g., Struhsaker, 1997). Community forests are heavily used and in poor condition because the local community can legally use these forests, but there are also fewer Community forests, they are closer to homesteads, and they are mostly small in area. It is inevitable that the larger State forests will have to be used by local communities, if for no other reason than it is difficult to justify the existence of exclusive protected areas when

faced by burgeoning community needs (e.g., Vermeulen, 1996). To ban the use of State forest resources is not a viable option because this would effectively alienate rural communities, making it difficult to both implement sustainable community forestry practices and to control resource use. Thus, although the use of Group 4 species in State forests is recommended, the survival of State forests is contingent upon (1) the implementation of suitable control measures, and (2) identifying and securing suitable alternative sources for forest wood products (e.g., woodlots and agroforestry initiatives) to lessen the harvesting pressure on these forests.

Although knowledge of a forest's resource base and its ecological functioning are prerequisites for sustainable forest management, equally important is the attitude of communities towards a management system. Where attitudes are negative, mainly because access to resources is limited, conservation initiatives often fail (Obua et al., 1998; Banana and Gombya-Ssembajjwe, 2000; Gibson and Becker, 2000). Because the communities in our study have a generally positive attitude towards State management we are optimistic that the current resource overexploitation can be reversed. The somewhat negative attitude toward community management may partly be explained by a reduction in local chiefs' control over Community forest caused by new political dispensations. With the introduction of the Transitional Rural Council, Chiefs' powers have dwindled (Pycroft 1996, 1999) and so have forests under their control (DWAF, 1997). This loss of influence may have reduced local confidence in the ability of the traditional tenure system to manage forests. Notwithstanding these attitudes and perceptions, participation of chiefs and the community in forest management is highly important (Poffenberger, 1996, Gibson and Becker, 2000) and we suggest a joint management forum for this region.

Conclusions

Sustainable development is critical to the survival of indigenous forests in Umzimvubu district. Based on the ecological findings of this study we can conclude that forests under State authority in Umzimvubu district are in better condition and more biodiverse than Community forests. Community forests had (1) higher stump densities, (2) lower densities of seedlings and saplings in important species such as *H.*

caffrum, *C. africana* and *Z. davyi*, (3) lower densities of the highly sought after 10-20 cm dbh size-class trees, and (4) fewer high-value species, than State forests. Shackleton (1993b) and Higgins et al. (1999) have reported similar trends in community-managed woodlands in South Africa.

This study provides an ecological basis for the sustainable harvesting of high-value tree species from forests of the Umzimvubu district. Direct recommendations to arise from this study are that:

- (1) harvesting of pole-sized stems should be limited to fine-grained species, currently one species only in Community forest (*E. natalense*) and two in State forest (*E. natalense* and *M. grandis*) can support moderate use. Some fine grained species such as *T. lanceolata*, *B. natalensis* and *V. lanceolata* may be harvestable in State forest under strictly controlled conditions;
- (2) pole and post harvesting and the overall state of the standing stock can be rapidly monitored by estimating the abundance of trees in the highly desired 10-20 cm dbh size class;
- (3) since households far from forest were willing to plant indigenous trees, there is potential for establishing community forestry initiatives, such as woodlots (Ahlback, 1995). Emphasis could be placed on indigenous trees, such as *A. karroo* and *P. obliquum*, which are popular with the communities; and
- (4) the establishment of suitable integrated community-based institutions and management practices are required for forests in the region to have any future at all.

While our data provide a strong case for State control over forests, community management options should not be dismissed. Effective protection of either tenurial category of forest is impossible without the co-operation of local users and collaboration between State and Community institutions is required if indigenous forest is to survive. A joint forest management system steered by the local chief, representatives of the Transitional Rural Council and resource managers from the Forestry Department would appear to be the most effective way of moving forward.

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CHAPTER IV

THE SUSTAINABILITY OF FUELWOOD AND MEDICINAL TREE SPECIES USE IN THE COASTAL FORESTS OF EASTERN CAPE PROVINCE, SOUTH AFRICA

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Abstract

Ecological and social surveys were conducted to determine the types, availability and harvesting dynamics of fuelwood and medicinal tree species in state and community-controlled forests in Umzimvubu District of the Eastern Cape Province of South Africa. Sampling plots of 1000 m² were used to estimate the standing forest biomass and the natural deadwood yield from the forests while questionnaires were administered to investigate the tree species used and amounts harvested.

Eighteen species were identified with fuelwood and medicinal uses and their densities were higher in state forest than community forest. Fuelwood species were selectively harvested and some highly desired species e.g. *Ptaeroxylon obliquum* were scarce. Fuelwood usage was supplemented mainly with paraffin whose use increased as household-forest distance increased. Estimates of deadwood yield were higher in state forests (1.24-2.36 tonnes.ha⁻¹.yr⁻¹) than in community forests (0.64 – 1.22 tonnes.ha⁻¹.yr⁻¹). State forests provided 92% of the total collectable fuelwood while community forests (fewer and smaller in area) provided only 8%. The average estimate of deadwood produced in all state and community forests (1898 tonnes.yr⁻¹) marginally balanced that removed through fuelwood harvesting (1532 tonnes.yr⁻¹). Thus, fuelwood use was unsustainable. Very little deadwood is left behind after fuelwood harvesting as residual fuelwood and a build up of fuelwood stocks is unlikely.

Medicinal tree use, largely through stem debarking, is unsustainable and mostly involves eight species whose potential to regenerate from either stem coppices or bark regrowth is poor. Only one species, *Macaranga capensis*, had substantial bark regeneration that could sustain the harvesting intensity. The intensity of tree debarking is significant and at least 28% of the debarked trees, particularly the ring-barked individuals, die. We recommend the establishment of woodlots around both community and state forests using indigenous tree species that have the potential for medicinal and fuelwood use.

Keywords: deadwood, forests, medicinal trees, sustainable use, fuelwood, bark use

Introduction

Current levels of use of fuelwood and medicinal tree species in the coastal forests of the Eastern Cape Province (ECP) of South Africa are thought to be unsustainable (DWAF 1997). This study addresses the proposition that the demand for fuelwood and medicinal components from trees is greater than the regenerative capacity of the trees in these forests. Over-harvesting will lead to a simultaneous ecological and economic crisis in the area. In this study medicinal tree and fuelwood use are analysed together, because most medicinal tree species are ring-barked, subsequently die, and form part of the deadwood component that is collected as fuelwood.

The collection of fuelwood and medicinal products is recognised as a significant threat to forest habitats in many parts of South Africa (Shackleton 1993, DWAF 1997, Mander 1998, Netshiluvhi 1999, Lawes and others 2000). The management of the use of these resources, essentially limiting access to them, is complicated by their important role in the socio-economic livelihoods of the rural poor (Cunningham 1989, 2001, Shackleton 1996, Lawes and others 2000). Traditional medicines worth US\$61million and an estimated 11 million tonnes of fuelwood (66% from natural forests and woodlands) are harvested annually by rural communities in South Africa (DWAF 1997, Mander 1998). For example in the ECP alone, some 50 forest plant species are used for traditional purposes such as craftwork (woodcarving, basketry, and mat-weaving), medicines, fuelwood and construction posts and poles (Cawe and Ntloko 1997, van Eck and others 1997, Bokwe and Bhat 1998, Cawe 1999, Obiri and others in press, see Chapter III).

In ECP the increasing scarcity of fuelwood and medicinal resources is largely linked to three factors. First, impoverished rural communities (range in household income = US\$ 32-160.month⁻¹, Kieplei and Quinlan 1996) continue to increase in population size, yet the forests upon which they are heavily dependent are shrinking in area, or their wood resources have been selectively harvested (Cooper and Swart 1992, Obiri and others in press, Chapter III). Second, the use of other forms of non-commercial fuels besides firewood, for example, agricultural residues and livestock

refuse (dry cow dung), is declining (Mander and Quinn 1995) and thus more pressure is being directed toward harvesting forest wood. Thirdly, conflicting views on forest ownership and management (DWAF 1997) and the difficulties associated with the policing of forest resource use (Cooper and Swart 1992, Castley and Kerley 1996, DWAF 1997, Obiri 1997) results in unchecked harvesting that exacerbates fuelwood scarcity. Underlying these factors is a shift in type of fuelwood species used, possibly in response to dwindling supplies of preferred species (c.f. Brouwer and others 1997, Kirubi and others 2000), and is a feature common to the use of open access resources (Hardin 1995).

Two types of tenurial or forest ownership systems occur in ECP; state management and community control. Harvesting of deadwood for fuelwood is authorised in both state and community forests. Medicinal tree harvesting however is limited and harvesters require permits. Most of the popularly harvested trees e.g. *Ptaeroxylon obliquum*, *Trichilia dregeana* and *Cassipourea gerrardii* are protected species and their harvesting is illegal. Although harvesting of fuelwood and medicinal tree species is common in forests under both tenurial systems, it is rampant in the community-controlled forests (Cooper and Swart 1992, DWAF 1997, Obiri and others in press, Chapter III). Here we compare the patterns of use of fuelwood and medicinal tree species between state forests and community forests. We further investigate the forest resource use by: (1) identifying the popular and intensively used fuelwood and medicinal tree species; (2) estimating the availability of fuelwood species (in terms of the densities and basal areas of the standing crop and estimate off-take rates) under the two tenurial systems; (3) estimating the regenerative potential of fuelwood and medicinal tree species as judged by their coppicing ability (Bond and Midgley 2001), bark regrowth and their spatial scale of regeneration and size-class distributions (Obiri and others in press; Chapter III); and lastly (4) suggest management options for the sustainable use of fuelwood and medicinal tree species from coastal forests in Eastern Cape Province.

Methods

Social survey

This study was undertaken in the Umzimvubu District of the Pondoland coast of Eastern Cape Province in South Africa ($31^{\circ} 30' - 31^{\circ} 45'S$ and $29^{\circ} 15' - 29^{\circ} 30'E$) during 1999 and 2000. We used open-ended questionnaires to compare rural tree use in state and community controlled forests. Questionnaires were administered to four local communities (Caguba, Vukandlule, Sicambeni and Tombo villages) neighbouring on the forests surveyed in this study. The questionnaires were structured to obtain information about (1) what tree species were harvested and their uses, (2) the intensity of fuelwood harvesting and medicinal tree exploitation, and (3) the influence of household to forest distances on exploitation level. Questions on the use of medicinal trees and the amount of fuelwood harvested were focussed on the tree species common to forests only and not the open grassland species such as *Acacia karroo*. Questionnaires were administered to heads of 60 households, with households being selected randomly across the four villages. There were 400 households (Mgudlwa *pers. comm.* 1999) and our sample represented 15% of the total.

The amount of fuelwood collected in the forests was measured in wood head-loads (iiyanda) collected per household per month. Head-loads were weighed using a 100 kg Salter hanging scale. A head-load (HL) consisted of small- to medium-sized tree branches and split stems from dead or damaged trees that were fastened into a bundle using forest vines. The quantity of fuelwood harvested from state and community forests was obtained from monthly estimates of the number of headloads collected by households from each forest. The uses of energy sources other than wood fuel (e.g., agricultural waste, livestock residues, paraffin, gas, and electricity) were also recorded.

Quantification of the forest stock

Stocking or standing crop densities and basal areas of the fifteen key fuelwood and medicinal trees species were estimated in seven community forests and ten state forests using seventy 0.1 hectare sample plots (50 m x 20 m). The plots were randomly located in two strata of the forest – forest margins and the forest interior. Forest margins were considered as those areas of the forests that were within 60m from the forest edge (Butchart 1989, Obiri 1997) while areas > 60m as the forest interior area. The fifteen species represented the most widely used fuelwood and medicinal tree species of the forest (see later, Table 3 and 4). Two other species also known for fuelwood use in the area (*Acacia karroo* and *Dalbergia obovata*) were not considered in this study. *A. karroo* does not make part of the head-load fuelwood collected in the forests. It is not a forest species but a typical savanna grassland tree (Grant and Thomas 1998). The other species, *D. obovata*, is a small forest climber whose twigs and small branches, although collected as fuelwood, were considered of negligible weight in a headload. Also not inventoried was the medicinal tree *Macaranga capensis*, which in these forests was rarely found in the forest but occurred at the forest margin. This species was seldom used for fuelwood, however the intensity of debarking (see later) was assessed for *M. capensis* trees.

Of the 70 sample plots, 50 were sited in state forests and 20 in community forests. Fewer plots were sited in community forests because these forests were fewer in number and smaller in size compared to state forests. Estimates of the density of the standing stock (stems.ha⁻¹) and basal areas (m².ha⁻¹) were calculated for all woody saplings and trees > 5 cm dbh. The basal area, considered as the cross sectional area of a tree at breast height, was obtained by summing the cross sectional areas of all individual trees and saplings in a hectare.

Medicinal tree assessment

Only those tree species whose bark was collected for medicinal purposes are included in our analysis of medicinal tree use. Trees were debarked from a height of

1.0 m above the ground to varying heights on the stems. The severity of debarking was assessed on a five-point scale (Table 1a) while the trees' regenerative potential or degree of bark recovery in previously debarked trees was measured on a three-point scale (Table 1b).

Table 1a: Five-point scale of severity of stem debarking of medicinal trees

Debarking scale	State of tree stem
1	Tree not ring debarked but some small stem portion(s) debarked
2	Tree ring debarked to 25% stem length
3	Tree ring debarked to 50% stem length
4	Tree ring debarked to 75% stem length
5	Over 75% stem length ring debarked

Table 1b: Three-point scale of bark recovery of stems of medicinal trees

Stem recovery scale	State of tree stem
1	No stem recovery- no regrowth on debarked stem portion
2	Partial stem recovery-regrowth only on edges of debarked portion
3	Complete stem recovery- regrowth of debarked stem portion

Estimation of forest wood volume

To our knowledge, no ecological studies quantifying the standing crop of forest biomass and production of harvestable deadwood have been done in coastal Pondoland forests (see also Cawe 1999). It is possible to obtain a reliable estimate of the growing stock and annual yield of forest deadwood if the forests' basal area and average height are known (Wass 1995, Uma Shankar and others 1998). The growing stock was defined as the volume of wood in a hectare of forest ($\text{m}^3 \cdot \text{ha}^{-1}$), and estimated as the product of basal area ($\text{m}^2 \cdot \text{ha}^{-1}$) and mean tree height (m) multiplied by a form factor (Uma Shankar and others 1998). The form factor describes a tree's shape with respect to volume and is measured as the ratio of a tree's volume to that of a cylinder of equal basal cross-sectional area and height (Philip 1998). In the southern Cape forests of

South Africa, Geldenhuys (unpublished data) has estimated an average form factor value of 0.7. We used the same value in our study although many trees bear more branches than those in the southern Cape forests and are therefore expected to have a lower form factor value. Our estimate of standing wood volume should therefore be regarded as approaching the upper limit. The volume estimates were converted to tonnage by equating 1.25m^3 to 1 tonne (Banks 1976).

Estimation of forest deadwood

There are no estimates of annual deadwood yield from South African natural forests of mixed species and ages. However, in wooded savannas of South Africa, Shackleton (1998) estimated the annual harvestable deadwood yield as varying between 1.5-1.7% of the growing stock. We adopted three estimates of deadwood yield in our forests. First, an uppermost limit of deadwood yield was estimated using a high yielding estimate of 1.9% of the growing stock of forests. This estimate is derived from the highly productive tropical rainforests of southwest Asia (Yoneda and Tamin 1990) and the Amazon (Korning and Balslev 1994). However, because our coastal forests are sub-tropical in origin and are thus less productive than tropical forests (Lugo and Brown 1991, Ricklefs and Miller 1999), we argue that they yield less deadwood than 1.9% of the growing stock. The second deadwood estimate was estimated as 1.45% of the forest stock, or the average of the upper- and lower-most possible deadwood yields in the forests i.e. 1.0% and 1.9%. A third and lower-most estimate was estimated as 1.0% of the growing stock. This estimate is derived from the coastal tropical forest of East Africa (Wass 1995) and is likely to be closest to deadwood yields from the subtropical coastal forests in this study. The three estimates were used to ascertain if the deadwood production in these forests balanced the fuelwood consumption rate. Deadwood yields from the 1.9%, 1.45% and 1.0% estimates are hereafter referred to as the maximum, average and minimum deadwood estimates respectively.

Results

Important fuelwood and medicinal tree species

Indigenous tree species were used for several purposes (Obiri and others in press; Chapter III). Ninety-five percent of households (n = 57) used fuelwood for cooking (Table 2). *Ptaeroxylon obliquum*, *Duvernoia adhatodoides* and *Englerophytum natalense* were the most widely used fuelwood tree species (Table 3) with respondents stating that these species produced an intense, slow burning and smokeless fire.

Ten species were commonly used in traditional medicines where extracts were derived from either their bark or roots. The most popular medicinal trees were *Cassipourea gerrardii* (used in 80% of households), *Trichilia dregeana* (62%), *Harpephyllum caffrum* (58%) and *M. capensis* (56%) (Table 4).

Patterns of fuelwood harvesting and use

In 90% of households (n = 54) women collected fuelwood and used small implements like machetes to cut off dry branches from dead trees. Most households (95%, n = 57) indicated that they collected fuelwood from dead forest wood resulting from (1) wind fallen trees, (2) snapped trees, (3) snags (dead standing trees), and (4) the remains of trees that had been cut for other purposes.

Eighty-seven percent of households (n = 52) supplemented fuelwood use with paraffin, 23% (n = 14) with butane gas and 7% (n = 4) had access to electricity (Table 2). Livestock residues and agricultural crop waste were each used as fuel by 3% of households.

Sixty percent of the households obtained their fuelwood needs from community forests and 40% from state forests. These data suggest that state forests experience lower levels of fuelwood exploitation than community forests.

Table 2: Fuel types and their current use (%) as compared to previous studies in Eastern Cape Province

Fuel type	Amount of fuel used (% of households, N=60)			
	Viljoen (1994)	Ward (1994)	Mander and Quinn (1995)	Current study (2001)
Paraffin	95%	89%	95%	87%
Gas	7%	8%	7%	23%
Wood	59%	-	59%	95%
Dung	31%	89%	32%	3%
Crop waste	22%	-	-	3%
Electricity	-	-	-	7%

The amount of fuelwood harvested per household per month was independent of the distance of households from the forest ($r = 0.045$, d.f. = 52, $P = 0.7$). Thus, although state forests are further from households (3.19 ± 0.65 km, $\bar{x} \pm 1S.D.$, $n = 23$) than community forests (1.46 ± 0.25 km, $\bar{x} \pm 1S.D.$, $n = 34$), the number of headloads harvested from state and community forests did not differ ($\chi^2 = 1.87$, d.f. = 3, $P = 0.59$). A monthly mean of 13.0 ± 1.9 ($\bar{x} \pm 1S.D.$, $n = 31$) headloads per household were collected from community forests and 12.0 ± 1.8 ($\bar{x} \pm 1S.D.$, $n = 22$) headloads per household from state forests.

The mean mass of a headload was 25.3 ± 1.5 kg ($\bar{x} \pm 1S.D.$, $n = 44$) and the mean total mass of fuelwood collected per household per month did not differ significantly between community forests (329.0 ± 46.7 kg) and state forests (304 ± 44.8 kg) ($t = 0.39$, d.f. = 50, $P = 0.78$). With a mean household size of 7 people (Mukolwe 1999) this gives a fuelwood harvest estimate of $564 \text{ kg} \cdot \text{capita}^{-1} \cdot \text{year}^{-1}$ in community forests and $521 \text{ kg} \cdot \text{capita}^{-1} \cdot \text{year}^{-1}$ in state forests, confirming slightly less use of state forests. Given

the study area had 400 households, 60% and 40% of whom collected 329 kgs.month⁻¹ and 304 kgs.month⁻¹ of fuelwood from community forest and state forest respectively, an annual estimate of 948 tonnes are collected from community forest and 584 tonnes from state forest.

Households that were farther from the forest used significantly more paraffin (Paraffin [litres] = 18.6+4.53*distance (km), $F_{1,48} = 9.57$, $r^2 = 0.17$, $P = 0.003$). On average each household requires 26.8 litres (± 2.9 litres, $\bar{x} \pm 1$ S.D., $n = 50$) of paraffin per month (at a cost of US\$12 per month) to subsidise wood use.

Table 3: Stocking densities and basal areas of fuelwood trees and medicinal trees (dbh>5cm) in state and community forests. Mean coppice shoots per stem (± 1 S.E.), are given.

Species	Uses	Mean coppices per tree (± 1 S.E.)	Community Forests		State Forests	
			Stocking density (stems.ha ⁻¹)	Basal area (m ² .ha ⁻¹)	Stocking density (stems.ha ⁻¹)	Basal area (m ² .ha ⁻¹)
<i>H. caffrum</i>	F/M	nc	6	1.13	37	4.20
<i>P. obliquum</i>	F/M	1.0 (0.31)	21	3.03	26	2.80
<i>M. grandis</i>	F	2.1 (0.16)	80	2.87	99	3.41
<i>Z. davyi</i>	M	nc	78	0.39	25	2.19
<i>D. gerrardii</i>	F	1.0 (0.24)	254	3.03	228	3.06
<i>V. lanceolata</i>	M	0.8 (0.37)	62	0.82	61	1.50
<i>C. spicata</i>	M	nc	15	1.15	31	1.19
<i>E. natalense</i>	F	2.1 (0.24)	328	2.96	1029	6.63
<i>C. sylvaticus</i>	M	0.3 (0.3)	45	3.77	87	0.89
<i>B. natalensis</i>	F	2.6 (0.25)	0	0	78	1.20
<i>C. myrtifolia</i>	M	2.3 (0.62)	0	0	131	1.03
<i>T. lanceolata</i>	F	2.3 (0.33)	0	0	89	1.29
<i>T. dregeana</i>	M	0.3 (0.30)	0	0	122	0.80
<i>C. gerrardii</i>	M	0.5 (0.28)	0	0	332	1.63
<i>D. adhatodoides</i>	F	1.0 (0.29)	0	0	15	0.16

F - Fuelwood tree; M- Medicinal tree; nc- Non coppicing tree

Table 4. The mean intensities of stem debarking and stem recovery in ten trees species commonly used for medicinal purposes. The medicinal uses and percentages of households that harvest and use the medicines are shown.

Species and abbreviations	Debarking Scale (mean)	Stem Recovery	Medicinal Uses	Household Use (%)
<i>Cassipourea gerrardii</i> Cg	4.1	1.1	skin protection	80
<i>Trichilia dregeana</i> Td	2	1.88	enema / STD ¹	62
<i>Harpephyllum caffrum</i> Hc	2.5	1.75	stomach ailments	58
<i>Macaranga capensis</i> Mc	2.7	2.95	skin ailments	56
<i>Cryptocarya myrtifolia</i> Cm	2.2	1.2	headache/fever treatment	55
<i>Croton syvaticus</i> Cy	1.8	1.3	stomach ailments	50
<i>Cussonia spicata</i> Cs	*	*	stomach-aches STD ¹	20
<i>Ptaeroxylon obliquum</i> Po	1.2	1.4	control of ticks, ritual sacrifices	20
<i>Zanthoxylum davyi</i> Zd	1.1	1.3	Purgative, tooth aches	20
<i>Vepris lanceolata</i> Vl	*	*	Influenza	9

¹ Sexually transmitted diseases;

*Roots, and not bark, are harvested

Availability of fuelwood and medicinal trees

The growing stock of fuelwood tree species was $155 \text{ m}^3 \cdot \text{ha}^{-1}$ and $80 \text{ m}^3 \cdot \text{ha}^{-1}$ in state and community forests respectively (Table 5a), and comparable to $108.8 \text{ m}^3 \cdot \text{ha}^{-1}$ in similar state controlled coastal forests in tropical East Africa (Wass 1995). Maximum estimates of annual deadwood yield in state forests ($2.36 \text{ tonnes} \cdot \text{ha}^{-1} \cdot \text{yr}^{-1}$) and community forests ($1.22 \text{ tonnes} \cdot \text{ha}^{-1} \cdot \text{yr}^{-1}$) (Table 5a) were much larger than those of mesic areas in wooded savannas of South Africa ($0.44\text{-}0.59 \text{ tonnes} \cdot \text{ha}^{-1} \cdot \text{yr}^{-1}$ Shackleton 1998). The average and minimum estimates of annual deadwood yield were 1.8 and $1.24 \text{ tonnes} \cdot \text{ha}^{-1} \cdot \text{yr}^{-1}$ in state forests and 0.93 and $0.64 \text{ tonnes} \cdot \text{ha}^{-1} \cdot \text{yr}^{-1}$ in community forests (Table 5a). Multiplying the annual deadwood yield by the total forest cover in the study area (state forest = 972 ha and community forest = 156 ha) gave a maximum estimate of deadwood yield of $2296 \text{ tonnes} \cdot \text{year}^{-1}$ in state forest and $191 \text{ tonnes} \cdot \text{year}^{-1}$ in community forest (Table 5b). Alternatively, the average and minimum estimates of deadwood yield were 1752 and $1208 \text{ tonnes} \cdot \text{yr}^{-1}$ in state forest and only 146 and $100 \text{ tonnes} \cdot \text{yr}^{-1}$ in community forest. Thus 92% of collectable deadwood is in state forest while only 8% is in community forest.

The maximum estimate of deadwood production in community forest ($191 \text{ tonnes} \cdot \text{year}^{-1}$) fell far short of the consumption rate ($948 \text{ tonnes} \cdot \text{year}^{-1}$) indicating a worrying imbalance in the supply-demand ratio. This imbalance was worse for average estimates (i.e. $146 \text{ tonnes} \cdot \text{year}^{-1}$ against $948 \text{ tonnes} \cdot \text{year}^{-1}$) and critical for minimum estimates ($100 \text{ tonnes} \cdot \text{year}^{-1}$ against $948 \text{ tonnes} \cdot \text{year}^{-1}$) (Table 5b). Although the maximum estimate of deadwood production in state forests ($2296 \text{ tonnes} \cdot \text{year}^{-1}$) was higher than its extraction ($584 \text{ tonnes} \cdot \text{yr}^{-1}$), overall, the maximum estimate of deadwood produced in all state and community forests ($2487 \text{ tonnes} \cdot \text{yr}^{-1}$) was only one and a half times larger than that harvested as fuelwood ($1532 \text{ tonnes} \cdot \text{yr}^{-1}$). These estimates suggest that on an annual basis only a small stock of deadwood is left behind in forests as residual fuelwood after harvesting is completed. Thus a considerable build up of forest deadwood is unlikely. The situation was worse when the average deadwood estimate was considered since the overall deadwood yield ($1898 \text{ tonnes} \cdot \text{yr}^{-1}$)

marginally balanced fuelwood harvested from the forest (1532 tonnes.yr⁻¹). Using the minimum estimate, the annual deadwood yield was much less than that harvested as fuelwood (i.e. 1308 against 1532 tonnes.yr⁻¹). However, because these estimates are based on liberal estimates of forest deadwood production (i.e. the assumptions of a high form factor of 0.7 and high annual deadwood yields - 1.9% and 1.45% of the growing stock), in reality deadwood production is expected to be much less.

Significantly higher fuelwood species densities were found in state forests compared to community forests ($\chi^2 = 246$, $P < 0.001$, d.f. = 7; Table 3). Similarly, the density of medicinal tree species was significantly higher in state forests than community forests ($\chi^2 = 439$, $P < 0.001$, d.f. = 8; Table 3). While all the tree species used for fuel and medicines were present in state forests, three fuelwood species and three medicinal species (among those 15 species sampled) were absent from community forests (Table 3) as a result of previous intense exploitation (Obiri and others in press; Chapter III).

Table 5a: Maximum, average and minimum estimates of the growing forest stock and deadwood yield in state and community forests.

Forest type	Density (trees.ha ⁻¹)	Basal area (m ² .ha ⁻¹)	Mean height (m) Trees >5cm dbh	Growing stock		Deadwood yield (t.ha ⁻¹ .yr ⁻¹)		
				Volume (m ³ .ha ⁻¹)	Weight ¹ (t.ha ⁻¹)	Min. Estimate	Ave. estimate	Max. estimate
State	131	31.71	7	155	124	1.24	1.80	2.36
Community	89	19.15	6	80	64	0.64	0.93	1.22

¹ The volume stocks have been converted to dry weight by using $1.25\text{m}^3 = 1 \text{ tonne}$ after Banks C.H. (1976)

Table 5b: Maximum, average and minimum estimates of annual production, extraction and net deadwood in state and community forests.

Forest type	Area (ha)	Annual deadwood yield estimate (t.ha ⁻¹ .yr ⁻¹)			Total deadwood yield estimate (t.yr ⁻¹)			Deadwood total yield (%)	Total deadwood extracted (t.yr ⁻¹)	Net available deadwood estimate (t.yr ⁻¹)		
		Min.	Ave.	Max.	Min.	Ave.	Max.			Min.	Ave.	Max.
State	972	1.24	1.80	2.36	1208	1752	2296	92	584	624	1168	1712
Community	156	0.64	0.93	1.22	100	146	191	8	948	-848	-802	-757

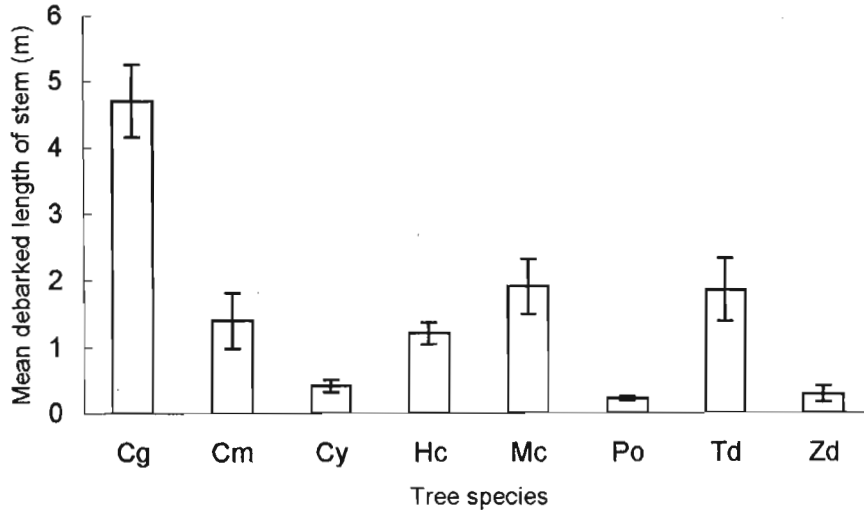


Fig. 1. Mean length of stem debarking in eight different species commonly used for bark medicines. Species abbreviations are as in Table 4.

Patterns of medicinal tree harvesting

Sixty-seven percent ($n = 55$) of the debarked trees were ring-barked and stripped over at least one quarter of their entire stem lengths, i.e. a debarking scale of 2 and above (Table 6). Thirty-three percent ($n = 27$) of the debarked trees were not ring-barked but had some portions of their stem debarked. The mean length of debarking in trees was $1.55 \text{ m} \pm 0.19$ ($n = 82$) and differed significantly across the tree species ($F_{7, 74} = 8.97$, $P < 0.001$, Figure 1). Twenty-eight percent ($n = 23$) of the debarked trees were dead (snags) and all had been ring-barked. Medicinal tree use, through ring barking, clearly contributes to tree deaths and subsequently forest deadwood.

The percentage of dead trees among the debarked trees increased with the intensity of debarking on the trees (Spearman correlation, $r_s = 0.9$, $P = 0.05$, Table 6.) Debarking was noted in eight species of which *C. gerrardii* and *M. capensis* were the most affected (Table 4). In *C. gerrardii* debarking had reached a critical level where the bark was entirely stripped from the main stem of standing trees or, in the case of tall trees, cut down

first then debarked from the logs. The levels of debarking were all the more critical for the survival of *C. gerrardii* given its poor coppicing ability (Table 3).

Table 6. The percentage of dead trees occurring within the different debarking scales.

State of tree and its debarking scale	No. of debarked trees	% of debarked trees found dead
1 – Tree not ring debarked but small stem portion(s) debarked	27	0
2 – Tree ring debarked to 25% stem length	15	27
3 – Tree ring debarked to 50% stem length	25	48
4 – Tree ring debarked to 75% stem length	11	36
5 – Over 75% stem length ring debarked	4	75

Regenerative potential of fuelwood and medicinal trees species

Apart from *H. caffrum*, all fuelwood species showed considerable regeneration through coppice stems (e.g. *Buxus natalensis* 2.60 ± 0.25 coppice stems.tree⁻¹, *Millettia grandis*, 2.1 ± 0.16 coppice stems.tree⁻¹; Table 3). Only one medicinal tree, *C. myrtifolia*, showed a substantial level of coppicing (2.55 coppice.tree⁻¹ ± 0.62). Although intensively debarked *M. capensis* showed good bark regeneration (Table 4) and may sustain current harvesting levels. In contrast, debarked individuals of other medicinal tree species, for example *C. myrtifolia*, *P. obliquum*, *Croton sylvaticus* and *Zanthoxylum davyi* showed limited recovery (Table 4) and most died, remaining as snags in the forest.

Discussion

Fuelwood harvesting

Per capita fuelwood use in this study area ($564 - 521$ kg.capita⁻¹.yr⁻¹) was similar to many rural areas in Africa and other developing countries (Table 7). Estimates of fuelwood use in Africa vary geographically and range over an order of magnitude. Differences in

levels of consumption depend upon wood availability, climatic variability and socio-economic parameters of a region (Leach 1992). Muir (1990) suggests that wood availability is particularly important in cases where considerable effort is associated with its collection e.g. long walking distances to collect the wood. In other cases more wood may be used because of certain cultural activities (e.g. traditional fish smoking in Malawi, Abbot and Homewood 1999) or due to climatic conditions such as the very cold winters in Kashmir, India (Tara and others 1999). The relatively large amount of fuelwood used by communities in our study is in part related to the fact that they live close to forests (< 2 km). However, Brouwer and others (1997) suggest that distance to forests is not the only factor determining fuelwood use but also the availability of labour in households. The effect of this factor on fuelwood harvesting is not considered here. Nonetheless, Wass (1995) suggests that the critical cut-off distance is 1.5km and this accords well with our findings. The lower fuelwood tree stocking densities in community-controlled forest as compared to state forests agrees with similar studies in southern Africa woodlands (Chidumayo 1989, Shackleton 1993, Vermeulen 1996, Nhira and others 1998, Higgins and others 1999) and is largely linked to the high levels of tree use in community forests.

The size of fuelwood preferentially collected is limited by two main factors. First, the tools (hand machetes) used by women restrict collection to branch-sized (dbh <10 cm) wood from the crowns of fallen trees (although once this has been collected the main stems are broken up). Second, headloads preclude the collection and transport of heavy fuelwood loads at one time. In cases where fuelwood is transported by sled or on pack animals (e.g. oxen, Malawi - Abbot and Lowore 1999) there are fewer limitations on fuelwood size collected. At this time in Umzimvubu District fuelwood is collected by headload, mostly by women, although changes to the transport system should be anticipated as levels of fuelwood use increase.

That certain species are popular and are highly selected (e.g. *P. obliquum*) is consistent with patterns of fuelwood use in rural Africa (Abbot and Lowore 1999, Nomtshongwana 1999, Kirubi and others 2000). Selection is based on the woodfuel properties of the species with choice depending on whether a species is slow or fast burning, smoky or smokeless and whether or not it burns intensely i.e. with high or low

calorific values (Wickstead 1984, Nomtshongwana 1999). The popularity and overuse of *P. obliquum* could be associated with its high calorific value and the smokeless fire it produces, which is ideal for cooking. Furthermore, *P. obliquum* is readily harvested since it grows on forest edges and in forest gaps (Obiri and other in press; Chapter III) that are accessible to fuelwood collectors.

Selective harvesting of species can in itself create problems related to overuse and loss of the preferred species. As certain species become over-used and scarce, progressively more effort is required to search for and harvest the species until a point is reached where further exploitation becomes uneconomical (Sutherland and Reynolds 1998). At this point, users turn their efforts towards alternative species, which are usually less used because of their inferior quality. This process is exemplified in the classical “tragedy of the commons” model of the use of open access resources (Hardin 1995), which predicts that by the time an overused species is replaced by another, the damage to the preferred species is irreversible. Such a trend has been reported in the Sahel region (Ki-Zerbo 1981) and East Africa (Bradley 1988). In this study the scarcity of *P. obliquum*, a popular fuelwood species, has resulted in the use of other previously less popular species such as the subcanopy species *E. natalense* and *B. natalensis*. Scarcity of *P. obliquum* and declining densities of other fuelwood species also explains the current widespread use of the non-forest inhabiting *A. karroo*. In fact *A. karroo* deserves careful consideration as an easily grown alternative source of fuelwood that could take some pressure off forest tree species.

Although more households currently use gas and electricity, and less paraffin, than in the mid 1990s (see Table 2) most still have a high preference for fuelwood. It is clear that a change in energy preference is occurring in the region since more households do not use lower grade bio-fuels (livestock residues and agricultural waste) to the extent they did before the mid 1990s. A recent cutback in the state control and management of forests in South Africa (DWAF 1997, National Forest Act 1998) has effectively made these forests more accessible to local communities and may be responsible for the shift away from low-grade biofuels. If rural populations continue to increase and the current uncontrolled

Table 7. Factors related to fuelwood dynamics in various rural regions of Africa and other developing countries.

Region	Fuelwood yield kg.ha ⁻¹ .yr ⁻¹	Fuelwood consumption kg.capita ⁻¹ .yr ⁻¹	Collection distances (km)	Fuelwood situation	Strategies to combat fuel shortage	Sources
AFRICA						
Shinyanga, Tanzania		100	8	Unsustainable	MFU	Boberg, 2000
Korr, Marsabit, Kenya		200	>100	Unsustainable		Walter and Herlocker, 1981
Jozanna Nek, South Africa		270	2 - 4	Unsustainable		Best, 1979
Malefilone, Lesotho		290		Unsustainable		Best, 1979
Kano, Nigeria	680 ^a	320 - 350 ^b		Sustainable		Collin, 1977 ^a ; Cline et al, 1990 ^b
Ntcheu, Malawi		320 - 730	4 - 5	Unsustainable	SLQW	Brouwer and others, 1997
Dar-es-Salaam, Tanzania		390	2 - 5	Unsustainable	MFU	Boberg, 2000
Mbeya, Tanzania		400		Unsustainable	MFU	Boberg, 2000
Hola/Bura Kenya		430 - 560	1 - 5	Unsustainable	Plant woodlots	Cline and others, 1990
Lake Malawi Park, Malawi	1390	520	1 - 3	Unsustainable	Plant woodlots	Abbot and Homewood, 1999
Umzimvubu, South Africa	1310-2600	521-564	< 2.0	Unsustainable	SLQW	Current study
Ngaoundere, Cameroon		550 - 720	1 - 4	Unsustainable	MFU	Cline and others, 1990
Transkei region, South Africa		560	1 - 4	Unsustainable	MFU/SLQW	Mander and Quinn, 1995
Matsheng villages, Botswana		600		Unsustainable	Plant woodlots	White, 1979
Mahlabatini, South Africa		620 - 740		Unsustainable		Gandar, 1983
Sigombeni, Swaziland	900	630	< 2.5	Unsustainable	Plant woodlots	Allen and others, 1988
Angola		720	1 - 3	Unsustainable	MFU	O'keefe and Munslow 1984a
Ranga, Bukina Faso		840		Unsustainable		Ernst, 1978
Mutanda, Zimbabwe	750	930	< 1.0	Sustainable		Grundy and others 1993
Maputo, Mozambique		1100 - 1700		Unsustainable	Plant woodlots	O'keefe and Munslow 1984b
Marsabit, Kenya		1500	1 - 4	Unsustainable		Kirubi and others, 2000
ASIA						
Kashmir, India		544 - 712		Unsustainable		Tara and others 1999
Himachal Pradesh, India		778	6 - 7	Unsustainable	Plant woodlots	Sharma and others 2000
Mysore, India		800 - 1180	4 - 6	Unsustainable	MFU	Uma Shankar and others, 1998
SOUTH AMERICA						
Jaracuaro, Mexico		690 - 730	1 - 2	Unsustainable	MFU	Masera and others, 2000

MFU - Multiple fuel use

SLQW - Shift to low quality wood

harvesting remains in place, it is possible that fuelwood scarcity could cause a shift of emphasis back to low-grade biofuels. This sort of cyclical preference for fuel types has been reported in rural India (Qureshi, and Kumar, 1997) where scarcity of fuelwood increased the time spent fetching fuelwood and cost of purchased fuelwood, causing a shift back to low-grade biofuels.

Is fuelwood use sustainable?

Although there is a large body of literature on fuelwood consumption and extraction rates from different types of forests, very few studies estimate deadwood production in forests (see Table 7). Studies quantifying biomass in natural forests, particularly dead wood matter, are scarce (see also Frost 1996) and where available are largely limited to single surveys and rarely based upon long-term surveys. Thus, comparisons between production and consumption of forest deadwood and the estimation of what can be sustainably harvested remain elusive.

Estimates of rates of fuelwood consumption and deadwood yield in this study indicate that the annual forest deadwood yield cannot sustain the current rate of consumption. Even with the highest possible fuelwood production - i.e. the assumption of a large annual deadwood yield (1.9% of growing stocks) and form factor (0.7) - deadwood production falls short of consumption. This emphasises that the local fuelwood demand and supply from the forest species, is in excess of a balanced cycle of use. Indeed, live trees are cut (as opposed to dead or senescing trees) to meet the fuelwood deficit in spite of claims to the contrary

Most studies in developing countries suggest that fuelwood use is unsustainable and recommend the establishment of woodlots to ameliorate fuelwood shortages (see Table 7). Dewees (1993) and Nhira and Fortmann (1993) link unsustainable use to increasing tendencies of communities to harvest fuelwood for sale. Although a number of studies recommend planting woodlots to reverse this situation, Chidumayo and others (1996) point

to the implementation difficulties and argue that while there are still indigenous trees to exploit, woodlots are not a priority in the minds of most rural communities. In Umzimvubu District we anticipate that the communities will shift to harvesting lower quality species. Such a trend has been reported in rural communities within southern Africa e.g. in Ntcheu District in Malawi (Brouwer and others 1997). There are some potential species in our forests that are currently unused but could substitute for existing species. These include *Millettia sutherlandii* and *Celtis africana* which occur as large biomass trees (Obiri and others, in press; Chapter III) and are commonly found as fallen logs or gapmakers in the forest (Obiri and Lawes, unpublished manuscript; see later Chapter V).

Medicinal tree use

Although we do not have estimates of bark removal from forests, the high intensities of debarking indicate unsustainable use. For example, the debarking of *C. gerrardii* is proceeding at an alarming rate with all accessible forest trees having been affected. Despite most of the harvested species being legally protected species (e.g. *H. caffrum*, *T. dregeana*, *C. gerrardii*), illegal harvesting continues sporadically throughout the forests. *H. caffrum* and *C. gerrardii* face a major threat since they are intensively debarked, yet they are poor bark regenerators. Furthermore, Obiri and others (in press; see Chapter III) recommend controlled harvesting of *H. caffrum* and *C. gerrardii* because these species possess a coarse spatial scale of regeneration that cannot tolerate intense harvesting. Unlike other species, the survival of *M. capensis* is not threatened by debarking since it can replace removed bark even after being intensely debarked.

Silvicultural practices such as severing stems to encourage new shoot growth is a viable management practice for fuelwood species (Kennedy 1998). This is possible for *M. grandis*, *B. natalensis* and *P. obliquum*, which coppice vigorously, however, it is not feasible for most medicinal trees in this study because of their generally poor coppicing and bark regrowth after debarking. Only *C. myrtifolia* shows an ability to coppice

sufficiently to be managed in the latter way. Coppice management must be used to sustain stocking densities in *C. myrtifolia* because use of this tree for medicine has increased enormously and has replaced the highly sought after, but extremely scarce, *O. bullata* (Mander 1998).

This study confirms the findings of Van Eck and others (1997) that *H. caffrum* is a popular medicinal tree that is also used for fruit. We recommend it as an agroforestry tree for the rural farming communities of coastal ECP. The same goes for *M. grandis*, *P. obliquum*, and *Vepris lanceolata*, which are multipurpose trees in this region.

Conclusion

Current exploitation of the fuelwood and medicinal tree species in the coastal forests of Umzimvubu District is not sustainable. The rate of fuelwood consumption is much higher than, and cannot be met by, the natural rate of deadwood production in the fuelwood tree species. Fuelwood deficit is largely met by pre-existing stocks of residual deadwood and the use of other inferior quality fuelwood tree species. Fuelwood demand and collection are likely to increase in the forests given (1) an increasing rural population, and (2) declining economic status in the area (Carter and May 1999).

Community forests, which occur as small isolated patches neighbouring homesteads, with limited stocks of growing trees, are the most intensely used forests and harvesting rates in these forests are clearly unsustainable. The following recommendations for fuelwood and medicinal tree management are suggested:

1. Woodlots should be established next to both community and state forests particularly those close to villages.
2. Fuelwood strategies need to be aligned with the socio-economic needs of communities. For instance, instead of establishing woodlots of *Eucalyptus* spp, that only provide wood, popular multi-purpose indigenous tree species such as *P. obliquum* and *M. grandis* could be used since these provide medicines and building posts as well as fuelwood.

3. *P. obliquum*, *H. caffrum*, *M. grandis* and *V. lanceolata* should be used as agroforestry species in the local farming system.
4. Only *M. capensis* is capable of sustaining current bark harvesting levels. Harvesting of this species can continue but under a monitored programme.
5. Bark harvesting in all other species should be strictly limited and cease in the case of *C. gerrardii*, *T. dregeana* and *H. caffrum*.

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CHAPTER V

REGENERATION IN CANOPY GAPS: IMPLICATIONS FOR SELECTIVE TREE HARVESTING IN SUBTROPICAL FORESTS OF SOUTH AFRICA

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Abstract

We investigate gap formation and tree regeneration pattern in a subtropical forest (South Africa) to determine (1) the causes and characteristics of gaps, and (2) whether or not tree regeneration in gaps occurs randomly. Understanding the mechanisms underlying forest regeneration is pertinent for sustainable forest management since these may be affected by activities such as tree harvesting.

Ninety-one gaps were sampled of which 61 were used to analyse the regeneration of only those gap-filler species whose trees dominate the canopy level and gap-formation, while in 30 gaps all gap-filler species were considered. Eight percent of the forest area was under gaps caused by four modes of disturbances of which windthrow was most common (50%). *Millettia sutherlandii* was the most frequent gap-maker, accounting for 43% of the gaps, and was also the most dominant gap-filler species, occupying 46% of the gaps.

Gap-filler species whose trees dominate the forest canopy and gap formation did not show gap-size niche-partitioning, had low replacement transition probabilities, and appeared to be randomly assembled in gaps. Similarly, where all gap-filling species were considered there was no nested pattern of species assemblage implying random regeneration or establishment in gaps. Although a putative gap-filling guild comprising nine species occurred more abundantly in large than small gaps, the four species confined to large gaps were shrubs, vines, exotic weeds or edge adapted species. The remaining five species occurred across a wide range of gap sizes. The role of these nine species in gap-size niche-partitioning was considered to be slight.

Given the random nature of regeneration in gaps and no determined effect of gap size on species regeneration it is unlikely that tree composition in coastal scarp forests of the Eastern Cape will be affected by moderate selective harvesting of tree species.

Keywords: gap dynamics, selective harvesting, species replacement, tree diversity, assemblage structure, sub-tropical forest, sustainable use.

Introduction

Forest communities change over time and space as individual trees die and are replaced. The regeneration of new individuals into tree populations is often controlled by the appearance of gaps (Runkle 1982, 1985; Denslow 1987; Zagt and Werger 1998; Brokaw and Busing 2000). Consequently the dynamics and abundance of tree species depends to a large degree on the dynamics of gap formation (Brokaw and Busing 2000), which in turn can have significant management implications for sustainable forest use (Attiwill 1994, Coates and Burton 1997, Coates 2000). Canopy gaps are common in the scarp forests studied here because of strong coastal winds (Everard *et al.* 1995). Here we consider the role and importance of gaps in these forests' dynamics. We address two aspects of the gap phase in sub-tropical coastal forests of South Africa: (1) what are the causes and characteristics of sub-tropical forest gaps, and (2) how do tree species in these forests recruit or replace themselves? Our aim is to understand the putative effect of gap dynamics on forest tree diversity so that we might predict tree responses to selective harvesting.

Forest gap formation, through treefall or tree death, is initiated by natural disturbances such as windstorms, fungal infections and insect attacks on trees (Pickett and White 1985, Runkle 1985, Pontailier *et al.* 1998, Yamamoto 1998, 2000). In many subtropical and tropical forests a further common cause of gaps is tree harvesting by communities neighbouring forests (Struhsaker 1997, Bruenig 1998, Myers *et al.* 2000). In our study area communities selectively cut trees for construction wood, fencing or carving materials (Cooper and Swart 1992, Everard *et al.* 1994, Obiri *et al.* in press; see Chapter III). These harvesting activities have resulted in declining densities of trees in the forests (DWAF 1997, Timmermans 2000) and the loss of some species altogether from local community forests (Obiri *et al.* in press; Chapter III). However, another more important and largely overlooked consideration is that tree cutting increases the forest area under gaps. The effect of the latter upon regeneration or recruitment pattern and thus forest species composition is unknown.

Gap dynamics theory suggests that spatial and temporal patterns of species recruitment within canopy gaps may be due to differences in shade tolerance (Brokaw 1985, Whitmore 1989, Martinez-Ramos 1994, Kneeshaw and Bergeron 1999). Shade tolerance constrains the establishment of species in canopy gaps in two ways, each with potentially different consequences for the maintenance of species composition in forest in response to harvesting pressure. First, if niche partitioning occurs in gaps through a gradient of differential resource availability, to which tree species respond differently with individuals establishing from seed according to their specific physiological needs (Grubb 1977, Denslow 1980, Brown and Jennings 1998), then a highly predictable gap-filling guild of pioneer species may arise (Brokaw 1985). Typically, light demanding (pioneer) species recruit into large gaps, although the threshold gap size for pioneers varies greatly (Brokaw 1985), while shade tolerant species establish under the forest canopy or in very small gaps (Denslow 1980, 1987; Whitmore 1989; Brown and Jennings 1998). On the other hand, the stochastic nature of gap formation in time and space may facilitate recruitment by advanced regeneration usually suppressed under the forest canopy. In other words, previously established plants exist as suppressed juveniles under the canopy and experience accelerated growth when a gap opens above them (Brokaw 1985). These primary or shade-tolerant species are at the opposite end of the regeneration behaviour continuum from shade-intolerant pioneer species (Brokaw 1985, Whitmore 1989, Alvarez-Buylla and Martinez-Ramos 1992). Of course these two “processes” are coincident to varying degrees in all gaps depending on gap size, microclimate and seed input to a gap (Denslow 1980, 1987; Lieberman *et al.* 1995, Brown and Jennings 1998, Brokaw and Busing 2000).

Depending on the rate of gap formation and size of gaps, forest composition and structure is predicted to vary over time. For instance, although pioneer species die out as forest matures after disturbance or a gap-forming event, intermediate logging intensities in Queensland forests have been observed to reverse this loss of diversity (Nicholson *et al.* 1988). However, at high rates of gap formation caused mainly by harvesting activities, the

putative equilibrium between pioneer and primary species composition in a forest may shift toward and become dominated by shade-intolerant species because the gap-filling guild is favoured (e.g., Mckenzie 1988, Whitmore 1989, Raich and Khoon 1990, Cannon *et al.* 1994, Johns 1997). In the case of extreme levels of disturbance this can give rise to single or few species stands of shade-intolerant species that are almost even-aged. Thus, regeneration in gaps appears to be maintaining coexistence between species in mixed stands (Kohyama 1993, Rebertus and Veblen 1993, Attiwill 1994).

In this study we examine the extent to which regeneration in gaps is a random process in relation to gap size and the likelihood that a gap-filling guild may come to dominate intensively harvested subtropical forest. Indirectly we investigate the notion that the maintenance of tree species diversity in sub-tropical rain forest depends on random processes. We predict that if trees recruit randomly then the number of gaps in the canopy, whether man-made or not, is less likely to affect tree replacement or the overall species composition in forests and selective tree harvesting could proceed in moderation.

Methods

Assessment of gap characteristics

This study was done in the coastal forests of Umzimvubu District in the Eastern Cape Province of South Africa ($31^{\circ} 30' - 31^{\circ} 45'S$ and $29^{\circ} 15' - 29^{\circ} 30'E$). These forests are subtropical in nature and described as coastal scarp forests (Cooper and Swart 1992). They have a high species diversity that is considered a composite of the Afro-montane forests and the Indian Ocean coastal belt forests (Macdevette *et al.* 1989, Lawes 1990). They are largely composed of evergreen high-canopy trees (mean height = $19.7\text{m} \pm 0.45$, $n = 96$) that form a continuous closed canopy of mixed species. Historically most of these forests experienced very little human disturbance (Maggs 1980, Feely 1986) however presently many have been exploited at least to some degree. Straight-line transects were walked in

ten forests along a compass bearing and parallel to the long axis of the forests. Those forests that had experienced relatively little anthropogenic disturbance were selected. Transects were 25 m wide and all canopy gaps or part of a gap falling within this belt were included in the sample. A gap was defined after Runkle (1981), Barden (1989) and Liu and Hytteborn (1991), as an opening in the upper canopy, extending to the bases of canopy trees surrounding the canopy opening, and caused by tree wind-fall, branch-or bole-snapped trees, or dead standing trees (snags). In this definition gaps were taken to be indistinguishable from the background overstorey when the regenerating trees attained two-thirds of the mean canopy height

Because a gap can only exist within a matrix of mature canopy trees with a recognisable and continuous border of gap edge trees, a gap perimeter was defined as a line joining the stems of live canopy trees surrounding the gap. Gap area (A) was calculated as $\pi LW/4$ (Runkle 1982), where L = gap length (largest distance between gap edges) and W = gap width (largest distance perpendicular to length). The tree that created a canopy gap was referred to as a gap-maker while saplings or small trees regenerating in the gap were gap-fillers. Gap-fillers were subcanopy individuals ranging in height from 0.2 m above the forest floor to two-thirds of canopy height.

Gap-makers were identified to species level and for each gap-maker the diameter at breast height (dbh), height (along the fallen stem) and mode of mortality was noted. Three modes of death in canopy trees are the main causes of gaps (Liu and Hytteborn 1991, Midgley *et al.* 1995, Mabry *et al.* 1998, Myers *et al.* 2000). These include (1) snags, (2) wind-throw trees, and (3) branch- or bole-snapped trees; all of which are adopted in this study. All branch-or bole-snapped trees are hereafter referred to as snapped trees. These modes of death result in three gap types: (1) snag gaps, (2) wind-throw gaps and (3) snapped tree gaps respectively. A fourth gap-type (cut tree gaps) resulting from selectively cut tree, was considered.

Compass directions of the fallen gap-makers were recorded. Mean wind speed and wind direction data were obtained from the Port St Johns meteorological station, 5km from the study area, and correlated with the wind-throw tree data.

Tree recruitment processes

We undertook two sampling surveys to test if recruitment in forests was a deterministic process with gap-size niche partitioning (Denslow 1980, 1987, Swaine and Whitmore 1988, Brown and Jennings 1998) or if it was a random event akin to the lottery paradigm (Sale 1977, 1979, Hubbell and Foster 1986, Hubbell *et al.* 1999).

a. Analysis of regeneration of dominant canopy-species and gap-fillers

Ten gap-filler species whose fully-grown trees dominated the forest canopy and gap formation were censused in 61 gaps. Only those individuals with gap-filler characteristics (i.e. 0.2 m to 2/3rds overstorey height) were counted. The gap-filler data were used to test if the tree recruitment or replacement pattern in the gaps was a directed, deterministic process or if it was a random event. To establish the former we examined recruitment in gaps and non-gap areas of forest and also checked for species that were unique to gaps. We used old gaps as far as possible because we anticipated that all prospective gap-filling species would have entered the old gaps and that species competition and replacement would be well underway. However, it is difficult to estimate gap age in tropical and subtropical forests because the climate signature is not clearly depicted in the tree rings (Lilly 1978, Norton and Ogden 1990). Thus, we used the size of gap-fillers and the decayed state of gap-makers to identify the old gaps. Gaps with large stem-sized gap-fillers or with logs (i.e. gap-makers) in an advanced state of decay were considered old. For each of the 61 sampled gaps, areas of equivalent size adjacent to the gaps were randomly selected under the canopy (i.e. non-gap area) and “gap-fillers” of the same 10 species counted. We

ensured that the second area had no recent evidence of disturbance and that it was at least 10 metres from the gap.

To determine if the recruitment of canopy-species gap-fillers follow the lottery paradigm of random replacement or not, two methods were employed.

- (1) We investigated whether a canopy tree was replaced by another tree of the same species (self-replacement) or not. Where a species was predominantly replaced by another of its own kind, replacement was considered deterministic or non-random. The probability of one species replacing another species (transition or replacement probability) was assumed to depend only on the relative abundance of subcanopy individuals of the first species in gaps created by the second species. Thus, transition probabilities were computed on the basis of the relative abundance of subcanopy individuals beneath forest canopy gaps (Runkle 1981, Midgley *et al.* 1995).
- (2) We further checked if species recruiting into tree gaps showed a nested distribution pattern (Patterson and Atmar 1986), whereby species composition in depauperate gaps collectively comprised nested subsets of those in richer gaps. Here forest gaps were considered as islands whereby species established themselves in a pattern dependant on their dispersal abilities and other chance factors such as mortality. A common pattern in natural and anthropogenic habitat island is the tendency for fauna and flora to form nested subsets; that is, within an archipelago, species compositions of smaller flora or fauna tends to form subsets of larger floral or fauna (Cutler 1991). This is a feature Pattern and Atmar (1986) term as the “nested subset model”. Where the pattern of species assemblage is nested, species replacement in gaps is assumed to proceed in predictable, directed or deterministic fashion. The extent of nestedness among gaps was measured as the degree of “unexpectedness” from a perfectly nested matrix using Cutler’s (1991) *U* statistic. Patterson and Atmar’s RANDOM1 program (Cutler 1991), a Monte-Carlo simulation, was used to test the results for significance against 100 simulations of randomly assembled matrices of the same size as our gap matrix data. The

RANDOM1 was preferred to an alternative simulation program (RANDOM0) because in calculating nestedness it constrained the data more, in order to match reality, by weighting the occurrence of individual species by their own abundance in the matrix of gaps (see Patterson 1987).

b. Analysis of recruitment of all gap-filler species

All gap-filling species were sampled in 30 gaps in a second survey. In each gap the densities of all the gap-fillers were estimated using 0.25m² quadrats placed 5m apart along a transect line within the gap. The length and width of the gap formed the first two transects and subsequent transects were placed parallel to these two but at a distance of 5m apart. We first tested for deterministic recruitment by checking for gap-size niche partitioning in the 30 gaps. This was done by:

- (1) using canonical correspondence analysis (CCA) where the density data of 53 species, sampled in the 30 gaps, was ordinated with gap size as an environmental variable. This direct gradient analysis technique tested if there was a pattern of species sorting or distribution along the gradient of gap size;
- (2) examining the frequency distribution of the gap-filling species by gap size in which a species was most abundant. If niche-partitioning occurred we expected the frequency distribution of species to show a bimodal distribution between small and large gaps with primary and pioneer species respectively; and
- (3) identifying those species that were unique or confined to either small or large gaps.

Next we tested for random recruitment by conducting a nested-subset analysis (Patterson and Atmar 1986, Cutler 1991). Here we again checked if species recruiting into tree gaps showed a nested distribution pattern (Patterson and Atmar 1986), using the RANDOM1 programme as in the previous analysis of canopy-species gap-fillers.

Results

Causes and characteristics of gaps

Thirty gaps were analysed to ascertain forest gap characteristics. Four types of gaps were observed; wind-throw gaps (50%, $n = 15$), snapped tree gaps (20%, $n = 6$), cut tree gaps (17%, $n = 5$) and snag gaps (13%, $n = 4$). The mean gap size was $57.6 \pm 5.4 \text{ m}^2$ (mean \pm 1SE, $n = 30$). These gap types differed in area ($F_{3,26} = 5.43$, $P = 0.003$; Figure 1). Wind-throw gaps were largest in area ($75 \pm 7.4 \text{ m}^2$, $n = 15$), while gaps created by snags were the smallest ($22.2 \pm 7.2 \text{ m}^2$, $n = 4$; Figure 1). Transect counts showed that an estimated $7.8 \pm 0.8\%$ ($n = 11$ transects) of the forest area was under gap phase with a mean of 13.5 ± 1.53 gaps. ha^{-1} (mean \pm 1SE, $n = 11$ transects).

Gap-makers resulted from four causes: wind-throw trees, snapped trees, snags and cut trees. Apart from cut trees, snags, wind-throw and snapped trees had no signs of previous anthropogenic interference and they solely resulted from natural disturbances. These four types of gap-makers differed in their mean diameter at breast height ($F_{3,63} = 5.37$, $P = 0.002$; Figure 2). The largest diameters were noted in gap-makers caused by wind-throw ($46.7 \pm 2.1 \text{ cm}$, mean \pm 1SE, $n = 51$) while the smallest were those resulting from cut trees ($35.5 \pm 2.0 \text{ cm}$, $n = 22$) and snags ($35.0 \pm 2.9 \text{ cm}$, $n = 12$). Larger trees are clearly prone to wind-throw. *Millettia sutherlandii* was the most common gap-maker with 4.8 ± 1.2 gap-makers ha^{-1} ($n = 11$ transects; Figure 3) and accounting for 43% (Table 1) of the total forest gap-makers. Most windthrows (50%) fell in a north-east to east direction ($\chi^2 = 9.35$, d.f. = 3, $p = 0.03$) commensurate with direction of the prevailing north-easterly winds in this region.

Recruitment of dominant canopy-species and gap-makers

All the canopy-level tree species that were found as gap-fillers regenerating in gaps were also found under the canopy or non-gap (shaded) areas of the forest (Figure 4). No species

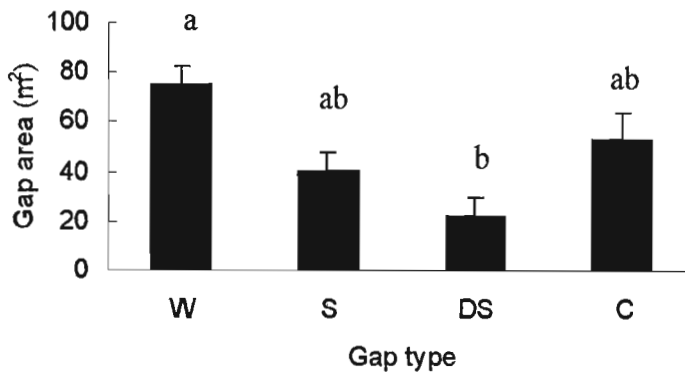


Figure 1: Mean \pm 1S.E gap area (m²) of the four types of forest gaps. W, S, DS and C are abbreviations for gapmakers resulting from wind-throw, snapped trees, dead standing trees (snags) and cut trees respectively. Gap types with different letters significantly differ in area ($p = 0.05$)

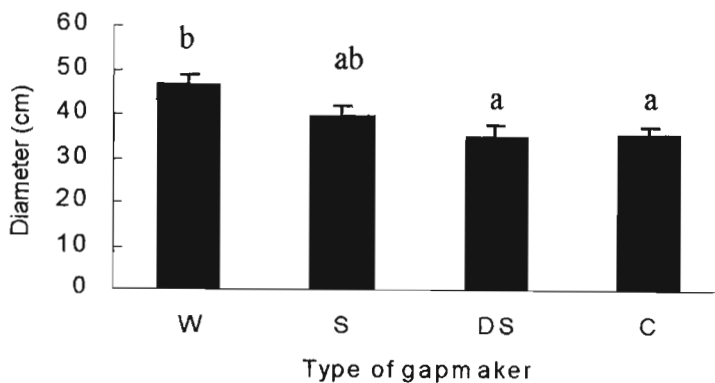


Figure 2: The mean diameters (dbh – cm) of four different types of gapmakers. Abbreviations of gap types are as in Figure 1. Gapmakers with different letters have diameters that differ significantly ($p = 0.05$).

Table 1: Dominant gap-makers species and their characteristics. Full Latin binomials of species are as in Figure 7.

Gap-makers	Number of gap-makers	Mode of death in gap-maker				DBH (cm)			Percentage of total forest gap-makers
		Wind throw	Snapped tree	Cut tree	Dead standing	Mean	SD	Range	
<i>M. sutherlandii</i>	43	26	14	0	3	42.0	12.4	23.0-69.0	43
<i>M. grandis</i>	11	3	0	8	0	35.9	8.50	23.0-51.0	11
<i>D. gerrardii</i>	11	4	5	1	1	39.1	10.9	23.0-61.0	11
<i>Z. davyi</i>	8	0	0	0	8	35.4	11.4	22.5-54.0	8
<i>T. dregeana</i>	8	6	1	0	1	46.9	12.6	23.9-62.0	8
<i>C. syvaticus</i>	5	3	2	0	0	52.4	17.3	38.5-80.0	5
<i>C. africana</i>	4	3	1	0	0	38.9	16.7	25.7-68.0	4
<i>H. lucens</i>	3	3	0	0	0	32.5	10.6	25.0-40.0	3
<i>C. spicata</i>	3	0	2	0	1	79.7	14.6	75.0-96.0	2
<i>C. kraussii</i>	2	1	1	0	0	48.6	7.9	43.0-54.2	2
<i>V. lanceolata</i>	1	1	0	0	0	44.7	-	-	1

was unique to either gaps or forest canopies suggesting no special gap recruitment guild among the canopy tree species. The probabilities of canopy species replacing themselves were low with all species showing a probability <0.3 (Table 2) except for *M. sutherlandii* (0.49). *Millettia sutherlandii* is a common large species and this attributes to its high probability however other equally large and frequently occurring species e.g. *Millettia grandis* and *Drypetes gerrardii* (see Obiri *et al.* in press; Chapter III), thus expected to have a high self-replacement, had low self-replacing probabilities. This suggests that replacement was not likely to be a deterministic or non-random process but a chance process. Gap-fillers of *H. lucens* and *Z. davyi* had the lowest gap occupancy and thus limited probability of 'capturing' a gap i.e., replacing a gap-maker (Table 2). For example, *H. lucens* gap-fillers were not observed under any gap-maker or gap except in those gaps created by *H. lucens* trees.

Table 2: The probabilities of gap-makers being replaced by gap-filler of species that occur in the forest canopy level. Probabilities in bold indicate self-replacements and dashes indicate a probability of zero. The ten species represent the most dominant canopy trees each of which contribute at least 2% of the total forest gaps (see Table 1).

Gap-makers		Gap-fillers									
		Ca	Ck	Csp	Csy	Dg	Hl	Mg	Ms	Td	Zd
<i>Celtis africana</i>	Ca	-	-	-	0.33	-	-	-	-	0.67	-
<i>Combretum kraussii</i>	Ck	-	0.20	-	0.20	0.27	-	-	0.33	-	-
<i>Cussonia spicata</i>	Cs	-	-	-	-	0.63	-	0.25	0.13	-	-
<i>Croton syvaticus</i>	Cy	-	-	-	-	0.17	-	0.58	0.25	-	-
<i>Drypetes gerrardii</i>	Dg	-	0.13	0.06	0.25	0.25	-	-	0.25	-	0.06
<i>Heywoodia lucens</i>	Hl	0.15	-	0.15	-	-	0.15	-	0.52	0.04	-
<i>Millettia grandis</i>	Mg	0.04	0.17	0.15	0.16	0.15	-	0.27	0.06	-	0.01
<i>Millettia sutherlandii</i>	Ms	-	0.07	0.04	0.10	0.19	-	0.10	0.49	-	-
<i>Trichilia dregeana</i>	Td	0.27	0.04	-	0.31	0.08	-	-	0.23	0.08	-
<i>Zanthoxylum davyi</i>	Zd	0.15	0.31	0.31	0.08	-	-	-	0.15	-	-
<i>Total occupancy under different gap-makers</i>		4	6	5	7	7	1	4	9	3	2

Nested-subset analysis of species distribution pattern in gaps supported the trend of random recruitment described by the transition probabilities above (Table 2). There was no difference between the calculated nestedness scores of our gap assemblage matrix ($Ua = 24.5$, $Up = 49.5$, $Ut = 74$) and that of randomly generated assemblages (RANDOM1: $Ua =$

30.7, $U_p = 47.1$, $U_t = 78$, $P > 0.05$), suggesting a random pattern of species assemblage in canopy gaps.

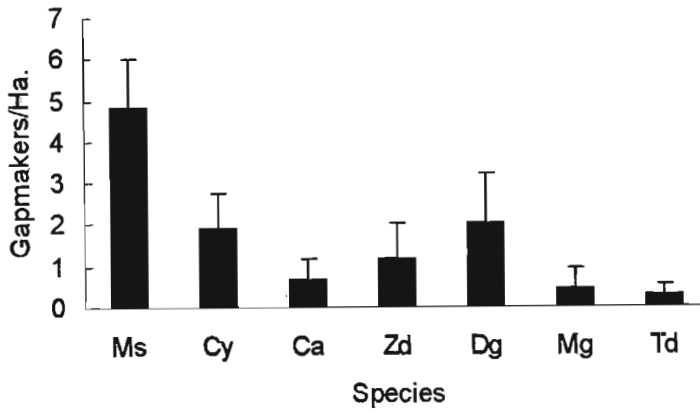


Figure 3: Mean number of gap-maker trees per hectare for seven main gap creating tree species (each of which contributes at least 4% of the forest gaps). See Table 2 for Latin binomials of the abbreviated species.

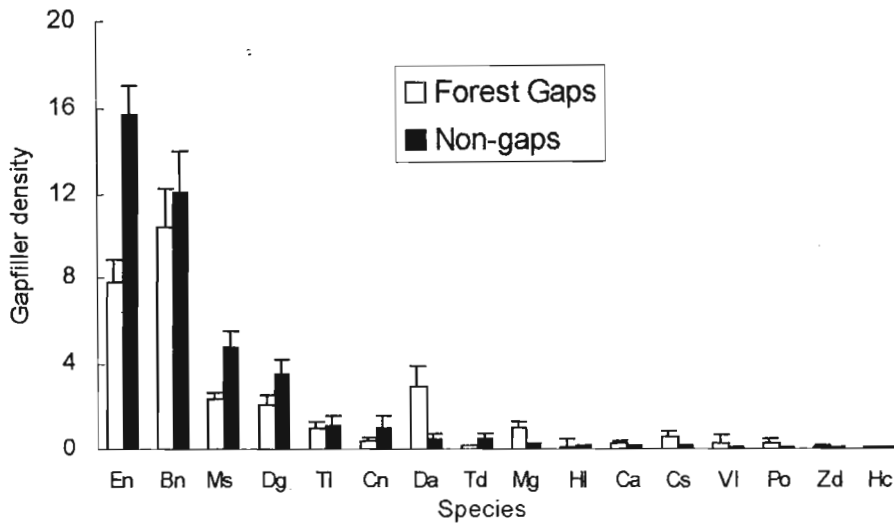


Figure 4: Mean gap-filler densities (individuals per 100m², n = 61) of canopy tree species recruiting into forest gaps (open bars) and the non-gap or forest canopy areas (shaded bars). For the Latin binomial of the abbreviated species see Table 2 and Figure 7.

Recruitment of all gap-filler species

A biplot of the CCA analysis showed that all the gap-filler species were distributed along the gap-size gradient resulting in two specific clusters - species occupying small and large sized gaps (Figure 5).

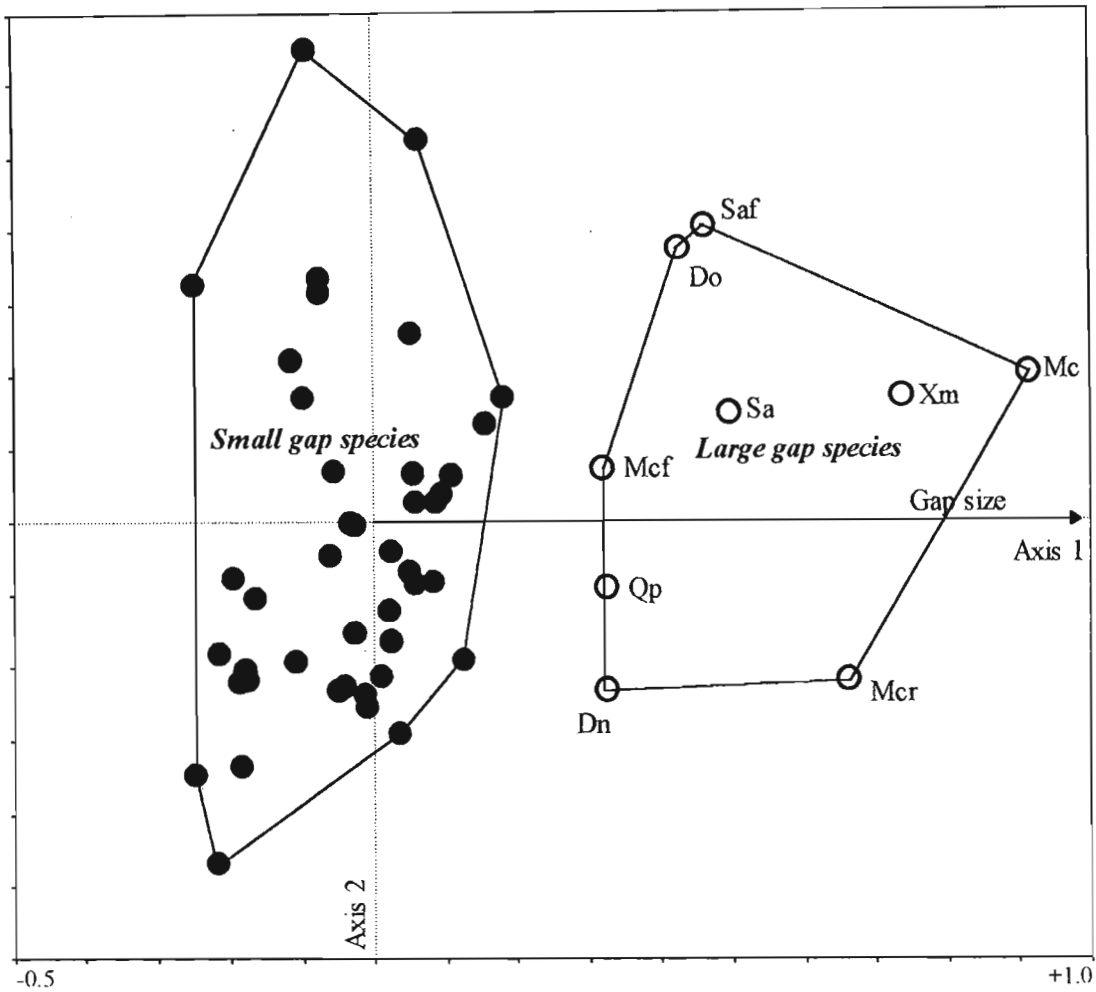


Figure 5: Correspondence canonical analysis (CCA) showing the distribution of species (n=53) along the gradient of forest gap size and the separation of large gap occupants (open circles) from species occupying small gaps (close circles). Species showing occupancy in large gaps are indicated (Dn- *Drypetes natalensis*, Do- *Dalbergia obovata*, Mc- *Macaranga capensis*, Mf- *Mimusops caffra*, MCr- *Maytemus cordata*, Qp- *Quisqualis parviflora*, Sa- *Solanum aculeastrum*, Saf- *Suregada africana*, Xm- *Xymalos monospora*).

The first two axes of the CCA analysis contributed to 14.5% of the variance in species data. All the species-environmental variance (100%) was contributed by the first axis of the CCA analysis and gap size contributed significantly to the fit of species data (canonical coefficient for gap-size variable = 1.0, $t = 7.06$). Nine of the 53 species were identified with large gaps (*Dalbergia obovata*, *Solanum aculeastrum*, *Maytemus cordata*, *Macaranga capensis*, *Suregada africana*, *Mimusops caffra*, *Xymalos monospora*, *Drypetes natalensis* and *Quisqualis parviflora*, see Figure 5).

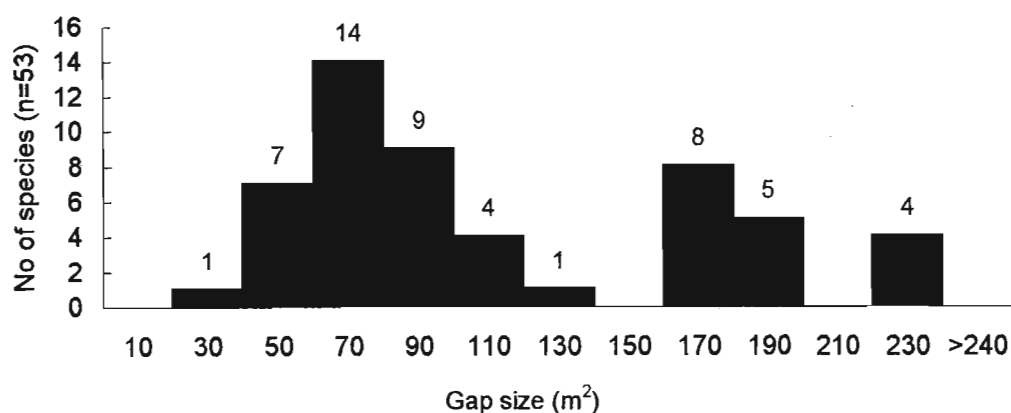


Figure 6: Frequency distribution of gap sizes by species for size at which species are most abundant. Species show bimodal distribution in small- and large-sized gaps. Figures on top of the bars indicate the number of species occurring in the particular gap sizes.

A frequency distribution of those gap sizes in which species were most abundant showed a bimodal distribution of gaps into two groups of small and large-sized gaps (Figure 6). Seventeen species were common in the large-sized gaps, nine of which occurred in the largest gaps (>180m², see Figure 6) and were the same species grouped as large-gap occupants in the CCA analysis (Figure 5). Thus, the bimodal distribution of gap size confirmed the CCA results that some gap-filler species were large gap occupants. The bimodal distribution further indicated a lower cut-off area for large gaps at 140 m² and thus a putative critical gap size threshold for gap-dependent (pioneer) species.

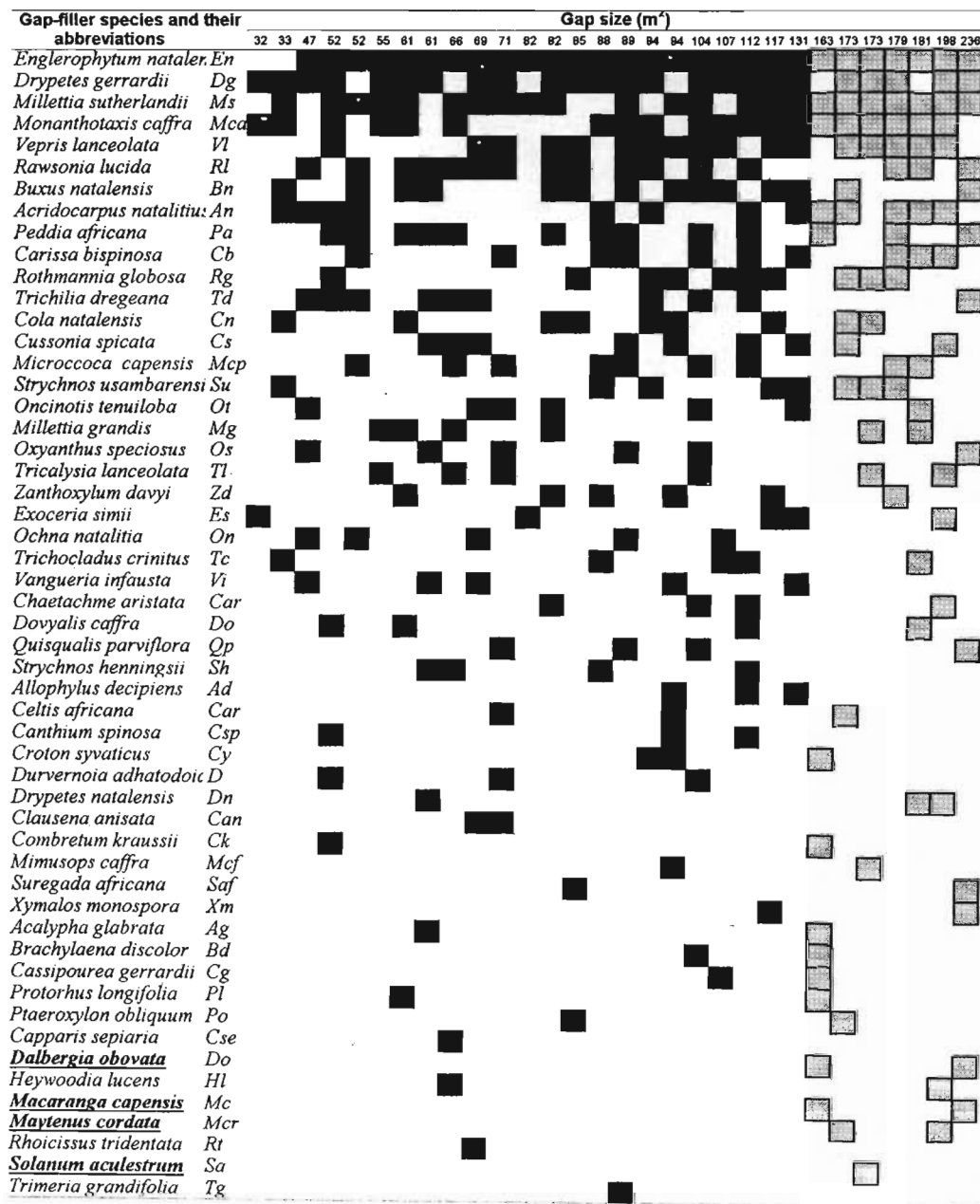


Figure 7: The distribution of all gap-filler species (n=53) in small gaps (black boxes) and large gaps (grey boxes). Gap-filler species unique to large gaps (n=4) are in bold and underlined.

Most gap-fillers species were found across all the gap sizes, and only four of 53 gap-filler species (7.5%) were confined to large gaps (Figure 7). Thus, gap-size niche differentiation is arguably weak. This conclusion is supported by the fact that of these four species, one is a vine (*Dalbergia obovata*), one is an aggressive weed (*Solanum aculestrum*), another is an understorey shrub (*Maytemus cordata*) and the only tree species (*Macaranga capensis*) is most commonly found at the forest edge. These four species are among the nine species identified as large gap occupants in both the CCA analysis and the bimodal gap distribution pattern. The other five large-gap-occupying species are not strictly large-gap occupants because they occurred in a range of gap sizes from small to large gaps (e.g. *Suregada africana*, 85 - 236m²; *Mimusops caffra*, 94 - 193m²; *Xymalos monospora*, 117 - 236m²; *Drypetes natalensis*, 61-198m² and *Quisqualis parviflora*, 71 - 236m²; see Figure 7).

The assemblage structure derived from all gap-fillers ($U_a = 56.5$, $U_p = 185.5$, $U_t = 242$) did not differ significantly from a randomly generated assemblage ($U_a = 69.3$, $U_p = 176.8$, $P > 0.05$). Species appear to be generally randomly assembled in gaps and tree replacement is a chance event.

Discussion

Causes and characteristics of gaps

Studies undertaken in various South African forests (e.g. southern Cape Forests; Geldenhuys and Maliepaard 1983, Midgley *et al.* 1995) have found that gap formation is rarely caused by wind disturbance but is mostly a result of senescence and death of standing trees. The coastal scarp forests of KwaZulu-Natal and the Eastern Cape Provinces are particularly affected by strong north-easterly on-shore winds (Everard *et al.* 1995). Thus, it is no surprise to discover that the key causes of gap formation in the subtropical coastal scarp forests of Pondoland are windthrows (50%) and wind-snapped trees (20%).

Strong gales and storm-level winds exceeding 100 km.hr^{-1} are often reported at the Port St. Johns weather station. The preponderance of tree falls in a north-east to easterly direction confirms the relationship between wind throw and wind direction and underpins the significance of wind disturbance in gap dynamics of these forests. Similar results have been reported in other tropical coastal forests (e.g., in Costa Rican forests, wind-throw and snapped trees cause 41% and 31% of gaps respectively; Lawton and Putz 1988). Wind disturbance largely explains why a higher percentage of our forests (8%) are under gap phase as compared to other forest facing different disturbance regimes (e.g. 5% Ituri Forest, Democratic Republic of Congo, Hart and Hart 1989; 2-10% southern Cape forests, Midgley *et al.* 1995; 4.8% Venezuelan Amazon forest, Uhl and Murphy 1982).

Gap size (mean area = 57.6 m^2) was similar to that in the forests of Panama and Costa Rica (86 m^2 and $87\text{-}125 \text{ m}^2$ respectively; Hartshorn 1978, Brokaw 1982, Foster and Brokaw 1982) but otherwise smaller than most tropical forest gaps (range = $140 - 628$; in Brokaw 1985) and larger than gaps in the southern Cape forests of South Africa ($20.5\text{-}37.7 \text{ m}^2$; Midgley *et al.* 1995). Forests, like the southern Cape forests, that have small gaps and a low percentage of their area under gaps are dominated with trees that typically die standing and which gradually drop their limbs and form small gaps (Brokaw 1985; Midgley *et al.* 1995). This is not the case in our forests where gaps largely occur through wind disturbances, although the relatively small gap size is most likely a consequence of shorter gap-maker tree size than in tropical forests.

Gap formation in the coastal scarp forests is largely caused by one species (*M. sutherlandii*, 43%). This is not surprising because in the least undisturbed forests *M. sutherlandii* is among the densest species (Obiri *et al.* in press; see Chapter III). The phenomenon of a single species dominating gap dynamics is common to forests. For example, Rebertus and Burns (1997) report 39% of the gaps in savannas and dry forest in central America being caused by *Quercus stellata* while in the southern Cape forest as many as 60% of the gaps have been attributed to *Olea capensis* spp. *macrocarpa* alone (Midgley *et al.* 1995).

Is recruitment a random or deterministic process?

Various studies suggest that species recruitment in forests can be categorised into two guilds: pioneer (gap-dependent) species and primary or non-pioneer (gap-independent) species (Brokaw 1985, Swaine and Whitmore 1988, Whitmore 1989, Yamamoto and Nishimura 1999). Swaine and Whitmore (1988) and Whitmore (1989) proposed the gap / non-gap paradigm whereby differences in gap size determine the species composition in subsequent cycles of recruitment. In other words species are able to partition the gap-size niche and in doing so recruit and coexist in the forests. Based on this paradigm, pioneer species are expected to establish in large gaps while non-pioneer (primary species) under the canopy and in small gaps (Tomlinson and Zimmerman 1978, Bazzaz and Pickett 1980, Brokaw 1985, Denslow 1987, Raich and Khoon 1990). On the other hand Lieberman *et al.* (1995) argue against gap-size niche-partitioning or the existence of guilds in forests because species possess wide ecological amplitude with large interspecific overlap instead of narrow tolerance and high specificity. In fact, they make a case for an alternative recruitment model that is akin to the lottery or random model of species replacement (*sensu* Sale 1977, 1979). They advocate that gaps are filled on a random 'first come first served' basis from a pool of potential propagules from all tree species (Zobel 1997). However, Grossman (1982) and Zagt and Werger (1998) argue that both random and deterministic (niche partitioning) processes of recruitment occur in nature. Zagt and Werger (1998) further suggest that in forests these processes vary during the different stages of seedling recruitment in the understorey and gaps.

Our results strongly favour a random pattern of species replacement in gaps. First, the assemblage structure in gaps of typical canopy species was not nested and no different from a pattern expected if their had been random recruitment in gaps. Second, the probability of a canopy tree replacing itself in a gap (i.e. non-random replacement) was low and non-existent in some tree species. In addition, at best there was only slight gap-

size niche-partitioning and gap-fillers were found in both gaps and the under the canopy of the forest. Random (lottery) recruitment models have also been shown in the southern Cape forest (Midgley *et al.* 1995), some tropical forests (Clark and Clark 1992, Lieberman *et al.* 1995) and mature conifer forests (Gray and Spies 1997).

The evidence for gap-size niche partitioning and the categorisation of recruitment by tree species into pioneer (gap dependent) and primary (gap-independent) species has been debated widely over the past three decades (Whitmore 1975, Tomlinson and Zimmerman 1978, Denslow 1980, Brokaw 1985, Swaine and Whitmore 1988, Raich and Khoon 1990, Alvarez-Buylla and Martinez-Ramos 1992). More recent studies (e.g., Lieberman *et al.* 1989, Canham *et al.* 1990, Grubb 1996, see Brown and Jennings 1998 for a review, Vera *et al.* 1999) suggest that the dichotomy between pioneer and primary species is an arbitrary typology given the continuum of variation of light intensities in forests, which causes most species to be generalists in their response to light (Lieberman *et al.* 1995). The notion of forest trees as a random assemblage of loosely co-evolved generalist species has gained increasing support (Hubbell and Foster 1986, Grubb 1996, Brown and Jennings 1998). Certainly, in the subtropical coastal forests of the Eastern Cape tree species do not display gap-size niche-differentiation, appear to have wide overlapping niches, and the majority are capable of comfortably establishing themselves in gaps and under the forest canopy.

The random recruitment of species occurs through the stochastic events associated with a forests' dynamics and which act on the available pool of species trying to colonise gaps. These stochastic events are manifested largely through (1) unpredictable tree fall and thus gap occurrence (2) the fact that gap are filled from a species pool of potential propagules on a random basis and the influence of environmental variables are negligible (Zobel 1997, see later Chapter VI).

Why is the lottery replacement model important for conservation?

The lottery hypothesis proposes that species are limited by a common short supply of living space. When vacant space is created, (through death of a previous occupant or disturbance in the ecosystem) it occurs in places and times that are unpredictable to the prospective (new) space occupants and as such the space is occupied by whichever species that gets there first (Sale 1979, Zobel 1997). An important aspect of random recruitment is that competitively inferior species have increased chances of establishing themselves in forests at the expense of other species with more competitive vigour. The distinction between deterministic and random processes of tree recruitment describes the relative influences of species guilds and lack of them on the forest community structure. The results of this study suggest that forest tree species undergo random replacement when recruiting into gaps. Indeed, regeneration in gaps appears to be maintaining coexistence between species in these mixed forests. These findings have considerable implications for the management of forests in the study area, in particular harvesting strategies. An important consideration is that these forests are exploited mainly through selective tree harvesting (Cooper and Swart 1992, Obiri 1997, Van Eck *et al.* 1997, Obiri and Lawes 2000, Obiri *et al.* in press; Chapter III). Forests under selective harvesting practices tend to have larger gaps that affect the available quantity of resources (light, space, water, nutrients) for saplings and thus species composition on the forest floor (Struhsaker 1997). If forests experiencing selective tree harvesting have larger (and more frequent) gaps on average, then according to the gap/non-gap model we would expect an increase in the abundance of gap-dependent (pioneer) species. That is to say, selective harvesting would facilitate the establishment of a gap-filling guild. However, given a very weak gap-filling guild in this forest, selective tree harvesting is unlikely to affect species composition. More importantly, the tree species are not gap-dependent, but recruit randomly, thus the current system of selective tree harvesting in these forests is unlikely to affect the species composition structure level unless the harvesting level surpasses the rate of recruitment (i.e., are unsustainable; see Obiri *et al.* in press; Chapter III).

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CHAPTER VI

CANOPY GAPS IN SUB-TROPICAL FOREST: SIZE OF THE SPECIES POOL AND NOT THE NUMBER OF AVAILABLE NICHES LIMITS SPECIES RICHNESS

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ABSTRACT

We consider whether species richness in canopy gaps in coastal scarp forest (South Africa) is constrained by competition for a limited number of available niches or is a chance effect of recruitment and dispersal limitation on species derived from the surrounding species pool. In so doing we provide the first test of the findings of Duncan *et al.* (1998) from relatively species-poor New Zealand *Nothofagus* forest. We investigate the contrasting roles of determinism and chance, and the putative assembly trajectory of woody plants in gaps, using a standard technique in which the species richness of small, replicate local assemblages (0.25 m² quadrats) are sampled within regions (the surrounding gap) that vary in total species richness. The form of the regression of local species richness against regional species richness indicates the extent to which community assembly is under local-ecological control (niche limited). Species richness of local assemblages was not niche limited and increased as a constant proportion of the size of the species pool in a gap. We argue that, in general, species assembly in forest gaps is chance-driven, and discuss the management implications this has for selective pole-harvesting in these forests.

Keywords: gap-phase regeneration, harvesting, local-regional processes, niche limitation, species availability, species pool, subtropical forest

INTRODUCTION

It is commonly argued that by promoting the coexistence of competing species, canopy gaps in forested habitats play an important role in the regeneration and the maintenance of tree diversity (Brokaw & Busing 2000, Hartshorn 1978, Zagt & Werger 1998). Although the importance of canopy gaps *per se*, as opposed to gaps in the understorey or leaf-litter, has been questioned (Brown & Jennings 1998, Grubb 1996), the process of gap-phase regeneration is generally accepted (Brokaw & Busing 2000). Recently, attention has focussed on two aspects of gap regeneration: (1) are gaps filled by chance occupants or by best adapted species? (Brokaw & Busing 2000, Hubbell & Foster 1986a, Midgley *et al.* 1995, Zagt & Werger 1998, Zobel 1992); and (2) is species richness in a gap constrained by the limited number of available niches or the size of the species pool (Duncan *et al.* 1998, Silvertown & Law 1987, Wilson *et al.* 1987)? In the case of the first question, it appears that given the stochastic availability of gaps, and the limited recruitment of juveniles, gaps are filled mostly by chance occupants (Brokaw & Busing 2000, Zagt & Werger 1998, Obiri & Lawes, unpublished manuscript; see Chapter V) drawn from a pool of species that are capable of surviving canopy gap conditions (Zobel 1997). In fact, this chance establishment and survival may well slow competitive exclusion and maintain tree diversity in forests ("community drift model"; Hubbell 1998, Hurtt & Pacala 1995).

In terms of the second question above, proponents of the niche limitation view argue that, within gaps, local and interspecific interactions, such as competition and predation, restrict the number or identity of species that can co-occur in a given area (Brokaw 1985, Denslow 1987). The implication of this proposition is that biological "assembly rules" involving niche, not chance, determine community richness and composition (Terborgh *et al.* 1996). However, recent ecological theory suggests that the availability of species from the surrounding region (species pool) is as important a determinant of the level of species richness in a community (Mueller-Dombois & Ellenberg 1974, Ricklefs 1987, Srivastava 1999, Zobel 1992, 1997). Here, dispersal and recruitment limitation, rather than competition for a limited number of niches, constrains the number and identity of species that co-occur in a local area (Hubbell & Forster 1986a).

Gaps may differ in species richness because different numbers of species are available in the surrounding region to colonize a locality (Srivastava 1999, Zobel 1997).

We test these alternative models of community assembly (niche limitation vs. the size of the species pool) by applying a standard method in which the species richness of small, replicate local assemblages (0.25 m² quadrats within forest gaps) are sampled in regions (the surrounding gap) that vary in total species richness (see Srivastava 1999). The form of the regression of local species richness (derived from the quadrats) against the number of species in the surrounding region (the whole gap) indicates the extent to which community assembly is under local-ecological control and thus niche limited (see methods; Lawes *et al.* 2000, Ricklefs 1987, Srivastava 1999). In this way we provide the first test from coastal scarp forest in South Africa, of the findings of Duncan *et al.* (1998) from New Zealand *Nothofagus* forest.

Finally, the consideration of species richness and the putative processes influencing regeneration in gaps has practical relevance within the context of the use of coastal scarp forest by local people. It is mostly the fast-growing, light-demanding trees that are harvested from gaps in coastal scarp forests (Obiri *et al.* in press; Chapter III). The influence of harvesting on recruitment and competition within gaps remains to be investigated but will be aided by an appreciation of whether or not species richness within gaps is limited by the size of the species pool or by the number of available niches.

METHODS

Study site

This study was conducted in the Mt. Thesiger Reserve (31°37'S, 29°31'E) of Umzimvubu District in the Eastern Cape Province of South Africa. The area receives an annual rainfall >1000 mm, mostly October through March, and has a mean annual temperature of 19 °C. The forests are subtropical in nature and classified as coastal scarp forests (Cooper and Swart 1985). Fundamentally, this forest type comprises a mixture of floral elements from both Afromontane and Indian Ocean coastal belt forests (Lawes 1990,

MacDevette *et al.* 1989); consequently it is the most species rich forest type in southern Africa. The indigenous forest in the Mt. Thesiger Reserve is approximately 1400 ha in extent and includes a number of smaller fragments (range = 4 to 101 ha) in addition to a large forest block. In this study, gaps in the main forest block were sampled.

Data collection

We sampled 30 forest gaps. These gaps were representative of the range of gap sizes in the forests. Although species composition in gaps is sometimes related to differences in gap age, we were not able to reliably estimate the ages of the gaps, and were thus unable to control for the age-effect in subsequent analyses. However, to account for this problem we sampled only those gaps that were clearly old and definitely not recent in origin. These were gaps that had in them large-sized gap-fillers i.e. gaps with oldest gap-fillers. All gaps sampled were canopy gaps, characterised by a lack of canopy cover, and caused by tree-fall either due to wind-throw or senescence. Gaps obviously caused by harvesting of large trees were not included to ensure that as far as possible natural gaps were sampled. The length and width of each gap was measured. The length (L) was measured as the longest distance between the boles of trees at the gap edges. Width (W) was measured as the longest perpendicular distance to the length between tree boles. The heights of the trees surrounding the gap were also measured and the mean height (H) calculated. The identity and number of gap-makers i.e. those trees whose fall created the gap, were recorded.

In each gap we sampled the local species richness of woody plants (seedlings and saplings) in 0.25 m² (0.4m x 0.625m) quadrats. Rectangular quadrats were used in preference to square quadrats to maximise the perimeter to surface area ratio and thus sampling contact with species (Stohlgren *et al.* 1995). Only seedlings > 10 cm high were counted because below this height seedlings face high mortality and are not all recruited into the forest (Brown & Whitmore 1992, Chapman *et al.* 1999). Our definition of seedlings includes saplings ≤ 3m high with a diameter ≤ 5 cm. Each quadrat was spaced 5 m apart along transects in the gap. Applying a method used by Duncan *et al.* (1998) two to

six transects were laid out in a gap, the number depending on gap size. The first two transects were positioned along the length and width of a gap and subsequent transects parallel to these two transects but positioned at 5 m intervals on either side. Sample quadrats that coincided with surfaces other than the normal forest floor (e.g. rocky surfaces, large logs or stumps, or root cavities or mounds) were generally bare of plants and not sampled. In this way we assessed 413 quadrats from 30 gaps, with the number of quadrats in a gap varying from 8 to 26.

Different species compositions in forest gaps primarily arise from differences in gap sizes. This is because the environmental conditions in a gap, such as the intensity of light, temperature and humidity, vary with gap size (Chazdon 1986, Dirzo *et al.* 1992, Kyereh *et al.* 1999). Consequently, we used gap size ($\pi LW/4$, Runkle 1982) along with gap orientation (degrees of the longest axis from north), gap light intensity (L/H, Midgley *et al.* 1995, Qinghong & Hytteborn 1991), gap shape (L/W), and the number of gapmakers per gap, as proxy variables to characterise environmental differences between gaps (see Duncan *et al.* 1998). In addition, since variation in light intensity within a gap varies from the gap centre towards the gap edge (Bazzaz & Wayne 1994, Chazdon 1986), and small-scale differences in species richness may be associated with this variation and influence local species richness, we therefore noted whether quadrats were located under overhanging canopy trees on the edge of the gap (i.e. gap-edge quadrat) or in the open canopy gap (i.e. open-canopy quadrat).

Previous studies have estimated regional richness of gaps (i.e., total gap richness) cumulatively from the local species richness within gaps (i.e. cumulative quadrat richness) (see Srivatsava 1999 for a review). Both Stohlgren *et al.* (1995) and Srivastava (1999) point out that in the latter method estimates of local and regional richness are not independent, since they are estimated from the same data set of site-specific species lists. This can result in spurious correlations. Therefore, gap species richness (regional richness) was estimated by first dividing each gap into four quadrants then randomly sampling each quadrant using between four to eight 0.25m² quadrats – depending on the size of the quadrant. We also searched for any species that were within a gap but not detected in the

sampling quadrats (e.g. large trees) and added these to the sum from gap quadrants to estimate the total gap species richness (regional richness).

Testing for niche limitation

We tested for niche limitation in forest gaps by regressing local species richness in a gap (dependant variable) against the overall (regional) species richness of a gap (Caley & Schluter 1997, Cornell & Lawton 1992, Duncan *et al.* 1998, Lawes *et al.* 2000, Ricklefs 1987). Two generalised outcomes are possible (for a review see Cornell & Lawton 1992):

- (1) If competition for a limited number of niches constrains local species richness then local richness may be constant or independent of regional richness and is described by either no relationship or an asymptotic curvilinear relationship. In other words there should be a ceiling to the number of locally co-occurring species (Cornell 1993, Cornell & Lawton 1992, Duncan *et al.* 1998, Ricklefs 1987, Zobel 1997).
- (2) On the other hand if local richness is limited by the availability of species then local richness should increase as a constant proportion of regional richness (i.e. proportional sampling of the regional pool) (Caley & Schluter 1997, Cornell & Lawton 1992, Duncan *et al.* 1998, Zobel 1997). A strong linear relationship results (straight-line, with maximum slope of 1). This is taken as evidence that local richness is independent of biotic interactions (niche competition) that occur within the local habitat (or that such local interactions are insufficient to limit local richness).

In order to avoid apparent spatial pseudoreplication in the regression analysis, where each regional value (species richness of the gap) is expressed several times for different localities (quadrats within a gap), quadrat species richness of a gap was expressed as a single mean (Srivastava, 1999).

Data were plotted as scattergrams and the likelihood of niche limitation was tested by fitting linear ($L = a + b \cdot R$) and polynomial (quadratic $L = a + b \cdot R + c \cdot R^2$) curves (where L and R are local and regional species richness, respectively, and a , b and c are parameters). Least-squares regression techniques were used. In order to test for

curvilinearity we chose the quadratic model over the power model ($L = aR^b$) because the linear model is nested within the second order (quadratic) polynomial model. This makes it easy to test whether the quadratic model is a significantly better fit than the linear model by a simple t -test on the extra term, or by an F -test on the reduced residual variance. Among the competing models the best fit was considered to be that model which had the lowest mean sum of squares of residuals, provided that the regression overall was statistically significant (F ratio test; $P < 0.05$) and that every coefficient in the model was statistically significant (t -test, $P < 0.05$).

Several studies suggest that regressions of this type should be forced through the origin because regions with a species pool of zero must also have local richness of zero (Caley and Schluter 1997, Hawkins and Compton 1992). However, as Srivastava (1999) has pointed out, constraining the intercept in this way (1) effectively extrapolates beyond the range of the data and (2) may overestimate the regression curvature causing an artefactual resemblance to a saturation (niche limited) curve. For these reasons we did not anchor our regression through the origin.

Cresswell *et al.* (1995) demonstrate that autocorrelation between estimates of local and regional species richness compromises the significance levels of regressions of local on regional richness. They argue that estimates of regional and local species richness contain a common term represented by the species that are endemic to a locality. Thus, where a non-linear, niche limited relationship is accompanied by a peak in levels of local endemism at intermediate to high values of regional richness, a linear relationship between local and regional species richness may result. Cresswell *et al.* (1995) suggest that this autocorrelation can be rectified by removing those species that are found in one sample quadrat only within a gap from estimates of species richness for the whole gap or region in which the quadrat or locality is embedded (e.g. Caley, 1997). We adopt this suggestion and explore the effect of endemism (species unique to a quadrat in a gap) on the model fits.

RESULTS

Local quadrat richness was significantly associated with regional gap richness in both the linear and curvilinear models (Table 1). However the quadratic term was not significant in the polynomial model ($t = -0.93$, $P = 0.36$) and thus the linear model was the best fit to the data. The influence of an outlying data point (Fig. 2; gap species richness = 24, mean quadrat species richness = 3.8) that was exerting leverage on the fit of the linear model was investigated. Although the fit of the polynomial model was improved when this data point was removed, the quadratic term was nevertheless not significant ($t = -1.90$, $P = 0.07$). Thus, local richness is proportionally enriched from the species pool in gaps and is not limited by the number of available niches (Fig. 1).

Although the gap environment and size can influence species composition of a gap, none of the environmental variables characterising the gap environment entered a multiple regression model (Table 2). Area did, however, enter the model (accounting for 20% of the variance in local richness) and local richness is clearly influenced by the area of the gap. Interpreting the fitted curves is difficult where the size of the regions is allowed to vary (Srivastava, 1999). A strong positive regional species-area relationship, as found in this study ($F_{1,28} = 7.1$, $P = 0.013$, $R^2 = 17.3\%$), may increase the probability of detecting a spurious curvilinear relationship because the equal sized local samples may include a decreasing proportion of larger and more species-rich regions (Caley and Schluter, 1997, 1998). We addressed this problem largely by increasing the number of quadrats sampled with increasing gap size. Although, the size of the gap was not fixed, and indeed could not be fixed, in this study, a curvilinear relationship was not detected. Nevertheless, the local-regional regressions were rerun controlling for the effect of gap area. The data fit the models better (linear: $F_{1,28} = 25.7$, $P < 0.0001$, $SS_{\text{error}} = 52.3$; quadratic: $F_{2,27} = 13.35$, $P < 0.0001$, $SS_{\text{error}} = 49.8$) but confirmed the fit of the linear (species pool) model since the quadratic term was not significant ($t = 1.16$, $P = 0.26$) in the curvilinear (niche limitation) model.

The fit of the data to the species pool model (linear) could be an artefact of species variation between gap-centre ($n=154$ quadrats) and gap-edge quadrats ($n=259$ quadrats), as

well as the different proportions of these quadrats in gaps. There were significantly more species in gap-centre quadrats (2.57 ± 0.13 , $\bar{x} \pm 1$ SE) than gap-edge quadrats (1.84 ± 0.12 , $\bar{x} \pm 1$ SE; paired t -test = 5.76, $P < 0.001$). However, very few species ($n=6$, $N=64$ species) were found in edge quadrats that were not present in centre quadrats and visa versa ($n = 12$ centre species). In addition, of the 18 species either found exclusively at the gap edge or centre, 12 species were found in one quadrat only ($n = 410$ quadrats).

Table 1. The evaluation of the fit of linear and curvilinear regressions of mean quadrat (local) species richness in a gap and total gap species richness (regional). MSS_R = mean residual sum of squares regional species. Note that the quadratic term in the polynomial model was not significant ($t = -0.93$, $P = 0.36$) and the linear model is selected in preference to the polynomial.

Regression type	Model	R^2	MSS_R	F -ratio	P
Linear	$y = 1.072 + 0.068x$	0.232	2.82	9.76	0.004
Polynomial	$y = -0.223 + 0.224x - 0.004x^2$	0.228	2.84	5.29	0.01

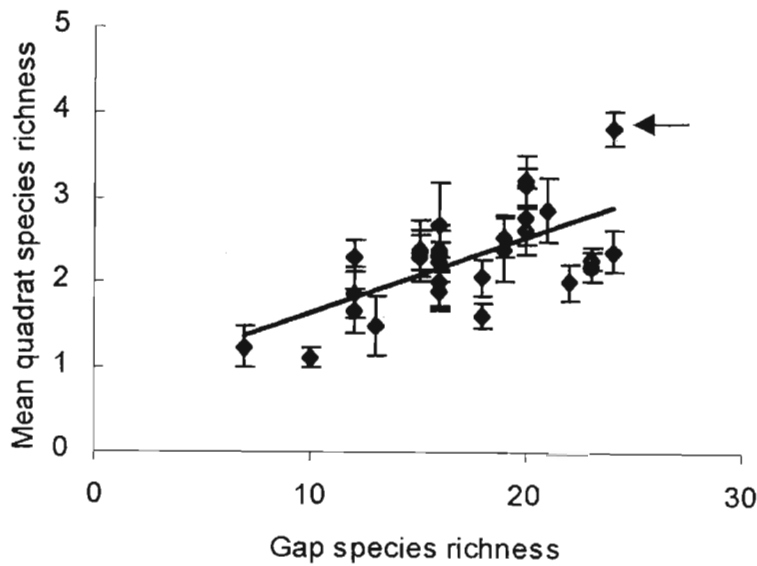


Figure 1. The relationship between mean quadrat species richness (local richness) within gaps and total gap species richness ($n=30$). The best-fit linear regression line is significantly positive (see Table 1) and local richness increases proportionately with regional richness. Mean quadrat richness, standard error bars and the outlying point (see arrow) are shown.

Table 2. Gap characteristics and their fit to a multiple regression model. Mean quadrat species richness within canopy gaps (n=30) is the dependent variable and gap characteristics and gap species richness are the independent variables. The coefficient estimates and their *t*-values are shown. See text for explanation of the computation of the various gap environment indices.

Gap characteristic	Mean	SE	Range	Coeff. estimate	Coeff. SE	<i>t</i>	P
Dependent variable							
Quadrat species richness	2.3	0.1	1.1-3.8				
Independent variables							
Constant				1.257	0.523	2.40	0.026
Gap species richness	17.1	0.8	7-24	0.112	0.023	4.77	0.0001
Gap area (m ²)	102.8	9.7	31.8-235.5	-0.006	0.002	-3.36	0.003
Gap shape index	2.16	0.13	1.09-4.73	-0.179	0.123	-1.46	0.159
Gap light index	1.43	0.47	0.39-14.0	0.018	0.032	0.57	0.57
Gap orientation	128.1	28.1	0-340	-0.0003	0.0006	-0.52	0.61
No. gapmakers	1.16	0.08	1-3	0.172	0.268	0.64	0.53

To examine the influence of these gap-edge or -centre species we removed them from the data and reran the regression models. The linear (species pool) model was again the best fit ($t = 2.82$, $P = 0.01$; quadratic term $-t = -0.8$, $P = 0.43$) to these data.

Allied to the latter analysis we further explored the effect of endemism (species unique to a quadrat in a gap) and possible autocorrelation (*sensu* Creswell *et al.* 1995; see methods) on the model fits. Once again the quadratic term was not significant ($t = -0.11$, $P = 0.91$) and the linear (species pool) model ($t = 2.52$, $P = 0.02$) was chosen.

DISCUSSION

This study reveals no evidence of a ceiling marking an upper limit to the number of locally co-occurring species in gap quadrats. We therefore reject the hypothesis that competition for a limited number of niches within canopy gaps constrains the number of co-existing species over a small-scale. Indeed, localities within forest gaps appear to be unsaturated with species (*sensu* Cornell 1993), and are a proportional sample of the species pool in a

gap. Rather than competitive effects it is the size of the species pool that limits small-scale species richness in gaps.

Local-regional curves are determined by the way total diversity is partitioned between its α and β components and hence the scale at which the heterogeneity of the interactions between organisms and their environment manifests itself (Cornell 1993, Loreau 2000, Shurin *et al.* 2000). Thus, as pointed out by Duncan *et al.* (1998) our conclusions derived from local-regional analyses are based upon the vital assumption that the sample quadrat size is sufficiently large, or at an appropriate scale, for detecting potential competitive effects among species. Duncan *et al.* (1998) discuss this problem and use a 0.2 m² quadrat size to detect niche limitation in temperate *Nothofagus* forest. Similarly, we use a 0.25 m² quadrat size, which is a compromise between a fine scale, that was likely to detect species saturation (Wilson *et al.* 1995), and a size large enough to sample as many species as possible.

The contribution of chance and determinism to successful regeneration are not constant during the different stages of the regeneration cycle. Duncan *et al.* (1998) argue that disturbed systems like young and transient forest canopy gaps, may not have reached competitive equilibrium, making it unlikely that we would observe niche limitation. Recently formed gaps are unlikely to show niche limitation because (i) gap localities may still be accumulating species and local niches would not be saturated, or (ii) localities are oversaturated but insufficient time has passed for competitive sorting to have eliminated inferior competitors (Duncan *et al.* 1998). Taking the latter into account, and based on the state of decay of the gap-makers (Obiri & Lawes, unpublished data), we were careful not to choose young gaps (< 4 years). Hence it is likely that sufficient time would have passed for competitive sorting to be well established in the gap localities. Furthermore, if the gaps are relatively old (i.e. approaching equilibrium), then we can reasonably assume that the total number of species in a canopy gap describes the pool of species available to occupy a quadrat within that gap. We can further extend this reasoning to conclude that in this study, because species richness in a quadrat scales in proportion to increasing species richness within the gap as a whole, that natural levels of competitive exclusion are weak at best, and insufficient to limit local richness.

That gap assemblages were limited in composition only by the size of the species pool in the gap (i.e. the influence of other gap environment variables was negligible) in this study, supports our previous findings of chance establishment and random replacement of species in gaps in coastal scarp forests (Obiri and Lawes, unpublished manuscript; Chapter V). Like Duncan *et al.* (1998) we argue that it is the history of dispersal to a gap and recruitment limitation within a gap, rather than competition for a limited number of niches, that constrains the number and identity of species that co-occur in a local area. Similarly, Midgley *et al.* (1995) found no correlation between species composition and gap size or forest structure in a subtropical forest in South Africa. They showed that random replacement (lottery model, Chesson 1986) was the dominant pattern for the more common species.

Both our study and that of Duncan *et al.* (1998), at forest sites that are continents apart, thus suggest a reduced role for niche limitation in determining the number and composition of gap species assemblages. Indeed, there is growing evidence that species interaction in gaps plays little role in influencing tropical forest plant community structure (see review in Brown & Jennings 1998). While further testing is required, in general, it seems that gap species assemblages are more likely to be driven by chance than determinism (species interactions). Rather than being highly specialised, most tropical forest species appear to be generalists, capable of adequate growth under a range of canopy conditions (e.g. Lieberman *et al.* 1995, also Grubb 1996, Hubbell & Foster 1986b, Welden *et al.* 1991, Zobel 1997). Thus, given the stochastic availability of gaps, and the limited recruitment of juveniles, gaps are filled mostly by chance occupants (Brokaw & Busing 2000, Hubbell & Foster 1986a, Midgley *et al.* 1995, Zagt & Werger 1998, Obiri & Lawes, unpublished manuscript; Chapter V). Being in the right place at a favourable time might be more important to a tree's success than competitive superiority in that particular environment (Hubbell & Foster 1986a). Furthermore, because the unpredictable identity of competing neighbours results in no constant selective pressure fostering niche-differentiation or specificity (Hubbell & Foster 1986a, Zagt & Werger 1998), chance establishment and survival may well slow competitive exclusion and maintain tree diversity in forests ("community drift model"; Hubbell 1998, Hurtt & Pacala 1995).

Implications for conservation and management

The importance of gap dynamics to the maintenance of forest plant diversity in the coastal scarp forests of the Eastern Cape Province remains to be confirmed. However, while canopy gaps *per se* are not all-important for regeneration in tropical forests (Grubb 1996), in scarp forests that experience high levels of canopy disturbance, canopy gap dynamics may have an important role in regulating species composition. Coastal scarp forests experience a fairly intense natural disturbance regime as a consequence of strong coastal winds (Everard *et al.* 1995), and the frequency of canopy gaps caused by windthrow is high (50%, Obiri & Lawes, unpublished manuscript; Chapter V). As species regenerating in gaps grow quickly and tend to be straight and tall, they are commonly harvested for poles (Obiri *et al.* in press; see Chapter III). If gaps were dominated by gap-adapted species, then such selective harvesting might lead to successional collapse (McKenzie 1988). However, since there is no dominant gap-filling guild (Obiri & Lawes, unpublished manuscript; Chapter V), but rather most species occur randomly in gaps, this is unlikely to occur. With this understanding of forest dynamics we can manage subtropical forest tree diversity. Provided the direction of control of species richness is from regional to local (limited by the size of the regional species pool), and not local to regional (limited by the number of available niches), species diversity is likely to be maintained by the stochastic nature and frequency rhythm of large-scale environmental processes such as gap forming events. Under these circumstances, even moderate harvesting levels are unlikely to create conditions of specific competitive advantage for plants, especially if species that have high densities and a fine-grained pattern of regeneration are harvested (Obiri *et al.* in press, Chapter III; Lawes & Obiri, unpublished data). It is perhaps worth pointing out that the putative process of regulation of species richness, described here for coastal scarp forests, may not apply to the higher altitude Afromontane *Podocarpus* forests where canopy gaps are infrequent, although could be influential in the supposedly disturbance driven Indian Ocean coastal belt forests of the eastern seaboard of southern Africa (i.e., Maputaland and Mozambique). Either way this

study calls attention to the usefulness of considering seedling and understory ecology as an aid to explaining the processes that promote the coexistence of large numbers of species in forests.

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CHAPTER VII

CONCLUSIONS AND RECOMMENDATIONS

CONCLUSIONS AND RECOMMENDATIONS

This study examines the ecological challenges facing the sustainable use of indigenous forests in Eastern Cape Province of South Africa. It integrates ecological and socio-economic approaches to forest management - a current global trend to solving environmental problems (Lubchenco 1998, Bradshaw and Bekoff 2001).

The study area (coastal region of Umzimvubu District, Eastern Cape Province of South Africa) is typical of rural life in South Africa and is characterised by poverty, high unemployment (80%; Risdale and Kallam 1996), the inability of agriculture to meet subsistence household needs, and a reliance on natural resources for income. Households are large (mean size = 7 persons; Mukolwe 1999) and with poor incomes (US\$ 32-160 month⁻¹; Kieplei and Quilan 1996), largely from pensions and remittances from migrant workers. Forests are an important source of income and provide a widespread supply of goods and services. Fuelwood, traditional medicines, raw materials for the craftwork and curio industry, grazing and timber are among the key goods provided, while services include the maintenance of the hydrological cycle, promotion of tourism through the provision of habitats for flora and fauna, and the provision of sacred and spiritual places of cultural importance. Over the past decade the demand for forest products has increased. For example, Cawe and Ntloko (1997) and Obiri (1997) report on the increased harvesting of *Flagellaria guineensis* vines for basketry and carving wood from *Millettia grandis* trees, respectively.

Seventy per cent of the forests in Umzimvubu District are under state management while 30% are community-controlled. In both of these forest types the use of key forest products (i.e. fencing posts, building posts and medicinal trees) is unregulated and unsustainable. Trees are harvested irrespective of whether the species is scarce or not and with little consideration for the forests' ecological integrity and capacity to sustain these harvesting levels. Wise and sustainable use of forest must first consider the ecological processes driving forest biodiversity and ecosystem integrity (i.e. ecological sustainability) because without these forests cannot establish or maintain themselves. Allied to this ecological requirement is the role of local communities in

conservation efforts, sound management policies and a strong political will to conserve forests.

The primary objectives of this study were to (1) establish an understanding of the local people's perceptions to different forest management practises, (2) establish the existing quantities of the exploited forest resource, (3) identify the key forest resources exploited and the views toward resource use (4) establish the ecological nature of tree recruitment and how it may be influenced by tree cutting activities.

Important Findings and Recommendations

Forest management practices

Options for forest management practice occur in a continuum beginning with fairly uncontrolled resource use at one end and grade through community management, and the restricted use of resources under state management to complete protection of forests, at the other. Three recommendations arise from this study that relate to ensuring best forest management practice.

- (1) Participatory forest management (PFM) should be implemented. Forest stakeholders (i.e., resource users and managers) supported the notion of PFM although from their experience they preferred state to community management when asked to choose. PFM would allow the contribution by state structures to be balanced by community involvement and has more chance of being a stable management structure than either (State Forest Management) SFM or (Community Forest Management) CFM on their own.

Community forest management has lost favour among resource users largely because forests under this system have been heavily encroached and degraded and community management structures have failed in recent times to prevent indiscriminate use of forests. A reduction in the authority of local chiefs' under the new political dispensation since 1994 (Pycroft 1999) has much to

do with this failure and the preference for more state involvement by local communities in the management of forests.

- (2) Because the involvement of the traditional leadership is important for the successful implementation of any forest management structure in Umzimvubu district, government should look for ways of strengthening the authority of traditional management institutions with respect to community forms of forest management. Currently, community involvement in PFM is meaningless since community leaders and their institutions have no real powers in the partnership. Should this not change, the 'intended beneficiaries' in PFM are likely to remain unnoticed and this will cause the demise of the PFM initiative. Similar trends have been reported in rural communities in Ethiopia (Campbell 1991).
- (3) If PFM is implemented, low-income earners in the community should be given high priority and be placed at the core of the PFM forum, since they have the most impact on forest conservation and face the highest costs of lost opportunity if forest use is restricted.

However, it should be noted that PFM is no panacea for forest management. In South Africa its success depends on a variety of factors that are linked to the participating stakeholders. These include the recognition of various interest groups, reconciling the views and needs of the different groups, raising the awareness of participatory initiatives, providing viable and sustainable alternatives to forest products, as well as the political will to support PFM processes (Mjwara *et al.* 2000). A key factor though, is the economic incentive offered to communities which, if reduced or removed, can lead to a reversal of views or even the abandonment of PFM on the whole – a factor reported in other PFM case studies with socio-economic circumstances similar to those in Umzimvubu District (e.g. Norton-Griffiths and Southey 1995, Hackel 1998). A further concern is that in forestry, unlike the case of wildlife-orientated PFM, which are often cited as successful PFM initiatives (e.g. the Zimbabwean CAMPFIRE programme and Zambia's Administrative Design for Game Management - see Western *et al.* 1994), economic incentives are few and less profitable (Campbell *et al.* 1996, Nhira *et al.* 1996), and where available they often cannot be sustained (Brigham 1996).

Resource Availability

The forest resource base in Umzimvubu District is limited and only *Englerophytum natalense* is recommended for exploitation under the current unregulated harvesting intensities. This species is largely used for poles and posts and displays high regeneration under the forest canopy.

Quantification of the resource base in Umzimvubu forests and South African indigenous forests in general has been impaired by a lack of long-term data on population dynamics of forest trees (Cooper and Swart 1992, Shackleton 1996, Cawe 1999). Instead, estimates of the resource availability has largely relied on the description of size-class frequency distributions of species population structure from single sampling surveys (e.g. Midgley *et al.* 1990, Shackleton 1993, Van Wyk and Everard 1993, Everard *et al.* 1994, 1995, Cawe 1999, Obiri *et al.* in press). Although Geldenhuys and Daalen (1992) have established permanent sample plots (PSPs) in selected regions in South Africa these are few, still in their infancy and none are in Umzimvubu District or in the Pondoland Region of Eastern Cape Province. It is imperative that these PSPs are established in the latter region.

In the absence of longitudinal data on growth dynamics of tree species the concept of the spatial grain of regeneration remains important in determining what species can be harvested. Grain is a spatial form of analysis that compares the number of individuals of a species in the canopy and subcanopy level. In fine-grained species the canopy and sub-canopy individuals are well represented in a small forest area and their regeneration is high over a small spatial area. These fine-grained species are most ideal for harvesting as they have an abundant supply of understorey trees whose exploitation would have less effect on the forest structure and composition than the exploitation of the more coarse-grained species. In this study *E. natalense* was the most fine-grained species while *Harpephyllum caffrum* and *Heywoodia lucens* were among the most coarse-grained and their exploitation should be discouraged.

Fuelwood and medicinal tree species and the highly sought pole-sized (10-20 cm dbh) trees were more abundant in state than community forests. Similarly, deadwood production was higher in state forests (2.36 tonnes.ha.yr⁻¹) than in community forests (1.22 tonnes.ha.yr⁻¹). Clearly state forests have a higher abundance of resources than community forests. This fact has not gone unnoticed by villagers and may be a factor in persuading them that SFM is in some ways better than CFM. Nevertheless, and villagers are likely to target state forests for resources as the resources in community forests become exhausted. Thus, conservation of state forest should be taken as a high priority if these forests are to survive into the next decade.

Resource use

All forests in Umzimvubu District face some form of resource exploitation. Twenty tree species were actively exploited for nine major uses by the community, with building poles, fencing posts, medicines and fuelwood being the most dominant. The current use of these products is unsustainable particularly in community-controlled forests. The use of medicinal tree species is of the gravest concern since only one species (*Macaranga capensis*) occurred in quantities that were harvestable yet several species were intensively exploited by the communities. *Cassipourea gerrardii* is highly threatened by medicinal harvesting practices as it is debarked when standing and also frequently cut down to facilitate maximum bark stripping. Apart from *M. capensis* all medicinal tree species had poor bark regeneration and limited coppicing ability and their use showed all the signs of pending over-exploitation. The high mortality levels of trees (>28%) arising from debarking lends weight to this conviction.

Harvesting of posts and poles affected a wide range of species as it was not species specific but depended on tree size. Trees ranging from 5-55 cm (dbh) were harvested but those of 10-20 cm dbh were the most frequently targeted. In community forests the harvesting of these stems is unsustainable and certain species are absent due to overuse. This selective harvesting practice is consistent with harvesting patterns reported from other studies within southern Africa (Shackleton 1993, Vermuelen 1996) and where woodlots of *Eucalyptus spp* have been established to reduce the

cutting of indigenous forests (Allen *et al.* 1988, Nhira *et al.* 1998, Abbot and Homewood 1999).

Unlike the harvesting of poles and posts, fuelwood harvesting is species selective and targeted a small number of species. Nonetheless, it is unsustainable since the average deadwood biomass produced by the forests (1898 tonnes.yr⁻¹) marginally balanced that removed as fuelwood through harvesting (1532 tonnes.yr⁻¹). Furthermore, being a species-specific harvesting practice, fuelwood use has resulted in overuse and the loss of some preferred species from forests. Continuous use of preferred species such as *P. obliquum* increases their scarcity and progressively more effort is needed to search for and harvest them until a point is reached where it is uneconomical to continue their exploitation (Sutherland and Reynolds 1998). At this point, alternative species, usually less used due to their inferior quality, are exploited. In the study an example includes *Englerophytum natalense* which is increasingly being used in place of the scarce *P. obliquum*. Typically this use of open access resources results in irreversible damage to the preferred species by the time it is replaced by an alternative species (Hardin 1995). The signs of overuse are evident in the forests of Umzimvubu district and require prompt management action to ensure the survival and ongoing use of the forests.

Recognising that tree use is unsustainable in the district I recommend that:

- (1) the debarking of trees for medicinal purposes should be stopped or tightly controlled in all species except for *M. capensis* where regulated use can continue. Although the forest department has limited resources to adequately police the forests the locally driven PFM or CFM programmes should be encouraged to take up this initiative. Urgent and special protection is needed for *C. gerrardii*, which is highly sought after and intensively exploited;
- (2) prioritising which tree species and size to cut for poles and posts should be based at this time, and until better data are available, upon the model of the spatial grain of regeneration of favoured trees; and
- (3) fuelwood harvesting in indigenous forests should be closely regulated. Incentives could be created for users to harvest from the existing *Eucalyptus* woodlots that neighbour indigenous forests. Other areas within southern Africa and with similar fuelwood problems have successfully used woodlots

to reduced pressure on indigenous forests (Allen *et al.* 1988, Muir 1990, Abbot and Homewood 1999). There are other sources of energy and energy saving measures that should be investigated too (Rubab and Kandpal 1996, Loxton 1997, Wallmo and Jacobson 1998). In fact, there is no doubt that the future of forests in the region will depend upon the identification and provision of suitable alternative products for forest products.

Although forest tree exploitation is rampant in the study area, it was encouraging that all resource users support the need for controlled or regulated harvesting. The difficulty is to put this into practice. Community-led forest regulation is difficult and often fails due to the actions of rule-breakers, even when supported by the state (Hackel 1998, Poffenberger and Singh 1998, Campbell *et al.* 2001). Rather than villagers solely relying on forest products from state and community forests, I suggest that they be encouraged, through forest extension services, to establish small stands of multipurpose trees in their gardens and farms. Or at least plant some multipurpose trees on the boundaries of their properties when land is scarce. This is essentially an agroforestry practice whose potential in the study area is high given the enormous demand for forestry products and the willingness of villagers to plant useful trees such as *P. obliquum* and *M. grandis*. Van Eck (1997) and Mukolwe (1999) have expressed similar views.

Ecological Factors

Forests in this study undergo gap formation through natural events of windthrow (50%), tree branch snapping (20%) and tree death (i.e. snags; 13%). Seventeen percent of gaps result from tree cutting activities. A high percentage of these forests' area was under gaps (7.8%) relative to other tropical and sub-tropical forests (e.g. 5% Ituri Forest, Democratic Republic of Congo, Hart and Hart 1989; 4.8% Venezuelan Amazon forest, Uhl and Murphy 1982; 2-10% southern Cape forests, Midgley *et al.* 1995).

Gaps create living space under the forest canopy into which prospective gap occupants recruit either by chance (Hubbell and Foster 1986, Lieberman *et al.* 1995)

or through a directed or deterministic process (Denslow 1987, Whitmore 1989, Alvarez-Buylla and Martinez-Ramos 1992). In the Scarp forests of the Eastern Cape Province I have shown that species recruit randomly into forest gaps.

The random recruitment in these forests has conservation and management implications for selective tree harvesting in the Umzimvubu forests. Although selective tree cutting creates gaps, their creation does not significantly favour the recruitment of gap-dependent (pioneer) species over gap-independent (primary) species, and thus is unlikely to cause forests to become dominated by pioneer species with time. Nor are artificially created gaps likely to affect the species composition of the forest given the general absence of a gap-filling species guild. Of course, intensive logging or harvesting practices will alter forest structure and composition, but under moderate levels of tree harvesting and with greater control over what species and how much fuelwood is extracted, the ecological integrity of the forest is expected to be maintained. As a rule of thumb I recommend that tree-cutting practices that create gaps should avoid exceeding the current area of forest under gaps (7.8%) by more than 5% (Hart and Hart 1989, Uhl and Murphy 1982).

The key theme throughout this study is the reconciliation of socio-economic requirements from forests with the ecological constraints on the sustainable use of the forests. Managing of indigenous forest involves more than determining and controlling what species are harvested. One must ensure that the spatial and temporal scales of harvesting are within the forest's capacity to recover i.e. to the point where processes maintaining forest biodiversity and ecosystem integrity are not compromised. Clearly there is still a lot to learn about factors determining sustainable harvesting and this calls for further long-term ecological studies quantifying the growth rates of different tree species, net primary productivity and forest biomass.

As important to the management of forests as the above recommendations on forest policy, resource use and ecology are, without the political will to protect the forests and their irreplaceable resources, forests are doomed. It is unlikely that real progress can be made in forest conservation and sustainable management until the political will exists and is harmonised with the multifaceted social and ecological issues of indigenous forest management.

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