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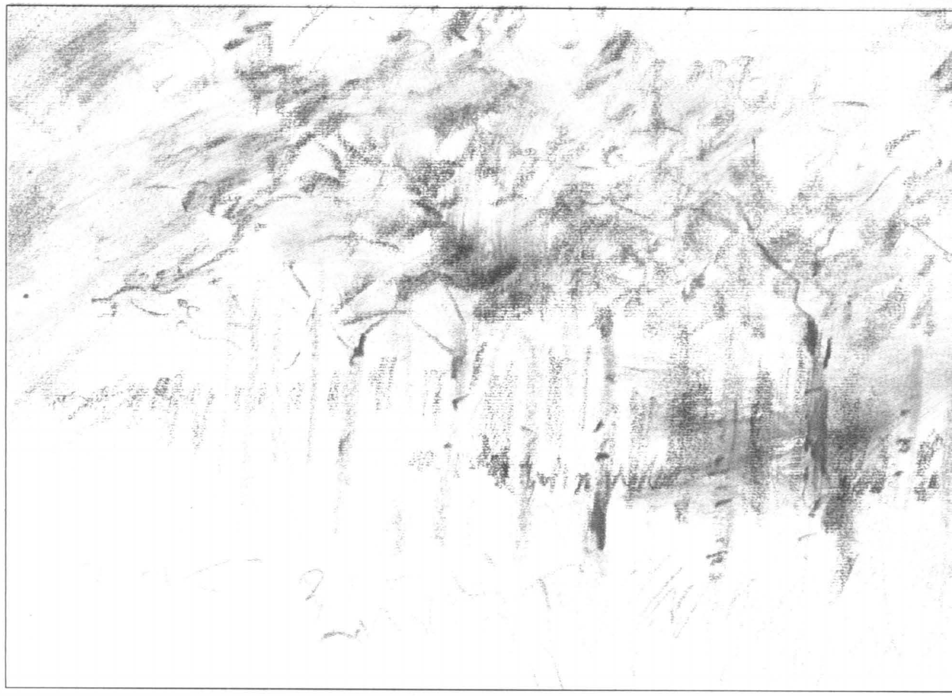

Vesa Kaarakka

Management of Bushland Vegetation Using Rainwater Harvesting in Eastern Kenya

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Management of Bushland Vegetation Using Rainwater Harvesting in Eastern Kenya

The Finnish Society of Forest Science — The Finnish Forest Research Institute

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Microcatchment water harvesting (MCWH) improved the survival and growth of planted trees on heavy soils in eastern Kenya five to six years after planting. In the best method, the cross-tied furrow microcatchments, the mean annual increments (MAI; based on the average biomass of living trees multiplied by tree density and survival) of the total and usable biomass in *Prosopis juliflora* were 2787 and 1610 kg $ha^{-1}a^{-1}$ respectively, when the initial tree density was 500 to 1667 trees per hectare. Based on survival, the indigenous *Acacia horrida*, *A. mellifera* and *A. zanzibarica* were the most suitable species for planting using MCWH. When both survival and yield were considered, a local seed source of the introduced *P. juliflora* was superior to all other species. The MAI in MCWH was at best distinctly higher than that in the natural vegetation (163–307 and 66–111 kg $ha^{-1}a^{-1}$ for total and usable biomass respectively); this cannot satisfy the fuelwood demand of concentrated populations, such as towns or irrigation schemes.

The density of seeds of woody species in the topsoil was 40.1 seeds m^{-2} in the *Acacia-Commiphora* bushland and 12.6 seeds m^{-2} in the zone between the bushland and the Tana riverine forest. Rehabilitation of woody vegetation using the soil seed bank alone proved difficult due to the lack of seeds of desirable species.

The regeneration and dynamics of woody vegetation were also studied both in cleared and undisturbed bushland. A sub-type of *Acacia-Commiphora* bushland was identified as *Acacia reficiens* bushland, in which the dominant *Commiphora* species is *C. campestris*. Most of the woody species did not have even-aged populations but cohort structures that were skewed towards young individuals. The woody vegetation and the status of soil nutrients were estimated to recover in 15–20 years on Vertic Natrargid soils after total removal of above-ground vegetation.

Keywords Drylands, Kenya, land rehabilitation, rainwater harvesting, seed bank, vegetation dynamics.
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Contents

Preface

The sight of a broken canopy of trees and bushes intermingled with dry, tawny-coloured and very scattered grass tussocks is the essence of the landscape in much of Kenya's drylands. The scene is completed with a distant cloud of dust created by the hooves of cattle, goats, sheep or camels in their search for graze and browse. Somewhere in the dust there is the lone herdsman, often a young boy, looking after the animals. It is the people of these areas, and the way how they move about in the bush with their herds that has continuously inspired me in this work. It is the upright and proud posture, the vigilant sight, and the thorough knowledge of their environment that make the pastoralists the masters of that landscape. I have tried to remember this when moving around in the bushland.

The present study is based on trials and experience compiled in the semi-arid bushland of Bura, Kenya. It forms part of the research under the theme "Forest Management and Sustainable Development in Dry and Degraded Tropical Lands", which was started at the Department of Forest Ecology of the University of Helsinki in the 1980's. The thesis is the outcome of my engagement in the Bura Fuelwood Project, which started in 1986 in Finland and 1988 in Kenya. The study was funded by the Academy of Finland, FinnIDA and the University of Helsinki.

There are a number of people whom I would like to thank for much appreciated support on many occasions. I would particularly wish to acknowledge the decisive role of Prof. Olavi Luukkanen who introduced me into the absorbing world of tropical forestry and tropical cultures in the early 1980's. His discreet but supportive guidance has helped me through the ups and downs of the exercise, especially during the writing up. The manuscript was reviewed by Prof. Veli Pohjonen and Prof. Tuomo Niemelä the comments of whom were highly encouraging. Prof. Anders Persson and an anonymous reviewer helped me to concentrate on the essentials in the final manuscript. Prof. Matti Leikola and Prof. Juhani Päivänen, the former and current Heads of the Department, provided for the inspiring atmosphere. Colleagues at the Department of Forest Ecology have aided in various ways. Joint work in dryland forestry with Stig Johansson in both Kenya and Finland brought along new ideas and challenges. Riikka and Antti Otsamo assisted in trial establishment and maintenance in Kenya. During the recent years, Kari Tuomela continuously and willingly shared his experience on statistical and methodological aspects. Consultations or discussions with Abdelrasoul Fadlelmula Mustafa, Hannu Ilvesniemi, Jarkko Koskela, Kari Leinonen, Eero Nikinmaa and Pekka Nygren offered firm background in moments of doubt. Aija Niskanen provided timely backstopping over the years. Antti Vertanen kindly allowed me to use his vegetation data. I would also like to extend my appreciation to the KEFRI researchers and other Kenyan staff for assisting in Bura.

The rapid development of computer networks has not reduced the indispensable role of scientific libraries in providing researchers with up-to-date references. In this regard, I attach a great value to the services offered by the Forestry Library, the Botany Library and the Science Library of the University of Helsinki.

Finally, I would like to express the gratitude to my beloved ones. My mother always supported my pursuits and showed that new cultures and new continents are always worth of acquaintance. My wife, Eva, and our children, Lilli and Samu, created an atmosphere of delight and harmony both here in Finland and at the second home in Kenya. I hope that we will enjoy many more pleasant moments together under the African skies in the years to come.

Helsinki, October 1996

Vesa Kaarakka

List of Symbols and Abbreviations

ANOVA	analysis of variance
ASAL	arid and semiarid lands
BFP	Bura Fuelwood Project
BISP	Bura Irrigation Settlement Project
C	carbon
CILSS	Comité permanent inter-Etats de lutte contre la sécheresse dans le Sahel
d_0	stem diameter at ground level
$d_{0.15}$	stem diameter at stump height (0.15 m)
dg_0	pooled diameter at ground level
$dg_{0.15}$	pooled diameter at stump height (0.15 m)
DM	dry matter
E_0	potential evaporation
E_t	potential evapotranspiration
FAO	Food and Agricultural Organisation of the United Nations
FinnIDA	Finnish International Development Agency
$g_{0.15}$	basal area at stump height (0.15 m)
ICRAF	International Centre for Research in Agroforestry
ILCA	International Livestock Centre for Africa
IUCN	World Conservation Union
KEFRI	Kenya Forestry Research Institute
KFMP	Kenya Forestry Master Plan
MAI	mean annual increment
MCWH	microcatchment water harvesting
MDS	multidimensional scaling
N	nitrogen
n	number of observations
$n_{0.15}$	number of stems at stump height (0.15 m)
NFTA	Nitrogen Fixing Tree Association
NG	Natric Grumustert
P	phosphorus
R	rainfall
RCB	randomised complete blocks
RUE	rain use efficiency
S.E.	standard error of the mean
TARDA	Tana and Athi River Development Authority
TC	Typic Camborthid
TFB	total fresh biomass
TH	Typic Halorthid
TN	Typic Natrargid
UFB	usable fresh biomass
UFV	usable fresh volume
UNEP	United Nations Environment Programme
UNESCO	United Nations Educational, Scientific, and Cultural Organisation
VN	Vertic Natrargid
WH	water harvesting
WUE	water use efficiency

1 Introduction

1.1 Definition and Coverage of Tropical Drylands

Tropical drylands are diverse in terms of climate, soils, vegetation, animals and people's activities. Because of this diversity, no fully covering and practical characterisation or definition of drylands can be made, but the environments are primarily characterised by insufficient and erratic rainfall.

An accepted measure for aridity is the moisture availability ratio used by the United Nations Educational, Scientific, and Cultural Organisation (UNESCO 1979), expressed as the ratio of average annual rainfall (R) to average potential evapotranspiration (E_t). Consequently, the drylands of the world can be delineated into hyperarid ($R/E_t < 0.03$); arid (0.03–0.20); semi-arid (0.20–0.50); and sub-humid (0.50–0.75) zones. Le Houérou (e.g. 1989) divided dryland into deserts (receiving less than 100 mm of rainfall); very arid (100–200 mm); arid (200–400 mm) and semi-arid (400–600 mm) areas. Other classifications based on the amount of rainfall include e.g. that of Heathcote (1983). Zonation systems combining rainfall with other climatic or floristic parameters have been developed by Köppen (1936), Thornthwaite (1948) and Holdridge et al. (1971). In tropical lowlands low temperatures play only a marginal role; drought periods are the primary concern for land use practice in seasonal tropical climates.

The drylands cover more than a third of the earth's land surface. They are mainly found in two belts approximately centred on the Tropics of Cancer and Capricorn. Various figures have been presented for the total coverage depending on the inclusion or exclusion of the above rainfall zones. Dregne (1983) presented a total area of 47.1 million km^2 for hyper-arid, arid and semi-arid zones. According to UNEP (1991), the area is 51.7 million km^2, covering arid, semi-arid and dry sub-humid but excluding hyper-arid zones. Conflicting figures have been presented on the extent of degradation in these areas. UNEP (1991) estimated that 36 million km^2 (or 69 %) of the total have been affected by degradation, of which 93 % are rangelands. In contrast, Thomas and Middleton (1994) presented that 10 million km^2 (or 20 %) are areas, where the soils are degraded. Murphy and Lugo (1986) and Janzen (1988) pointed out that tropical dry forest may be the most threatened major forest type in the world.

Based on the UNESCO (1979) scheme, the arid and semi-arid lands (ASAL) constitute about 88 % of land area in Kenya, occurring in areas between 250 and 1000 mm of average annual rainfall (KFMP 1994). They support roughly 20 % of the human and about two thirds of the livestock population of the country.

1.2 Forms of Degradation in Tropical Drylands

The main physical features of degradation are soil and vegetation degradation. Soil degradation occurs in four main ways: (1) water erosion; (2) wind erosion; (3) compaction; and (4) waterlogging, salinization and alkalinization (Grainger 1990). Changes in dryland soils are usually harder to identify and monitor as compared to vegetation degradation (Thomas and Middleton 1994). Degradation of vegetation follows two main pathways: the reduction in vegetation cover and the change towards a less productive type of vegetation. The former is caused by the clearance of trees for cropping and grazing, cutting of trees for fuelwood and fodder, overbrowsing or overgrazing of rangelands. The latter form involves a change in species composition, and possibly also in the types of plants, e.g. bush encroachment is a process in which an open, often grassy, vegetation changes into bushland or sometimes thicket (Grainger 1990).

Degradation in drylands (Fig. 1) cannot be separated from desertification, which usually refers

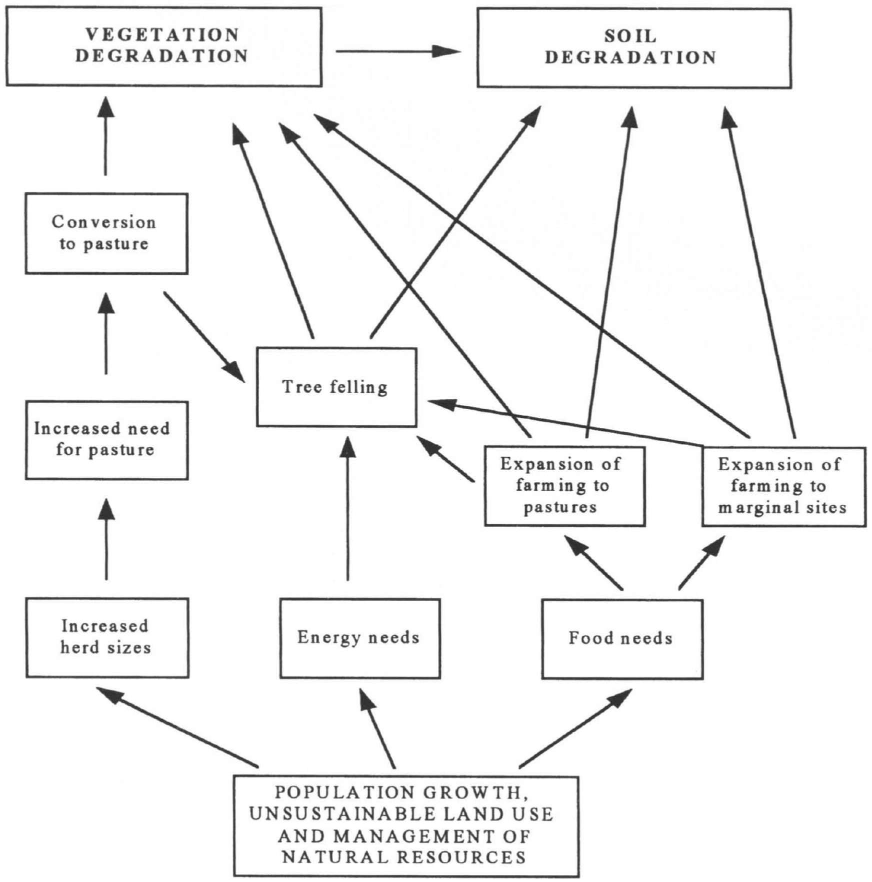

Fig. 1. A general problem analysis of land use in arid and semi-arid lands with respect to woody vegetation (modified from Esser-Winckler 1992 and Johansson 1995).

to the gradually advancing, dispersed and continuous process mainly taking place far from desert fringes (Grainger 1990). The International Convention to Combat Desertification (United Nations 1994) signed in 1994 defines desertification as "land degradation in arid, semi-arid and dry sub-humid areas resulting from various factors, including climatic variations and human activities".

There is a clear difference between this and the former definition by UNEP (1991) according to which desertification refers to "land degradation in arid, semi-arid and sub-humid areas resulting from adverse human impact". According to Dregne (1983), desertification is the impoverishment of terrestrial ecosystems under the impact of man. It is the process of deterioration in these ecosystems that can be measured by reduced productivity of desirable plants, undesirable alterations in the biomass and the diversity of the micro and macro fauna and flora, accelerated soil deterioration and increased hazards for human occupancy. According to Le Houérou (1995), it is the irreversibility of desertification, which separates it from land degradation. Several other definitions exist (cf. e.g. Ahlcrona 1988). Therefore it is not surprising that there has been considerable ambiguity in the meaning and use of the term desertification (e.g. Warren and Agnew 1988, Helldén 1991, Olsson 1991, 1993), to the extent that some authors (Ahlcrona 1988, Warren and Agnew 1988) prefer using the term degradation.

The threat of desertification, understood as irreversible collapse of the vegetation cover has been over-emphasised in the past. This was largely based on the misinterpretation of natural variation in rainfall and vegetation cover parameters obtained by aerial reconnaissance by Lamprey (1975) in the Sudan, which received much publicity and was hence widely quoted, but later strongly contested (cf. Warren and Agnew 1988, Olsson 1991). Now it is apparent, that there are not sufficient data to substantiate the hypothesis of a mainly anthropogenic trend towards desert-like conditions in the Sahel (Helldén 1991).

The resilience of the dryland ecosystems has often been underestimated or ignored in the desertification debate (Olsson 1993). Climatic variation explains most of the annual deviations from the mean annual production of major natural and rainfed agricultural production systems of the Sudan, without considering possible adverse impacts by man (Helldén 1991). The main problem is the degradation of vegetation around population centres rather than advancing dunes along forest edges. As Thomas and Middleton (1994) pointed out, "the advancing desert concept may have been useful as a publicity tool", but it does not represent the real nature of degradation. The problem is not less serious, but it requires different measures that are less spectacular than in combating direct desertification (Skarpe 1991). Nevertheless, the multidimensional natural and man-made dryland degradation poses a threat to biodiversity. The loss of biodiversity may, in turn, undermine the resilience in drylands thereby threatening the potential for recovery (Bie and Imevbore 1995).

Deforestation between 1981 and 1990 at pan-tropical level was estimated to have been 15.4 million ha a^{-1}, of which 2.2 million ha a^{-1} occurred in the dry and very dry zones (Singh 1993). Deforestation, as defined in the FAO (Food and Agricultural Organisation of the United Nations) Assessment of Forest Resources in Developing Countries (Singh 1993) refers to change of land use with the depletion of tree cover to less than 10 % of the land area. It does not include changes within the forest class, e.g. from closed to open forest, which negatively affect the stand and site and lower the production capacity. In the FAO assessment such changes were termed forest degradation that is not reflected in the above estimates (Singh 1993). The ratio between deforestation and reforestation and afforestation combined is 8.5:1 for all tropical areas and 45:1 for Africa (Singh 1993), but there is wide variation among and between countries.

1.3 Dynamics and Productivity of Natural Vegetation

1.3.1 Structure and Function

Dry woodlands and savannas are patchy ecosystems composed of mixed herbaceous, shrub and tree communities, with variation in the abundance and distribution of both species and community types across the landscape. Patchiness is related to climate seasonality (e.g. Walker and Menaut 1988), but can also result when savannas occur in ecological tension zones where trees are frequently under climate, water or nutrient stress, and where successful regeneration is irregular and infrequent (Coughenour and Ellis 1993). Plant establishment in dry areas often occurs in pulses that are related to high precipitation years (Noy-Meir 1979/1980) or a set of successive wet seasons (Agnew and Waterman 1989). Lamprey (1986) presented that, due to favourable conditions occurring very infrequently, the stands of woody species in ASAL are often even-aged and separated by intervals of 15–40 years, depending on the species and the recent climatic history of the region.

Dynamic changes in dry wooded ecosystems are also attributed to biotic and human disturbances, such as overgrazing (which creates opportunities for woody invaders); overbrowsing (which kills trees and shrubs and limits regrowth); seed dissemination by herbivores; and wood-cutting by people (Sinclair and Norton-Griffiths 1979, Lamprey 1983, Skarpe 1991).

The structure and function of semi-arid and arid savannas are governed by four major determinants: soil water, available nutrients, fire and herbivory (Skarpe 1991, Scholes and Walker 1993). The physiognomy of savannas is determined primarily by soil water and nutrient conditions (Walker 1987, Scholes and Walker 1993). With increasing aridity, the importance of moisture is superimposed (Penning de Vries and Djitéye 1982). Fires may affect population dynamics by killing seeds; promoting germination of seeds; damaging or killing seedlings; damaging adult plants; promoting growth of burned plants; and by affecting flowering, either by promoting or hindering it (Solbrig et al. 1992). Herbivory is an essential element in the savannas for maintaining the structure, either directly (physically) or by reducing the fuel load available to fires (Scholes and Walker 1993). Walker (1981) mentioned the high diversity and biomass of the large-mammal fauna, many of which are browsers (Bergström 1992), as the most distinguishing feature of the African savannas.

Trees affect the micro-climate under their canopies primarily by intercepting rain and solar radiation (Tiedemann and Klemmedson 1977, Belsky and Amundson 1992). Trees and herbaceous plants are not only competitors; trees also have positive impacts on the herbaceous biomass production and composition, and on soil nutrient status basically by shading the flora and soil which prevents evaporation and transpiration (Tiedemann and Klemmedson 1977, Belsky et al. 1989, 1993, Weltzin and Coughenour 1990, Belsky and Amundson 1992, Breman and Kessler 1995).

Trees and shrubs have also been found to improve the nutrient status of their close surroundings in other dryland ecosystems (Vetaas 1992b, Scholes and Walker 1993, Campbell et al. 1994). Grass biomass affects minimally the growth rate of mature trees, but it has a strong reducing effect both on the survival and growth of woody plant seedlings (Knoop and Walker 1985). The presence of a dense herbaceous layer thus exerts a strong control on the rate of recruitment of mature trees.

1.3.2 Regeneration

Successful establishment of a given tree species depends on a combination of factors that guarantee water availability to allow the seedling to grow roots beyond the superficial soil layers. These layers are very drought prone in all dry areas. The main factor favouring establishment is the occurrence of at least one, but preferably a series of relatively humid years, regarding both the amount of precipitation and the duration of the wet season (Agnew and Waterman 1989, Tybirk 1991a). According to Tybirk (1991a) the critical stages in the natural regeneration of tree legumes in the Sahel are seed dispersal, predation and germination as well as seedling growth. Trees often become established in depressions with some accumulation of water and organic material (Kennenni 1990, Gijsbers et al. 1994), emphasising the role of soil moisture as the main limiting factor in regeneration.

The rainy season is the period of germination of tree seeds, and thus of crucial importance for the overall tree and shrub dynamics (Lawesson 1990, Lieberman and Li 1992). The pattern of precipitation also affects the regeneration dynamics. Rains trigger the germination of many herbs that will perish if a long, dry period follows before the bulk of the rain falls, thereby allowing for later germinating herbs and woody saplings (Lawesson 1990). Lawesson (1990) mentioned *Boscia senegalensis*, *Balanites aegyptiaca*, *Guiera senegalensis* and *Combretum* spp. as examples of Sahelian woody species that could resist such a discontinuity of rains. Cissé (1986) concluded from his germination studies on herbs in Mali that species with soft seeds, quick germination, a short biological cycle and drought-tolerant seedlings would dominate during sequential dry years with irregular rains. During good rain years with regular and early rains, species with hard seeds, slow germination, long biological cycles, large size and less drought tolerance of seedlings would dominate. It is not known how the above strategies apply to woody species.

Browsing has both positive and negative effects on seed germination and plant regeneration. Most trees and shrubs in arid areas will tolerate recurrent heavy browsing and burning: their aerial parts may be removed annually, while their rootstocks continue to grow with each season's regrowth of twigs and foliage (Lamprey 1986). Ungulates digest *Acacia* seeds that pass through the alimentary system and end up in nutritious environment of the droppings. This may result in increased dispersal and germination (Bourlière and Hadley 1983, Pellew and Southgate 1984, Ahmed 1986, Coughenour and Detling 1986), but contradicting observations, in which the germination was not enhanced (Coe and Coe 1987), also exist. In addition, a high degree of destructive digestion has also been reported (Tybirk et al. 1994). Therefore the overall effect of browsing ungulates on *Acacia* seed survival remains uncertain (Miller 1994, Miller and Coe 1993).

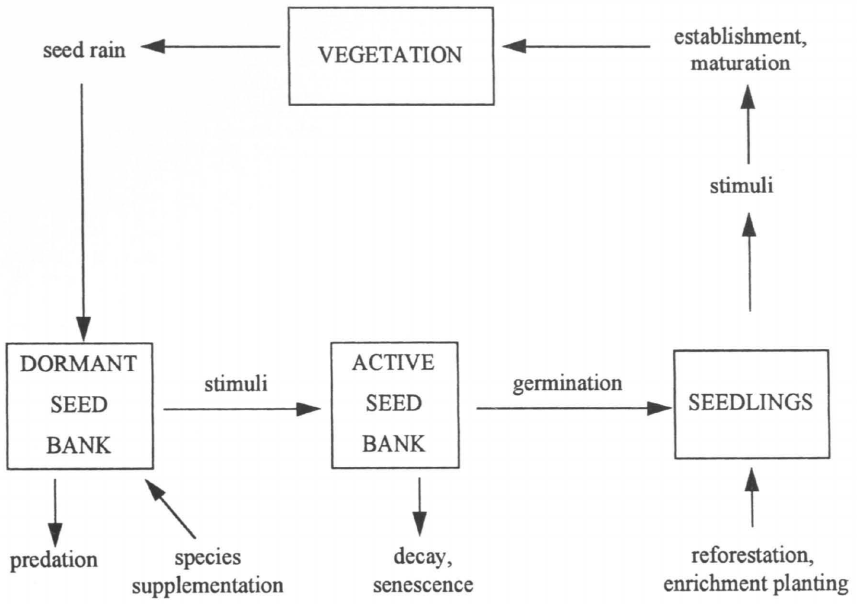

Fig. 2. Schematic representation of state variables and transfers in relation to soil seed bank (Adapted from Bell 1988).

A seed bank is formed by viable but dormant seeds in the soil. It is potentially useful in restoration attempts where establishment of a plant cover is targeted (Fig. 2). Recruitment from the seed bank is restricted to periods with favourable conditions of those soil parameters that may control seed germination (Skoglund 1992), particularly soil moisture (Reynal and Bazzaz 1973). Superficial layers of soil with an intact seed bank removed from nearby vegetated areas can be used to initiate plant succession locally (Skoglund 1992).

1.3.3 Primary Production and Browsing

There is a close relationship between average rainfall and the average primary production over large geographic areas (Le Houérou and Hoste 1977), specially in the dry savannas (Scholes and Walker 1993). Tropical savannas are characterised by relatively high levels of net primary production as compared with the standing biomass (Bourlière and Hadley 1983), specially when water and soil fertility are not limiting. Both periodic water shortage (Knoop and Walker 1985, Scholes and Walker 1993) and low soil nutrient content, particularly of nitrogen and phosphorus (Breman and de Wit 1983, Belsky et al. 1993), have been considered as major limiting factors for biomass production. In conditions of extreme aridity with severe degradation, the available vegetation becomes the main limitation (Breman and Kessler 1995).

In savannas, the efficiency of conversion, as expressed by the ratio of primary production to radiant energy, is low as compared to other terrestrial ecosystems, indicating that there are insufficient quantities of water for carbon assimilation to occur for most of the year (Scholes and Walker 1993). Fertilisation trials indicate that the availability of nitrogen and phosphorus may become limiting on rangelands, especially on sandy soils with good water absorption capacity and clay depressions (Breman and de Wit 1983).

Menaut et al. (1985) found a decreasing water stress and increasing nitrogen stress from arid to humid savanna in the Sahel. In the unimodal rainfall conditions of the Sahel the transition from growth in the herbaceous layer mainly determined by water to that determined by nutrients corresponds to an annual rainfall of 300 mm (Breman and Kessler 1995). There the importance of nutrients as the limiting factor is related to the low fertility of most West African soils (Breman and Kessler 1995); higher soil fertility increases the critical rainfall limit. Woody plant production is also related to rainfall, but not linearly, which is attributed to the precipitation of the previous season causing tree production to vary less than either rainfall of grass production (Scholes and Walker 1993).

Arid and semi-arid silvopastoral systems cover approximately a third of intertropical Africa and sustain about 60 % of the livestock population of the continent (Le Houérou 1987). Browse forage can be a significant part in the diet of the livestock and wild herbivores in these areas (Le Houérou 1980a, b, Coughenour et al. 1985, 1990, Mwalyosi 1990). The role of browse in the silvopastoral context is to provide stability and productivity of livestock production, which is the main source of livelihood and income in ASAL in Africa (Le Houérou 1987). Browsers can consume various parts of woody plants, such as leaves, twigs, thorns, bark, wood, bulbs, tubers, roots, flowers, seed pods, and fruit (Bergström 1992). The nutritive importance of browse was grossly overlooked in the past, although it provides an important and nutritious supplement for the energy-rich dry herbaceous forage, which is only available for a few months annually (Le Houérou 1989). Among the woody species, Capparidaceae, including genera such as *Maerua*, *Cadaba*, *Boscia* and *Capparis*, are particularly important due to their high nitrogen and mineral content (Le Houérou 1987).

Camels and goats are particularly dependent on browse species. Le Houérou (1980b, 1987) estimated that woody plants represent 10–20 % of the annual DM intake in the Sahel. The importance is superimposed due to the high nutritional quality, particularly that of the seeds, and during critical dry season (Le Houérou 1989). In the semiarid *Acacia-Commiphora* bushland (rainfall 800 mm) in Kenya *Acacia* pods contributed 37 and 30 % of the diet of goats and cattle respectively (Scholte 1992); protein-rich pods (of *A. senegal* and *A. tortilis*) and browse enabled especially goats to collect a more balanced diet. Leaf litter is important for herbivores in semi-arid bushland areas with the cover of trees and shrubs dominating over the herbaceous layer, such as in most *Acacia-Commiphora* bushlands of East Africa (Coughenour et al. 1985, Scholte 1992).

Knowledge on the utilisation of browse in African dryland ecosystems is limited. The effects of browsing on trees and shrubs have been studied mainly with acacias (Bergström 1992). *Acacia nigrescens, A. tortilis* (du Toit et al. 1990) and *A. xanthophloea* (Pellew 1984) exhibited a high degree of resilience against browsing. This was explained as positive selection for tolerant genotypes (*A. xanthophloea*) or by a fast growth of the browsed trees to compensate for the recurrent browsing (*Acacia nigrescens, A. tortilis*). Seif El Din and Obeid (1971) found the effect of simulated browsing on *A. senegal* to be age-dependent.

The total production of shrubs *Combretum aculeatum* and *Cadaba farinosa* decreased, whereas that of *Feretia apodanthera* increased following leaf stripping (Cissé 1980). Although some species can survive one browsing of the leaves, recurrent browsing during the same growing season will be lethal for the plant (Tybirk 1991a). Continuous browsing by an absolute browser (e.g. giraffe) may theoretically reduce the woody vegetation to the advantage of the herbaceous layer.

1.4 Management of Natural Vegetation

Natural forest management is a land-use system aiming at a sustainable production of forest products, based on the regeneration potential of the existing vegetation (Sabogal 1992). The state of knowledge and experience in management of natural forests and woodlands by means of systematic research are limited (Burley et al. 1989, Fries and Heermans 1992, Breman and Kessler 1995, Ffolliott et al. 1995). This is consequent from the fact that past efforts concentrated on plantation forestry to introduce fast-growing exotic species rather than managing existing forests and woodlands (e.g. Heermans and Minnick 1987, Tybirk 1991a, Shepherd 1992), to the extent that the indigenous "useless brush" was cleared and uprooted to give way for industrial plantations or village woodlots (Heermans and Minnick 1987, Fries and Heermans 1992).

Piearce (1993) noted that plantations may have been considered "more impressive" than natural regeneration. Jackson (1983), reviewing experience from the Sahel, found only one example of successful formal forest management up to 1980, namely that of *Acacia nilotica* in the Sudan. Similarly, Wormald (1984) in a review of semiarid east and Southeast Africa, pointed out the production of gum arabic from *Acacia senegal* in the Sudan as the sole representative of intensive natural forest management. Shepherd et al. (1993) assumed that most of the other related undertakings had failed, since they were not based on the indigenous technical knowledge of the local people.

There are also other factors emphasising the role of natural vegetation management. The yields of dryland vegetation may have been underestimated (Jackson 1983). The results obtained from tree planting in drylands have often not been satisfactory mainly due to inappropriate species selection, poor survival of the seedlings, high establishment cost, overestimated yield and insufficient involvement of the local people (Fries and Heermans 1992, Shepherd 1992). The negative attitude of the forestry administration has also been mentioned; the only approach towards natural forests in the past was that of complete protection (Fries and Heermans 1992). Other socio-political obstacles to the development of woody perennials and the participation of the people include the lack of settlement on land ownership, control and tenure (Heermans and Minnick 1987).

Ffolliott et al. (1995) defined the objective of (natural vegetation) management as to assure multiple uses of natural resources rather than to maximise returns on any one product and, in doing so, risk further degradation and destruction. The goal of all natural interventions is to assure a sustained yield of useful products from trees and shrubs produced in a manner that maintains the balance of the existing ecosystem (Breman and Kessler 1995). Another justification is that goods and services can be produced more effectively by retaining land under forest than by conversion to other land uses (Palmer and Synnott 1992). Grainger (1990) pointed out that one way of conserving valuable species may be by managing natural woodlands sustainably.

With increasing aridity, management should become more integrated, concerned with the whole space and placing more emphasis on the pastoral aspects. It becomes in practice a set of directives on exploitation and management (Baumer 1990). In managing natural vegetation in drylands it is important to consider the balance between the role of trees and forests as an ongoing producer of firewood and as a provider of food, fodder and other products upon which the people depend for survival and economic welfare (Heermans and Minnick 1987). Sabogal (1992) considered that management of natural vegetation enables production according to multiple-use objectives, combining also biodiversity conservation, environmental functions and social services. Shepherd (1992) pointed out that local woodland management would be more likely to succeed in relatively remote areas with a low population density and often with a low rainfall as compared to areas adjacent to population centres.

The methods for management of natural vegetation need not be intensive: management involves, in broad terms, any kind of activity in connection with the utilisation of the natural forests (Fries 1992). Mere protection can be regarded as a type of management when the main aim is the preservation of the existing vegetation (Fries 1992). Effective protection for a period of several years may be sufficient for recovery (Weber and Stoney 1986, Fries 1992); stands respond readily to protection against grazing and fire due to elasticity and good regenerative capacity of particularly the woody perennials (Baumer 1990). The protection of seedlings and saplings until they become established could be more effective than attempts to plant trees (Lamprey 1986, Oba 1994).

Assisted natural regeneration involves methods whereby the regeneration and early development of favoured indigenous species is improved. For example, interfering plants may be removed around young seedlings or additional seed could be sown in spots where successful regeneration may occur. In addition, methods to speed up the succession and to divert the vegetation development into a direction that is desirable for the intended land use include controlled burning, mulching, turning over turfs of soil, sowing and the use of viable seeds in the soil, i.e. the soil seed bank (Backeus 1992, Skoglund 1992).

1.5 Afforestation and Reforestation

1.5.1 Advantages and Disadvantages

Tropical forest plantations are being established at slightly more than 10 % of the rate of deforestation world-wide, and only about 2.5 % of that in Africa (Singh 1993). There are, however, notable exceptions to this gloomy trend. Significant increase in woody biomass has been achieved in Kenya primarily by tree growing besides agricultural fields, in grazing and marginal lands (Holmgren et al. 1994, KFMP 1994). The standing volume of planted trees increased annually by 4.7 % in 1986–92, and as the amount of natural vegetation was approximately constant and the decrease in closed-canopy forest biomass small in absolute terms, the wood balance remained positive (Holmgren et al. 1994).

Plantation forestry is a well-justified option for the otherwise unproductive woodlands or abandoned agricultural and pasture lands (Evans 1992), especially in areas with poor natural forest regeneration (Gladstone and Ledig 1990, Parrotta 1992). Plantations usually consist of fast growing, light-demanding tree species possessing valuable wood properties or other useful products (Lugo et al. 1993). Intensive silviculture based on fast-growing, multipurpose tree species, provenances and other genetically improved material offers an alternative to meet the needs for wood products at the farm, local and regional levels, and also contributes to alleviate the pressure on the remaining natural vegetation (e.g. Zobel et al. 1987, Evans 1992), but it cannot be widely applied in ASAL due to various ecological and socio-economic reasons (Lamprecht 1989, Tybirk 1991a, Shepherd 1992). While tropical lands respond to intensive management, such as site preparation, fertilisation, irrigation, the cost of such manipulations usually limits the extent of their applications (Evans 1992). The possibilities for site improvement also generally decrease with increasing distance from population centres due to funds and constraints in labour (von Maydell 1986).

Evans (1992) indicated the lack of satisfactory natural regeneration as one of the main obstacles for sustainable management of natural forests. Enrichment planting involves the planting of seedlings of the indigenous species raised in a nearby nursery. This is a mixed regeneration system (Evans 1992) combining reforestation and natural vegetation management. It can be applied to accelerate the recovery of a degraded site when natural regeneration is lacking or insufficient. Swaine (1992) noted that it is expensive in dry degraded forests and requires often supportive measures, such as application of mulch or irrigation. However, where woody species have disappeared, active artificial regeneration is necessary preferably with local species (Breman and Kessler 1995).

Success or failure of tree plantations depends on many factors, but technical constraints are usually less important than political, social and economic constraints (Poore et al. 1990, Persson 1995). The failures in tree planting in drylands have been caused by e.g. lack of clear objectives for planting, inappropriate species selection, lack of adequate protection, poor survival of the seedlings, high establishment cost, overestimated yield, unsettled land tenure and insufficient involvement of the local people (von Maydell 1986, Lamprecht 1989, Fries and Heermans 1992, Shepherd 1992, Persson 1995). Some form of water management is required, unless there is a permanent water source nearby or the site is seasonally flooded (e.g. Lamprecht 1989, Evans 1992). It is apparent that large intensive plantations established and managed by forest administration are inappropriate solutions to shortages of fuelwood and other tree-related products and services of wood in ASAL (Grainger 1990). Tybirk (1991a), referring to the Sahel, argued that the traditional method of planting legume species can rarely succeed on a long-term basis.

Barrow (1991) considered tree nurseries and planting as "cures for the ailment" rather than a means of preventing the problem (of dryland degradation), thereby undermining the wider issue of natural resource management. Tybirk (1991a) pointed out that the regeneration of trees is closely connected to the human management, and hence the re-establishment of the ecological balance by tree planting is not possible without adaptation in the living habits of the people concerned. The incorporation of management practices allowing for tree planting to fit in with the traditional livestock management of the pastoral people requires time and efficient extension. Grainger (1990) considered social forestry, tree planting by local people outside regular forest areas, as the only viable long-term strategy to meet afforestation targets and to cover the rising demand for fuelwood.

Barrow (1988) noted that there is a high susceptibility to change, and subsequently adoption of new practices, if there is an extensive degree of traditional knowledge and tangible benefits to be expected, such as in the utilisation and management of trees. On the other hand, as tree planting is neither perceived as important nor does it provide useful products or services in the short term, it is not easily adopted. Shepherd et al. (1993) noted that rural people will engage in tree planting only on land for which they have permanent tenure, using trees capable of providing cash crops, and only when land is short in supply to the extent that there are no alternative sources, e.g. government-owned forests from which the products could be collected.

1.5.2 Species Selection

Few local species have so far been planted in pure or dense stands, while plantations with exotic species often occupy favourable sites and are managed intensively and with high inputs (von Maydell 1986). Indigenous species were earlier often omitted from the plantation programmes and experimenting, until in the late 1970's when FAO, followed by the national and international organisations concerned with forestry changed some of their principles for selecting species for dry areas, emphasising the indigenous species and the multipurpose role of trees. Since then several compendiums and lists have been compiled (e.g. Goodin and Northington 1979, National Academy of Sciences 1980, Le Houérou 1980a, Wickens et al. 1985, Felker 1986, von Maydell 1986, Rocheleau et al. 1988, FAO 1995) on dryland tree and shrub species and their uses. This has gradually geared the emphasis from a few multipurpose species, assumed to cover most of the needs of the rural people, to a wider selection of mainly indigenous ones (Hughes 1994). The key criteria of species selection for planting are presented in Fig. 3.

Several factors have earlier favoured exotic over indigenous species. Inadequate knowledge of local trees and shrubs often leads to the selection of exotic species (Zobel et al. 1987, Ffolliott et al. 1995). There is a wide choice of species, prove-

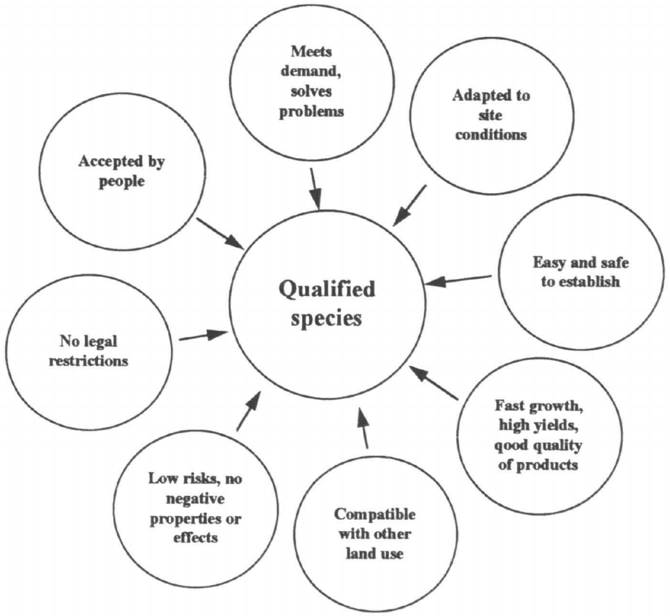

Fig. 3. Relevant criteria for the selection of plantation species (adapted from von Maydell 1986).

nances and other genetically improved material among the exotics (Zobel et al. 1987). Their plantation management regimes are well-known, seeds sources are well-documented and productivity is higher (Evans 1992), albeit often overestimated (Breman and Kessler 1995, Persson 1995) and difficult to sustain without compensating for the high uptake of limited nutrients (Breman and Kessler 1995).

Seed availability, problems of storage, and pretreatment requirements, in turn, regularly limit the use of indigenous species (Piearce 1993). Lugo (1988) claims that exotics generally succeed in human-created environments, since these environments usually include new conditions to which indigenous species adapt poorly. Exotic tree plantations can also be considered as foster ecosystems for endangered species (Lugo 1988, Lugo et al. 1993), and for the rehabilitation of highly damaged sites (Zobel et al. 1987, Lugo et al. 1993), which by definition require human intervention if their productivity is to be restored (Lugo 1992).

Arguments favouring indigenous species have been summarised by von Maydell (1986) as follows: (1) they involve less risk from the ecological point of view; (2) they are known and traditionally accepted by the local people; (3) they have a proven ability to grow and yield products in the prevailing conditions; (4) seeds are readily available; and (5) their use enables the full use of their potential yet insufficiently known or recorded.

The good performance of exotics in tree planting programmes has partially been attributed to their resistance to pests, diseases and herbivorous predators (Hughes and Styles 1987). This advantage may be temporary, since the pests and diseases are likely to reach the areas of introduction within time, as has been demonstrated by the leucaena psyllid (*Heteropsylla cubana*), which seriously affects the planting of *Leucaena* spp. first in Asia and gradually also elsewhere. The insect spread from Hawaii through Asia to Kenya between 1984 and 1992 (van der Beldt and Napompeth 1992). Risk for similar incidents remain, as many successful exotics result from casual introduction with narrow genetic bases and geographic locations (Burley et al. 1986).

Altieri and Merrick (1987) paralleled the promotion of exotic agroforestry trees over indigenous alternatives to the loss of traditional crop varieties following promotion of green revolution varieties. Differences which exist within the natural range of a species usually become more evident and important, when a species is grown as exotic (Zobel et al. 1987).

Critical reviews by Hughes and Styles (1987) and Hughes (1994), highlighting the invasiveness involved in many species introductions, particularly in open forest types, savannas, fire-dominated ecosystems and many semi-natural habitats, have also emphasised the role of local species that have the advantage of not turning into weeds. However, in most cases species selection is not a matter between a native or an exotic, but rather, as Le Houérou (1987) pointed out, choosing the best adapted and most productive under the circumstances and for the objective that is being pursued. Armitage (1985) warns against the uncritical use of a species that is successful under one set of conditions in areas to which it is less well adapted. In addition, narrowing interest on individual species can misdirect the concern from forest or woodland ecosystems as a whole (Piearce 1993).

1.6 Rainwater Harvesting in Dryland Rehabilitation

In ASAL the limited availability of water is usually the major constraint to plant growth (see above). In arid regions the rainfall is insufficient, whereas in semi-arid regions the uneven distribution of rainfall turns rainfed agriculture and tree planting risky (Reij et al. 1988). Part of the rainfall may also be lost from the plant root zone as surface runoff, particularly under high rainfall intensities and poor infiltration properties of the soil. In conditions, where total seasonal rainfall is insufficient for plant growth, the focus on management practices is to increase the amount of surface runoff towards and into the plant-soil system (Reij et al. 1988).

Water harvesting encompasses a range of methods of inducing, collecting and concentrating various forms of runoff (Boers and Ben-Asher 1982). Rainwater harvesting usually refers to methods entirely dependent on local rainfall and runoff and is thus differentiated from the irrigation systems, including the tapping of groundwater (Reij et al. 1988). Many farming communities used rainwater harvesting traditionally to provide water seasonally or, with storage, over longer periods to grow agricultural crops and trees for fruit, amenity and other purposes (Armitage 1985) especially in arid and semiarid areas, where runoff has an intermittent character (Boers and Ben-Asher 1982). A notable example is from the Negev desert in Israel where rainwater harvesting provided a livelihood for a substantial population more than 2000 years ago, continuing until about 700 A.D. (Pacey and Cullis 1986).

Runoff capacity may be improved either physically, by removing the rocks and vegetation or compacting the surface, or both; chemically, by using sodium salts, silicones, latex or wax; or by using waterproof covers, rubber, plastic or light metal (Armitage 1985). Due to the ephemerality of surface and sub-surface water flow, storage is an integral element of water harvesting (Myers 1967). The advantages of the systems are low energy and capital investments and simple maintenance (Zohar et al. 1988). Total dependency on rainfall, on the other hand, results in low reliability of the systems (National Academy of Sciences 1974, Armitage 1985).

In addition, for water harvesting to be effective, the water retention capacity of the soil profile needs to be high to allow the storage of the incidentally high amounts of harvested water. Therefore water harvesting is often practised in combination with measures to increase infiltration, suppress evaporation and prevent runoff (Reij et al. 1988). Useful criteria for assessing the potential of water harvesting are the frequency of individual precipitation events and the probability of a certain minimum amount and intensity rather than the total amount of rainfall (Kutsch 1983).

Water harvesting has received considerable attention in increasing the yield of agricultural and horticultural crops, particularly in combination with other moisture conservation practices, such as mulching and tillage (e.g. Pacey and Cullis 1986, Gupta 1994), but knowledge e.g. on traditional soil and water conservation in Sub-Saharan Africa is limited (Reij et al. 1988). Reij et al. (1988) indicated that there is very limited evidence to support the assumption that the use of WH in semi-arid sub-Saharan Africa leads to significant and sustained yield increases in crop production, although some examples of improved production and reduced soil degradation also exist (Rapp and Håsteen-Dahlin 1990). In countries with an adverse climate and limited water resources, such as Kenya, rainwater harvesting needs to be promoted to supplement the existing water supplies, especially for agricultural and livestock needs (Bambrah 1994).

Armitage (1985) reported that water harvesting has been tried in areas with an annual rainfall of 25–80 mm in various countries in Africa and Asia. Pacey and Cullis (1986) set a minimum requirement for water harvesting of 100 mm of average annual rainfall in winter rainfall areas such as the Negev desert, but 200 mm would be more viable. The commonly used applications include contour ridges with furrows, semicircular or trapezoidal bunds and microcatchments. Microcatchment water harvesting (MCWH) – collection of runoff from the immediate vicinity of the seedling and stored in the rooting zone beneath it – is perhaps the most widely used method in tree planting.

A microcatchment usually consists of two elements: the runoff or contributing area and the infiltration basin (Boers and Ben-Asher 1982). Wairagu (1994) noted, as the advantages of microcatchments, the ease of installation, compatibility with most land tenure systems and the low level of damage due to failure as opposed to large engineering works. The ratio between the two areas is a key parameter in microcatchment design depending on climate, soil conditions and crop water requirement (Boers and Ben-Asher 1982). Boers (1994) listed the three most important factors in designing water harvesting methods in tree planting in ASAL as: (1) seasonal distribution of rainfall; (2) soil hydraulic properties; and (3) tree hydrological characteristics.

The main use of MCWH in tree planting is to help in the critical stage of tree establishment, i.e. from the time of planting to when the tree is well established. Heermans and Minnick (1987) added that MCWH permits the introduction of species that would normally be planted in higher rainfall zones. Examples of the designs are e.g. the square or diamond-shaped "Negarim" and semicircular microcatchments, "demi-lunes"; the former are used in Israel for fruit tree (Ben-Asher and Warrick 1987) and the latter for forest and fodder tree production in West Africa (Reij et al. 1988) respectively.

Conditions where trees are active throughout the year, facilitating the taking up of water as soon as rainfall and runoff are stored in the root zone, are better suited for MCWH than those that experience an inactive period, since the latter require a larger storage capacity (Boers 1994).

In Israel the construction of microcatchments is incorporated in attempts to establish small-scale patches of trees in the desert (Tenbergen et al. 1994). Boers et al. (1986) reported MCWH suitable for areas of 250 mm of rainfall and loess soils that form a surface crust. Zohar et al. (1988) reported successful growing of introduced trees for fuelwood utilising ‘limans’ (water collection ponds) with annual rainfall of 115 mm in Israel. A financial cost-benefit analysis of the MCWH technique by Oron et al. (1983) revealed that the predicted income is negative with annual rainfall below 150 mm, but by introducing inserts (perforated vertical drainage pipes) the net income increased and (as compared to the option of not using them), turned positive in the above 250 mm rainfall zone (Oron et al. 1983).

Until recently, water harvesting methods have not been widely applied to trees, with the exception of microcatchments on short slopes (Reij et al. 1988). Furthermore, the knowledge on the performance of tree species planted under MCWH is very limited.

1.7 Aim of the Study

The principal aim of the present study was to develop the methods for sustainable management of the woody vegetation in arid and semi-arid conditions in Kenya. This was studied by focusing on the biomass production and regeneration of trees and shrubs in natural (non-disturbed) and cleared, but naturally regenerated (disturbed) *Acacia-Commiphora* bushland. The biomass production of natural and disturbed bushland was compared to that obtained under a heavily manipulated bushland, i.e. using planted seedlings under various microcatchment water harvesting designs. In addition, the indigenous and exotic species were compared during establishment and early growth. Three hypotheses were derived:

(1) Natural regeneration is sufficient for rehabilitation of degraded bushland.
(2) The soil seed bank can be utilised to improve the regeneration of woody species.
(3) Microcatchment water harvesting improves the establishment and early growth of planted tree seedlings, but species respond differently to this treatment.

The present study, which was part of a larger investigation on developing sustainable tree production systems on arid, semi-arid and degraded tropical lands, is based on the material compiled during the assignment of the author to the research component of the FinnIDA-funded (Finnish International Development Agency) Bura Fuelwood Project (BFP) in eastern Kenya. The practical aim of the research was to provide directly applicable information for BFP implementation.

2 Materials and Methods

2.1 The Study Area

The study area falls within the Bura Irrigation Settlement Project (BISP), which is situated just south of the equator at the west bank of Tana river at a latitude of 1°08’S and a longitude of 39°45’E and at an elevation of 80–110 m above sea level in Tana River District, Coast Province of Kenya (Fig. 4). The Tana is Kenya’s largest river with a total catchment area of 94 700 km^2 (Ministry of Water... 1978), and the only one that can support large-scale irrigation development.

2.2 Climate

The rainfall pattern in the study area, like in most parts of Kenya occurring East of the Rift Valley, is controlled by the North-South movements of the intertropical convergence zone. This zone is known for the year to year variation in rainfall amounts in addition to the wide variation in the time of year at which the rain falls (Griffiths 1972). Rainfall is bimodal, averaging 372 mm annually in Bura for 1983–1992. The annual minimum and maximum in the same period were 177 and 610 mm respectively. The rainfall was measured with a standard raingauge located at the Bura meteorological station (see Fig. 7).

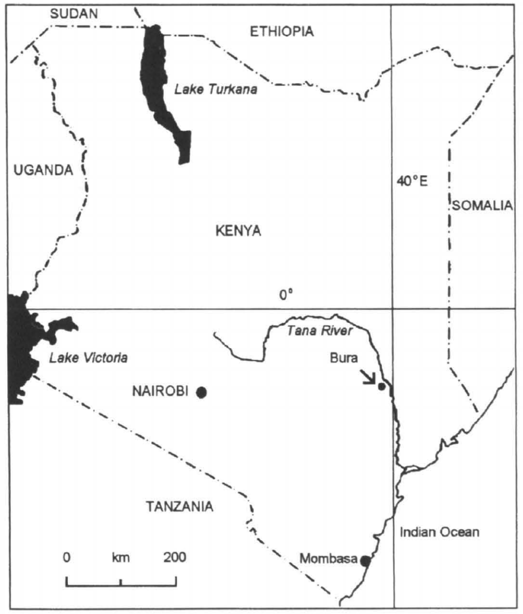

Fig. 4. Map of Kenya showing the location of the Bura research area (arrow).

Rain events can be very intense, and a high proportion of the monthly and annual rainfall can occur in a few occasions. The rainy seasons roughly correspond to equinoxes; the “long rains” come during a period from March to May, and the “short rains” fall from October to December (Muchena 1987). Mean, minimum and maximum annual potential evaporation (class A pan) for the same period were 2336, 1773 and 2798 mm (A. Otsamo et al. 1993) respectively. The mean annual moisture deficit for 1983–1992 was 1964 mm, with a range of 1162–2639 mm. Low rainfall and high potential evaporation together imply that the vegetation in the area is subjected to a considerable moisture deficit for most of the year. The rainfall and evaporation data are presented in Figs. 5–6.

The temperatures in the study area were high throughout the experimental period with little annual and seasonal variation. The absolute maximum and minimum for 1983–1992 were 39.5° and 17.5° respectively, while the mean monthly maximum and minimum were 33.4° and 22.5° for the same period. Mean annual duration of daily sunshine was approximately 7 hours with relatively little seasonal variation. Relative air humidity, which was also fairly constant, averaged from 72 % at 9.00 a.m. and to 47 % at 3.00 p.m. (A. Otsamo et al. 1993).

On the agro-climatic zone map of Kenya (Sombroek et al. 1982), Bura is located on the border between the zones VI and VII. The classification

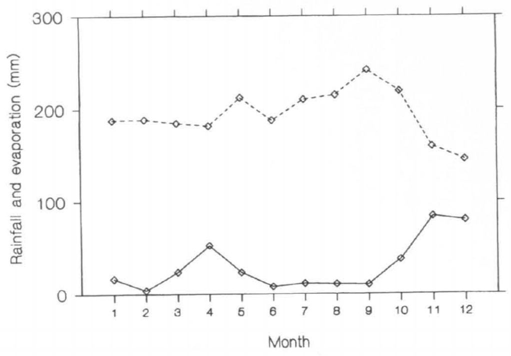

Fig. 5. Mean monthly rainfall (solid line) and potential evaporation (class A pan; dashed line) in Bura 1983–1994.

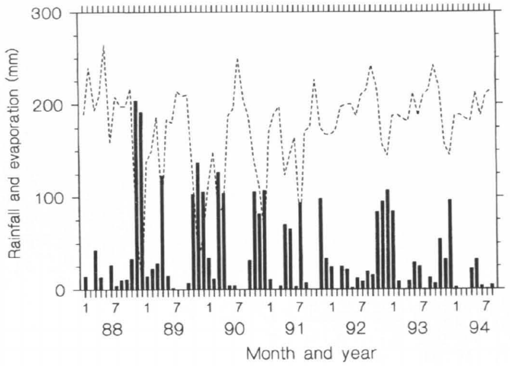

Fig. 6. Monthly rainfall (bars) and evaporation (class A pan; dashed line) in Bura in 1988–1994.

depends on ratio of precipitation (R) and potential evaporation, E_0. At Bura this ratio was 0.16 for 1983–1992 (A. Otsamo et al. 1993), which would mean that it narrowly falls within zone VI. From the agricultural potential point of view, zones VI and VII are suitable for nomadic pastoralism only (Sombroek et al. 1982).

2.3 Soils

Geologically the study area forms part of the sedimentary basins of the East Kenya. It falls under alluvial fan plains, where the bay sediments were remodelled under fluviatile conditions (Muchena 1987). Physically the area can be divided into two parts: (1) a lower terrace bordering the Tana river, covered by young alluvial soils, "the flood plain" and (2) a higher terrace, further west from the river, covered by old alluvial soils, "old alluvial terrace" (ILACO 1977). A slight escarpment may occasionally be found on the boundary between these two terraces. The old alluvial terrace is a gently sloping plain with a dominant slope running from west to east, towards the river. Its gradient varies from 1–3 %. The gradient of the overall north-south slope that runs parallel to the course of the Tana river amounts to 0.6 % only (ILACO 1977).

The soil types for the experimental sites were derived from the soil survey undertaken by ILACO (1977). This survey, which also incorporated the results of an earlier survey by ACRES-ILACO (1967a) was primarily conducted to assess the irrigation suitability of the soils. The Great Group Halorthids and the Subgroup Natric do not exist in the US Soil Taxonomy, but were introduced by ACRES-ILACO (1967a), the former to cater for soils which were saline, but not enough to qualify them as having a saline horizon, and the latter to describe soils which had non-saline but sodic horizons.

Typic Natrargids (TN) are characterised by a 10–20 cm thick, natric horizon which is covered by a coarse-textured, lime-free non-saline/non-alkaline layer, which is mostly 30–60 cm thick. The natric horizon has a sandy clay loam texture and contains practically no lime. Below this there is usually a lime-containing, saline/alkaline, sandy clay loam to sandy clay layer. The vertical distribution of both roots and water may be prevented below these two layers. These soils form slightly elevated areas in the landscape (ILACO 1977).

Vertic Natrargids (VN) exhibit some vertisol tendencies, have a rather high clay content, some weak cracks and more or less prismatic structure. A natric horizon is often found within 20 cm of the surface. The subsoil is saline/alkaline and the clay content varies between 35 and 65 %. The topsoil contains more sand and the textural classes range between sandy clay loam and clay (ILACO 1977). Vertic Natrargids are found between the higher-lying Typic Natrargids and the lower-lying Natric Grumusterts (NG), often in a close association with the latter type.

Typic Camborthids (TC) are deep and porous, and within 50 cm of the soil surface non-saline/non-alkaline. Textures are mostly medium, but become heavier (sandy clay and clay) at greater depths. Typic Halorthids (TH) occur between the higher-lying Natrargids and the lower-lying Grumusterts. They contain a non-saline/non-alkaline top-layer with a sandy clay texture. Lime is found throughout the profile and salt crystals occur occasionally at a depth of 50 cm below soil surface. Natric Grumusterts form part of the heavy clays that expand and contract markedly depending on the moisture content. They also have a non-saline/non-alkaline topsoil layer. As compared to the other soil units, the silt content is rather high, 10–20 %. During the dry season part of the surface soil disappears in the cracks (ACRES-ILACO 1967a).

2.4 Vegetation

The overall natural vegetation in the study area is very sparse, except on a narrow belt on the floodplain along the Tana River, where mainly evergreen forest with closed canopy can be supported (Gachathi et al. 1995). The extent of the forest depends on the water table which subsides rapidly as one moves further from the river (Hughes 1988, 1990). The width of the riverine forest extends approximately 1–3 km on either side of the river (Gachathi et al. 1995). Away from the floodplain the vegetation consists of drought-deciduous thorny bushland or wooded grassland, which according to Pratt et al. (1966) can be classified as *Acacia-Commiphora* bushland. This vegetation type (dry *Acacia-Commiphora* bushland and thicket) is one of the major ones in Kenya covering 36 % of the land area (Millington et al. 1994).

The basic distinction in the functional determinants of the *Acacia-Commiphora* bushland of eastern Kenya, as compared to the savannas of eastern and southern Africa, is the infrequency of fire (Pratt et al. 1966), although burning occurs occasionally in areas S of Bura (ACRES-ILACO 1967b). In addition, grasses are represented only by few annual and short-lived perennial species thereby contributing little to the phytomass in many places.

Several vegetation classifications schemes have been presented covering either the whole of Bura area (ACRES-ILACO 1967b, FAO 1973) or only some vegetation types (Andrews et al. 1975, Marsh 1976, Homewood 1978, Allaway 1979, Hughes 1985), particularly the evergreen riverine forest. Marsh (1976), using the criteria of Pratt et al. (1966), divided the plains outside the floodplain in the Tana River Primate Reserve into four categories; (1) woodland; (2) bushland; (3) bushed grassland; and (4) dry plains grassland. The classification of ACRES-ILACO (1967b), applying the work of Greenway (1973), distinguished 10 vegetation associations with three sub-types (Table 1).

Johansson (1995) delineated the vegetation into three main types for forest management purposes: (1) riverine forest; (2) transitional zone between the riverine forest and the dry bushland; and (3) dry bushland, including the ephemeral streams. This division had already been applied by Gachathi et al. (1987, 1995) and Gachathi (1993) when describing the vegetation, and by Pukkala (1989, 1992), for a classification during

Table 1. Classification of the vegetation of the Bura-Galole area (ACRES-ILACO 1967b).

No.	Description of the association
1	Semidesert scrub
2	Deciduous scrubland
3	Shrub thicket
4	Open bushland dominated by *Cordia* sp. and *Acacia reficiens*
4a	*Acacia* bushland, usually *Acacia paolii* and *Acacia reficiens*
4b	Middle terrace grassland, dominated by grass and herb species identical to those of association 4
4c	Closed bushland, dominated by *Acacia reficiens*, *Commiphora* spp., *Euphorbia grandicornis*
1/4	Mosaic of associations 1 and 4, often dominated by *Commiphora* spp
5	Thicket dominated by *Terminalia orbicularis*
6	Shrub grassland
7	Valley grassland, dominated by *Sporobolus helvolus*
8	Floodplain, dominated by nongraminaceous herbs
9	Floodplain bushland
10	Tana riverine forest

the inventory of the natural vegetation in the Bura area. The transitional zone is situated on a strip between the riverine forest and the bushland. The flora is composed of species from both zones, such as *Acacia tortilis*, *Dobera loranthifolia*, *Lawsonia inermis*, *Grewia plagiophylla* and *Terminalia brevipes*.

The general feature of the dry bushland is of predominantly deciduous thorny bushes and shrubs usually branching from near the ground with few well-defined trunks and a patchy ground cover consisting of tufted grass or the salt-resistant shrub of *Salsola dendroides*. The vegetation is in drought-dormant condition for most of the year, but leaves appear soon before or after the onset of the rainy season (Gachathi 1993, Gachathi et al. 1995). Characteristic shrubs include *Acacia reficiens*, *A. bussei*, *A. mellifera*, *Cadaba glandulosa*, *Commiphora incisa*, *Commiphora campestris* and *Salsola dendroides*. Scattered deciduous trees are represented by *Acacia tortilis*, *A. zanzibarica*, *Cordia sinensis*, *Euphorbia robecchii*, *Salvadora persica* and *Dobera glabra* (Gachathi 1993).

White (1983) described the Somalia-Masai *Acacia-Commiphora* deciduous bushland and thicket as dense, 3–5 m tall, with scattered emergent trees up to 9 m. It can locally form impenetrable thickets. The dominant *Acacia* spp. and some of the *Commiphora* spp. are spinous. Most species are deciduous; notable exceptions are evergreen *Dobera glabra* and *Salvadora persica*. In many places grasses contribute little to the phytomass and are represented by a few annual and short-lived perennial species.

Gachathi et al. (1995) prepared a check-list of 228 primarily woody species by their scientific and local names covering all the three vegetation zones defined by Johansson (1995). This check-list was used in the identification of species in the present study. Nomenclature follows Beentje (1994); for genera not treated there, Boland et al. (1984) and von Maydell (1986) have been used.

2.5 Experimental Design

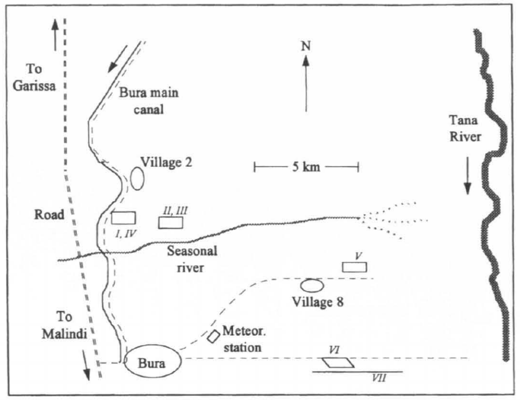

Fig. 7. Map of the Bura area showing the location of the experiments (*I–VII*).

The study consisted of eight separate experiments, which are referred to by their roman numerals (I–VIII). Experiments I–V dealt with water harvesting and Experiment VIII with the soil seed bank. Experiment VI concentrated on cleared and VII on undisturbed natural *Acacia-Commiphora* bushland. The location of the experiments in the Bura area are presented in Fig. 7.

2.5.1 Microcatchment Water Harvesting

The seed was sown in germination beds after which the seedlings were transplanted into polythene tubes and raised in the nursery approximately six months before outplanting. This was applied to all experiments involving planting apart from the sowing and natural regeneration treatments in Experiment V. The soil mixture in the tubes contained compost manure and soil from Tana riverine forest in an approximate ratio of 5 : 3. All the experiments were established immediately after the onset of the rainy season. The species were mostly indigenous. Two seed sources of *Prosopis juliflora* (Bura and Mombasa) were included in the experiments as general controls, because they were widely planted in Bura area. The experiments were guarded throughout the experimental period to avoid browsing and other damage by animals. Summary of all the species included in rainwater harvesting experiments are displayed in Table 5 (p. 27).

Table 2. Treatments in long-furrow microcatchments (Experiment I).

No.	Species	Seed source[1]
1	*Tamarindus indica* L.	Katilu, Lodwar
2	*Acacia mellifera* (Vahl) Benth.	Kibwezi
3	*Prosopis juliflora* (Swartz) DC	Mombasa
4	*Acacia nubica* Benth.	Bura
5	*Commiphora pseudopaolii* Gillett	Bura
6	*Prosopis juliflora* (Swartz) DC	Bura
7	*Cordia sinensis* Lam.	Turkana
8	*Acacia reficiens* Wawra	Bura
9	*Garcinia livingstonei* T. Anders	Bura
10	*Acacia tortilis* (Forssk.) Hayne	Katilu, Lodwar
11	*Azadirachta indica* A. Juss.	Bura
12	*Acacia xanthophloea* Benth.	Kibwezi
13	*Faidherbia albida* (Del.) A. Chev.	Katilu, Lodwar
14	*Berchemia discolor* (Klotzsch) Hemsl.	Bura
15	*Acacia zanzibarica* (S. Moore) Taub.	Bura
16	*Lawsonia inermis* L.	Turkana
17	*Acacia aneura* F. Muell. ex. Benth.	SETROPA, Holland[2]
18	*Parkinsonia anacantha* Brenan	Bura

[1] In Kenya unless otherwise indicated
[2] Seed supplier, seed origin unknown

Long-Furrow Microcatchments

Experiment I was set up in 0.3 m deep and 0.6 m wide long, straight furrows prepared by a bulldozer-driven plough on a sandy clay soil (TH). The soil preparation followed the standard design applied in rainfed afforestation in the area until 1987. This method was treated as the control to which the other micro-catchment designs were compared. The seedlings were planted in the bottom of the furrow (Fig. 8). A plot consisted of ten trees with an in-row spacing of 3 m, and the distance between furrows was 5 m. Thus the total theoretical spacing was 666 ha^{-1}. The design was in randomised complete blocks (RCB) in five replications incorporating 18 species (Table 2). Apart from *Prosopis juliflora*, *Acacia aneura* and *A. xanthophloea* all other species are indigenous to Bura area. The seedlings were planted in November 1988. Survival from the whole plot and stem height of eight sample trees were assessed in May 1989 at the age of six months. Total and effective rainfall, estimated as that exceeding 30 mm month^{-1} for the 12-month experimental period was 709 mm and 502 mm a^{-1}. The total amount is almost twice the annual average of the area in 1983–1992.

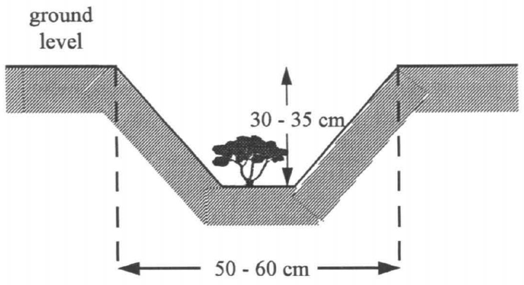

Fig. 8. Schematic side view of a long-furrow microcatchment (Experiment I, not to scale).

Deep Triangular Microcatchments

Experiment II was laid out in microcatchments modified from those presented by Boers et al. (1986). The front blade of a bulldozer was used to form the two edges of the catchment of 45° across the contour. The other edge was prepared at an angle of 45° across the contour and perpendicularly to the first edge to form a triangle. The length of both edges was approximately 5 m and the catchment descends 50–70 cm towards the lower end, i.e. towards the tip of the triangle, each catchment providing water for a single tree (Fig. 9). The seedlings were planted close to the deepest point of the catchment in November 1988. Several planting spots were re-established three weeks after the initial planting due to the breaching of the catchment edges and subsequent burial of the seedlings. The spacing was 10 · 10 m (100 ha^{-1}) in 12-tree plots (Fig. 10) and the design was in randomised complete blocks incorporating 12 species (Table 3) in four replications.

The trial was assessed 18, 36, 48 and 70 months after planting. Height was measured from the tallest stem and stem diameters over bark were measured at stump height ($d_{0.15}$). Stem numbers ($n_{0.15}$), including all major stems of an individual tree, were counted at the corresponding height. All

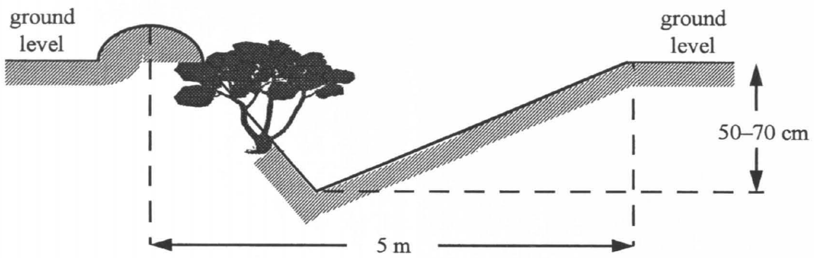

Fig. 9. Schematic side view of a deep triangular microcatchment (Experiment II, not to scale).

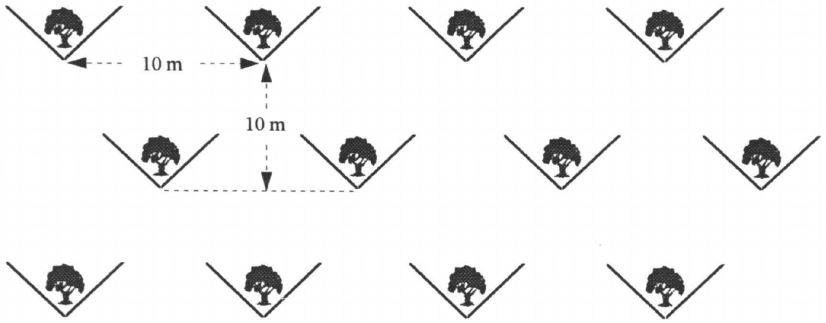

Fig. 10. Schematic aerial view of deep triangular microcatchments (Experiment II, not to scale).

stems were measured separately in multiple stemmed trees. The diameters of multi-stemmed trees were converted into basal area ($g_{0.15}$) and pooled diameter ($dg_{0.15}$); this was required for statistical tests and the calculation of total biomass. All trees in a plot were included in the calculation of survival, height, diameter and volume. Total rainfall during the experimental period was 2779 mm equalling to an annual mean of 476 mm a^{-1}. The effective rainfall (above 30 mm month^{-1}), for the same period was 1622 mm or 278 mm a^{-1}.

Shallow Triangular Microcatchments

Experiment III was similar to II except that the depth of the catchment was approximately 30–40 cm, the length of the edges was 4 m (Fig. 11), and the plots consisted of eight trees only. *Acacia nilotica* substituted *Lawsonia inermis* in Experiment III (Table 3). The soils in Experiments II and III were heavy clays (Natric Grumusterts). The trial was established in November 1989 and assessed 13, 24, 36 and 59 months after planting using the same parameters as in Experiment II. Total and mean annual rainfall for the experimental period were 2069 mm and 420 mm a^{-1}. The rainfall exceeding 30 mm month^{-1} totalled 1119 mm and averaged 228 mm a^{-1}.

Cross-Tied Furrow Microcatchments

Experiment IV was laid out in a split-plot design on a sandy clay soil (TH). The furrows were dug with a land preparation plough pulled by a bulldozer approximately along the contours 0.35 m deep. The furrows were blocked and rectangular catchments were manually separated with cross ties in 10 m intervals. There were five seedlings in a plot spaced 4 m on alternate side of the furrow, planted on the upper part of the furrow close to the edge of the shoulder (Fig. 12). The uphill ridge of the furrow was partly opened to

Table 3. Treatments in deep and shallow triangular microcatchments (Experiments II and III).

No.	Species	Acronym	Seed source[1]
1	*Acacia horrida* (L.) Willd.	Aho	Bura
2	*Acacia mellifera* (Vahl) Benth.	Ame	Kibwezi
3	*Acacia nubica* Benth.	Anu	Bura
4	*Acacia tortilis* (Forssk.) Hayne	Ato	Katilu, Lodwar
5	*Acacia xanthophloea* Benth.	Axa	Kibwezi
6	*Acacia zanzibarica* (S. Moore) Taub.	Aza	Bura
7	*Commiphora pseudopaolii* Gillett.	Cpa	Bura
8	*Cordia sinensis* Lam.	Csi	Hola
9	*Lawsonia inermis* L. / *Acacia nilotica* (L.) Del.	Lin / Ani	Turkana / Katilu, Lodwar
10	*Parkinsonia anacantha* Brenan	Pan	Bura
11	*Prosopis juliflora* (Swartz) DC	Pjm	Mombasa
12	*Prosopis juliflora* (Swartz) DC	Pjb	Bura

[1] All in Kenya

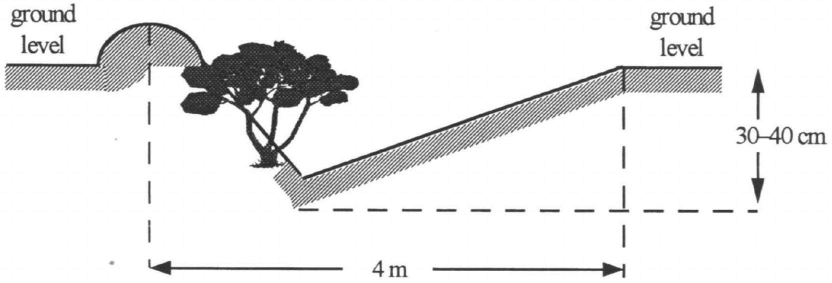

Fig. 11. Schematic side view of shallow triangular microcatchments (Experiment III, not to scale).

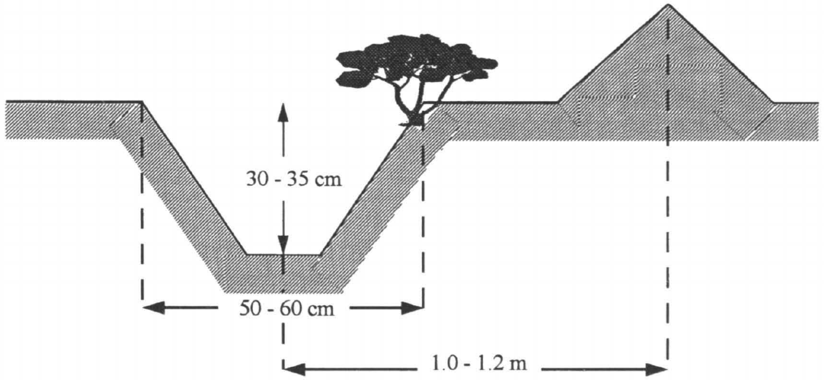

Fig. 12. Schematic side view of the furrow in rectangular, cross-tied furrow microcatchments (Experiment IV, not to scale).

Table 4. Treatments in cross-tied furrow microcatchments (Experiment V). Ratio represents the catchment to cultivated area ratio.

Main plot treatment			Sub-plot treatment			
Catchment size / m^2	Spacing n ha^{-1}	Ratio	No.	Species	Acronym	Seed source[1]
30	1667	3 : 1	1	*Prosopis juliflora* (Swartz) DC	Pjb	Bura
40	1250	4 : 1	2	*Acacia tortilis* (Forssk.) Hayne	Ato	Kitui
60	833	6 : 1	3	*Acacia zanzibarica* (S. Moore) Taub.	Aza	Bura
80	625	8 : 1	4	*Commiphora pseudopaolii* Gillett	Cpa	Bura
100	500	10 : 1	5	*Acacia nilotica* (L.) Del.	Ani	Bura
			6	*Acacia horrida* (L.) Willd.	Aho	Bura
			7	*Lawsonia inermis* L.	Lin	Bura
			8	*Azadirachta indica* A. Juss.	Ain	Bura
			9	*Cordia sinensis* Lam.	Csi	Bura
			10	*Terminalia brownii* Fresen.	Tbr	Bura

[1] All in Kenya

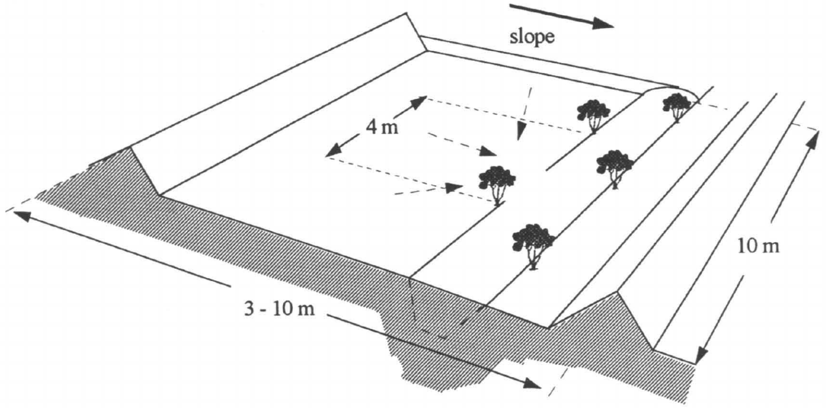

Fig. 13. Schematic side view of a rectangular microcatchment with cross-tied furrows (Experiment IV, not to scale). Dashed arrows demonstrate the flow of water from the catchment into the furrow.

divert rainwater collected in the catchment into the furrows (Fig. 13). Five catchment sizes and ten species (Table 4) were used as main and sub-plot treatments in four replicates. Apart from *Prosopis juliflora* and *Azadirachta indica* the species were indigenous. Furrow and catchment embankments were restored three times during the study period due to breaching caused by cattle and rains. Livestock damaged several *A. indica* trees during the first year. The trial was established in November 1989 and assessed annually (for assessment cf. Experiment II). The total and mean annual rainfall during the experimental period were 2084 mm and 424 mm a^{-1}. The effective rainfall estimates equalled to those of Experiment III.

Contour Furrow Microcatchments

The site of Experiment V was situated on a slightly sloping site (inclination less than 1 %) at the border of the floodplain bushland, 10 km away from Experiments I–IV. The soils are sandy clays (TH) and sandy clay loams to sandy clays (TN; ILACO 1977). The furrows were prepared approximately along the contours by a two-wing plough pulled by a tractor such that the soil heap was left on the downhill side (Fig 14). One plot consisted of an area of approximately 5 · 10 m, the (pairs of) furrows bordering a plot both on uphill and downhill side. The furrows were blocked manually after every 10 m to separate adjoining plots along the contours. There were 10 planting or seeding spots in a plot placed at a distance of 1 m apart on alternate furrows (2 m apart along one furrow). Ten seeds were applied for every seeding spot. The experiment was set in a split-plot design in three replicates with fencing (or no-fencing) as the main plot and the establishment method as the sub-plot treatment. The sub-plot treatments were:

Table 5. Summary of the species in water harvesting Experiments I–V.

Species	Microcatchment type				
	Long-furrow I	Deep triangular II	Shallow triangular III	Cross-tied furrows IV	Contour furrow V
Acacia aneura	X				
A. horrida		X	X	X	
A. mellifera	X	X	X		
A. nilotica			X	X	
A. nubica	X	X	X		
A. reficiens	X				
A. tortilis	X	X	X	X	X[1]
A. xanthophloea	X	X	X		
A. zanzibarica	X	X	X	X	X[2]
Azadirachta indica	X			X	
Berchemia discolor	X				
Commiphora pseudopaolii	X	X	X	X	
Cordia sinensis	X	X	X	X	X
Faidherbia albida	X				
Garcinia livingstonei	X				
Lawsonia inermis	X	X		X	
Parkinsonia anacantha	X	X	X		
Prosopis juliflora	X[3]	X[3]	X[3]	X	
Tamarindus indica	X				
Terminalia brownii				X	

[1] planting and sowing
[2] sowing only
[3] two seed sources

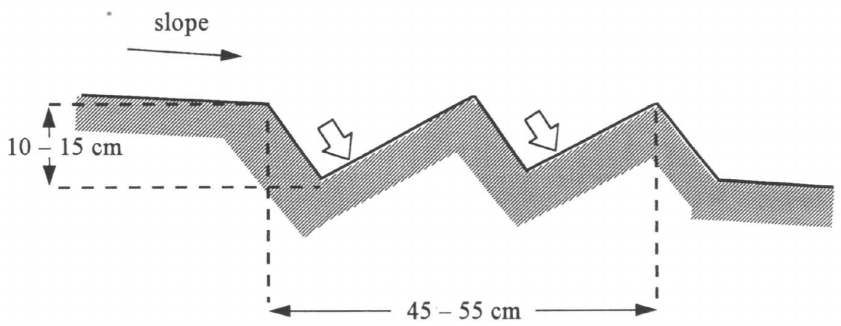

Fig. 14. Schematic side view of the contour furrows (Experiment V). The thick arrow denotes planting or sowing spot (not to scale).

(1) *Acacia tortilis*, planted;
(2) *Acacia tortilis*, sown;
(3) *Cordia sinensis*, planted;
(4) *Acacia zanzibarica*, sown; and
(5) control (natural regeneration)

The trial was established in November 1991 and assessed in December 1992. In addition, survival was counted at one, three and seven months after trial establishment. Survival in the sowing treatments was based on enumerating sowing spots with at least one live seedling; multiple seedlings in a single spot did not affect the survival rate. Height was measured from the tallest stem and stem diameter(s) over bark at ground level (d_0). Stem diameters were converted into pooled diameter (dg_0). Only the tallest seedling was assessed for one sowing spot in the sowing treatments.

2.5.2 Regeneration and Dynamics of Woody Vegetation in Cleared Bushland

Experiment VI was undertaken on a site, which covered 60.4 ha, and had been cleared of all vegetation in January 1982. The vegetation was cleared by a bulldozer moving forward in a straight line with the teeth of the rake (subsoilers) about 20 cm below the surface running through the topsoil. The aim of the clearing operation was to remove, from above and below the surface, all woody material likely to interfere with the process of irrigated cultivation. It was, however, not possibly to identify the depth to which roots had been removed. The area was never cultivated and had been gradually regenerated predominantly by *Acacia zanzibarica*. Most of the soils of the site constituted of VN, but small areas of TC, NG and TN were also encountered (Fig 15; ILACO 1977).

The site was assessed using a systematic sampling of 100 m² circular sample plots on a 50 · 200 m grid in September 1988 and in June 1992, 6 years 8 months and 10 years 5 months after clearing respectively (Fig. 15). The sample accounted for 1 % of the total cleared area. In 1992, seven additional (control) plots outside the cleared area aligned with the North-South grid lines were also assessed; these plots represented VN and TN soils (ILACO 1977). A check-list prepared for the area (Gachathi et al. 1987, 1995) was used for species identification. The total number of observed individuals was used to determine the frequency of occurrence. Difficulties were occasionally encountered in distinguishing individuals of particularly the bushy species. Where stems were very close together and appeared to share the same root system, they were counted as one individual, but all the stems were measured. Height of the tallest stem was recorded for all individuals. For trees above 1.3 m in height, number and diameters of stems at 0.15 m and 1.3 m were also determined.

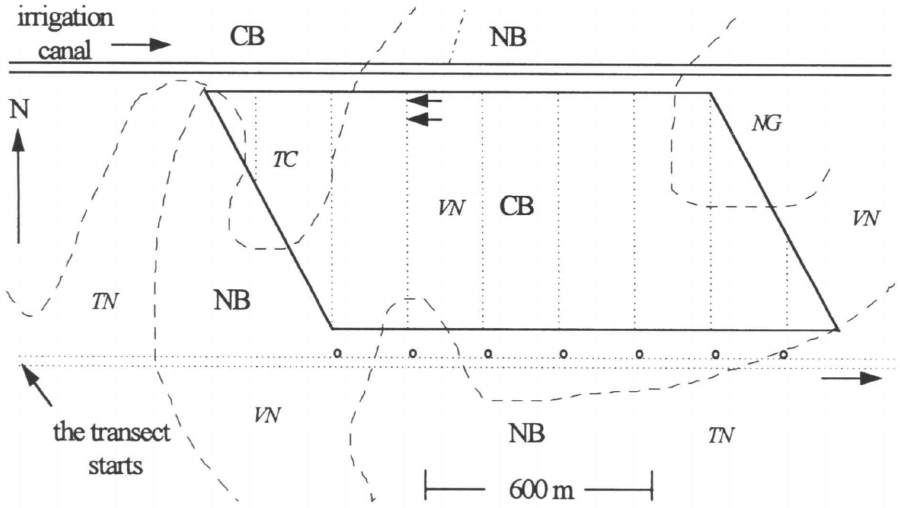

Fig. 15. Schematic view of the research site in cleared (CB; Experiment VI) and natural bushland (NB). The line spacing (dotted lines) and the distance between plots (represented by arrows inside the rectangle) were 200 m and 50 m respectively. The circles and the two parallel dotted lines South of the cleared area represent the location of the control plots assessed in 1992 and that of the transect (Experiment VII) assessed in 1985 and 1994 respectively. The soil types (ILACO 1977) are Typic Natrargid (*TN*), Vertic Natrargid (*VN*), Typic Camborthid (*TC*) and Natric Grumustert (*NG*).

For statistical tests and volume calculation, the diameters of multi-stemmed trees were converted into basal area ($g_{0.15}$) and pooled diameter ($dg_{0.15}$). Four to six soil samples (for pH, organic carbon and some other major elements) down to a depth of 30 cm were augered from all plots and mixed to form one composite sample per plot. The soil samples were analysed in the soil laboratory of the Kenya Forestry Research Institute (KEFRI). Before analysis all the samples were air dried, crushed and sieved through a 2-mm sieve. For pH, the soil:water (1:2.5) suspension was used. Organic carbon (C) was analysed by using the Walkley-Black method, nitrogen (N) by the Kjeldahl method, phosphorus (P) by the Olsen method and exchangeable cations by the Mehlich method. 0.1N HCl and 0.03N H_2SO_4 were used as extractants. Na, K, Ca and Mg were determined with atomic absorption spectrometry.

2.5.3 Composition and Dynamics of Woody Vegetation in Undisturbed Bushland

A transect running in West-East direction in the vicinity of Experiment VI was assessed in natural, undisturbed bushland in 1985 and 1994 (Experiment VII). The length of the transect was 4000 m; the width varied between 10 and 20 m, the former being applied in the dense vegetation section approximately parallel to Experiment VI (Fig. 15). The position, stem diameter(s) at 0.15 m and height were measured for all individuals of woody species at least 1.3 m in height for 20 m sections of the transect. In addition, all individuals below 1.3 m in height, which were regarded as seedlings, were enumerated and recorded by species in 1994. The separation of plant individuals was undertaken as above in Experiment VI. Trees were divided into 2-cm diameter ($dg_{0.15}$) classes up to 40 cm in diameter.

The diameter distribution of eight species in 1994 was studied in detail. All the seedlings were assumed to fall into the smallest diameter class (0–2.0 cm) in *Acacia tortilis*, *A. zanzibarica*, *Prosopis juliflora* and *Salvadora persica*, whereas in *Cordia sinensis* 95 % and in *Acacia reficiens* 80 % of the seedlings were assumed to belong to the smallest, and the rest to the next class (2.1–4.0 cm). In *Commiphora campestris* and *Dobera glabra*, the division into the two classes was assumed to be 50 % / 50 % each, due to the substantial tapering of the trees between 0.15 and 1.3 m in height. These estimates were based on the growth form and on the height-to-diameter relationship of the tree individuals at least 1.3 m in height, as well as on the observations on small trees (below 1.3 m in height) in experiment VI, in which the diameter was measured.

The species were identified by local informants in 1985, whereas the check-list by Gachathi et al. (1987, 1995) was used for identification in 1994. The standard of species identification among the shrub species increased between the two (transect) assessments, which could have affected the species distribution. This did not, however, affect the distribution and biomass estimates of the main species. TN and VN soils covered approximately 95 % of the soils along the transect (ILACO 1977); the respective areas included in the analyses were 4.20 and 2.06 ha. A small section of TH soil was also encountered, but it was excluded from the analyses. Several temporary or semi-permanent campsites of pastoralists were noted in the immediate vicinity of the transect in 1985. All the semi-permanent camps had been removed by 1994, although some of the sites were still clearly visible.

2.5.4 Seed Bank in Undisturbed Bushland

Sampling areas in Experiment VIII were selected from two vegetation types, namely *Acacia-Commiphora* bushland (Pratt et al. 1966) under TN soil and the transitional forest between the bushland and the Tana riverine forest (cf. Gachathi 1993, Gachathi et al. 1987, Johansson 1995). The soil types in the transitional forest were VN, NG and Vertic Hapludents. The dominant woody species in the sampling sites in the *Acacia-Commiphora* bushland were *A. tortilis*, *Boscia coriacea*, *Cordia* spp., *Grewia* spp., *Maerua* spp., *Commiphora* spp. and *Acacia elatior*, *A. zanzibarica*, *Salvadora persica*, *Cephalocroton cordofanus*, *Cordia sinensis*, *Phyllanthus somalensis and Terminalia* spp. in the transitional forest.

Tentative selection of the sites was based on aerial photographs taken in 1989. In field inspection and sampling, which was undertaken in 1992, some of the sites were rejected due to difficult terrain – poor visibility or dense thicket vegetation – or due to the absence of an unambiguous reference point. The exact sampling positions within each site were located randomly utilising compass direction and distance from a reference point or landmark at or near the site. Four sampling sites were included for both vegetation types, and ten samples were collected in each of the sites.

The samples were collected in July 1992 using a square, 50 · 50 cm, rigid metal frame which was inserted into a depth of 3–4 cm in the soil. The soil was collected by hand down to the lower edge of the frame, deposited into black polythene bags, transported to a nursery and spread over to plastic germination trays (size approximately 30 · 40 cm). Litter was not separated from the samples. Polythene shade (at 1.2 m height), which absorbs 50 % of direct sunlight, was used to cover the samples during the nursery stage.

The samples were watered daily and the germination, as defined by seedling emergence, was monitored for a period of 16 weeks after first application of water. Species were identified eight and 16 weeks after first watering. Species identification was done by a local informant, who had participated in the preparation of the species checklist prepared for the area (cf. Gachathi 1993, Gachathi et al. 1987). Seedlings of woody species were not removed during the monitoring period. Grass and herb shoots (not roots) were cut using scissors in connection with the identification after eight weeks. Various grass and herb shoots resprouted after cutting. After 16 weeks all samples were diluted in water and sifted using two sizes of wire mesh (approximate diameters 1–2 and 6–8 mm) for identifying the ungerminated seeds of the woody species.

2.6 Volume Estimation

Tree volume or biomass (in Experiments II–VII) were estimated using genus-, species- or form-specific functions (Table 6) developed at Bura by Pukkala (1989). These functions were based on 209 sample trees, which represented nine of the most common trees in the area. When necessary, biomass estimates were converted to volume equivalents by utilising species-specific conversion factors (fresh or dry density; Table 7). The shrinkage of wood during drying and subsequent possible decrease in volume was not considered in the conversion. The general model of the volume or biomass functions was (Pukkala 1989):

$$\text{UFV or UFB} = a \cdot d_{0.15}\,(n)^{b} \qquad (1)$$

where UFV is usable volume (dm^3) or UFB is usable fresh mass (kg) of stem and branches between stump (0.15 m) and top diameter of 2 cm, $d_{0.15}$ diameter at stump height, and a and b are regression coefficients. Usable volume corresponds to those parts of tree which are generally collected for fuelwood. Biomass and volume for each separate stem at stump height were calculated separately. Volume was calculated for individuals of at least 1.3 m in height. The specific models are displayed in Table 6. In Experiments II–IV the volume/biomass estimates per unit area were estimated by multiplying the average volume/biomass of living trees with tree density (n ha^{-1}) and survival; this was termed transformed volume or biomass.

Table 6. Regression coefficients (a, b) of volume and biomass equations (Pukkala 1989). S_f denotes standard error of estimate, R^2 coefficient of determination, UFB usable fresh biomass, TFB total fresh biomass and UFV usable volume.

Model	*a*	*b*	S_f	R^2	Estimated parameter
(2)	0.02796	2.9412	0.551	0.922	UFB
(3)	0.01144	3.0489	0.445	0.950	UFB
(4)	0.06392	2.4596	0.351	0.954	UFV
(5)	0.00171	3.7485	0.335	0.985	UFV
(6)	0.10259	2.5930	0.290	0.960	UFB
(7)	0.19109	2.5210	0.240	0.980	TFB

For calculation of rain use efficiency (RUE; the total aerial dry matter wood production per unit area as a function of time and precipitation) total biomass was estimated by using model 7 by Pukkala (Table 6) for *Prosopis juliflora*, and those by Vertanen et al. (1993) for *Acacia* sp. (models 8–9).

Acacia nilotica, *A. tortilis*, *A. xanthophloea*, *A. zanzibarica*

$$\text{TFB} = 0.153 \cdot dg_{0.15}{}^{2.4831},\ \text{R}^2 = 0.968 \qquad (8)$$

other *Acacia* sp.

$$\text{TFB} = 0.15777 \cdot dg_{0.15}{}^{2.4376},\ \text{R}^2 = 0.944 \qquad (9)$$

where TFB is total fresh mass above stump and $dg_{0.15}$ represents pooled diameter at stump height. For other species total volume was roughly approximated by utilising the total biomass to usable biomass ratio (model 10; Pukkala 1989).

$$\text{ratio} - 1 = 4.61 \cdot d_{0.15}(n)^{-0.709} \qquad (10)$$

2.7 Data Analyses and Statistical Methods

Water Harvesting (Experiments I–V)

The normality of the distributions was tested with Lilliefors test. Parametric ANOVA (Analysis of variance) was also preceded by the Levene's test, which was used to test the variance equality of all variables except survival (percentage). The residuals from the ANOVA were transformed into their absolute values and the *F*-test was repeated for the absolute residuals. Logarithmic transformations (*log* x+1) were then undertaken for all variables (except survival) in which null hypothesis (variances are equal) using untransformed variables was rejected at $p<0.01$ (cf. Milliken and Johnson 1984). The *F*-test for the absolute residuals was done as above. The selection whether to use the initial variable or its logarithmic derivative in the eventual ANOVA was based on the *p*-value from the absolute residuals. For survival percentages arc-sin$\sqrt{x}$ transformations were used in parametric ANOVA.

Where the conditions of parametric ANOVA were met, the data were analysed using ANOVA for randomised blocks in rainwater harvesting Experiments II and III and for split-plot in Experiments IV and V respectively. ANOVA for randomised blocks was then also applied to Experiment V by pooling the main plot treatments of the split-plot design within replications, as there were no significant differences between the main plot effect or the interactive effect of main and sub-plots. Where appropriate, parametric ANOVA was followed by Tukey's HSD-test for comparisons by pairs. Non-parametric Kruskal-Wallis one-way ANOVA of ranked data was used for variables that were not normally distributed. In such cases comparisons by pairs was undertaken by Mann-Whitney U-test.

Natural Vegetation (Experiments VI–VIII)

Soil conditions were not uniform in the research site. All the samples / sample plots were tentatively grouped according to the soil types as evaluated for irrigation suitability by ILACO (1977). In the cleared area the differences between the three soil types (Fig. 15) were tested by subjecting the soil data gathered in the present study to Kruskal-Wallis one-way ANOVA followed by Mann-Whitney U-test for comparisons by pairs. Paired t-test was used to compare the differences in soil parameters over time in the cleared site.

In the undisturbed control area the similarities / dissimilarities between the two soil types, TN and VN (Fig. 15) were compared with the help of multidimensional scaling (MDS) ordination, based on a correlation matrix of soil data across all measured variables (pH, C, N, C/N-ratio, K, P, Ca, Mg, Zn, Fe, Mn, Cu). The variables were initially standardised to zero mean and unit variance due to different measurement units. In addition, the differences in single characteristics between the two soil types in the control area, as well as those between the VN samples in the cleared and non-cleared control areas were tested with the Mann-Whitney U-test.

The diameter distribution of *Acacia zanzibarica* trees in the cleared bushland 10.4 years after clearing (Experiment VI) and in the undisturbed bushland (Experiment VII) were compared using the non-parametric Kolmogorov-Smirnov test. The tested samples constituted of all *A. zanzibarica* individuals on VN and NG soil types in the cleared bushland and undisturbed areas. The individuals in the seedling category (below 1.3 m height) in the undisturbed area were all assumed to fall into the smallest diameter class (0.1–2.0 cm).

Table 7. Basic density and volume or biomass equations used in the computation of dry matter production of species in Experiments II–VII (basic densities from Pukkala 1989, except *Azadirachta indica*, from Pohjonen 1989).

Species	Basic density kg m^{-3}	Form	Biomass / volume model
Acacia horrida	720	bush	3
A. mellifera	720	bush	3
A. nilotica	720	tree, flat crown	2
A. nubica	640	bush	3
A. paolii	720	bush	3
A. reficiens	908	bush /small tree	3
A. senegal	720	bush	3
A. tortilis	744	tree, flat crown	2
A. xanthophloea	867	tree, flat crown	2
A. zanzibarica	867	tree, flat crown	2
Azadirachta indica	685	tree	4
Berchemia discolor	720	bush / tree	2
Cadaba farinosa	640	bush / climber	4
Cadaba gillettii	640	bush	4
Combretum constrictum	720	bush	6
Commiphora africana	320	bush /crooked tree	5
Commiphora campestris	346	crooked tree	5
Commiphora edulis	320	bush	5
Commiphora incisa	320	bush / crooked tree	5
Commiphora pseudopaolii	482	crooked tree	5
Commiphora rostrata	320	bush / crooked tree	5
Commiphora sp.	320	bush / crooked tree	5
Cordia sinensis	573	bush /crooked tree	4
Dobera glabra	681	crooked tree	4
Lannea triphylla	320	bush	5
Lawsonia inermis	689	bush	4
Maerua decumbens	640	bush	6
Maerua macrantha	640	bush	4
Maerua triphylla	640	bush / tree	4
Meyna tetraphylla	720	bush	6
Parkinsonia anacantha	320	bush / crooked tree	5
Prosopis juliflora	763	bush	6, 7
Salvadora persica	654	bush / crooked tree	4
Sterculia africana	200	bush / tree	4
Terminalia brownii	720	tree	4
Terminalia orbicularis	651	bush / tree	4
Wrightia demartiniana	640	bush	5

3 Results

3.1 Microcatchment Water Harvesting

3.1.1 Long-Furrow Microcatchments

In the long-furrow catchment the average and range of survival rate in the tree species tested six months after establishment were 51.7 % and 0–96 % respectively. The corresponding values for stem height for live seedlings were 0.31 m and 0.15–0.75 m (Table 8). The trial was terminated in November 1989, one year after establishment following the death of virtually all seedlings due to drought and subsequent termite attacks.

Table 8. Long-furrow microcatchment: Mean tree survival and height, with standard errors of the mean (S.E.) in 18 species six months after outplanting (Experiment I; n=5).

Species (ranked by survival)	Survival %	S.E.	Height m	S.E.
Azadirachta indica	96.0	2.4	0.55	0.04
Acacia nubica	94.0	4.0	0.31	0.01
Acacia zanzibarica	92.0	3.7	0.38	0.04
Prosopis juliflora	88.0	7.3	0.42	0.03
Acacia reficiens	88.0	5.8	0.21	0.01
Acacia mellifera	86.0	5.0	0.33	0.30
Garcinia livingstonei	86.0	8.7	0.22	0.02
Cordia sinensis	86.0	9.2	0.16	0.01
Acacia xanthophloea	78.0	7.3	0.75	0.06
Commiphora pseudopaolii	46.0	14.3	0.27	0.04
Acacia aneura	42.0	10.6	0.18	0.01
Berchemia discolor	28.0	3.7	0.22	0.02
Prosopis juliflora	10.0	6.3	0.15	0.05
Acacia tortilis	4.0	2.4	0.30	0.10
Tamarindus indica	4.0	4.0	0.20	–
Faidherbia albida	0	–	–	–
Lawsonia inermis	0	–	–	–
Parkinsonia anacantha	0	–	–	–

3.1.2 Deep Triangular Microcatchments

When testing the deep triangular rainwater harvesting design, statistically significant differences were found between species in survival, height, diameter ($dg_{0.15}$), number of stems ($n_{0.15}$) and volume 70 months after outplanting (Tables 9–11). No statistically significant differences were noted between the replicates (blocks). The survival rate was higher than 50 % for *Acacia horrida, A. zanzibarica, A. mellifera, Parkinsonia anacantha* and *Commiphora pseudopaolii*. Local species rated on the average much higher in survival (49.7 %), as compared to those not indigenous to the area (11.8 %).

All species were single-stemmed at the ground level, but nevertheless the multiple-stem character was peculiar in most of them. Branching below stump level was most pronounced in *Acacia horrida* and *Prosopis juliflora* (Pjm seed source), and apart from *Acacia zanzibarica* and *Commiphora pseudopaolii* all other species had generally developed two or more stems at stump level (Table 10).

At the age of 70 months, the tree height ranged from 0.9 m (*Commiphora pseudopaolii*) to 3.5 m (*Prosopis juliflora*, Pjm seed source). The mean diameter and mean volume of *Prosopis juliflora* (Pjm) were the highest among all the species (10.2 cm and 24.2 dm^3 tree^{-1}), but the respective survival was only 2.1 % (Table 11). The usable biomass (DM) per tree for the three best performing species, *P. juliflora* (Pjb seed source), *Acacia zanzibarica* and *A. horrida*, were 7.6, 4.7 and 3.7 kg tree^{-1} respectively. Mean survival over all species was 40.9 %, and the mean height, diameter and volume were 2.1 m, 5.1 cm and 3.7 dm^3 tree^{-1} respectively. If *P. juliflora* (Pjm) which only had one single tree alive, were excluded from the analysis, the respective mean values for height, diameter and volume would decrease to 1.9 m, 4.7 cm and 2.7 kg tree^{-1} respectively.

The mean survival of the seedlings planted in

Table 9. Deep triangular microcatchments: Mean tree survival and height, with S.E., in 12 species 70 months after outplanting (n=4). Results of Kruskal-Wallis and parametric ANOVA are presented for survival and height respectively. Differences of mean values which are not followed by the same letter are statistically significant at $p<0.05$ as determined by Mann-Whitney U-test and Tukey's HSD-test for survival and height respectively.

Species (ranked by survival)	Survival[1] %	S.E.	Tukey's HSD	Height[2], m	S.E.	Tukey's HSD
Acacia horrida	97.2	2.8	a	3.3	0.1	ab
Acacia zanzibarica	79.2	8.0	ab	3.1	0.2	ab
Acacia mellifera	66.7	12.3	bc	1.4	0.1	e
Parkinsonia anacantha	60.8	10.4	bcd	2.4	0.1	acd
Commiphora pseudopaolii	52.1	11.5	bcd	0.9	0.1	e
Acacia tortilis	35.4	10.4	cd	1.3	0.2	e
Cordia sinensis	35.4	14.6	cde	1.3	0.1	e
Prosopis juliflora (Pjb)	33.3	5.9	cd	2.9	0.2	abc
Acacia nubica	16.7	16.7	abcde	1.5	–	cde
Lawsonia inermis	4.2	4.2	ae	1.2	–	de
Prosopis juliflora (Pjm)	2.1	2.1	e	3.5[3]	–	a
Acacia xanthophloea	0	–	e	–	–	

[1] $H = 33.50$; $df = 11$; $p = 0.000$
[2] $F = 28.41$; $df = 10/19$; $p = 0.000$
[3] Based on one observation only.

Table 10. Deep triangular microcatchments: Mean tree diameter ($dg_{0.15}$) and mean stem number at stump height ($n_{0.15}$), with S.E. in 12 species 70 months after outplanting (n=4). Kruskal-Wallis and parametric ANOVA results are presented for diameter and number of stems respectively. Differences of mean values which are not followed by the same letter are statistically significant at $p<0.05$ as determined by Mann-Whitney U-test and Tukey's HSD-test for diameter and number of stems respectively.

Species (ranked by $dg_{0.15}$)	$dg_{0.15}$[1] cm	S.E.	Tukey's HSD	$n_{0.15}$[2]	S.E.	Tukey's HSD
Prosopis juliflora (Pjm)	10.2[3]	–	a	4.0[3]	–	a
Acacia horrida	8.8	0.8	ab	3.2	0.4	a
Parkinsonia anacantha	7.9	0.5	abc	2.2	0.1	ab
Prosopis juliflora (Pjb)	6.4	0.3	ad	2.3	0.3	ab
Acacia zanzibarica	5.8	0.7	acde	1.3	< 0.05	b
Cordia sinensis	3.9	0.2	ef	2.4	0.1	ab
Acacia nubica	3.4	–	abcdefg	1.8	–	ab
Acacia mellifera	3.0	0.4	afg	1.8	0.3	b
Acacia tortilis	2.8	0.4	af	2.0	0.3	b
Commiphora pseudopaolii	2.5	0.2	g	1.2	0.1	b
Lawsonia inermis	2.1	–	cdefg	2.0	–	ab
Acacia xanthophloea	–[4]			–[4]		

[1] $H = 28.23$; $df = 10$; $p = 0.002$
[2] $F = 7.65$; $df = 10/19$; $p = 0.000$
[3] Based on one observation only.
[4] no live trees

deep triangular microcatchments declined steadily over time (Fig. 16). At the end of the observation period of 70 months, there was only one *Prosopis juliflora* (Pjm) tree alive, and *Acacia xanthophloea* had perished completely. Height growth during the first 18 months was rapid, with *A. horrida* and *P. juliflora* (Pjb) attaining a height above 2.5 m. Thereafter the growth curve clearly levelled, except in *Acacia zanzibarica* which continued a rather rapid growth throughout (Fig. 17).

The differences between species were most pronounced in volume. *P. juliflora* (Pjb seed source) grew steadily and rather rapidly from the beginning, whereas the growth rate of *A. horrida* declined after 36 months and that of *Acacia zanzibarica* exhibited a clear increase between 49 and 70 months after outplanting (Figs. 18–19). The application of transformed volume (tree volume multiplied by survival percentage) at the age of 70 months changed the ranking of the species: the performance of *P. juliflora* (Pjb) declined and that of *A. horrida* improved, but the exact order varied over time. Generally, the indigenous species performed better when transformed volume was incorporated; this was due to their higher survival rate as compared to exotic species.

The transformed MAI of usable volume and biomass (DM) per unit of area for the best three species at 70 months were 0.06–0.09 m³ ha⁻¹a⁻¹ (Fig. 19) and 43–63 kg ha⁻¹a⁻¹ respectively. For the sake of clarity, only five local species and the standard control species, *P. juliflora* (Pjb), are presented in figures comparing the performance over time.

Table 11. Deep triangular microcatchments: Mean usable tree volume (with S.E.) and MAI of transformed tree volume (average usable volume of living trees multiplied by tree density and survival) in 12 species 70 months after outplanting (n=4). Results of Kruskal-Wallis ANOVA are presented for volume. Differences of mean values which are not followed by the same letter are statistically significant at $p<0.05$ as determined by Mann-Whitney U-test.

Species (ranked by MAI)	Volume[1] dm^3tree^{-1}	S.E.	Tukey's HSD	MAI $m^3ha^{-1}a^{-1}$
Acacia horrida	5.1	1.5	acd	0.09
Acacia zanzibarica	5.4	2.1	abc	0.07
Prosopis juliflora (Pjb)	10.0	0.8	ab	0.06
Parkinsonia anacantha	2.9	0.7	acde	0.03
Cordia sinensis	1.9	0.2	ace	0.01
Prosopis juliflora (Pjm)	24.2[2]	–	a	0.01[2]
Acacia mellifera	0.3	0.1	ef	< 0.01
Acacia tortilis	0.6	0.2	af	< 0.01
Acacia nubica	0.5	–	abcdefg	< 0.01
Commiphora pseudopaolii	0.1	<0.05	g	< 0.01
Lawsonia inermis	0.3	–	abcefg	< 0.01
Acacia xanthophloea	–[3]			–[3]

[1] $H = 29.27$; $df = 10$; $p = 0.001$
[2] Based on one observation only
[3] no live trees

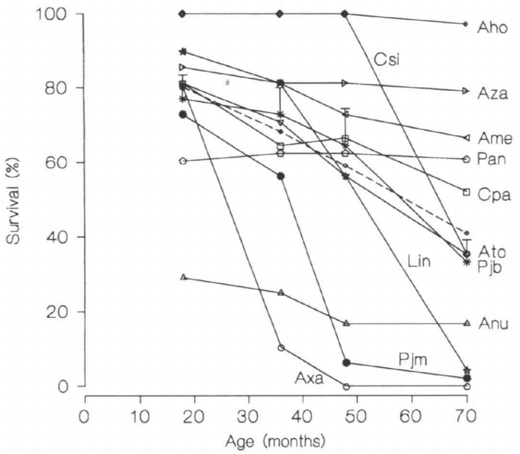

Fig. 16. Deep triangular microcatchments: Mean tree survival as a function of time in 12 species. The dashed line displays the mean over all species (n=4). For the sake of clarity, S.E. is displayed only for Pjb. (Aho = *Acacia horrida*; Ame = *A. mellifera*; Anu = *A. nubica*; Ato = *A. tortilis*; Axa = *A. xanthophloea*; Aza = *A. zanzibarica*; Cpa = *Commiphora pseudopaolii*; Csi = *Cordia sinensis*; Lin = *Lawsonia inermis*; Pan = *Parkinsonia anacantha*; Pjb = *Prosopis juliflora*, local seed source; Pjm = *P. juliflora*, Mombasa seed source).

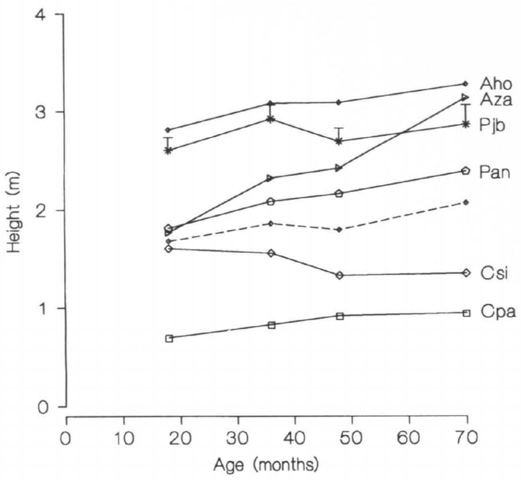

Fig. 17. Deep triangular microcatchments: Mean tree height in six species as a function of time (n=4). For the sake of clarity, not all species are displayed; S.E. is presented for Pjb only. The dashed line represents the mean over all species. (Aho = *Acacia horrida*; Aza = *A. zanzibarica*; Cpa = *Commiphora pseudopaolii*; Csi = *Cordia sinensis*; Pan = *Parkinsonia anacantha*; Pjb = *Prosopis juliflora*, local seed source).

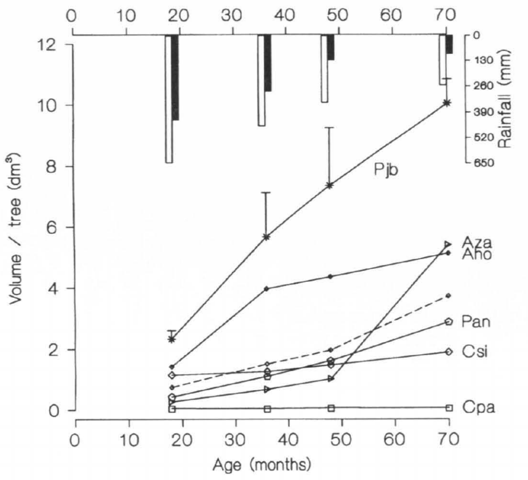

Fig. 18. Deep triangular microcatchments: Mean usable tree volume in six species as a function of time (n=4). White bars stand for the total rainfall and black ones for the amount exceeding 30 mm month^{-1} during the 12-month period prior to assessment. For the sake of clarity, not all species are displayed; S.E. is presented for Pjb only. The dashed line represents the mean over all species. (Aho = *Acacia horrida*, Aza = *A. zanzibarica*, Cpa = *Commiphora pseudopaolii*, Csi = *Cordia sinensis*, Pan = *Parkinsonia anacantha*, Pjb = *Prosopis juliflora*, local seed source).

Expressed as rain use efficiency (RUE), the total aerial wood production (DM) per unit area as a function of time and precipitation, the productivity of the three best species ranged between 0.57–0.83, 0.08–0.30 and 0.15–0.28 kg ha–^{1}a^{-1}mm^{-1} over time for *A. horrida, A. zanzibarica* and *P. juliflora* (Pjb) respectively. The RUE of *A. zanzibarica* increased and that of *P. juliflora* declined between 49 and 70 months after trial establishment, whereas that of *A. horrida* increased at the beginning up to 35 months and thereafter remained rather stable.

3.1.3 Shallow Triangular Microcatchments

For the experiment which tested the effect of shallow triangular microcatchments, the data on mean survival, height, diameter ($dg_{0.15}$), number of stems ($n_{0.15}$), tree volume and transformed MAI 59 months after planting are presented in Tables 12–14. Statistically significant differences existed between species in survival, height, $dg_{0.15}$, $n_{0.15}$ and volume at that stage. *Acacia xanthophloea* ranked highest in survival; also four local species, *A. zanzibarica*, *A. horrida*, *A. mellifera* and *Commiphora pseudopaolii* showed an above-50 % survival (Table 12). The mean survival of local species (33.3 %) was only slightly higher than that of the exotic species (28.1 %). For the sake of clarity, only the best single species, *A. xanthophloea*, three local ones and the standard control, *P. juliflora* (Pjb seed source), are presented in figures comparing performance over time.

At stump level *Acacia tortilis* and *A. mellifera* had developed more than two stems, whereas all other species had an average of 1–2 stems per individual tree (Table 13). At ground level all species were practically single stemmed. Height ranged from 0.4 m (*Commiphora pseudopaolii*) to 2.6 m (*A. horrida*; Table 12). *A. horrida* performed best in diameter, whereas *A. xanthophloea* ranked highest in volume (Table 14). Usable tree biomass (DM) for the three best species, *A. xanthophloea*, *A. horrida* and *P. juliflora* (Pjb) were 1.6, 1.1 and 0.9 kg tree^{-1} respectively. In transformed volume, *A. nilotica* replaced *P. juliflora* (Table 14).

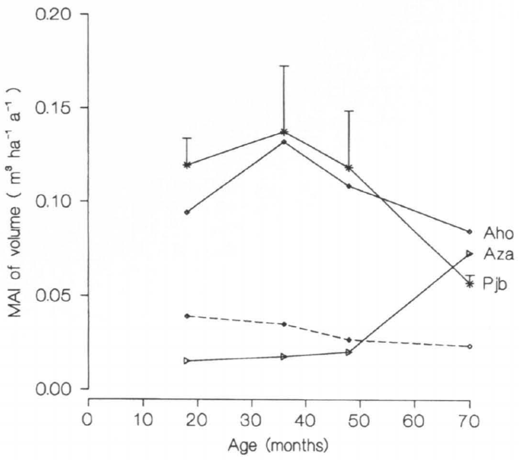

Fig. 19. Deep triangular microcatchments: MAI of transformed tree volume (average usable volume of living trees multiplied by tree density and survival) of the three most promising species as a function of time. For the sake of clarity S.E. is presented for Pjb only. The dashed line displays the mean over all species (n=4). (Aho = *Acacia horrida*; Aza = *A. zanzibarica*; Pjb = *Prosopis juliflora*, local seed source).

The equivalent of MAI of transformed usable volume and biomass (DM) per unit area for the best three species 59 months after planting were 0.01–0.03 m^{3} ha^{-1}a^{-1} and 3.6–23.4 kg ha^{-1}a^{-1} respectively. Mean survival over all species was 32.5 %, and mean height, diameter and volume were 1.5 m, 3.0 cm and 0.8 dm^{3} tree^{-1} respectively.

When testing the effect of shallow triangular microcatchments it was observed that there was a gradual decline in mean survival among the species over time (Fig. 20). It is notable that for both of the seed sources of *Prosopis juliflora* the mean survival was low as compared to that of the native species; one *P. juliflora* source (Pjm) as well as one local species, *Acacia nubica* perished completely soon after planting. In contrast, *A. xan-*

Table 12. Shallow triangular microcatchments: Mean tree survival, height (with S.E.) in 12 species 59 months after outplanting (n=4). Results of Kruskal-Wallis and parametric ANOVA are presented for survival and height respectively. Differences of mean values which are not followed by the same letter are statistically significant at $p<0.05$ as determined by Mann-Whitney U-test for survival and Tukey's HSD-test for height respectively.

Species (ranked by survival)	Survival[1] %	S.E.	Tukey's HSD	Height[2] m	S.E.	Tukey's HSD
Acacia xanthophloea	68.8	10.8	a	2.1	0.1	a
Acacia horrida	62.5	13.5	ab	2.6	< 0.05	a
Acacia zanzibarica	62.5	22.2	abc	1.7	0.2	ab
Commiphora pseudopaolii	59.4	7.9	abcd	0.4	< 0.05	c
Acacia mellifera	53.1	10.7	abcde	0.9	0.1	bc
Acacia nilotica	21.9	10.7	bcef	1.7	0.4	ab
Prosopis juliflora (Pjb)	12.5	8.8	cfg	1.9	0.7	ab
Acacia tortilis	12.5	12.5	bcefg	1.8	1.8	abc
Cordia sinensis	12.5	8.8	cfg	0.8	0.2	bc
Parkinsonia anacantha	6.3	6.3	acefg	1.1	1.1	abc
Acacia nubica	0	–	cg	–		
Prosopis juliflora (Pjm)	0	–	cg	–		

[1] $H = 29.39$; $df = 11$; $p = 0.002$
[2] $F = 10.64$; $df = 9/15$; $p = 0.000$

Table 13. Shallow triangular microcatchments: Mean tree diameter ($dg_{0.15}$), mean stem number at stump height ($n_{0.15}$), with S.E. in 12 species 59 months after outplanting (n=4). Results of parametric and Kruskal-Wallis ANOVA are presented for diameter and number of stems respectively. Differences of mean values which are not followed by the same letter are statistically significant at $p<0.05$ as determined by Tukey's HSD-test and Mann-Whitney U-test for diameter and number of stems respectively.

Species (ranked by $dg_{0.15}$)	$dg_{0.15}$[1] cm	S.E.	Tukey's HSD	$n_{0.15}$[2]	S.E.	Tukey's HSD
Acacia horrida	5.1	0.3	a	1.6	0.2	abc
Acacia xanthophloea	4.7	0.3	ab	1.9	0.2	abc
Acacia tortilis	3.4	–	abc	2.7	–	a
Acacia nilotica	3.3	0.8	abc	1.2	0.2	ac
Parkinsonia anacantha	2.5	–	abc	1.0	0.0	bc
Prosopis juliflora (Pjb)	2.3	1.2	abc	1.0	0.0	bc
Acacia zanzibarica	2.2	0.4	c	1.0	< 0.05	ac
Cordia sinensis	2.0	<0.05	bc	1.5	0.5	abc
Acacia mellifera	1.7	0.2	c	2.4	0.2	ab
Commiphora pseudopaolii	1.4	0.2	c	1.1	< 0.05	ac
Prosopis juliflora (Pjm)	–[3]			–[3]		
Acacia nubica	–[3]			–[3]		

[1] $F = 8.03$; $df = 9/15$; $p = 0.000$
[2] $H = 19.59$; $df = 9$; $p = 0.021$
[3] no live trees

Table 14. Shallow triangular microcatchments: Mean usable tree volume, (with S.E.) and MAI of transformed volume (average usable volume of living trees multiplied by tree density and survival) in 12 species 59 months after outplanting (n=4). Results of Kruskal-Wallis ANOVA are presented for volume. Differences of mean values which are not followed by the same letter are statistically significant at $p<0.05$ as determined by Mann-Whitney U-test.

Species (ranked by MAI)	Volume[1] dm^3tree^{-1}	S.E.	Tukey's HSD	MAI $m^3ha^{-1}a^{-1}$
Acacia xanthophloea	1.9	0.4	a	0.03
Acacia horrida	1.6	0.3	a	0.02
Acacia nilotica	1.1	0.5	abc	0.01
Acacia zanzibarica	0.3	0.1	bcd	< 0.01
Prosopis juliflora (Pjb)	1.2	1.1	ab	< 0.01
Acacia tortilis	0.8	–	abc	< 0.01
Cordia sinensis	0.4	<0.05	abc	< 0.01
Acacia mellifera	0.1	<0.05	b	< 0.01
Commiphora pseudopaolii	< 0.05	<0.05	b	< 0.01
Parkinsonia anacantha	0.1	–	abcd	< 0.01
Prosopis juliflora (Pjm)	–[2]			–[2]
Acacia nubica	–[2]			–[2]

[1] $H = 22.14$; $df = 9$; $p = 0.008$
[2] no live trees

thophloea, which is not native to the area, performed well both in survival and growth. The height growth of *A. horrida* was rapid and stable throughout, whereas that of *A. xanthophloea* declined and that of *P. juliflora* (Pjb) stalled after 36 months (Fig. 21).

P. juliflora (Pjb seed source) grew fastest in volume from the beginning up to 36 months, but was thereafter outperformed by *A. xanthophloea* and *A. horrida* (Fig. 22). In transformed MAI and RUE the difference in favour of the acacias became more pronounced (Fig. 23) due to the low survival rate in Pjb. The mean volume across all species started to increase after 36 months (Fig. 22), but this was not reflected in the transformed MAI due to a concurrent decrease in the mean survival (Fig. 23).

At the age of two years, the growth and RUE of *A. xanthophloea* and *A. horrida* were superior as compared to all other species, and it remained like that throughout the assessment period. RUE of the best species ranged between 0.05–0.19, 0.03–0.18 and 0.01–0.03 kg $ha^{-1}a^{-1}mm^{-1}$ over time for *A. horrida, A. xanthophloea* and *A. nilotica* respectively. In the shallow triangular microcatchments the average growth and RUE figures were distinctly lower than in the deep microcatchments of this type (Experiment II).

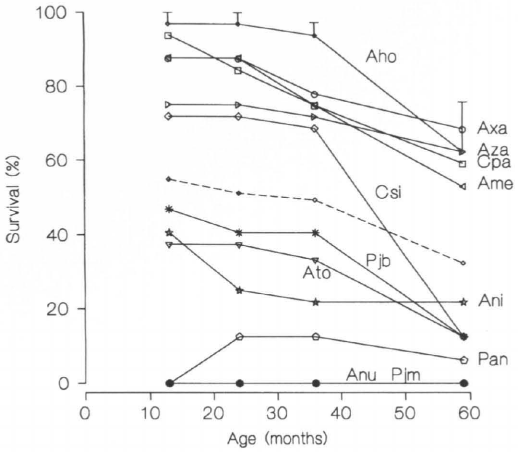

Fig. 20. Shallow triangular microcatchments: Mean tree survival in 12 species as a function of time (n=4). For the sake of clarity, S.E. is displayed only for Aho. The dashed line displays the mean over all species. (Aho = *Acacia horrida*, Ame = *A. mellifera*, Ani = *A. nilotica*, Anu = *A. nubica*, Ato = *A. tortilis*, Axa = *A. xanthophloea*, Aza = *A. zanzibarica*, Cpa = *Commiphora pseudopaolii*, Csi = *Cordia sinensis*, Pan = *Parkinsonia anacantha*, Pjb = *Prosopis juliflora*, local seed source, Pjm = *P. juliflora*, Mombasa seed source).

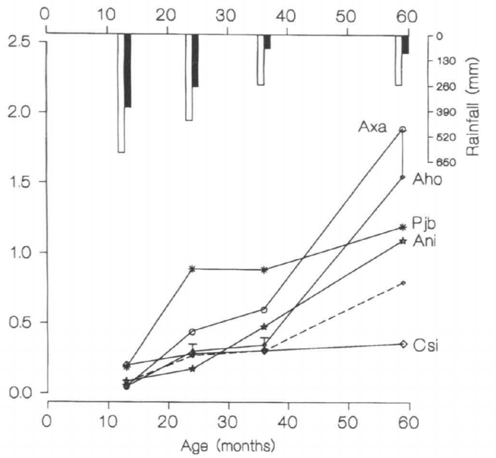

Fig. 22. Shallow triangular microcatchments: Mean usable tree volume in five species as a function of time (n=4). White and black bars represent the total rainfall and the proportion exceeding 30 mm month^{-1} during the 12-month period prior to assessment. For the sake of clarity, not all species are displayed; S.E. is presented for Aho only. The dashed line represents the mean over all species. (Aho = *Acacia horrida*; Ani = *A. nilotica*; Axa = *A. xanthophloea*; Csi = *Cordia sinensis*; Pjb = *Prosopis juliflora*, local seed source).

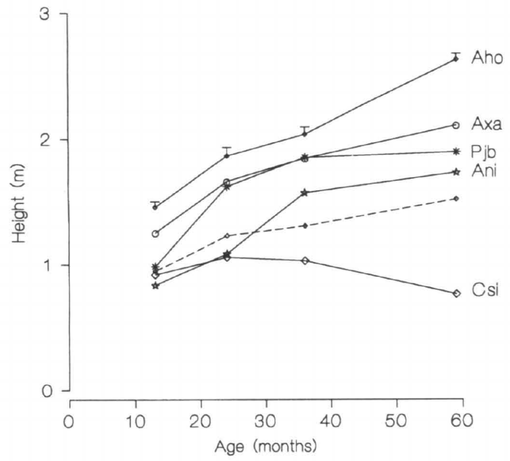

Fig. 21. Shallow triangular microcatchments: Mean tree height in five species as a function of time (n=4). For the sake of clarity, not all species are displayed; S.E. is presented for Aho only. The dashed line represents the mean over all species. (Aho = *Acacia horrida*, Ani = *A. nilotica*, Axa = *A. xanthophloea*, Csi = *Cordia sinensis*, Pjb = *Prosopis juliflora*, local seed source).

Table 15. Cross-tied furrow microcatchments: Mean tree survival, height (with S.E.) in 10 species 59 months after outplanting (n=20). Results of Kruskal-Wallis and parametric ANOVA are presented for survival and height respectively. Differences of mean values which are not followed by the same letter are statistically significant at $p<0.05$ as determined by Mann-Whitney U-test for survival and Tukey's HSD-test for height.

Species (ranked by survival)	Survival [1] %	S.E.	Tukey's HSD	Height[2] m	S.E.	Tukey's HSD
Prosopis juliflora (Pjb)	95.0	2.0	a	3.3	0.1	a
Acacia tortilis	91.1	2.9	a	1.5	0.1	c
Lawsonia inermis	88.4	6.2	a	1.3	0.1	cd
Acacia zanzibarica	84.0	4.5	a	2.6	0.1	b
Acacia nilotica	55.7	6.7	b	2.3	0.1	b
Commiphora pseudopaolii	55.0	6.2	b	0.6	<0.05	e
Cordia sinensis	43.3	8.4	bc	1.0	0.1	d
Acacia horrida	35.4	5.8	c	2.4	0.1	b
Terninalia brownii	0	–	d	–		
Azadirachta indica	0	–	d	–		

[1] $H = 134.53$; $df = 9$; $p = 0.000$
[2] $F = 84.69$; $df = 7/78$; $p = 0.000$

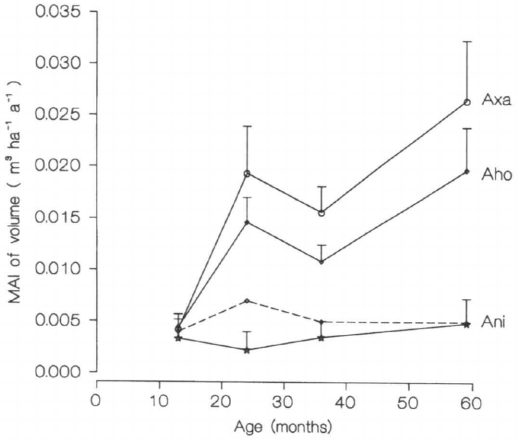

Fig. 23. Shallow triangular microcatchments: MAI of transformed tree volume (average usable volume of living trees multiplied by tree density and survival), with S.E., of the three most promising species as a function of time. The dashed line displays the mean over all species. (Aho = *Acacia horrida*; Ani = *A. nilotica*; Axa = *A. xanthophloea*).

3.1.4 Cross-Tied Furrow Microcatchments

There were no differences in survival caused by catchment sizes 59 months after outplanting, whereas the species differed significantly in their responses (Table 15). The mean survival over all species varied from 59.4 % (catchment size 80 m²) to 51.9 % (60 m²). At the species level, *Prosopis juliflora* (Pjb seed source), *Acacia tortilis*, *Lawsonia inermis* and *A. zanzibarica* exhibited a mean survival rate above 80 %. The indigenous *Terminalia brownii* and *Azadirachta indica*, an exotic widely planted in ASAL, had perished completely at 49 months.

In the cross-tied furrow microcatchments, height differed significantly between species (Table 15), but not between catchment sizes. Height growth at 59 months in the catchment size of 30 m² averaged 1.7 m, whereas that of all other treatments ranged from 1.9–2.0 m. *Prosopis juliflora* (Pjb), the only exotic species in this experiment, grew best among the species and differed from all others ($p<0.05$); mean height ranged from 2.7–3.6 m, whereas in the other species above average, *A. zanzibarica*, *A. horrida* and *A. nilotica* the respective range was 2.3–3.0 m, 2.3–2.5 m and 2.1–2.8 m.

Statistically significant differences in mean diameter ($H = 9.19$; $df = 4$; $p = 0.046$) were detected as an effect of catchment size: the diameter in the 30 m² catchment was significantly smaller ($p<0.05$) than in all other treatments, except 100 m². The respective diameters were 3.6, 4.3, 4.7, 4.3 and 4.2 cm for 30, 40, 60, 80 and 100 m² of catchment size. Between species the differences were again highly significant, with means ranging from 2.0 cm (*Commiphora pseudopaolii*) to 6.7 cm (*P. juliflora*, Pjb seed source; Table 16).

Table 16. Cross-tied furrow microcatchments: Mean diameter ($dg_{0.15}$), mean stem number at stump height ($n_{0.15}$) and S.E. of 10 species 59 months after outplanting (n=20). ANOVA results are presented for each variable. Differences of mean values which are not followed by the same letter are statistically significant at $p<0.05$ as determined by Tukey's HSD-test.

Species (ranked by $dg_{0.15}$)	$dg_{0.15}$[1] cm	S.E.	Tukey's HSD	$n_{0.15}$[2]	S.E.	Tukey's HSD
Prosopis juliflora (Pjb)	6.7	0.3	a	2.1	0.1	b
Acacia horrida	5.9	0.3	ab	1.9	0.2	bc
Acacia nilotica	5.0	0.3	bc	1.6	0.1	bcd
Acacia zanzibarica	4.4	0.3	cd	1.6	0.1	cd
Acacia tortilis	3.5	0.2	de	2.5	0.1	a
Cordia sinensis	2.8	0.2	ef	1.6	0.2	bcd
Lawsonia inermis	2.7	0.2	f	3.0	0.1	a
Commiphora pseudopaolii	2.0	0.1	g	1.1	0.1	d
Terminalia brownii	–[3]			–[3]		
Azadirachta indica	–[3]			–[3]		

[1] $F = 45.21$; $df = 7/78$; $p = 0.000$
[2] $F = 27.80$; $df = 7/78$; $p = 0.000$

Table 17. Cross-tied furrow microcatchments: Mean usable tree volume (with S.E.), MAI of transformed volume (average usable volume of living trees multiplied by tree density and survival), and rain use efficiency (RUE) of 10 species 59 months after outplanting (n=20). Results of Kruskal-Wallis ANOVA are presented for volume. Differences of mean values which are not followed by the same letter are statistically significant at $p<0.05$ as determined by Mann-Whitney U-test.

Species (ranked by MAI)	Volume[1] dm^3tree^{-1}	S.E.	Tukey's HSD	MAI $m^3ha^{-1}a^{-1}$	RUE $kg\ ha^{-1}a^{-1}mm^{-1}$
Prosopis juliflora (Pjb)	11.9	1.1	a	2.11	6.25
Acacia zanzibarica	2.0	0.2	bc	0.32	1.89
Acacia nilotica	3.1	0.5	b	0.30	1.53
Acacia tortilis	0.9	0.2	c	0.17	1.16
Acacia horrida	2.1	0.3	b	0.15	1.47
Lawsonia inermis	0.7	0.1	cd	0.11	0.41
Cordia sinensis	0.9	0.1	cd	0.08	0.19
Commiphora pseudopaolii	<0.05	<0.05	e	<0.01	0.01
Terminalia brownii	–[2]	–[2]	–		
Azadirachta indica	–[2]	–[2]	–		

[1] $H = 94.72$; $df = 7$; $p = 0.000$
[2] no live trees

No statistically significant differences were detected in mean $n_{0.15}$ between catchment sizes with means ranging between 1.8 and 2.0, whereas at the species level highly significant differences were found (Table 16) in the cross-tied furrow microcatchments. There was no branching at ground level. *Lawsonia inermis* and *Acacia tortilis* had developed most stems ($n_{0.15}$) and differed from all other species with means of 3.0 and 2.5 respectively (Table 16).

Differences in the mean volume of a single tree were also significant between species (Table 17) and catchment sizes ($H = 10.76$; $df = 4$; $p = 0.029$).

Table 18. Cross-tied furrow microcatchments: Mean usable tree volume (V, dm³, with S.E.) 59 months after outplanting in *Acacia horrida*, *A. nilotica*, *A. tortilis*, *A. zanzibarica* and *Prosopis juliflora* grown with different catchment sizes (n=4).

Species	Catchment size									
	30 m²		40 m²		60 m²		80 m²		100 m²	
	V	S.E.	V	S.E.	V	S.E.	V	S.E.	V	S.E.
A. horrida	1.67	0.34	2.71	0.61	2.33	0.57	1.70	0.97	2.43	0.91
A. nilotica	1.82	0.61	4.20	1.75	3.01	0.64	3.72	1.47	2.37	0.92
A. tortilis	0.60	0.34	1.25	0.54	1.28	0.25	0.69	0.12	0.81	0.15
A. zanzibarica	1.43	0.17	2.16	0.39	3.24	0.58	1.46	0.39	1.59	0.71
P. juliflora	6.80	2.17	12.20	1.67	18.59	4.67	11.84	2.16	10.32	1.41

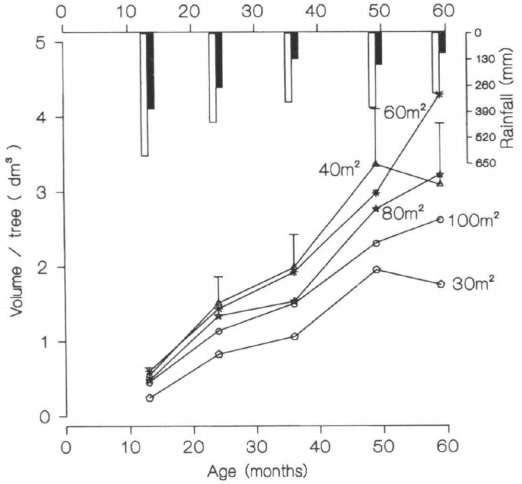

Fig. 24. Cross-tied furrow microcatchments: The effect of catchment size on the mean usable tree volume in all species 59 months after outplanting (n=4). For the sake of clarity, S.E. is displayed for the catchment size of 40 m² only.

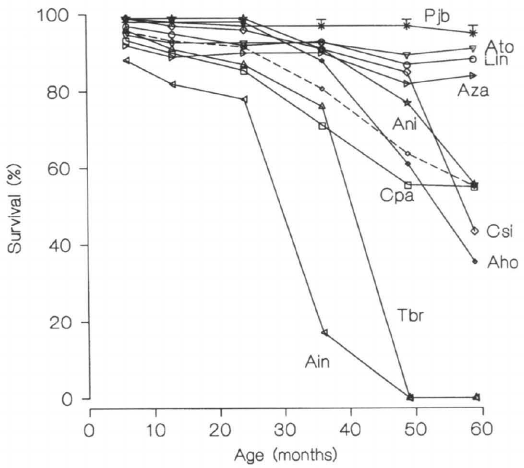

Fig. 25. Cross-tied furrow microcatchments: Mean tree survival in 10 species as a function of time (n=20). For the sake of clarity, S.E. is displayed only for Pjb. The dashed line displays the mean over all species. (Aho = *Acacia horrida*; Ain = *Azadirachta indica*; Ani = *Acacia nilotica*; Ato = *Acacia tortilis*; Aza = *Acacia zanzibarica*; Cpa = *Commiphora pseudopaolii*; Csi = *Cordia sinensis*; Lin = *Lawsonia inermis*; Pjb = *Prosopis juliflora*, local seed source; Tbr = *Terminalia brownii*).

Means of catchment size ranged from 1.8 dm³ (30 m²) to 4.3 dm³ (60 m²); the mean volume in the former catchment size differed ($p<0.05$) from that in 60, 80 or 40 m² catchments (Fig. 24). The mean volume in *P. juliflora* (Pjb) across all treatments 59 months after outplanting was approximately four times (11.9 dm³) that in *A. nilotica* (3.0 dm³), which rated next in the species ranking (Table 17).

The mean usable biomass (DM) for *P. juliflora* (Pjb), *A. nilotica*, *A. zanzibarica* and *A. horrida* were 9.1, 2.2, 1.7 and 1.6 kg tree⁻¹ respectively. Volume means of *P. juliflora* (Pjb) by catchment size varied between 6.8–18.6 dm³, while for the best three local species, *A. nilotica*, *A. horrida* and *A. zanzibarica* the range was 1.6–4.2 dm³ (Table 18). *A. horrida* and *A. nilotica* grew best in 40-m² catchment, while in *A. zanzibarica*, *A. tortilis* and *P. juliflora* the growth peaked at 60 m².

Mean transformed MAI peaked at 60 m² catchment size for *P. juliflora* (3.0 m³ ha⁻¹a⁻¹), but for *A. nilotica*, *A. horrida*, *A. tortilis* and *A. zanzibarica* the maximum growth was found in 40-m² catchments (Table 19). Corresponding MAI of usable biomass (DM) across all catchment sizes at 59 months was 1611, 281, 215, 126 and 110 kg ha⁻¹a⁻¹ for *P. juliflora*, *A. zanzibarica*, *A. nilotica*, *A. tortilis* and *A. horrida* respectively.

Table 19. Cross-tied furrow microcatchments: Mean MAI of transformed volume (average usable volume of living trees multiplied by tree density and survival; m³ha⁻¹a⁻¹), with S.E., 59 months after outplanting in *Acacia horrida*, *A. nilotica*, *A. tortilis*, *A. zanzibarica* and *Prosopis juliflora* in different catchment sizes (n=4).

Species	Catchment size									
	30 m²		40 m²		60 m²		80 m²		100 m²	
	MAI	S.E.	MAI	S.E.	MAI	S.E.	MAI	S.E.	MAI	S.E.
A. horrida	0.25	0.10	0.36	0.07	0.18	0.03	0.07	0.03	0.05	0.02
A. nilotica	0.41	0.25	0.58	0.33	0.32	0.10	0.40	0.21	0.13	0.08
A. tortilis	0.20	0.12	0.26	0.11	0.20	0.03	0.08	0.02	0.08	0.02
A. zanzibarica	0.40	0.08	0.50	0.11	0.43	0.08	0.17	0.05	0.16	0.08
P. juliflora	1.98	0.61	2.82	0.49	3.00	0.81	1.51	0.28	1.05	0.14

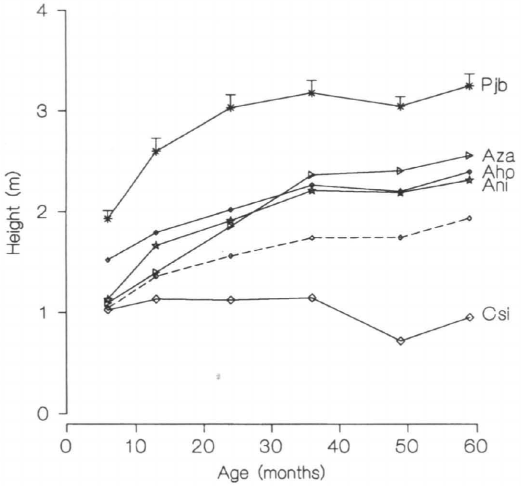

Fig. 26. Cross-tied furrow microcatchments: Mean tree height in five species as a function of time (n=20). For the sake of clarity, not all species are displayed; S.E. is presented for Pjb only. The dashed line represents the mean over all species. (Aho = *Acacia horrida*; Ani = *Acacia nilotica*; Aza = *Acacia zanzibarica*; Csi = *Cordia sinensis*; Pjb = *Prosopis juliflora*, local seed source).

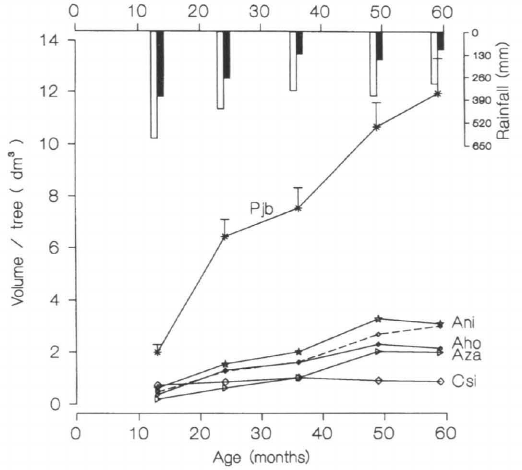

Fig. 27. Cross-tied furrow microcatchments: Mean usable tree volume in five species as a function of time (n=20). White and black bars represent the total rainfall and the proportion exceeding 30 mm month⁻¹ during the 12-month period prior to assessment. For the sake of clarity, not all species are displayed; S.E. is presented for Pjb only. The dashed line represents the mean over all species. (Aho = *Acacia horrida*; Ani = *Acacia nilotica*; Aza = *Acacia zanzibarica*; Csi = *Cordia sinensis*; Pjb = *Prosopis juliflora*, local seed source).

When the effects of catchment size were studied over time, the survival was found to be high in all species at year two, but it started to decline more rapidly thereafter. Throughout, it remained higher than in Experiments II and III (Fig. 25).

Height growth was rather fast during the first 13 months, particularly for *Prosopis juliflora*, but thereafter the increase in mean height declined and stabilised. Of the local species, *Acacia zanzibarica* exhibited the best growth (Fig. 26). Similar trend was apparent in volume growth over all catchment sizes: the volume growth of *P. juliflo-*

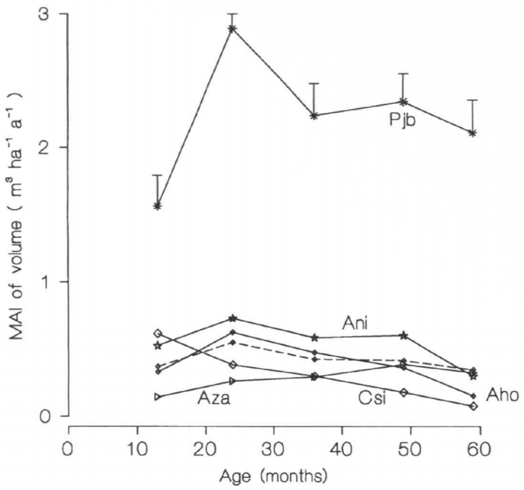

Fig. 28. Cross-tied furrow microcatchments: MAI of transformed tree volume (average usable volume of living trees multiplied by tree density and survival) in five species as a function of time. (n=20). For the sake of clarity, S.E. is presented for Pjb only. The dashed line displays the mean over all species. (Aho = *Acacia horrida*; Ani = *Acacia nilotica*; Aza = *Acacia zanzibarica*; Csi = *Cordia sinensis*; Pjb = *Prosopis juliflora*, local seed source).

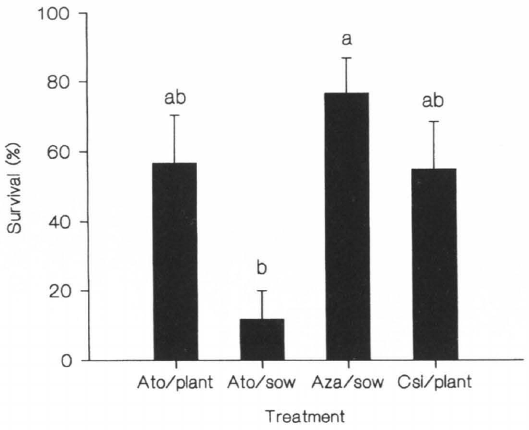

Fig. 29. Contour furrow microcatchments: Mean survival (with S.E.) of tree species in planting and sowing treatments at 13 months (n=3). ANOVA indicated significant differences between treatments ($F = 8.85$; $df = 4/8$; $p = 0.006$). Differences of mean values which are not designated with the same letter are statistically significant ($p<0.05$) as determined by Tukey's HSD-test. (Ato = *Acacia tortilis*; Aza = *A. zanzibarica*; Csi = *Cordia sinensis*).

ra was superior to that in all other species. Among the local species, *Acacia nilotica* performed best in volume growth (Fig. 27).

In viewing the effect of catchment size on tree volume in all species, it is noteworthy that the trees in the smallest catchment size (30 m^2) had the lowest mean volume throughout; these differences were statistically significant at 59 months. Furthermore, the volume growth in the two smallest catchment sizes (30 m^2, 40 m^2) stalled at 49 months, whereas that in 60, 80 and 100 m^2 catchments continued to increase (Fig. 24).

The mean MAI per unit area, when all catchment sizes were combined, ranged between 1.7–2.8 m^{3}ha^{-1}a^{-1} in *Prosopis juliflora* and between 0.1–0.7 m^{3}ha^{-1}a^{-1} in *A. nilotica*, *A. horrida* and *A. zanzibarica* (Fig. 28) during the experiment. RUE peaked in all the best species at two years, except for *A. zanzibarica* which reached its maximum (2.0 kg ha^{-1}a^{-1}mm^{-1}) at 49 months and declined slightly to 1.9 kg ha^{-1}a^{-1}mm^{-1}at 59 months. The RUE of *P. juliflora* increased rapidly to 6.7 kg ha^{-1}a^{-1}mm^{-1} at two years and remained over 6.7 kg ha^{-1}a^{-1}mm^{-1} thereafter, whereas that in *A. horrida* and *A. nilotica* decreased from 4.8 and 2.9 kg ha^{-1}a^{-1}mm^{-1} at two years to 1.5 kg ha^{-1}a^{-1}mm^{-1} at 59 months, mainly due to a decreasing survival rate.

3.1.5 Contour Furrow Microcatchments

Experiment V, carried out in order to clarify the effect of contour furrow microcatchments and fencing on the establishment of planted, sown or naturally regenerated seedlings, showed that the survival, height and diameter (dg_0) over all treatments, were 50 %, 28.2 cm and 6.7 mm respectively, excluding natural regeneration. There were no significant differences in any of the tested variables caused by protection or by the interaction between protection and establishment method. ANOVA of randomised blocks over the main plot effect detected significant differences in survival (Fig. 29), height and diameter (Table 20).

Table 20. Contour furrow microcatchments: Effect of stand establishment method (planting vs. sowing) on tree height and diameter (dg_0; with S.E.) at 13 months (n=3). ANOVA results are presented for each variable. Differences of mean values which are not followed by the same letter are statistically significant ($p<0.05$) as determined by Tukey's HSD-test.

Treatment (ranked by height)	Height^1 cm	S.E.	Tukey's HSD	dg_0^2 mm	S.E.	Tukey's HSD
C. sinensis, planted	47.7	4.7	a	11.2	1.2	a
A. tortilis, planted	37.0	5.2	ab	7.3	0.8	a
A. zanzibarica, sown	8.4	1.5	c	1.5	0.2	b
A. tortilis, sown	7.4	5.6	bc	3.5	2.0	ab

[1] $F = 19.57$; $df = 3/4$; $p = 0.007$
[2] $F = 29.23$; $df = 3/4$; $p = 0.004$

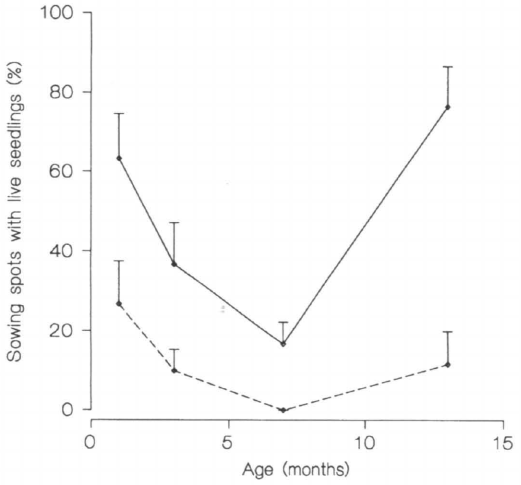

Fig. 30. Contour furrow microcatchments: Proportion of seeding spots with live seedlings (with S.E.) in *Acacia tortilis* (dashed line) and *A. zanzibarica* (solid line) as a function of time (n=3).

When planted, *Acacia tortilis* and *Cordia sinensis*, performed well in height and diameter growth (Table 20), whereas when sown *A. zanzibarica* ranked highest in survival at 13 months (Fig. 29). The germination was, however, not uniform over time (Fig. 30). Natural regeneration of the local species produced poor results: only one seedling of *Prosopis juliflora* (the local, naturalised seed source, Pjb) had emerged in 13 months. However, a total of 21 naturally regenerated seedlings of *P. juliflora* and one of *A. zanzibarica* were found in three separate plots (with a total area of 150 m^2) of the planting treatments.

3.2 Regeneration and Composition of Woody Vegetation in Cleared Bushland

The study on factors determining the pattern of regeneration on cleared bushland revealed that there were statistically significant differences ($p<0.05$) in the original soil types (ILACO 1977) and in soil characteristics caused by clearing (undertaken in 1982) as a function of time. (Table 21). The amount of organic carbon (C) increased significantly in VN and NG soils, but decreased in TC between 1988 and 1992. Organic C levels were also higher in VN and NG than in TC, particularly in 1992, although no significant differences were noted. The amount of total nitrogen (N) decreased in all soil types; the level was significantly lower in VN and TC in 1992 than in 1988 (Table 21).

TC exhibited the highest levels of total N in both assessments. Consequently, the C/N ratio increased in VN and NG over time, and the ratio was significantly lower in TC as compared to VN and NG both in 1988 and 1992. The phosphorus (P) levels generally decreased from 1988 to 1992; the P level was significantly lower in TC than in VN or NG in 1992. Based on the differences between the soil types the sample plots representing TC were excluded when comparing the biomass productivity between the cleared and non-cleared area.

Table 21. Soil nutrient characteristics of three different soil types (ILACO 1977) in cleared bushland, with S.E.(in parentheses), at 6.7 and 10.4 years after clearing (VN = Vertic Natrargid; NG = Natric Grumustert; TC = Typic Camborthid). Differences of mean values of the same year which are not followed by the same letter are statistically significant ($p<0.05$) as determined by Mann-Whitney's U-test. Asterisks (*) indicate statistically significant differences ($p<0.05$) of mean values between two assessments of the same soil type as determined by Tukey's HSD-test.

Variable	Unit	1988 VN n=46	1988 NG n=4	1988 TC n=4	1992 VN n=46	1992 NG n=4	1992 TC n=4
pH		7.80^{a} (0.33)	7.85^{a} (0.05)	7.83^{a} (0.11)	7.85^{a} (0.03)	7.65^{ab} (0.22)	7.45^{b*} (0.05)
Organic Carbon (C)	%	0.393 a (0.018)	0.410^{a} (0.050)	0.330^{a} (0.041)	0.511^{a*} (0.031)	0.663^{a*} (0.072)	0.236^{a} (0.170)
Total Nitrogen (N)	%	0.051^{a} (0.001)	0.047^{a} (0.003)	0.067^{b} (0.005)	0.039^{a*} (0.001)	0.033^{a} (0.005)	0.054^{b*} (0.002)
Potassium (K)	ppm				29.2^{a} (1.77)	36.3^{a} (6.38)	22.1^{a} (3.93)
Phosphorus (P)	ppm	3.01^{a} (0.18)	3.46^{a} (0.46)	2.62^{a} (0.76)	2.49^{a} (0.23)	2.83^{a} (0.64)	1.50^{b} (0.94)
Sodium (Na)	ppm	10.62^{a} (0.78)	11.90^{a} (0.96)	6.50^{a} (1.50)			
Calcium (Ca)	ppm				1605 a (33.1)	1696^{a} (135)	1468^{a} (389)
Magnesium (Mg)	ppm				401^{a} (9.9)	434^{ab} (62.1)	576^{b} (57.4)
Zinc (Zn)	ppm				4.53^{a} (0.21)	3.78^{a} (0.54)	3.70^{a} (0.29)
Iron (Fe)	ppm				88.3^{ab} (5.31)	120^{a} (12.2)	47.5^{b} (18.0)
Copper (Cu)	ppm				3.00^{a} (0.11)	2.88^{a} (0.24)	3.45^{a} (0.25)
Manganese (Mn)	ppm				258^{a} (18.6)	331^{a} (36.1)	357^{a} (50.4)
C/N -ratio		7.83^{a} (0.42)	8.75^{a} (1.17)	5.00^{b} (0.60)	14.1^{a} (0.99)	21.4^{a} (4.15)	4.44^{b} (3.21)

Characteristics of the two soil types (TN and VN) in the non-cleared area, studied by using the two-dimensional linear MDS ordination, showed clear differences: all TN plots formed a distinct group which was clearly different from another group formed by the VN plots (Fig. 31). Separate testing of the amounts of single nutrients did not reveal statistically significant differences (Table 22) between the two soil types.

The first MDS dimension separated the TN soil samples from those of VN, while the second dimension revealed differences among the VN soil samples (Fig. 31). The first MDS dimension was associated with a decrease in the contents of K, P and Fe and an increase in those of Ca and Mg, while the second dimension was associated with a decrease in C. Based on the soil types by ILACO (1977) and the MDS ordination, only the VN plots were included when comparing the regeneration and biomass production in the cleared and non-cleared areas.

The amounts of Fe and Cu were significantly higher in the VN soil samples in the cleared area 10.4 years after clearing, as compared to the non-cleared area.

A total of 11 and 16 species (including in some cases two different species when locally identified as one) were encountered in the cleared site in 1988 and 1992 respectively, whereas 18 species were identified in the undisturbed (control) site in 1992. *Acacia zanzibarica* was clearly the most abundant species, followed by *Acacia reficiens*, although the relative density of the former decreased in the cleared area between 1988 and 1992. The remaining species only accounted for 10.5 % of the total density in 1988, but increased to 22.7 % in 1992. In the undisturbed area *A. zanzibarica*, *Salvadora persica* and *Solanum incanum* were the most abundant species (Table 23).

Table 22. Soil nutrient characteristics of two different soil types, with S.E., in undisturbed bushland (VN = Vertic Natrargid; TN = Typic Natrargid) in 1992. *p* refers to the values obtained in the Mann-Whitney's U-test.

Variable	Unit	TN (n=2) mean	TN (n=2) S.E.	VN (n=5) mean	VN (n=5) S.E.	*p*
pH		6.95	0.45	7.74	0.051	0.051
Organic Carbon (C)	%	0.530	0.050	0.534	0.087	0.329
Total Nitrogen (N)	%	0.036	0.011	0.044	0.002	0.554
Potassium (K)	ppm	85.0	1.00	25.4	3.32	0.053
Phosphorus (P)	ppm	30.9	6.8	2.98	0.50	0.053
Calcium (Ca)	ppm	133	42.0	1706	100	0.053
Magnesium (Mg)	ppm	163	92.5	462	22.0	0.051
Zinc (Zn)	ppm	5.55	0.25	4.88	0.17	0.108
Iron (Fe)	ppm	135	25.0	62.0	10.2	0.051
Copper (Cu)	ppm	1.25	0.25	2.20	0.26	0.076
Manganese (Mn)	ppm	137	2.50	170	8.52	0.051
C/N -ratio		16.7	6.49	12.5	2.56	0.329

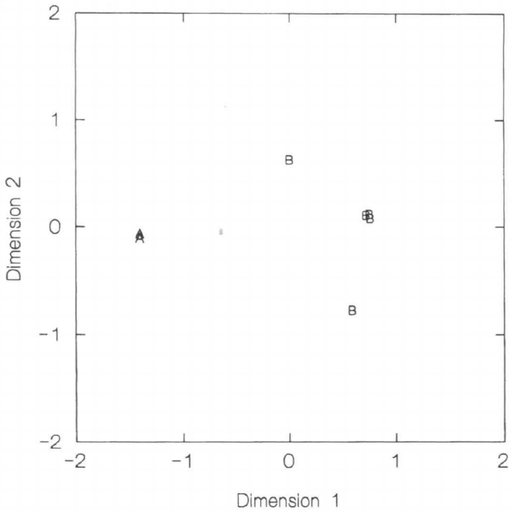

Fig. 31. Multidimensional Scaling Ordination of the first two axes describing the similarities /dissimilarities of the sample plots in the undisturbed bushland in relation to soil nutrients. A and B refer to plots on Typic Natrargid and Vertic Natrargid soil.

The mean height calculated from plot averages in the cleared area (NG and VN soil types only) increased from 1.68 m (n=50; S.E.=0.10) in 1988 to 2.62 m (n=50; S.E.=0.12) in 1992, while that for the control area (VN soil type only) averaged 2.89 m (n=5; S.E.=0.16) in 1992. The respective ranges were 0.45–2.93 m in 1988 (cleared), 0.85–4.25 m and 2.41–3.32 m (cleared and control plots) in 1992. The mean height in different species varied considerably (Table 24).

In the cleared area, the mean number of stems per tree at 0.15 m over all species was 1.32 (n=46; S.E.=0.05) in 1988 m and 1.50 (n=49; S.E.=0.06) in 1992. The number of stems at 1.3 m decreased from 2.31 (n=45; S.E.= 0.11) in 1988 to 1.85 (n=48; S.E.= 0.11) in 1992. The corresponding values for undisturbed control plots in 1992 (n=5) at 0.15 m and 1.3 m were 1.47 (S.E.= 0.16) and 1.57 (S.E.= 0.15).

In *Acacia zanzibarica*, the mean stem number at 0.15 m decreased slightly from 1.25 (n=44; S.E.= 0.04) to 1.16 (n=48; S.E.= 0.03) between 1988 and 1992, whereas the respective decline at 1.3 m was from 2.24 (n=44; S.E.= 0.11) to 1.92 (n=47; S.E.= 0.11). The corresponding means in the control site were 1.28 (n=5; S.E.= 0.09) and 1.55 (S.E.= 0.14) at 0.15 m and 1.3 m levels respectively in 1992.

The tree density increased from 1360 trees ha^{-1}

Table 23. Tree species composition in cleared bushland in 1988 and 1992 (6.7 and 10.4 years after clearing respectively) and in undisturbed bushland in 1992. The total sampled area was 0.6 and 0.05 ha for cleared and undisturbed bushland respectively. The soils in the cleared area comprised of Vertic Natrargid (VN), Typic Natrargid (TN), Natric Grumustert (NG) and Typic Camborthid (TC), while those of the undisturbed area included VN and TN only. Species totals refer in some cases to two different species when locally identified as one.

Species	Cleared (1988)		Cleared (1992)		Undisturbed	
	n	%	n	%	n	%
Acacia zanzibarica (S. Moore) Taub.	588	76.3	670	65.6	214	67.0
Acacia reficiens Wawra	102	13.2	119	11.7	2	0.6
Cephalocroton cordofanus Hochst.	48	6.3	88	8.6	4	1.3
Acacia mellifera (Vahl) Denth.	13	1.6	8	0.8	–	
Phyllanthus somalensis Hutch.	9	1.2	12	1.2	1	0.3
Salvadora persica L.	5	0.6	26	2.5	30	9.4
Grewia tenax (Forssk.) Fiori	2	0.2	1	0.1	16	5.0
Cadaba glandulosa Forssk.	2	0.2	1	0.1	3	0.9
Salsola dendroides Pall. var. *africana* Brenan	2	0.2	1	0.1	–	
Cordia sinensis Lam.	1	0.1	5	0.5	2	0.6
Maerua decumbens (Brongn.) De Wolf	1	0.1	4	0.4	2	0.6
Cadaba gillettii R.A.Graham / *Maerua macrantha* Gilg.	–		73	7.2	7	2.2
Asparagus africanus	–		8	0.8	1	0.3
Cadaba ruspolii Gilg.	–		2	0.2	2	0.6
Sericocomopsis pallida	–		1	0.1	1	0.3
Commiphora campestris Engl.	–		1	0.1	–	
Solanum incanum L.	–		–		28	8.8
Boscia coriacea Pax	–		–		3	0.9
Premna resinosa (Hochst.) Schauer	–		–		2	0.8
Commiphora pseudopaolii Gillett	–		–		1	0.3
Cadaba farinosa Forssk. / *Maerua triphylla* A.Rich.	–		–		1	0.3
Total abundance	773		1020		320	

(n=50; S.E.= 157) in 1988 to 1778 (n=50; S.E.= 156) in 1992 in the cleared area, while that of the undisturbed area was 5400 (n=5; S.E.= 1589) in 1992. In the former area the density ranged from 500 to 3500 trees ha^{-1} in 1988 and between 500–4000 trees ha^{-1} in 1992, with the highest number of trees found close to the southern outline of the site next to a dense natural stand dominated by *Acacia zanzibarica* (Fig. 32). In contrast, the highest densities in *A. reficiens* were obtained in the northern border of the area.

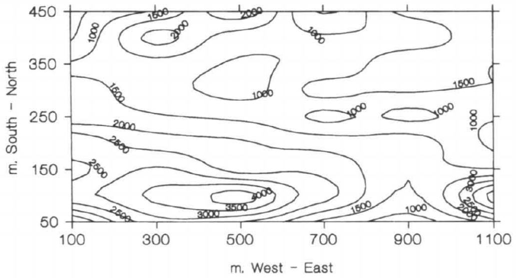

Fig. 32. Contour plot of tree density (trees ha^{-1}) in cleared bushland developed 10.4 years after clearing. Density interval 500–4000 ha^{-1}. Distances from the SW corner of the site.

Table 24. Mean tree height, with S.E., by species in cleared bushland in 1988 and 1992 (6.7 and 10.4 years after clearing) and in undisturbed bushland in 1992. Means are based on all observations across all soil types. Tree individuals at least 1.3 m in height of species marked with and asterisk (*) were included in the volume and biomass calculations. The soils in the cleared area comprised of Vertic Natrargid (VN), Typic Natrargid (TN), Natric Grumustert (NG) and Typic Camborthid (TC), while those of the undisturbed area included VN and TN only. Species refers in some cases to two different species when locally identified as one.

Species	Cleared (1988)			Cleared (1992)			Undisturbed		
	Height m	n	S.E.	Height m	n	S.E.	Height m	n	S.E.
Acacia zanzibarica *	2.05	588	0.05	3.17	668	0.05	3.22	214	0.07
Acacia reficiens *	0.94	102	0.04	1.57	119	0.06	2.15	2	1.15
Cephalocroton cordofanus *	0.54	48	0.02	1.02	88	0.04	1.58	4	0.45
Acacia mellifera *	1.05	13	0.13	1.70	8	0.26	–		
Phyllanthus somalensis *	1.12	9	0.11	1.49	12	0.09	1.40	1	–
Salvadora persica *	2.18	4	0.48	2.03	26	0.27	2.17	30	0.31
Grewia tenax *	1.20	2	0.70	0.40	1	–	1.89	16	0.15
Cadaba glandulosa *	1.40	2	0.70	0.50	1	–	0.60	3	0.15
Salsola dendroides *	1.60	2	0.15	1.60	1	–	–		
Cordia sinensis *	0.50	1	–	0.68	5	0.13	2.10	2	0.40
Maerua decumbens *	0.90	1	–	0.68	4	0.11	1.65	2	1.25
Cadaba gillettii / Maerua macrantha *	–			3.05	59	0.28	1.60	7	0.52
Asparagus africanus	–			1.00	6	0.14	1.40	1	–
Cadaba ruspolii	–			0.25	2	0.05	1.25	2	0.75
Sericocomopsis pallida	–			0.40	1	–	1.70	1	–
Commiphora campestris *	–			0.20	1	–	–		
Solanum incanum	–			–			0.76	28	0.09
Boscia coriacea *	–			–			3.47	3	0.61
Premna resinosa	–			–			2.00	2	0.50
Commiphora pseudopaolii *	–			–			4.60	1	–
Cadaba farinosa / Maerua triphylla *	–			–			0.60	1	–

The mean pooled diameter ($dg_{0.15}$) per plot over all species was 2.94 cm (n=46; S.E. = 0.12) in the cleared site in 1988 and 4.05 cm (n=49; S.E. = 0.16) in 1992, whereas that of the undisturbed site averaged 3.69 cm (n=5; S.E. = 0.25). In *Acacia zanzibarica* alone, the increase in diameter was fast, from 3.04 cm (n=44; S.E. = 0.13) in 1988 to 4.69 cm (n=47; S.E. = 0.23) in 1992. The respective mean diameter in the undisturbed site was lower, 3.75 cm (n=5; S.E. = 0.29).

The diameter distribution ($dg_{0.15}$; in 2-cm classes) of the two most abundant species in the cleared area in 1988 and 1992, *Acacia zanzibarica* and *A. reficiens*, are presented in Fig. 33.

Between 1988 and 1992, the biomass and volume of the cleared area increased more than threefold. In 1992, 10.4 years after bush-clearing, the mean total biomass and volume were 5626 kg ha^{-1} and 6.56 m^3 ha^{-1}; the respective means for usable parts were 2392 kg ha^{-1} and 2.78 m^3 ha^{-1}. Consequently, the MAI estimates also increased considerably between the two assessments (Table 25). Relative growth rates, expressed as the proportion of MAI to standing stock, were 14.7–15.1 % in 1988 and 9.5–9.7 % in 1992. The MAI variation within the site largely followed that of tree density; growth increased towards the S and SE edges which were located close to dense natural stands of *Acacia zanzibarica* (Fig. 34). This species constituted over 95 % of biomass and volume in the cleared area in 1988 and between 91–93 % in 1992, whereas in the undisturbed control area the proportion was only 69–77 % in 1992.

3.3 Composition and Dynamics of Woody Vegetation in Undisturbed Bushland

A total of 17 and 31 woody species above 1.3 m in height (including in some cases two different

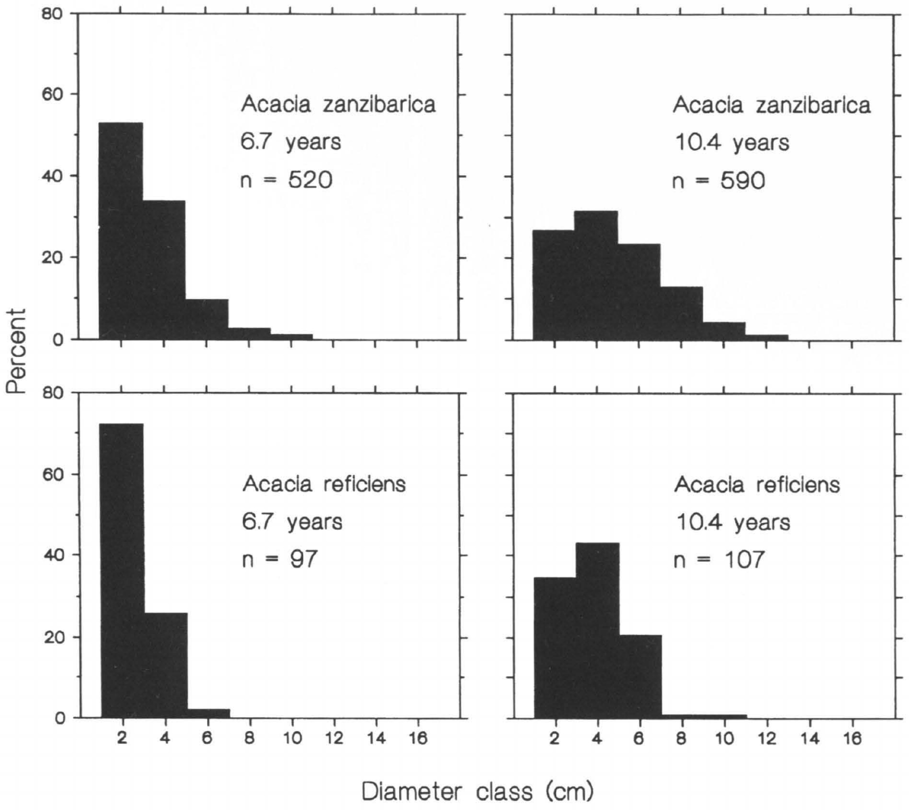

Fig. 33. Diameter ($dg_{0.15}$) class distribution (in 2-cm classes) of *Acacia zanzibarica* and *A. reficiens* in cleared bushland in 1988 and 1992 (6.7 and 10.4 years after clearing) on Vertic Natrargid and Natric Grumustert soils.

species when locally identified as one) were identified in the undisturbed bushland consisting of VN soil in 1985 and 1994 respectively. The woody vegetation was, however, dominated by one single species, *Acacia zanzibarica*, which accounted for 90.8 and 82.8 % of all observations in 1985 and 1994 respectively. Other important species in this soil type were *Salvadora persica* and *Grewia tenax* (Appendix 1). Canopy cover was casually estimated to vary between 50–80 %. The mean woody plant density per ha was 919 in 1985 and 1419 in 1994. Canopies were relatively low, and separate canopy layers could not be distinguished. The mean height was 3.1 m and 3.4 m for the two assessments. The proportion of trees exceeding 5 m in height was only 4.1 and 12.4 % in 1985 and 1994 respectively.

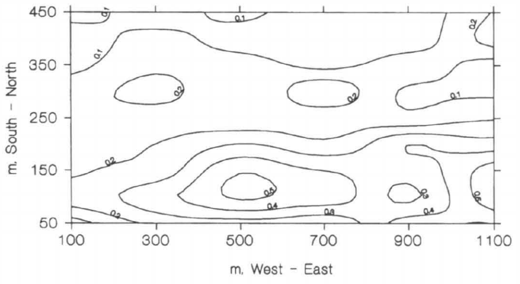

Fig. 34. Contour plot of MAI of usable tree volume (m^3 ha^{-1}a^{-1}) in cleared bushland 10.4 years after clearing. The plot scale refers to distances from the SW corner of the site (n=54).

Table 25. Mean tree biomass, volume and MAI, with S.E., in cleared bushland in 1988 and 1992 (6.7 and 10.4 years after clearing) and in undisturbed bushland in 1992 (n=50 and n=5 for cleared and undisturbed area respectively).

	Cleared[1] (1988)		Cleared[1] (1992)		Undisturbed[2]	
	mean	S.E.	mean	S.E.	mean	S.E.
ALL SPECIES						
Total biomass (kg ha^{-1})	1590	230	5626	546	15307	1459
MAI (kg ha^{-1}a^{-1})	238	35	540	52		
Usable biomass (kg ha^{-1})	587	88	2392	235	6467	399
MAI (kg ha^{-1}a^{-1})	88	13	230	23		
Total volume (m^{3}ha^{-1})	1.85	0.27	6.56	0.64	18.79	1.42
MAI (m^{3}ha^{-1}a^{-1})	0.28	0.04	0.63	0.06		
Usable volume (m^{3}ha^{-1})	0.68	0.10	2.78	0.28	8.02	0.51
MAI (m^{3}ha^{-1}a^{-1})	0.10	0.02	0.27	0.03		
Acacia zanzibarica						
Total biomass (kg ha^{-1})	1532	230	5164	530	11854	2617
MAI (kg ha^{-1}a^{-1})	230	34	496	51		
Usable biomass (kg ha^{-1})	571	87	2320	230	4793	971
MAI (kg ha^{-1}a^{-1})	86	13	214	22		
Total volume (m^{3}ha^{-1})	1.77	0.27	5.96	0.61	13.67	3.02
MAI (m^{3}ha^{-1}a^{-1})	0.27	0.04	0.57	0.06		
Usable volume (m^{3}ha^{-1})	0.66	0.10	2.57	0.27	5.53	1.12
MAI (m^{3}ha^{-1}a^{-1})	0.10	0.02	0.25	0.02		

[1] Natric Grumustert (NG) and Vertic Natrargid (VN) soil types
[2] Vertic Natrargid (VN) soil type only

Species distribution of seedlings (in 1994) followed the same pattern as that in taller trees, although the dominance of *Acacia zanzibarica* was less distinct; the species only accounted for 48.9 % (193 seedlings ha^{-1}) of all seedlings. The other species exceeding the mean density of 10 seedlings ha^{-1} were *Salvadora persica, Acacia mellifera, A. reficiens* and *Grewia tenax* (Fig. 36). Both *A. mellifera* and *A. reficiens* demonstrated a much higher abundance and relative density in seedlings than in tall trees. Regeneration was recorded for 27 species, five of which were absent in the above-1.3 m category. Mean seedling density per ha across all species was 395 (S.E. = 60) (Appendix 1).

The average size of woody plants in the VN soil type was small; over 50 % of all trees observed had a diameter below 4 cm both in 1984 and 1994. The mean diameter over all observations decreased slightly from 4.8 in 1985 to 4.7 cm in 1994. The increase in number of trees with a diameter between 4 and 12 cm was also conspicuous between the two assessments (Table 26). This may, however, be attributed to differences in the sampling between 1985 and 1994, particularly among the shrub species (cf. Methods). The diameter distribution ($dg_{0.15}$) of some of the most abundant species (those with n ≥ 30) in 1994 are presented in Fig. 35.

Between 1985 and 1994, the total mean volume of woody plants increased from 9.30 to 13.14 m^3 ha^{-1} and the corresponding biomass from 7495 to 10255 kg ha^{-1}. The respective estimates for usable components rose from 4.61 to 5.92 m^3 ha^{-1} and from 3712 to 4712 kg ha^{-1}. The mean annual growth was 3.3 % (0.15 m^3 ha^{-1}a^{-1}) and 3.0 % (111 kg ha^{-1}a^{-1}) in usable volume and biomass respectively for the nine-year period.

Acacia zanzibarica constituted over 60 % of the volume and biomass in 1994, and the proportion also grew from 1985 in relation to other species. The mean annual growth was 4.3 % and 4.4 % for usable volume and biomass between 1985 and 1994 (Table 27).

Table 26. Diameter ($dg_{0.15}$) class distribution of individual woody plants on a 4000-m transect in undisturbed bushland of two soil types in 1985 and 1994. Total percentages may differ from 100 % due to rounding.

Diameter class	Vertic Natrargid				Typic Natrargid			
	1985		1994		1985		1994	
cm	n	%	n	%	n	%	n	%
0–2	77	4.1	407	15.0	3	1.0	64	10.0
2–4	967	52.0	1000	36.8	42	14.5	134	20.9
4–6	505	27.2	712	26.2	50	17.3	115	17.9
6–8	135	7.3	320	11.8	30	10.3	86	13.4
8–10	70	3.7	137	5.0	28	9.6	79	12.3
10–12	47	2.5	63	2.3	18	6.2	40	6.3
12–14	25	1.3	34	1.2	30	10.4	20	3.1
14–16	11	0.6	21	0.8	24	8.2	22	3.4
16–18	9	0.5	6	0.2	17	5.8	16	2.5
18–20	3	0.2	5	0.2	8	2.7	13	2.0
20–22	3	0.2	6	0.2	7	2.4	10	1.6
22–24	–	–	2	0.1	8	2.7	7	1.1
24–26	1	0.1	3	0.1	4	1.4	10	1.6
26–28	1	0.1	–	–	6	2.1	5	0.8
28–30	–	–	2	0.1	4	1.4	3	0.5
30–32	–	–	1	< 0.05	2	0.7	3	0.5
32–34	–	–	1	< 0.05	1	0.3	3	0.5
34–36	–	–	1	< 0.05	3	1.0	1	0.2
36–38	–	–	–	–	1	0.3	–	–
38–40	–	–	–	–	1	0.3	1	0.2
40–45	1		1		2	0.7	6	1.0
45–50	1		–		1	0.3	1	0.2
>50	–	–	–	–	1	0.3	–	–
Total	1856	100	2722	100	291	100	639	100

Acacia tortilis and *Salvadora persica* also contributed substantially to volume production in VN soil type, although the proportion of *A. tortilis* decreased between the two assessments. This can be explained by the cutting of a few large individuals for fuelwood and forage by the local pastoralists.

The potential maximum growth, as assessed from individual trees, can be much higher (Table 28). Among the three species included, *Acacia zanzibarica* was best in relative and absolute growth in height, whereas it was outperformed by *Cordia sinensis* and *Salvadora persica* in diameter and volume growth. It is noteworthy that the latter two species grew much faster in diameter and volume in VN than in TN (Table 28).

A total of 25 and 39 species (including in some cases two different species when locally identified as one) were identified in TN soil in 1985 and 1994. The species distribution was more uniform as compared to that found on VN; the most frequently recorded species were *Grewia tenax* and *Acacia reficiens*, which formed 24.5 % and 18.7 % of all observations (of trees at least 1.3 m in height) in 1985 and 38.9 % and 15.5 % in 1994 respectively (Appendix 2).

Other abundant species included *A. tortilis*, *Cadaba farinosa / Maerua triphylla*, *Commiphora campestris*, *Cordia sinensis*, *Prosopis juliflora* (1994) and *Salvadora persica*. The canopy cover generally ranged between 10–30 % (casual estimation). Mean woody plant density increased from 102 in 1985 to 370 ha^{-1} in 1994; this increase in plant number was mainly attributed to a significant increase in the density of *Grewia tenax*, although differences in sampling between the two assessments may partially explain the increase (cf. Methods).

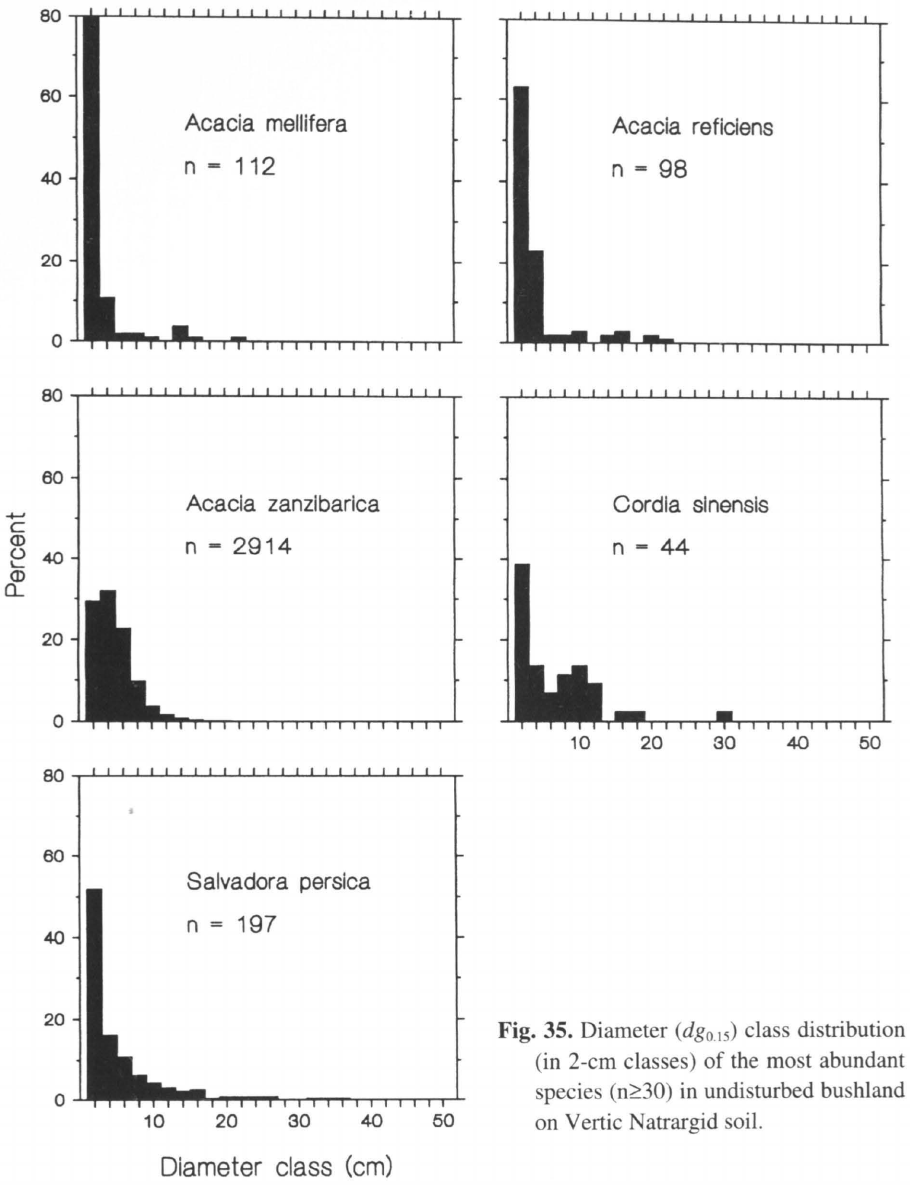

Fig. 35. Diameter ($dg_{0.15}$) class distribution (in 2-cm classes) of the most abundant species (n≥30) in undisturbed bushland on Vertic Natrargid soil.

Although the general pattern of the bushland vegetation on TN soil was patchy, it was possible in this case to distinguish two canopy layers. The upper layer (3–6 m) was mostly composed of *Acacia reficiens*, *A. senegal*, *A. tortilis*, *A. zanzibarica*, *Commiphora campestris*, *C. pseudopaolii* and *Terminalia orbicularis*, whereas the lower layer was dominated by *Grewia tenax*, accompanied by *G. villosa*, other *Grewia* spp., *Cadaba farinosa / Maerua triphylla* and *Cadaba gillettii / Maerua macrantha*. The tallest emergents of *Acacia tortilis* and *Commiphora campestris* attained the height of 10 m in 1994. The proportion of trees at least 5 m in height was 6 % in 1985 and 1994. The diameter distribution ($dg_{0.15}$) of the most abundant species (n ≥ 30) in 1994 are presented in Fig. 37.

Acacia reficiens featured conspicuously in the seedling category, making up to of 53.6 % (135 ha^{-1}) of all observations. *Grewia tenax* attained a seedling density of 49 ha^{-1}, whereas for all other

Table 27. Mean tree biomass, volume and MAI (1985–1994) with S.E,. in undisturbed bushland of two soil types in two successive inventories (n=105 and n=71 for Typic and Vertic Natrargid respectively).

	Vertic Natrargid				Typic Natrargid			
	1985		1994		1985		1994	
	Mean	S.E.	Mean	S.E.	Mean	S.E.	Mean	S.E.
ALL SPECIES								
Total biomass (kg ha^{-1})	7495	1131	10255	815	4457	753	5924	997
MAI (kg ha^{-1}a^{-1}) *			307				163	
Usable biomass (kg ha^{-1})	3712	768	4712	485	2012	402	2602	525
MAI (kg ha^{-1}a^{-1}) *			111				66	
Total volume (m^{3}ha^{-1})	9.30	1.50	13.14	1.12	8.39	1.55	11.65	2.24
MAI (m^{3}ha^{-1}a^{-1}) *			0.43				0.36	
Usable volume (m^{3}ha^{-1})	4.61	1.03	5.92	0.65	3.23	0.62	4.35	0.83
MAI (m^{3}ha^{-1}a^{-1}) *			0.15				0.12	
Acacia zanzibarica								
Total biomass (kg ha^{-1})	4907	516	6840	559				
MAI (kg ha^{-1}a^{-1}) *			215					
Usable biomass (kg ha^{-1})	2224	229	3113	254				
MAI (kg ha^{-1}a^{-1}) *			99					
Total volume (m^{3}ha^{-1})	5.66	0.60	7.89	0.64				
MAI (m^{3}ha^{-1}a^{-1}) *			0.25					
Usable volume (m^{3}ha^{-1})	2.58	0.26	3.59	0.65				
MAI (m^{3}ha^{-1}a^{-1}) *			0.11					

* Calculated for the period 1985–1994

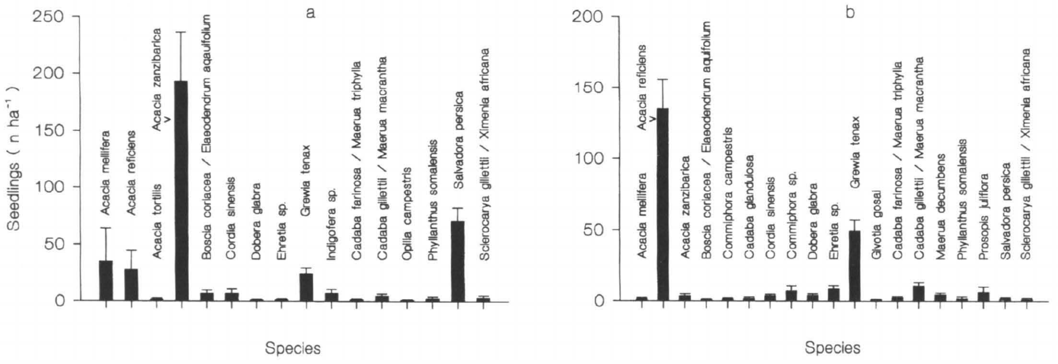

Fig 36. Seedling density (n ha^{-1}) in undisturbed bushland of Vertic Natrargid (a) and Typic Natrargid (b) soil types. Only species with a density above 1 ha^{-1} are presented. Some species include in some cases two different species when locally identified as one.

species the density remained below 10 ha^{-1} (Fig. 36). Seedlings were found in 28 species. Significant recruitment from the seedling category occurred also in this bushland type between the two assessments (Table 26). The number of trees increased considerably in all below 12 cm diameter classes, thereby contributing to a decrease in mean tree diameter, particularly in *Acacia reficiens*, *A. tortilis* and *Salvadora persica*. A few conspicuous *A. tortilis* and *Dobera glabra* and *S. persica* trees had also been cut between the two assessments.

On Typic Natrargid soil, the volume estimates

Table 28. Growth (S.E. in parentheses) and MAI of individual trees in undisturbed bushland on two soil types based on the individual trees identified in 1985 and 1994. Only species with a minimum of 10 observations are presented.

Soil type / Species	n	Mean height m 1985	Mean height m 1994	MAI %	MAI m	Mean diameter ($dg_{0.15}$), cm 1985	Mean diameter ($dg_{0.15}$), cm 1994	MAI %	MAI cm	Mean usable volume, dm^{3}tree^{-1} 1985	Mean usable volume, dm^{3}tree^{-1} 1994	MAI %	MAI dm^{3}tree^{-1}
VERTIC NATRARGID													
Acacia zanzibarica	11	3.9 (0.3)	5.5 (0.5)	4.4	0.17	9.1 (1.0)	11.2 (0.9)	2.2	0.23	18.2 (4.4)	30.0 (6.0)	7.2	1.3
Cordia sinensis	12	2.3 (0.1)	3.1 (0.2)	4.0	0.09	7.5 (1.0)	11.2 (2.0)	5.5	0.42	4.8 (1.2)	19.5 (11.6)	34.0	1.6
Salvadora persica	57	2.8 (0.1)	3.4 (0.1)	2.2	0.06	8.0 (0.8)	11.4 (1.0)	4.2	0.37	11.1 (2.5)	24.6 (5.6)	13.5	1.5
TYPIC NATRARGID													
Acacia reficiens	51	3.2 (0.1)	4.5 (0.2)	4.1	0.13	11.6 (0.6)	16.0 (0.7)	4.3	0.50	16.3 (2.9)	45.0 (6.5)	19.6	3.2
Acacia tortilis	18	4.2 (0.5)	5.3 (0.7)	2.8	0.13	16.5 (3.1)	18.7 (3.7)	1.5	0.24	243.4 (87.4)	360.5 (134.6)	5.3	13.0
Commiphora africana	12	3.0 (0.3)	3.9 (0.2)	3.3	0.10	9.3 (2.0)	12.8 (2.1)	4.2	0.39	6.7 (3.3)	17.0 (8.4)	17.1	1.1
Commiphora campestris	18	3.7 (0.3)	5.1 (0.5)	4.3	0.16	23.2 (2.7)	29.2 (2.7)	2.9	0.67	188.9 (60.7)	343.4 (79.1)	9.1	17.2
Commiphora pseudopaolii	10	3.2 (0.3)	4.7 (0.4)	5.0	0.16	13.6 (1.7)	16.9 (1.8)	2.7	0.37	26.0 (11.6)	49.1 (18.1)	9.9	2.6
Cordia sinensis	21	2.2 (0.1)	2.5 (0.1)	1.5	0.03	7.0 (0.5)	8.2 (0.6)	1.9	0.13	3.8 (0.6)	5.7 (1.2)	5.6	0.2
Salvadora persica	13	3.5 (0.3)	4.4 (0.3)	3.0	0.10	13.5 (2.8)	14.5 (2.1)	0.8	0.12	42.2 (19.9)	53.3 (11.0)	2.9	1.2

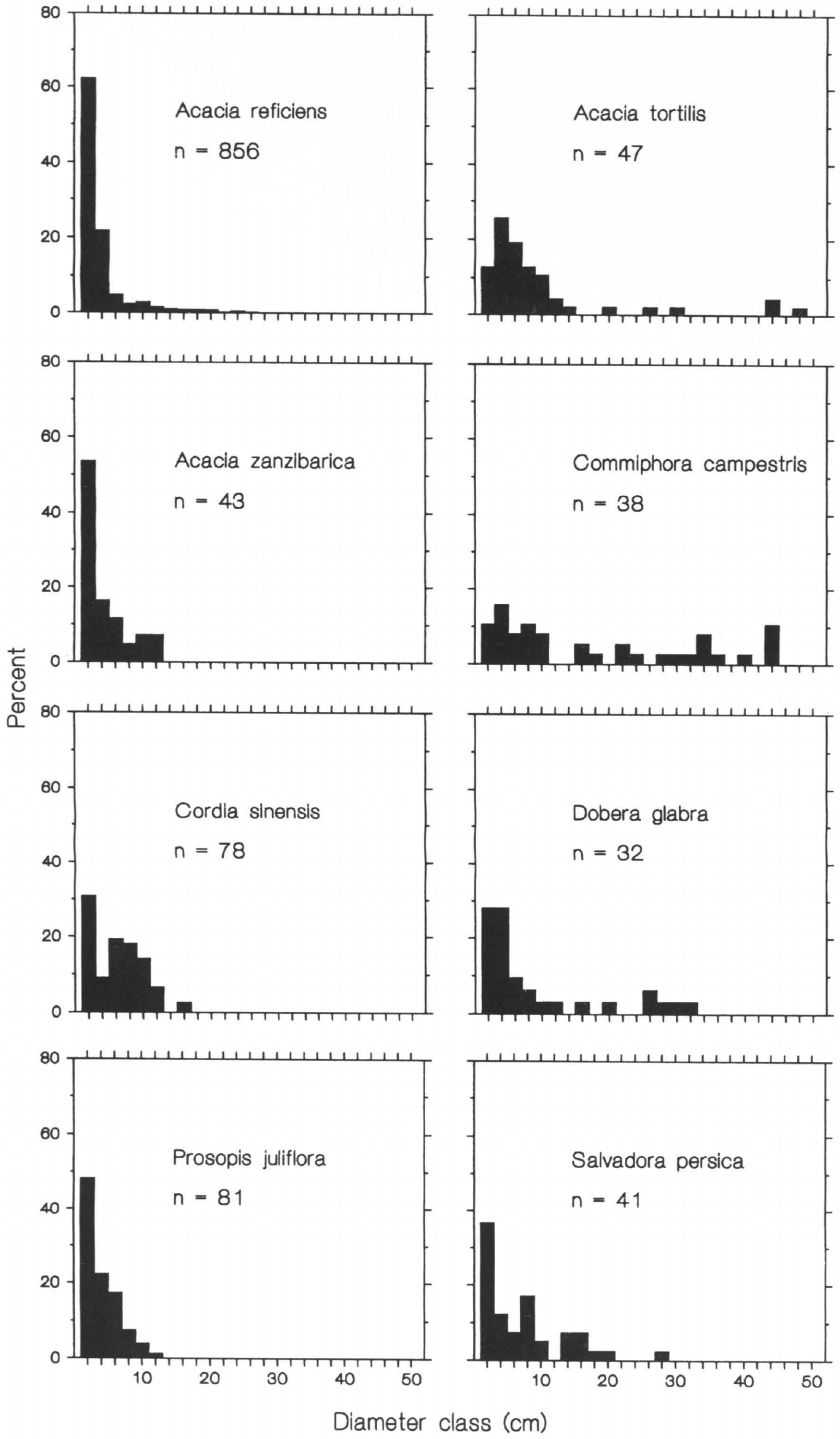

Fig. 37. Diameter ($dg_{0.15}$) class distribution (in 2-cm classes) of the most abundant species ($n \geq 30$) in undisturbed bushland on Typic Natrargid soil.

Table 29. Soil seed bank density (no. of identified seed with S.E.) in 0–3 cm topsoil layer from two vegetation types (n=4).

Vegetation and soil type	Germinated seed Species	Germinated seed seeds m^{-2} (S.E.)	Ungerminated seed Species	Ungerminated seed seeds m^{-2} (S.E.)	Total seeds m^{-2} (S.E.)
Acacia-Commiphora bushland	*Acacia zanzibarica*	0.2 (0.1)	*C. sinensis*	0.4 (0.4)	
	Cordia sinensis	0.2 (0.2)			
Typic Natrargid	*Indigofera* spp.	39.2 (25.9)			
	Ocimum spp.	0.1 (0.1)			
	Total woody species	39.7 (26.3)		0.4 (0.4)	40.1 (26.7)
	Grasses, herbs	257 (112)		n.a.	257 (112)
Transitional forest[1]	*A. zanzibarica*	2.5 (2.0)	*A. zanzibarica*	3.4 (2.7)	
	Indigofera spp.	6.2 (3.8)	*C. sinensis*	0.3 (0.3)	
different soils	*Sesbania quadrata*	0.1 (0.1)	*Oncoba spinosa*	0.1 (0.1)	
	Total woody species	8.8 (4.7)		3.8 (2.7)	12.6 (6.5)
	Grasses, herbs	60.2 (19.6)		n.a.	60.2 (19.6)

[1] Vegetation type between *Acacia-Commiphora* bushland and Tana riverine forest

were lower than in the previous VN soil type. The mean usable volume was 3.23 m^3 ha^{-1} in 1985, and it increased to 4.35 m^3 ha^{-1} in 1994. The mean usable biomass increased from 2012 to 2602 kg ha^{-1} respectively. The respective mean annual growth rates for volume and biomass were 3.7 % and 3.2 % for the nine-year period. *Acacia tortilis*, *A. reficiens* and *Commiphora campestris* constituted 72 and 77 % of the total usable volume in 1985 and 1994. In biomass, the proportion of *C. campestris* and other *Commiphora* spp. was less significant, due to the low specific gravity as compared to acacias (Appendix 2).

The growth rate of individual trees was much higher (Table 28). *Commiphora pseudopaolii* and *C. campestris* grew best in height, followed by *Acacia reficiens*. The best growth in diameter was attained in *A. reficiens* and *C. campestris*. *A. reficiens* and *C. africana* exhibited the highest relative volume growth, whereas *C. campestris* and *Acacia tortilis* grew best in absolute volume.

3.4 Seed Bank in Undisturbed Bushland

Seed densities of woody species in the topsoil (0–3 cm) averaged 40.1 and 12.6 seeds m^{-2} for *Acacia-Commiphora* bushland and the transitional forest respectively (Table 29). Most of the seeds belonged to shrubs (*Indigofera* spp., *Ocimum* spp., *Oncoba spinosa*); only two species of noteworthy trees, *Acacia zanzibarica* and *Cordia sinensis*, were found. The seed densities for herbs and grasses based on emerged plants only are gross underestimations, since it was not possible to separate and identify the ungerminated seeds in the samples. The ratio between germinated seeds of woody and non-woody species was approximately 1 : 6 in both vegetation types.

4 Discussion

4.1 Microcatchment Water Harvesting

4.1.1 Comparison Between the Methods

The study demonstrated the potential of rainwater harvesting in improving the survival and growth of planted trees in semi-arid conditions. Mean survival over all species was higher using the deep and shallow triangular and particularly the cross-tied furrow microcatchments (Table 30), as compared to the standard long-furrow method in which the trees perished completely one year after planting. The observation period in contour furrow microcatchments was apparently too short to assess the applicability of the method.

The survival rates compare well with other water harvesting studies. Sheikh et al. (1984) tested various methods in Pakistan with 250 mm of annual rainfall: the average rates of eight xerophytic species were 64–76 % 18 months after planting. Helin (1989) reported an average of 80 % at five years for ten best species or provenances among 101 tested, with equal survival on fine loams (Typic Haplargids) and Vertisols in Somalia with 290 mm of annual precipitation.

The growth rates of the best species, *Prosopis juliflora*, obtained under cross-tied furrow catchments in the current study, compare well with previous studies, but those in the other species and designs are lower than those reported earlier. The average height in eight species reported by Sheikh (1988) in the two best methods, sloping catchment with and without a trench 0.3 m deep, were 3.6–4.2 m at six years, while the maximum height was 3.5 m in the current study at 59 months (Table 30). Gupta (1994) recorded mean heights of 2.2–3.2 m in *Azadirachta indica* under various water harvesting designs at 26 months.

MAI of total biomass (above stump) in *Prosopis juliflora* in cross-tied furrows over all spacings 59 months after planting averaged 4.2 kg tree^{-1}a^{-1}, and that in the best treatment was 6.5 kg tree^{-1}a^{-1}. Brunori et al. (1995) recorded a MAI of total biomass of 13.0 and 6.5 kg tree^{-1}a^{-1} at seven years in *Eucalyptus sargentii* and *E. occidentalis* planted along contour ridges with average annual rainfall of 220 mm. Gupta (1994) obtained a MAI of 2940 kg ha^{-1}a^{-1} for the same biomass component in *Azadirachta indica* under long-furrow catchments at 26 months with 300 mm of annual rainfall. The respective MAI in *P. juliflora* across all spacings in cross-tied furrow microcatchments in the current study at 59 months averaged 2790 kg ha^{-1}a^{-1}, while that in the best spacing was 5120 kg ha^{-1}a^{-1}.

The best RUE values of the current study also compare well with other studies. Gupta (1994) recorded a RUE of 3.5–6.8 kg ha^{-1}a^{-1} mm^{-1} in *Azadirachta indica* under different water harvesting methods at 26 months. *Prosopis juliflora* maintained mean RUE rates of up to 8.9 kg ha^{-1}a^{-1} mm^{-1} for 59 months (Table 30), whereas the best indigenous species, *Acacia horrida*, *A. nilotica* and *A. zanzibarica* reached mean rates of 2.6–2.8 kg ha^{-1}a^{-1} mm^{-1} at their best.

When comparing the results in the two triangular microcatchment designs under the same soil type (NG), it is noteworthy that survival was lower in shallow catchments than in deep ones, although the latter experiment was established one year earlier. The deep catchments similarly produced better results in height, diameter and consequently volume growth. The differences may be attributed to a bigger volume, and subsequently a higher water collection and storage capacity of the deep catchments (approximately twice that of the shallow ones). The amounts of precipitation did not differ markedly between the experiments. In the first year following planting the total and effective rainfall (expressed as the amount exceeding 30 mm month^{-1}) in deep catchments were 709 and 502 mm, and 664 and 432 mm in shallow ones respectively. For the entire monitoring period, the total and effective rainfall were 2779 and 1617 mm in 70 months in deep and 2069 and 1114 mm in 59 months in shallow catchments.

Table 30. Mean values over all and the best single species (as assessed by MAI of transformed volume: average usable volume of living trees multiplied by tree density and survival) planted under different microcatchment water harvesting designs on clay and sandy clay soils in Kenya. In cross-tied furrows microcatchments also the best species under the best single catchment-to-cultivated area ratio is displayed.

Microcatchment type, depth and size	Species	Trial age months	Spacing n ha^{-1}	Soil type	Survival %	Height m	Volume* dm^{3}tree^{-1}	Biomass* kg tree^{-1}	MAI of volume^1 m^{3}ha^{-1}a^{-1}	MAI of biomass^1 kg ha^{-1}a^{-1}	Rain use efficiency^2 kg ha^{-1}a^{-1}mm^{-1}
Deep triangular, 50–70 cm, 20–25 m^2	All	70	100	clay	41	2.1	3.7	3.0	0.024	18	0.115
	Acacia horrida				97	3.3	5.1	3.7	0.085	61	0.777
Shallow triangular, 30–40 cm, 15–20 m^2	All	59	100	clay	33	1.5	0.8	0.6	0.005	4.6	0.038
	Acacia xanthophloea				69	2.1	1.9	1.6	0.027	23	0.175
Cross-tied furrows, 30–35 cm, 6–20 m^2	All	59	500–1667	sandy clay	55	1.9	3.0	2.2	0.348	246	1.291
	Prosopis juliflora		500–1667		95	3.3	11.9	9.1	2.111	1611	6.248
	Prosopis juliflora		833		95	3.5	18.6	14.2	2.997	2287	8.870

[1] refers to usable volume or biomass (above 2 cm in diameter)
[2] refers to total volume above stump height (0.15 m)

These rainfall amounts are, however, above the average of the area (372 mm a^{-1}), especially at the beginning of the experimental period.

The trials using furrowed microcatchments (long-furrow and cross-tied furrow) were both undertaken on a sandy clay soil (TH), where water percolation into the soil is estimated to be slightly faster than on clay (NG). In the latter type the remoistening of the soil is mainly caused by water that runs into the cracks rather than from water that percolates through the soil (ILACO 1977). This renders NG prone to waterlogging, which affects the species in a different way according to their tolerance to this property. The complete failure in the long-furrow design was most likely caused by the insufficient water collection capacity of the furrows rather than insufficiency in rainfall, which amounted 709 mm during the experimental period. In the long-furrow design the catchment area per planted seedling, assuming that rainwater that fell outside the furrow was not utilised, was only 1.8 m^2, as compared to that in cross-tied furrow catchments (6–20 m^2) and that in deep (20–25 m^2) and shallow triangular catchments (15–20 m^2).

In addition, planting position on the bottom of the furrow in the long-furrow design, as opposed to the shoulder of the furrow in cross-tied design, may have contributed to the poor performance, although the soil type was similar. However, neither the planting position nor the soil type is a likely cause for a complete failure of a wide range of species, several of which have performed well in other water harvesting designs under the same soil type in the present study and in similar conditions elsewhere in Kenya.

Boers (1994) indicated that rainwater harvesting is best suited on a soil with a low infiltration rate ensuring a maximum runoff supply, but it also requires a soil with a high infiltration rate to allow for the runoff water to infiltrate and prevent waterlogging over an extended period. In designs, where the water contributing and infiltration areas are clearly separate, such as along slopes, the former soil type would be ideal for the collection and the latter for the cultivation area. In MCWH, where the catchment size is small, and the contributing and infiltration areas are not separated, such as in the methods tried in this study, the ideal conditions can rarely be met. Rocheleau et al. (1988) regarded loams and sandy loams as the most suitable soil types for MCWH.

4.1.2 Spacing in Cross-Tied Furrow Catchments

All the major species in the cross-tied furrow microcatchment (*Acacia horrida*, *A. nilotica*, *A. tortilis*, *A. zanzibarica* and *Prosopis juliflora*) obtained the lowest volumes (per tree) in the 30 m^2-catchments with the lowest catchment-to-cultivated area ratio (3 : 1), indicating that the water collected was insufficient thereby limiting the growth. *A. horrida* and *A. nilotica* grew best, when the ratio was 4 : 1 (catchment size 40 m^2). The tree volume was highest in 60 m^2-catchment (ratio 6 : 1) in *P. juliflora* and *A zanzibarica*, which indicated that these species were better able to utilise the (additional) water supplied by a bigger catchment. *A. tortilis* grew almost equally in catchments of 40 and 60 m^2.

In the light of the current results, increasing the catchment size above 60 m^2 will not be beneficial for volume growth in any of the species; this feature will be superimposed since the growth per unit area will inevitably decline due to increased catchment-to-cultivated area ratio and consequently decreased spacing, unless compensated by increased per tree volume. The increasing competition for resources, particularly water, between the trees within a catchment (plot) will reduce the growth and may eventually increase mortality (Schönau and Coetzee 1989). This is demonstrated by the decrease in mean tree volume after 49 months in the two smallest catchments (30 m^2, 40 m^2).

Boers (1994) assumed that roots generally develop vertically rather than horizontally under rainwater harvesting, but (Gupta 1994, 1995), in contrast, found the main effect on lateral roots. On the other hand, there is also evidence that tree roots often align with the furrows under irrigated systems, both with excessive and insufficient watering (Armitage 1985), resulting in limited root growth in depth. Since the furrow design in the cross-tied microcatchments resembled that applied in continuous irrigation, it is possible that the root development mainly occurred along the furrows, thereby superimposing the effect of within-plot competition among the species with good growth. Low infiltration rate or the presence of impermeable layers limiting vertical water or root distribution have the same effect. Although many of the species included in the current study, such as *Acacia tortilis* and *Prosopis juliflora*, are capable of extending the deep tap root to reach groundwater in suitable conditions (Fagg and Stewart 1994), such development may be hindered after several years of mainly lateral root growth under rainwater harvesting.

Table 31. Classification of 14 species for afforestation under microcatchment water harvesting on heavy soils in arid and semiarid lowlands of Kenya. Species in parentheses were only included in one of the long-term experiments lasting at least 59 months.

Survival, %	Growth^1, dm^{3}tree^{-1}a^{-1} Good (>1.0)	Moderate (0.4–1.0)	Slow (< 0.4)
High (> 60)		*Acacia horrida* *A. zanzibarica*	*Acacia mellifera*
Moderate (20–60)	*Prosopis juliflora* (local seed source)	*Acacia tortilis* *A. nilotica*	*Acacia xanthophloea* *Commiphora paolii* *Cordia sinensis* *Lawsonia inermis* *Parkinsonia anacantha*
Poor (< 20)	*Prosopis juliflora* (Mombasa, Kenya seed source)		*Acacia nubica* (*Azadirachta indica*) (*Terminalia brownii*)

[1] Mean annual increment of usable volume, *i.e.* woody parts above 2 cm in diameter

4.1.3 Species Selection

Species selection for tree planting in drylands is a compromise between good growth with higher risk and slower growth but good survival. The species included in the current study have been grouped according to their survival and growth during the first 5–6 years (Table 31). From the survival viewpoint, indigenous *Acacia horrida*, *A. mellifera* and *A. zanzibarica* are most promising for planting with water harvesting in the area or in similar conditions. If both survival and yield are considered, the local seed source of introduced *Prosopis juliflora* is superior to all other species. This is consistent with previous experience in dryland afforestation in that the indigenous species generally have a high survival, but the best introduced species show a faster growth rate than the local ones.

Despite the performance of *Acacia nilotica* and *A. tortilis* was slightly inferior, as compared to the best species, these two can also be recommended for use in MCWH due to their highly valued products, such as browse, fodder, fuelwood and charcoal, in many land-use systems in ASAL in Africa (e.g. Fagg and Greaves 1990a, b, NFTA 1991, 1992, ICRAF 1992). Young plants of *A. nilotica* do not compete well, which may necessitate weeding (ICRAF 1992). The slow growth of *A. tortilis* can be considerably improved by using selected seed (von Maydell 1986). *A. tortilis* tolerates exceptional droughts, but generally prefers sandy, well-drained soils (NFTA 1991).

The poor performance of *Terminalia brownii* and *Azadirachta indica* was surprising. *T. brownii*, which is indigenous to the area, is the commonest and most wide-spread *Terminalia* sp. in Kenya. The species prefers well-drained soils (ICRAF 1992), which may explain its poor survival in the current study. *A. indica* is renowned for its drought-tolerance and suitability for rehabilitation of degraded sites, such as saline, shallow or nutrient-deficient soils (von Maydell 1986, ICRAF 1992, Gupta 1994). The species grew well during the first two years under cross-tied furrow catchments, but thereafter the survival declined rapidly.

Kimondo (1991) attained similar ranking of *Acacia nilotica*, *A. tortilis*, *A. mellifera*, *Cordia sinensis* and *Terminalia brownii* at two years using pit planting in Turkana, Kenya with 450 mm of average annual rainfall. *Prosopis juliflora* has also performed well on sand and clay under MCWH in Somalia up to three years (Mikkola and Jama 1991) and tolerated extreme aridity and goat browsing in Cape Verde (Sandys-Winsch and Harris 1992, Pasiecznik et al. 1995).

Mostly indigenous species have previously been recommended for planting using MCWH or other water conservation measures due to their good adaptation in prevailing conditions (von Maydell 1986, Rocheleau et al. 1988), higher survival (Mikkola and Jama 1991) or higher WUE (Tenbergen et al. 1995), as compared to exotics. Several introduced species have also performed well in Somalia and Kenya, such as *Parkinsonia aculeata*, *Prosopis chilensis*, *Prosopis juliflora* and *Senna atomaria* (Wieland et al. 1985, Rocheleau et al. 1988, Klein et al. 1990, Mikkola and Jama 1991).

Azadirachta indica exhibited good growth and survival both within (Gupta 1994) and outside the area of natural distribution (Boers 1994). According to Boers (1994), successful species for rainwater harvesting are simply deep-rooting and drought-resistant, such that they tolerate both short and prolonged spells of drought (Wairagu 1991).

Yield and survival are not the sole determinants, but species selection depends also on the expected end-use of the tree products, and on how well they fit in to the local land-use system, which is primarily pastoralism in the arid and semi-arid areas. The preferences of the people originating from other areas or ethnic groups are likely to differ from (Hughes 1987) or even contradict with those of the local herders. A mixture of people with various backgrounds and preferences exists e.g. in the current study area, which is located within a large irrigated agricultural and settlement scheme surrounded by dry bushland.

Among the species included in this study at least three indigenous, *Acacia tortilis*, *Cordia sinensis*, *Lawsonia inermis* and introduced *Prosopis juliflora* are used for fodder in the Bura area (Anttila et al. 1993). *A. tortilis* and *C. sinensis* have also been favoured for planting among the local people in other ASAL areas in Kenya (Rocheleau et al. 1988). Kireger and Blake (1994) recommended *A. tortilis* for planting in Baringo, Kenya. This was based on the better biomass production, WUE and survival of *A. tortilis* over *A. nilotica*, *A. xanthophloea* and *Faidherbia albida*. The observation period, however, covered only the first six months after planting, which renders the results merely tentative.

The invasion of exotic species following introductions may have reflections to the ecology and management of natural vegetation. Promotion of new species may reduce local use of, and thereby traditional knowledge about, the local flora, thus speeding up the loss of species diversity (Fagg and Stewart 1994). *Prosopis juliflora*, which exhibited the highest biomass growth in this study, is well-known for its invasiveness in the study area (Kaarakka 1993a, b) and elsewhere (e.g. Hughes and Styles 1987, Poynton 1990, Stewart et al. 1993).

The negative effects of invasion by *P. juliflora* are apparent (Reetu Sharma and Dakshini 1991, Kaarakka 1993a, Stewart et al. 1993, Hughes 1994, Fagg and Stewart 1994). Nevertheless, the spreading of this species has also been considered beneficial, especially when it occupies waste or salt-affected lands, or areas devoid of natural vegetation (Verma 1987, Singh 1995, El Fadl 1996), which are now common in ASAL in many countries.

The spread of *Prosopis juliflora* in the Bura area is mainly attributed to grazing of livestock within the irrigation network of BISP. Similar spreading is unlikely to occur from trees planted on sites far from irrigation schemes and other permanent bodies of water. *P. juliflora* can hence safely be planted using MCWH outside such areas, but only after alternative indigenous species have been found inferior for the particular service or end-use. *P. juliflora* does not require protection from cattle per se, since it is one of the few species the leaves and shoots of which are not consumed. However, since the pods are eaten and the seeds are spread by livestock, it may be necessary to restrict the access of animals.

Prosopis juliflora and other leguminous trees can considerably ameliorate soil properties, e. g. by increasing biological activity in the soil, through tree roots, litter fall, possible nitrogen fixation (cf. above) and by decreasing high pH on alkali soils (e.g. Singh 1995), such that the decrease in pH is faster than with non-leguminous trees (Singh et al. 1990, Gill and Abrol 1991). El Fadl (1996) indicated that the most important effect of *P. juliflora* is the production of litter. The beneficial effects of the trees may, however, be very local. Breman and Kessler (1995) concluded for Sudan and Sahel zones of West Africa, that the fertility improvement by woody plants for the whole region is very limited, probably concentrating on the build-up of organic matter in the soil and protective aboveground biomass.

4.2 Regeneration and Seed Banks of Natural Vegetation

The total density of seeds of all woody species in the soil was 40.1 seeds m^{-2} in the *Acacia-Commiphora* bushland (under TN soil) and 12.6 seeds m^{-2} in the transitional forest. Almost all seeds in the bushland (39.3 seeds m^{-2}) and approximately half of them in the transitional forest (6.4 seeds m^{-2}) belonged to shrubs. The only notable representative of trees in the seed bank was *Acacia zanzibarica*, which amounted for 5.9 seeds m^{-2} in the transitional vegetation zone. In addition, the seed density of *Cordia sinensis* was 0.6 and 0.3 seeds m^{-2} in the bushland and the transitional forest respectively. The absence of *Commiphora* seeds in the samples probably resulted from heavy predation by granivores (cf. Agnew et al. 1986), which can easily detect the large-sized seeds.

No previous records were available on the seed bank of *Acacia zanzibarica*. Johansson and Kaarakka (1992) assumed that seed stored in the soil may have an effect on regeneration, but could not establish the presence of a seed bank. Results of the current study verified that a viable seed bank of *A. zanzibarica* exists in the transitional forest, where this species is only one of the dominant species. Hence it may be assumed that there were viable seeds in the soil at the time of the vegetation removal or at least some seeds were dispersed at bush-clearing. It is therefore very likely that at least part of the regeneration in the species in the cleared bushland has come from seeds – either from the seeds that were dispersed at bush-clearing or from the stands adjacent to the cleared area. The seeds of *Acacia* and other dryland species may remain viable in the soil for periods up to 50 years (Tybirk 1991b).

In semi-arid conditions, deep disturbance favoured the regeneration of the sexually reproducing grass species by loosening and aerating the soil and increasing water infiltration – conditions that are favourable for seed germination and growth of seedlings (Belsky 1986a). Similar development may have occurred in the cleared bushland of this study, since most of the underground organs of the vegetatively reproducing species may have been removed by subsoiling during bush-clearing. It was, however, not possible to measure the relative effect of seed-borne regeneration with that from coppice in *Acacia zanzibarica*, although the former pathway was assumed to be more important (cf. above). The species coppices readily from stumps while young, but the vigour is reduced in mature trees. There is no information on the role of root sprouts in vegetative regrowth *A. zanzibarica*.

The seed densities obtained in this study are much lower than those reported for other African acacias. In Kenya, *Acacia mellifera, A. seyal* and *A. hockii* had seed banks from 810–2439 seeds m^{-2}, while in *A. reficiens, A. tortilis, A. seyal, A. senegal, A. nilotica* and *Faidherbia albida* it was 0–81 seeds m^{-2} (Tybirk et al. 1994). The largest seed banks were obtained in *A. hockii* and *A. seyal*, which are dispersed by birds (Tybirk et al. 1994). In South Africa 1250–2500 seeds m^{-2} of *Acacia* spp. have been found in the top 10 cm of soil (Milton 1980).

The lack of *Acacia tortilis* seeds in the soil is likely to be related to the high consumption of pods by domestic and wild herbivores, which has been reported by many authors (Pellew and Southgate 1984, Coughenour and Detling 1986, Ernst et al. 1989, Mwalyosi 1990). The indehiscent pods that reach maturity during the dry season and are available for most part of the year, provide an important food source for cattle, goats, sheep and camels in the current study area (Anttila et al. 1993).

Another cause for the absence of the seeds may also be the insect infestation, which is known to affect all *Acacia* spp. world-wide (e.g. Ernst et al. 1989). Considerable infestation of seeds of dehiscent *A. elatior* has been noted in the same area (pers. observ.). The complete absence of *A. refi-*

ciens seeds in the seed bank was surprising, since this species recorded the highest seedling density among all species in TN. The seeds of primarily anemochorous species, such as *A. reficiens*, are usually not spread far from the parent plant (Tybirk et al. 1994), and therefore the absence of mother trees at or near the sampling sites may simply explain the lack of seeds in the soil.

The difference in the densities can at least partially be explained by the systematic sampling employed in the present study and the subjective choice of samples (Tybirk et al. 1994) beneath the trees, where concentration of the seeds is likely to be highest. The average soil volume applied by Tybirk et al. (1994) was also bigger than that in the present study. In addition, the litter concentrated under the canopies of trees and bushes protect the seeds from being digested (Agnew and Waterman 1989), thereby superimposing the quantitative difference in the amounts of seed deposited under the canopy and in the open.

Soil physical characteristics also affect the seed bank, particularly in soils that expand and contract considerably depending on the moisture content, such as NG and to a limited extent also VN in the current study area. During the dry season, part of the surface soil, including the seeds deposited in the soil, may be relocated in the cracks and hence become a component of the passive seed bank.

The dominance of seeds of grasses and herbs over woody species (based on the identified seeds) was approximately similar in both vegetation types, although canopy cover and the number of tree species is higher in the transitional forest than in the *Acacia-Commiphora* bushland. Seed densities for grasses and herbs obtained in this study (60–257 seeds m^{-2}) are gross underestimates, since the figures are based on germinated plants only. According to Scholes and Walker (1993), one m^2 of savanna grassland contains tens of thousands of grass seeds in the top few centimetres of soil, but only about a hundred grass plants. Skoglund (1992) found an average of ca. 5500 seeds m^{-2} (depth 0–16 cm) in a Serengeti savanna soil, and all seeds belonged to grasses or herbs. For comparison, seed banks beneath humid tropical forest are almost entirely composed of woody species (Thompson 1992).

A thorough assessment of the role of the soil seed bank in the regeneration dynamics of an ecosystem would require repeated observations in time and in relation to rainy seasons, which primarily control the changes in soil moisture. An estimate of seed production or seed rain would facilitate the relating of the seed bank to the annual seed input (Thompson 1992), which fluctuates considerably from year to year, particularly in tropical arid and semi-arid ecosystems (Fenner 1982, Agnew et al. 1986).

The dispersal mechanism of *Acacia zanzibarica* has not been documented. Based on the morphological and ecological closeness of *A. seyal* and *A. zanzibarica* (dehiscent pods, preference of heavy soils and tolerance of occasional water-logging), it may be assumed that the dispersal patterns resemble each other. Johansson and Kaarakka (1992) suggested that browsing by livestock and wild herbivores could influence the dispersal of *A. zanzibarica*. At least elephants browse on the species; this is verified by the presence of its spines in the faeces (pers. observ.). Barkhadle and Ongaro (1992) mentioned *A. zanzibarica* as highly palatable for camels.

However, there are mechanisms that protect the species from browsing, at least by smaller herbivores. In addition to the armature, the ants inhabiting the swollen bases of the spines assist in deterring browsing. When intruders, such as man or herbivores, approach the trees, ants gather on the tips of branches and may thus limit further feeding. Similar phenomenon has been reported in other acacias accommodating symbiotic ants (Coe and Beentje 1991).

4.3 Size Structure of Woody Vegetation

4.3.1 Vertic Natrargid (VN) Soil Type

The diameter distribution of *Acacia reficiens* and *A. mellifera* in VN soil type (Fig. 35) exhibited a clear dominance of small individuals. It is possible that *A. mellifera* and particularly *A. reficiens* are short-lived, ecologically opportunistic species, which are capable of regenerating after any rainfall event at any season, as proposed by Agnew and Waterman (1989) in *A. senegal*. Schmidt (1988) noted a high appearance of *A. mellifera* seedlings in nine days after 25 mm of rain, indicating prolific regeneration stimulated by rainfall, but virtually no seedlings survived when the rains discontinued. The opportunistic strategy may also be related to seed dispersal, since *Acacia mellifera*, *A. reficiens* and *A. senegal* are all primarily anemochorous species (Tybirk et al. 1994), albeit prolific seed production is characteristic to all species in the genera, regardless of the dispersal mechanism (Ernst and Tolsma 1990).

The size distribution of *Salvadora persica* also indicated copious regeneration with a seedling density of 70 ha^{-1} (Fig. 35), but also moderate survival, since the number of trees declined gradually towards the biggest trees. In *Cordia sinensis* the diameter development roughly followed that of *S. persica*. *Acacia zanzibarica* showed an almost even distribution among the three smallest size classes (Fig. 35), and the highest seedling density of 193 ha^{-1} in VN. The size distribution in this species in VN differed from that in TN, suggesting that the population structure is more stable in VN, as expected, since *A. zanzibarica* is the dominant woody species in this bushland type.

It is noteworthy that the size distribution of *Acacia zanzibarica* in the cleared area 10.4 years after clearing and that in the undisturbed area resembled each other, with the exception of a few emergents in the latter area. In contrast, the size distribution of *A. reficiens* in the cleared area 6.7 years after clearing showed similar preponderance of small individuals as that in the undisturbed area. In both species the development of size structure between 6.7 years and 10.4 years in the cleared area suggests that very little regeneration occurred during the period. This situation is likely to remain until some of the emergents reach maturity and start seeding.

Seed rain from surrounding natural stands of *Acacia zanzibarica* was apparently also small. The possible remnants of seed in the soil from the bush-clearing may have remained in the passive seed bank, assuming that the seeds of *A. zanzibarica* do not readily germinate under the canopy, similarly with many other *Acacia* spp. (Obeid and Seif-El-Din 1970, Smith and Goodman 1986, Tybirk et al. 1994).

4.3.2 Typic Natrargid (TN) Soil Type

The density of *Acacia tortilis* seedlings in TN (< 1 ha^{-1}) and in VN measured in this study (1.4 ha^{-1}) and that obtained by Pukkala (1989) for areas roughly corresponding to VN (8 ha^{-1}), are lower than those obtained (16 ha^{-1}) by Agnew and Waterman (1989), who reported of regular regeneration of the species in Kora, located about 150 km NW of Bura, under a similar rainfall pattern. Although the low seedling density and the complete absence of *A. tortilis* in the soil seed bank in TN (on sites where it is one of the main species in the upper canopy) may suggest insufficient regeneration, the diameter distribution indicated reproduction in the past (Fig. 37). *A. tortilis* is typically a slow-growing species (e.g. von Maydell 1986), which may attain ages over 150 years and a height of 20 m in a riparian environment (Coughenour et al. 1990).

Poor regeneration of *Acacia tortilis* has been reported in areas with heavy pressure by domestic and wild ungulates. Kiyiapi (1994) found a very low proportion of young seedlings and saplings in the primarily monospecific stands of *A. tortilis* in Baringo, which he attributed to soil erosion, localised subsurface competition for moisture, drought, shade intolerance and intense browsing. Young *A. tortilis* saplings are also cut for fodder for small livestock in many pastoral areas of Kenya (Oba 1994); the collection of pods for the same purpose was also reported in Bura (Anttila et al. 1993). The Orma, the main pastoral group in the Bura area, considered *A. tortilis* as the best fodder species among the ones compared by Anttila et al. (1993).

The size class distribution of *Acacia zanzibarica* and *Prosopis juliflora* was clearly skewed towards the abundance of small individuals (Fig. 37). In *P. juliflora*, which is not indigenous to the area, such a distribution was expected, since the species started to spread naturally in the area in late 1980's, after it was selected as the main species in the irrigated fuelwood plantations of the BISP. The high palatability of the pods (Anttila et al. 1993), combined with the low palatability of the leaves (e.g. Fagg and Stewart 1994), has considerably contributed to its dispersal in the area, which is demonstrated by vigorous regeneration in abandoned cattle corrals and other sites frequently visited by livestock (Kaarakka 1993a, b).

The distribution of *Acacia reficiens* exhibited similar dominance of small individuals (Fig. 37) as *A. mellifera* and *A. reficiens* in VN. This suggests high reproduction, which was demonstrated by a high seedling density (135 ha^{-1}; over 50 % of all recorded seedlings in this soil type), but also high mortality in the small size classes. Such a distinctly skewed distribution is typical for plant weight–frequency distribution in increasing density stress (Cannell 1983a), although in ASAL conditions it is also related to the regeneration dynamics. The diameter distribution of *Cordia sinensis*, *Dobera glabra* and *Salvadora persica* did not exhibit as strong prevalence of small individuals in TN (Fig. 37), which indicated a rather constant, although slow regeneration rate (seedling densities 2–7 ha^{-1}), but recruitments to higher size classes had also occurred. All three species can attain considerable sizes with diameters ($dg_{0.15}$) up to 30–40 cm.

Commiphora campestris exhibited a rather even size distribution (Fig. 37), suggesting a relatively constant regeneration (seedling density 1.7 ha^{-1}) and mortality rate. This was reflected in the density of *Commiphora* spp. seedlings (9.6 ha^{-1}), which was low as compared to their contribution to the biomass production in TN (cf. Appendix 1). The data on size structure of other individual *Commiphora* spp. was insufficient, but similar size distribution in *C. africana* has been reported by Coughenour and Ellis (1993) in Turkana. Fenner (1982) reported of similarities in flowering, fruiting and leafing of seven species of *Commiphora* (including *C. africana* and *C. campestris*). *C. campestris* and *C. pseudopaolii* can also attain a diameter ($dg_{0.15}$) exceeding 40 cm.

Acacia-Commiphora bushland covers large areas in Kenya and adjoining Tanzania (Pratt et al. 1966, Millington et al. 1994). According to Agnew and Waterman (1989), *Commiphora africana* is the principal *Commiphora* species in the deciduous *Acacia-Commiphora* woodland in the Kora area 150 km NW of Bura. Agnew and Waterman (1989) further assumed that the same type is prevalent in large areas of Eastern and North-Eastern Provinces of Kenya, probably embracing also the current study area. However, a particular sub-type borne out by the current data under TN soil, could be classified as *Acacia reficiens* bushland, in which the dominant *Commiphora* is *C. campestris*. This observation is also supported by Pukkala's (1989) data, in which *C. campestris* featured conspicuously (among the *Commiphora* spp.) both in density and total biomass.

4.3.3 Comparison between VN and TN

The total seedling density of 252 ha^{-1} in TN is slightly lower than that recorded by Agnew and Waterman (1989) in Kora (377 ha^{-1}), although they did not specify their size range of seedlings. They observed the highest densities of 249 and 55.6 ha^{-1} in *Acacia senegal* and *Commiphora africana* respectively. The average density of 395 ha^{-1} of this study in VN compares well with 466 ha^{-1} recorded by Pukkala (1989), which was based on a same size range of seedlings and similar soil type, but on a bigger sample. He obtained the highest density in *Acacia zanzibarica* (210 ha^{-1}), followed by *Salvadora persica*, *Grewia tenax*, *Thespesia danis* and *Maerua macrantha*, all of which recorded densities of 25–37 ha^{-1}.

Although the seedling density was lower in TN than in VN, the proportion of seedlings of all woody plant individuals was much higher in the former type. The size distribution of the major species is skewed towards small individuals more in TN than in VN, indicating a higher mortality at early ages. This suggests a less stable woody vegetation structure in TN than in VN. One factor contributing to the high mortality in TN may be the hardcap in the soil, which may be detrimental for the critical seedling stage, particularly when located near the soil surface (ACRES-ILACO 1967b).

Much higher *Acacia* seedling densities (1440–1470 ha^{-1}) in *Acacia-Commiphora* woodland were observed by Hayashi (1992) in Kitui with a semi-humid climate. The most abundant species were *A. mellifera* (675–856 ha^{-1}), *A. senegal* (312–550 ha^{-1}), and *A. tortilis* (112–161 ha^{-1}). The climate in Kitui is more favourable (rainfall about 1100 mm) than in the current study area, but the woody vegetation has seriously degraded due to livestock raising, firewood collection and charcoal production.

The dominance of *Acacia* spp. in ASAL environment has been attributed to the ability to maintain high reproductive and moderate survival rates under heavy environmental stress (Coughenour and Ellis 1993). Such a pattern was apparent in *A. zanzibarica* and *Salvadora persica* in VN and probably also in *Grewia tenax* in both VN and TN in the current study area. *A. mellifera* and *A. reficiens* relied on a high rate of regeneration, which could counterbalance the high mortality, such that a few individuals survive to mature from most cohorts. Despite the dense wood, many *Acacia* spp. grow fast under arid conditions: Gourlay and Kanowski (1991) recorded a MAI of 1.2 cm in diameter for 21 African *Acacia* spp.

Revegetation of persistent seedlings is an appropriate strategy of woody species in areas where opportunities for recruitment from the seedling populations occur infrequently (Grime 1981). In Serengeti, Belsky (1986b) attributed the survival of *Acacia* and *Commiphora* to underground rootstocks. The species are burned or browsed down regularly, but regenerate by rootstocks, representing a bank of persistent juveniles that can quickly make use of favourable conditions. The trees survive with minimal annual growth until conditions allow them to grow tall enough to escape fire and browsing (Belsky 1984).

The dispersal of *Prosopis juliflora* from the irrigated fuelwood plantations of BISP is clearly borne out in the current data, particularly in the more open TN bushland. The species already constitutes a permanent element in the local flora and has also become a weed along the irrigation network (Kaarakka 1993a, b). On the other hand, the supply of fuelwood from the plantations and from the naturalised stands of *P. juliflora* has considerably decreased the demand from the natural bushland and the Tana riverine forest (Pukkala 1992, Johansson 1995).

In both soil types the species composition of seedlings largely followed that of the mature plants, except for some species with very few observations (Appendices 1–2), although distinct differences were noted in the respective densities between the two categories by species. This suggests that no major changes in the structure of the woody vegetation are currently occurring in the bushland area, apart from the spread of *Prosopis juliflora* and probably the reduced regeneration of *Acacia tortilis*. This is consistent with the results in the Tana riverine and transitional forest by R. Otsamo et al. (1993), who observed only small changes in the forest structure, but declined regrowth of species with narrow ecological tolerance in an eight-year period.

If the current results are considered to represent the entire area, it would imply that the direct effects of BISP on vegetation are less significant than those noted near other irrigation schemes, such as in Hola 40 km S of Bura or along the Turkwell in Turkana, which have become virtually devoid of all woody vegetation or at least the regeneration has seriously declined (Johansson 1995). However, as the current study area is located in the outskirts of BISP close to areas, which were initially prepared for irrigated agriculture, but were never cultivated, the direct impact of the scheme on vegetation may not be detected. Furthermore, the pressure on natural vegetation has recently decreased. It also appears that the current level of plant use by the pastoral peoples can be sustained, but the increasing sedentarisation of the pastoralists brought about by the BISP will considerably increase the number of livestock in the area thereby posing additional pressure on plant resources.

4.4 Factors Underlying Vegetation Structure and Dynamics

4.4.1 Spatial Structure of Woody Vegetation

The even-aged aspect of the tree populations and the lack of decaying individuals or seedlings may reflect the transitory structure of the vegetation, which would render it particularly susceptible to disturbance by large mammal populations (Glover 1963, Agnew 1968, Laws 1968). Agnew and Waterman (1989) reported of intermittent regeneration of *Commiphora boiviniana*, *Lannea triphylla* and *Maerua endlichii* in Kora. In this study, the size class distributions and seedling densities of species with diverse ecological requirements (*Acacia mellifera*, *A. reficiens*, *A. tortilis*, *A. zanzibarica*, *Cordia sinensis*, *Dobera glabra*, *Prosopis juliflora* and *Salvadora persica*) did not reveal even-aged populations, but rather cohort structures that were skewed towards young individuals. A notable exception to this general pattern occurred in *Commiphora* *campestris*, in which regeneration and mortality were approximately balanced over a wide range of sizes. The results do not support the assumption by Lamprey (1986) of even-aged cohorts of woody vegetation in ASAL.

Scholes and Walker (1993) pointed out that the clumped distribution of individuals suggest that competition is not an important force in structuring the community, while evenly spaced plants could result from competitive interactions. In the current study area this would imply that in the VN bushland, where woody plant individuals are rather evenly distributed, it is the competition between these plants that determines the vegetation structure. In this case it would especially refer to intraspecific competition in *Acacia zanzibarica*, which constituted about 80 % of all woody plants in this bushland type. Accordingly, in the TN bushland characterised by patchy vegetation, competition would not be a decisive factor.

Landscape properties and rainfall gradients provided the main environmental controls on vegetation composition and structure in northern Kenya (Coughenour and Ellis 1993, Patten and Ellis 1995). There precipitation and drainage patterns primarily determined the tree height, woody canopy cover was controlled by rainfall, and species composition by elevation, substrate and drainage. Precipitation is often spatially very variable in ASAL thereby influencing e.g. plant regeneration. It was, however, not possible to measure the small-scale variation in rainfall, and the current study area is too small for any rainfall gradients to be detected. In addition, the variation in topographic relief were small due to the very gentle slopes of the landscape (Muchena 1987). Hence substrate remained the main abiotic factor in determining the vegetation composition and structure within the area. The main soil factors in this respect were texture, structure, water-holding capacity and nutrients.

4.4.2 Soil Properties

The incomplete plant cover of TN soil is related to the presence of the hardcap that exists within 20–50 cm below the soil surface underlain with alkali or saline-alkali soil (ACRES-ILACO 1967b). These layers may impede root penetration and vertical distribution of water. Thus most of the vegetation has to rely on the moisture that remains on top of the hardcap, although some species, e.g. *Salvadora persica* and *Dobera glabra* are also able to tap saline groundwater (ACRES-ILACO 1967b), which could explain the presence of these species also in the bigger diameter classes. Agnew et al. (1986) found all the major roots of perennial species located at 10–30 cm depth in Kora, although the hardcap layer was not always present.

Acacia reficiens and *Grewia tenax*, which together constituted over 60 % of all woody plant individuals both in the seedling and in the 'above-1.3 m' category in TN, are typically shallow-rooted species (Coughenour et al. 1990, Patten and Ellis 1995). *A. tortilis* also adapts to such conditions by extending hose-pipe sub-surface roots over twice the width of the crown, although in sandy soils it forms a deep tap root (NFTA 1991). The abundance of the *Commiphora* spp. in this environment may be related to the high water content of the trunk (Hayashi 1992) and roots (Agnew et al. 1986), as an adaptation to drought avoidance (cf. Larcher 1983), or to the presence of underground rootstocks, which permit regeneration under favourable conditions (Belsky 1986b; cf. below).

Where there are expanding clays exhibiting moderate cracking when dry and swelling during the rainy season, the environment is very unfavourable for most trees and bushes, severely hindering root development and killing young seedlings. Deep cracking may also partially expose the roots. In addition, these soils may remain waterlogged for extended periods during and after the rainy seasons. This applies particularly to NG and to a lesser extent to VN. Often the vegetation develops to a grassland dominated by *Sporobolus helvolus*, which can tolerate saline groundwater (ACRES-ILACO 1967b). Notable exceptions among the woody plants are *Salvadora persica* and especially *Acacia zanzibarica*, which thrive on these seasonally waterlogged areas.

Nutrient amounts may have contributed to differences in vegetation composition, especially between TN and VN, although factors related to soil moisture are likely to remain decisive in determining the vegetation type in East Africa, as proposed by Pratt and Gwynne (1977). The major differences between TN and VN were detected in pH, and in the amounts of K, P, Ca, Mg, Fe and Mn (cf. Results). The small number of observations on the nutrient status in most soil types and their limited spatial representativeness did not, however, allow other but tentative conclusions.

The nutrient amounts observed in this study are mostly in the same range as those reported for various irrigated soils under agriculture or forestry in the area. The major differences between the values of the current study and those of Kamoni et al. (1987) in irrigated forestry, Muchena (1987) in irrigated agriculture and Mwendwa et al. (1993) in furrowed, but non-irrigated areas, are the low pH, Ca and Mg in the TN, and the generally low P and K in all soil types except TN.

Organic C content of the soils in the current study was low (0.24–0.66 %). This is consistent with Kamoni et al. (1987), Muchena (1987) and Mwendwa et al. (1993), who measured 0.03–0.50 % and 0.78 % in irrigated and undisturbed soils respectively in the area. Muchena (1987) attributed the low organic C proportions to rapid rate of decomposition under the prevailing high temperatures, although fine soil texture and especially lack of moisture may significantly reduce the rate (Scholes and Walker 1993, Breman and Kessler 1995). Low C content generally results in low levels of N and P in the soil, since the organic matter is the primary source of these elements. In the cleared bushland, the nutrient depletion was further aggravated by the removal of most of the vegetation from the site during bush-clearing, which usually inevitably reduces the nutrient stores in the soil (Young 1989). Low organic matter levels are known to affect soil features critical to seedling establishment and to water and root infiltration in ASAL (César 1989).

Although the nutrient levels were not measured at the time of bush-clearing, it is apparent that the rate of decomposition exceeded that of net release, until the replenishment of the nutrient pool began, e.g. by means of increased litter from shoots and roots. Woody legumes, such as most *Acacia* spp., are also potentially capable of symbiotic N fixation (e.g. Halliday 1984, Dommergues 1987), but they may not nodulate in the field in semi-arid regions, or the nodulation is limited (Dommergues 1987, Penning de Vries and Djitèye 1991, Miettinen et al. 1992).

Following the decline after the vegetation removal, the organic C content increased along with the increasing woody canopy cover in the more rapidly revegetated VN and NG between 6.7 and 10.4 years after clearing, but this was not reflected in the N and P levels which decreased in both types during the same period. In TC, the slow increase in the woody vegetation cover could not yet replenish the organic C content in the soil. In relative terms, the uptake of nutrients usually exceeds the accumulation of biomass and the release of nutrients by litterfall, at early ages in maturing woody plants in semi-arid regions (Breman and Kessler 1995). Declining levels of the soil organic C and total N are also typical in the beginning of secondary succession of old fields in temperate climates (Zak et al. 1990).

On the other hand, most nutrient levels, except Fe and Cu, in the cleared VN did not show marked differences as compared to the undisturbed area 10.4 years after clearing. This may indicate that the changes in soil nutrients brought about by abrupt disturbance in VN, such as vegetation removal, will be recovered within 15–20 years. The small number of observations in the control area and the lack of information on the nutrient status at bush-clearing, however, call for additional data to verify this assumption.

The drainage factor was not assessed in the current study, but its effect is clearly visible along the ephemeral streams in the area. These water courses affect the vegetation composition and structure primarily in two ways by spreading the seeds downstream and by extending the period of water availability in the soil in their lower reaches, especially in the floodplain. If pods are dropped during the period of strong current, long distance water dispersal may occur. Most of the bigger trees in the deciduous bushland are concentrated along the ephemeral streams (Gachathi 1993). R. Otsamo et al. (1993) assumed that the regeneration of *Acacia zanzibarica* may have been influenced by seasonal waterways in temporarily inundated areas.

Risk of Erosion

In its natural, undegraded stage TN soils are not particularly vulnerable for sheet erosion (ACRES-

ILACO 1967b). Firstly, the vegetation is, although patchy, usually dense enough to arrest spontaneous sheet erosion. Secondly, loose, sandy material tends to concentrate around the bases of particularly bush vegetation, establishing slightly elevated mounds, which are protected from erosion. Thirdly, the slopes are very slight in the area. However, mammalian herbivores show great preference to these areas due to the openness of the country and the attraction of halophytic plants such as *Boscia* spp., *Cadaba* spp., *Maerua* spp., *Dobera* spp., *Salvadora persica* and *Salsola dendroides* (ACRES-ILACO 1967b), with their high mineral content (Le Houérou 1980b). In addition, a substantial proportion of the best grazing lands of the Orma were set aside for the BISP (Johansson 1995). Although the gazettement was not fully enforced, and hence some grazing was allowed to continue within the irrigation scheme, it may have resulted in overgrazing locally.

Increased sedentarisation of the Orma pastoralists in the vicinity of the irrigation scheme also brought about a potential risk for overuse of vegetation. (Kelly, undated). The trampling of livestock destroys the brittle dry season vegetation and causes subsurface soil compaction. This could trigger local erosion, when the surface sand is removed by heavy rains and translocated into eroded gullies and natural depressions (ACRES-ILACO 1967b). These micro-habitats are favourable for regrowth (pers. observ.), whereas the vegetation can not reinvade the degraded upper parts of the landscape. Gully erosion can be seen in places where there is a combination of increased grazing pressure and an uneven microtopography, such as along the ephemeral streams and besides the structures made for irrigation or road construction.

Soils that exhibit Vertisol (FAO/UNESCO 1974) tendencies, such as NG and VN, are generally prone to erosion (Young 1989). The erodibility of VN in the Bura area is, however, assumed to be low. This is primarily due to the flatness of the landscape where these soils occur; this usually deters erosion (e.g. Huck 1983). The efficient recovery of the vegetation, particularly that of *Acacia zanzibarica*, may also contribute to low erodibility, since living herbaceous plants and plant litter reduce the risk for erosion (Young 1989). In addition, the woody plants occupying VN do not attract herbivores, and hence these soils are less subjected to overgrazing and surface compaction, as compared to some other soil types, such as TN.

This study is in line with Johansson (1995), who considered VN bushland resilient due to a good regenerative capacity of the woody species. Based on the current results both the woody biomass and the soil nutrient status in VN are estimated to recover in 15–20 years. In TN, the recovery of the woody vegetation and hence also that of the biomass would apparently require a longer period. The differences of the two types have repercussions in rehabilitation, since the recovery by means of restoration or rehabilitation may be inversely related to the structural and functional complexity of the ecosystem, as suggested by Aronson et al. (1993a)

4.4.3 Anthropogenic Factors

The size structure was most intensely affected in *Acacia tortilis*, in which several large individuals were cut for fuelwood and forage between the assessments. A few freshly cut trees were even found lying on the ground close to the measured transect in 1994. In addition, the mean diameter of *A. tortilis* trees decreased on both soil types, from 15.9 to 9.7 cm in TN, and from 21.5 to 14.9 in VN between 1985 and 1994. Since *A. tortilis* is one of the main species in the upper canopy layer in TN, the removal of large individuals may influence the micro-scale gradients in the field layer (Vetaas 1992a, b), thereby affecting the regeneration dynamics.

Young *Acacia tortilis* are apparently susceptible for browsing, particularly by goats and sheep, until they reach the height of 1.5–2 m (Kiyiapi 1994, Oba 1994), which roughly corresponds to the first 5–7 years. Rahman and Dafei (1977), however, observed that clipping of shoots to simulate browsing had no effect on the number of established seedlings and increased production of shoot and root DM and the rate of growth in *A. tortilis* in the Sudan. At the early stages, the species can present morphological adaptations, such as profuse branching and thorniness, until it is no longer susceptible for browsing (Pellew 1983).

The main species used for fencing the homesteads and cattle corrals in the area are *Acacia mellifera*, *A. reficiens* and *Commiphora* spp., particularly *C. pseudopaolii*. In addition, leaves and twigs of almost all *Commiphora* spp. in the area are browsed by camels and goats, and some also by cattle and sheep, but they are not used for fuel. The *Commiphora* spp. are leafless for 8–9 months of the year, often acquiring leaves after a short spell of rain and then shedding them within a few weeks. This implies that they tolerate rather heavy browsing of leaves. The absence of fire may also contribute to the abundance of *Commiphora* spp. in Kora (Agnew and Waterman 1989) and in the current study area.

Grewia tenax may have benefited from browsing. It is an endozoochorous species (Hilger and Schultka 1988) and an important source of browse among goats, sheep and camels in the area (Anttila et al. 1993). The protein content of the leaves of *G. tenax* was the highest among 10 browse species studied by Anttila et al. (1993). The fruits, which ripen once a year and are available for 1–3 months in the dry season, are also highly valued as human food (Anttila et al. 1993). There is no information on the browse value of *Acacia zanzibarica* for domestic ungulates in Bura, but it is highly palatable for camels in Somalia (Barkhadle and Ongaro 1992). In the study area, the camels mainly browse on TN soils, where *A. zanzibarica* is not abundant (cf. Results).

4.5 Biomass Production of Natural Vegetation

The rates in MAI of woody volume and biomass obtained in this study in cleared and undisturbed bushland fell within the ranges presented earlier (Table 32). When comparing MAI between the two soil types of the current study, the volume productivity was approximately 20 % higher in VN (0.43 $m^3 ha^{-1}a^{-1}$) than in TN (0.36 $m^3 ha^{-1}a^{-1}$). Biomass estimates differed more distinctly: VN produced 307 and 111 kg $ha^{-1}a^{-1}$ in MAI of total and usable biomass, and TN 163 and 66 kg $ha^{-1}a^{-1}$ respectively. These observations are consistent with those of Pukkala (1989) in the corresponding strata. It is hence evident that the difference in standing biomass and productivity in the two soil types is not caused by anthopogenic factors, such as fuelwood collection and browsing by domestic herbivores, for which the pressure is higher in TN than in VN, but it is rather attributed to water and nutrient conditions in the soil.

Insufficient data on soil nutrient status in one of the two main soil types, TN and VN, limited comparison between them. Nevertheless, it is assumed that the difference in biomass production is related to soil moisture, especially to the better water absorption and retention capacity in VN (ACRES-ILACO 1967a), as well as to the excellent adaptation of *Acacia zanzibarica* to these conditions. The species constitutes 60–70 % of woody biomass in VN (cf. Results).

The comparison of biomass and volume estimates in cleared and undisturbed bushland may have a slight bias due to the different treatment of some shrub species whose volume was calculated in Experiment VI, but not in Experiment VII. This applied particularly to *Grewia tenax*, which was one of the most abundant species accounting for 38.9 % of all observations in undisturbed bushland under Typic Natrargid soil in 1994. However, due to the small average size of the species, the usable volume and biomass were estimated to be approximately 0.1 m^3ha^{-1} and 60 kg ha^{-1} only in Experiment VII, which rendered the contribution of *G. tenax* to total volume and biomass of all woody species insignificant.

It is possible that nutrients are the limiting factor instead of water availability in woody plant production (Breman and de Wit 1983) in some soil types of the area, especially on sandy soils with good water absorption capacity, such as TC, and clay depressions such as NG and maybe also VN. In the Sahel, the transition from water-limited to nutrient-limited growth in the herbaceous layer was estimated to occur at a level of 300 mm of annual precipitation, corresponding to 250 mm of infiltrated water (Breman and Kessler 1995). They assumed that the threshold rainfall may be lower in East Africa due to the better spreading of rain under bimodal rainfall regime. The rainfall limit depends on soil texture and fertility (Breman and Kessler 1995), and it may be different in woody species.

Annual growth percentages of 2.8–3.0 % and 2.4–2.5 % were recorded in total and usable biomass in undisturbed TN and VN bushland respectively. Pukkala (1989) recorded growth percentages of 5.2–17.0 % depending on stratum, with

Table 32. Estimates of mean annual increment (MAI) of biomass (b) or volume (v) in very arid (average annual rainfall 100–200 mm), arid (200–400 mm) and semiarid (400–600 mm) areas in Africa. Biomass refers to total aerial woody DM, and volume to aerial woody parts unless otherwise indicated.

Rainfall mm	Country / region	MAI kg DM ha^{-2}a^{-1} (b) m^{3}ha^{-1}a^{-1} (v)	Notes	Reference
100–200	Sahel	50 (b)		Le Houérou 1989
n.a.	Somalia, Niger	40–150 (b)	air-dry	Openshaw 1982, Andeke Lengui 1987
		40–380 (b)	air-dry	Rutherford 1978
300	Senegal	200–700 (b)		Bille 1985
350	Kenya	238–540 (b)	cleared	Kaarakka, this study
350	ditto	88–230 (b)	cleared, usable[1]	ditto
350	ditto	163–307 (b)		ditto
350	ditto	66–111 (b)	usable[1]	ditto
200–400	Sahel	200 (b)		Le Houérou 1989
250–500	Niger	350 (b)	air-dry	Andeke Lengui 1987
400–600	Sahel	500 (b)		Le Houérou 1989
400	Burkina Faso	450 (b)		Toutain et al. 1983
480	S. Africa	2810 (b)		Scholes and Walker 1993
100–200	Sahel	0.075 (v)		Le Houérou 1989
200–400	Sahel	0.30 (v)		Le Houérou 1989
200–400	Sahel	0.3 (v)		Clément 1982
350	Kenya	0.28–0.63 (v)	cleared	Kaarakka, this study
350	ditto	0.10–0.27 (v)	cleared, usable[1]	ditto
350	ditto	0.36–0.43 (v)		ditto
350	ditto	0.12–0.15 (v)	usable[1]	ditto
350	ditto	0.38 (v)		Johansson and Kaarakka 1992
350	ditto	0.2–1.5 (v)	usable[1]	Pukkala 1989
400	Sahel	0.01 (v)		Goudet 1984
400–600	ditto	0.1 (v)		Goudet 1984
400–600	ditto	0.06–0.5 (v)		Clément 1982
400–600	ditto	0.10–0.75 (v)		Bonkoungou and Catinot 1986
400–600	ditto	0.75 (v)		Le Houérou 1989
< 500	ditto	1.0–1.5 (v)	logged	Catinot 1994
250–1000	Kenya	0.23 (v)	net	KFMP 1994
400–600	Sahel	0.5–2.0 (v)	industrial plantation	Catinot 1983

[1] refers to biomass or volume of woody parts above 2 cm in diameter

strata of low plant densities generally exhibiting higher relative growth rates. Wood increment as a proportion of standing biomass decreases with increasing maturity. In the cleared bushland, the relative growth rate of both total and usable biomass dropped from 15.0 to 9.5 % between 79 and 124 months after vegetation removal. Surprisingly, it is the former rate, which equalled that obtained by Pukkala (1989) in mainly undisturbed and mature stands of the same stratum. Low standing biomass and high net production to biomass ratio are characteristic to immature or stressed vegetation (Breman and Kessler 1995).

The potential maximum growth rates, as assessed by individual trees, can be much higher (cf. Results), although only a limited number of species were included in these comparisons. The sample was composed of easily identifiable tree individuals, which were often conspicuous or growing in isolation, such that they may not represent the average trees in the populations. Nevertheless, the rather slow relative (1.5 %) and absolute (2.4 mm a^{-1}) growth in diameter in *Acacia tortilis*, and the clearly higher rates in *Commiphora* spp., particularly in *Commiphora africana* (4.2 %; 3.9 mm a^{-1}) and *C. campestris* (2.9 %; 6.7 mm a^{-1}), as well as in *Acacia reficiens* (4.3 %; 5.0 mm a^{-1}), are borne out in these data. This is consistent with earlier observations in Kenya (Agnew et al. 1986, Agnew and Waterman 1989) and the Sahel (Breman and Kessler 1995).

Brown and Lugo (1984) indicated that the ratio of total woody biomass to stem wood biomass may be up to 2.9 in open forest formations. The ratio between total and consumable DM production would be about 2.5 : 1 for the Sahel and 2 : 1 for the Mediterranean zone. Relative proportion of the usable woody component generally increases with size and age of trees until canopy closure (Cannell 1983b): in the cleared bushland, usable biomass constituted 37 % of the total biomass at 79 months, but 43 % at 124 months after clearing.

Le Houérou and Hoste (1977) and von Maydell (1986) gave an approximation of 1 kg ha^{-1}a^{-1} mm^{-1} for the annual consumable DM and harvestable woody biomass (including branchwood) production respectively in the Sahel, while the equivalent for Africa North of Sahara is 2 kg ha^{-1} a^{-1}mm^{-1} (Le Houérou and Hoste 1977). Lamprey (1983) considered the latter estimate more appropriate for East African rangelands, due to a relatively long growing season and bimodal annual rainfall. The results of the current study (Table 32) do not, however, support this supposition, although this may be attributed to the limited coverage in area and vegetation types.

Total standing biomass (DM) of the woody component in undisturbed bushland in this study was 5924 kg ha^{-1} in TN and ranged between 10255–15307 kg ha^{-1} in VN. The corresponding values of usable biomass were 2602 and 4712–6467 kg ha^{-1}. In a forest inventory, Pukkala (1989) obtained mean total biomass of 1400–6300 and 8700 kg ha^{-1}, and mean usable biomass of 700–3800 and 4700 kg ha^{-1} for strata corresponding to TN and VN respectively. For comparison, the respective total woody biomass estimate of the transitional forest, situated on the Tana river floodplain, was 17300 kg ha^{-1} (Pukkala 1989).

Olang (1986), in turn, reported a distinctly higher range of 18000–62000 kg ha^{-1} for standing biomass of the woody layer in shrubland, bushland and thicket in the 300 mm a^{-1} rainfall zone in Kora. In the light of the other results, the values of Olang (1986) may be overestimations. In a 500 mm a^{-1} unimodal rainfall zone in Niger, the biomass of the woody component was reported to be 1958–9072 kg ha^{-1} (Heermans and Minnick 1987). South African tree savannas, with unimodal annual rainfall of 300–600 mm, have a much higher average standing biomass of 20000–45000 kg ha^{-1} (Walker 1980).

Total standing volume ranged from 11.7 m^{3}ha^{-1} in TN to 18.8 m^{3}ha^{-1} in VN, and the usable component from 4.4 m^{3}ha^{-1} in TN to 8.0 m^{3}ha^{-1} in VN respectively. These estimates are in line with those of Pukkala (1989), who recorded 0.7–2.3 and 6.6–10.6 m^{3}ha^{-1} for low and high density strata corresponding to TN, and 0.4–1.2 and 4.5–9.7 m^{3}ha^{-1} for low and high density strata corresponding to VN. KFMP (1994) presented an average growing stock of 15.9 m^{3}ha^{-1} of woody biomass in woodlands, bushlands and wooded grasslands within a rainfall range of 250–1000 mm a^{-1} in Kenya. In the Sahel, the standing woody biomass averages about 2 and 10 m^3 ha^{-1} in the zones receiving a rainfall of 150–350 mm a^{-1}, and 350–600 mm a^{-1} respectively (Breman and Kessler 1995).

Rain use efficiency (RUE), expressed as aerial DM production per unit area as a function of time and precipitation (Le Houérou 1989), is used as an indicator of ecosystem productivity, allowing comparisons between ecosystems of different climatic zones having different botanical and structural characteristics. Water-use efficiency (WUE), expressed in terms of aboveground biomass produced in relation to the amount of water lost by evapotranspiration, is more accurate, but in the absence of complete data on water balance, RUE serves as a good indicator, especially in drylands (Aronson et al. 1993a). RUE usually decreases with aridity, low intensity of useful rain and increasing potential evapotranspiration (Thomas and Squires 1991). It depends on soil conditions and particularly on the dynamic status of the ecosystem.

Properly managed arid and semiarid grazing lands are usually in the 3.0–6.0 value range (Thomas and Squires 1991). Cissé (1986), indicated RUE for total aerial primary productivity varying between 4.0 and 7.2 kg DM ha^{-1}a^{-1} mm^{-1} in the Sahel with less than 500 mm of rainfall. Distinctly lower RUE values for wood production were presented for the same region by Le Houérou (1989), namely 0.33, 0.66 and 1.00 kg DM $ha^{-1} a^{-1} mm^{-1}$ for the 100–200, 200–400 and 400–600 mm rainfall zones respectively.

The values recorded for woody species in the current study in VN soil, 0.98 and 1.53 DM $ha^{-1} a^{-1}mm^{-1}$ 6.7 and 10.4 years after bush-clearing, were slightly higher than those from the Sahel. This is partially attributed to the juvenile phase, which is characterised by rapid net growth. There are no estimates on RUE in TN, but the value is lower than in VN, since the standing biomass is also lower in the former type.

5 Conclusions

5.1 Productivity of Natural Vegetation vs. Microcatchment Water Harvesting

In areas with no tradition of end-use oriented management, the low productivity of the bushland does not provide a sustainable alternative in the supply of fuelwood (Johansson 1995). The estimated annual per capita fuelwood consumption in the irrigation schemes within the area (BISP, Hola) was 0.72–1.23 m^3 (Hughes 1982, 1984, Vainio-Mattila 1987), whereas that of the Orma pastoralists was 0.46–0.74 m^3 (Ensminger 1984) respectively. The sustainable harvest, (i.e. MAI of usable volume) in cleared and undisturbed bushland varied between 0.10–0.27 and 0.12–0.15 m^3 $ha^{-1}a^{-1}$ respectively. If the entire fuelwood demand of the BISP population of 20 000 people, with 1 m^3 of average per capita consumption, were to be supplied from the natural bushland, an area of 74 000–133 000 ha would be required for this purpose alone. Besides this, the needs of the indigenous population will also need to be covered. Assuming an optimistic threefold increase in MAI of biomass resulting from prescribed management, the production area would still occupy 25 000–44 000 ha, excluding the demand by the indigenous population.

The best (usable) biomass production rates obtained in MCWH are higher than those in the natural bushland. MAI of the best species, *Prosopis juliflora* in the best MCWH method averaged 2.11 $m^3ha^{-1}a^{-1}$, while that of the two best local species, *Acacia zanzibarica* and *A. nilotica*, was 0.30–0.32 $m^3ha^{-1}a^{-1}$. Applying the same consumption estimate as above, 9500 ha and 62 500–67 000 ha would be required to cover the fuelwood needs of 20 000 people using *P. juliflora* and the acacias respectively. Planting of trees using MCWH can thus neither suffice the fuelwood needs of concentrated populations, such as towns or irrigation schemes. The MAI can, however, be raised considerably from that of unmanaged vegetation.

5.2 Means to Improve Microcatchment Water Harvesting

The efficiency of MCWH can be improved by combining it with other water conservation measures. The top soil in the basin area of the microcatchment may be broken by tillage before the rainy season to improve water infiltration. This is particularly important in soils that contain a hardpan layer close to the soil surface. The incorporation of organic matter, such as mulch, acts in the same way by improving the water retention capacity of the soil and reducing the evaporation from the soil surface. In addition, mulching improves microbial activity and nutrient availability in the soil and reduces the root zone temperature (Gupta 1994). The use of mulch may become profitable, if suitable material becomes available from agricultural fields or elsewhere, e.g. crop and plant residues, such as straw, cotton stalks, grass, branches or wood bark. As the trees grow, they gradually begin to produce mulch themselves. Weeding in the catchment basin favours tree growth due to reduced competition for water, although increased regeneration and growth of natural vegetation near the catchments may also be targeted (cf. below).

The methods used in the current study may also be improved by selecting suitably sloping sites. This would reduce the volume of soil to be removed during catchment preparation, which would consequently reduce the establishment cost. Ideally, only small terraces are made between the catchments, such as in the contour furrow microcatchments in the current study. Ochieng et al. (1992) recommended a slope of 3–5 % in loams and sandy loams, but Barrow (1988) regarded 2.5 % as an appropriate upper limit. With this kind of terrain, there is little risk for erosion, and the little that may occur is usually contained by the catchment bunds. The application of MCWH on slopes lay more emphasis on the soil hydrological characteristics, such as the determination of runoff-rainfall ratio, and threshold rainfall, i.e. the minimum rainfall to generate runoff (Sharma 1986, Wairagu 1991, Boers 1994). In the current study appropriate slopes were not available due to the general flatness of the area.

The application of microcatchments does not require planting of trees. Small semicircular bunds can also be built to encourage the growth of young naturally occurring trees. The advantage is that the trees get more water through MCWH to allow them to grow quicker and escape browsing. This was found to favour the development of naturally regenerated *Acacia tortilis* seedlings, but also *Faidherbia albida, Cordia sinensis* and *Salvadora persica* benefited from MCWH in Turkana, Kenya (Barrow 1988).

Besides tree growing, rainwater harvesting may bring about beneficial effects on the local vegetation. Eyog Matig (1989) reported of spontaneous regeneration of the natural vegetation on sterile, degraded soils following land preparation. Tenbergen et al. (1995) noted increased diversity of local species in the vicinity of microcatchments on slopes. Tillage along contours combined with mulching significantly increased seed capture and development of ground cover during a single rainy season in Sahel (Chase and Boudouresque 1987). To assist regeneration of forage and grasses in over-grazed areas, grass seeds can also be spread along the contour bunds before the rains (Pacey and Cullis 1986). Heavy growth of grass was also noted in triangular and contour furrow microcatchments in the current study.

It is noteworthy that the mean RUE values across all species obtained using rainwater harvesting were generally lower (0.12–1.29 DM $ha^{-1}a^{-1}$ mm^{-1}; Table 30) than those of the natural vegetation, except in the cross-tied furrow microcatchments, in which close spacings of up to 1667 trees ha^{-1} were applied. This suggests that the productivity of the methods applying the triangular microcatchments (spacing 100 trees ha^{-1}) could be improved simply by planting more than one tree in a catchment or by increasing the number of catchments per unit area. Simulation models by Ben-Asher and Warrick (1987) indicated that increasing the number of trees per catchment may also improve WUE.

Based on the results of the current study, it is very easy to subscribe to what Boers (1994) pointed out, that in ASAL "...runoff from small areas such as microcatchments is an important potential source of water for the establishment, development and good growth of trees". The survival and growth of young plants is primarily determined by the amount and distribution of water. It is the survival during the first rainy season that is crucial for plant establishment, and it is in these conditions where rainwater harvesting is most beneficial.

Rainfall is the main factor controlling the movement of water in the terrestrial hydrological cycle, but the soil water status also depends on soil properties, particularly in the arid and semiarid tropics, where the role of plants is less pronounced than in the humid tropics. Therefore, the quantity of precipitation is of indirect importance only, but the amount of water that actually becomes available to plants in the soil remains important (e.g. Alatalo 1989). The extent to which water harvesting techniques increase the use of water made available for plants and promote deep root penetration may vary considerably from one location to another depending on site-specific conditions (Brunori et al. 1995). This calls for further experimenting with methods included in the present study over a range of soil types. In addition, more data on soil water status will be required.

5.3 Implications for Bushland Management

Rehabilitation of woody vegetation by utilising the soil seed bank is not possible, if the seed bank is absent or does not contain the seeds of the desired species. The results of the current study confirm earlier observations (Aronson et al. 1993b), that the seed bank may not be sufficient for revegetation of woody species, particularly in the key species used for energy, such as many acacias. Nevertheless, the presence or absence of certain keystone species may assist in selecting a suitable rehabilitation strategy. *Acacia tortilis*, which is one of the most desired woody species in Kenyan drylands, could be considered as an example. If a sufficient seed bank exists, the rehabilitation may be based on exclosures. If it does not exist, reintroduction of seeds, possibly combined with exclosures, may be necessary.

Direct seeding of extensive areas with a mixture of several species may offer a cheap alternative for planting, although the techniques are yet poorly known. The seeds are simply thrown in acacia bushes or fenced-in areas to protect the seeds and young seedlings from animals. There are several obstacles in this technique, albeit some success has been reported (Weber and Stoney 1986, Kerkhof 1990). Although the bush canopies protect the seeds from being digested (Agnew and Waterman 1989), the seeds of shade-intolerant species may not germinate under the canopy. This has been shown in many *Acacia* spp. (Obeid and Seif-El-Din 1970, Smith and Goodman 1986). More importantly, the costs in practice generally overweigh the results mainly due to the high quantities of seed required.

Gerhardt and Hytteborn (1992) suggested that, as wind-dispersed seeds enter easily into the natural restoration process, planting in dry tropical forests could be restricted to animal-dispersed trees. This may not apply to arid and semi-arid bushland where most of the woody species are animal-dispersed (Ernst and Tolsma 1990, Hilger and Schultka 1990.)

Large scale seeding and afforestation is not an economic proposition when considered in purely economic terms (FAO 1995). Konuche and Amuyunzu (1992) indicated that at the current technological level it is not economically feasible to carry out large-scale tree planting in the arid lands in northern Kenya. Management of woodlands for the production of fuelwood only may neither be economically justified in semi-arid areas. Furthermore, the socio-economic setting in ASAL makes it usually expensive and difficult to insist on the primacy of wood production over other uses (Wormald 1984)

Any considerable land preparation will inevitably be uneconomical, if the production of the native vegetation is the main purpose. From the economical viewpoint, mulching alone may possess more potential, as indicated by Chase and Boudouresque (1987). However, if simple and inexpensive MCWH methods can be combined with tree planting to yield products and services that are required by the local community, and in such a way that the production does not interfere with the traditional land-use, it may become feasible and socially accepted. As MCWH is most often used in pastoral areas, species yielding fodder for animals may be the most beneficial ones. Reintroducing local key species with MCWH may also facilitate the reintroduction and establishment of other native species thereby increasing the rate of rehabilitation (Aronson et al. 1993a).

A novel approach by Tybirk (1991a), as an application of silvopasture, may provide a new pathway in an attempt to re-establish natural mixed woody vegetation. This would involve the collection of seeds of the species to be re-established, and feeding the seeds to the animals that are kept in exclosures in good localities e.g. for one week. Cattle are released, but the fencing will be kept for 2–3 years to protect the seedlings. Ideally, some of the seedlings will develop into mature plants and start spreading the seeds over other areas in good years. The success will rely heavily on the co-operation of the local herdsmen, who may assist the regeneration, provided that they receive tangible benefits from the activities.

A wide array of fine-tuned indigenous management methods exist (e.g. Shepherd 1992), but there is little tradition of end-use oriented management of woody plant resources in most tropical semi-arid and arid areas. Further research is required in management techniques, as well as training of local personnel on these techniques. However, it may take up to 20–30 years for these methods to become significant in many countries (Grainger 1990). The lack of research is often not the limiting factor, but rather the application of the results. The proposition by Fries (1991) offers a pathway to combine traditional knowledge and research: one should first experiment on the management methods already developed within the region and then use the research as an instrument to overcome problems and to improve the methods. As Walker (1985) put forward, savanna and bushland management may also include deliberate manipulation of the system to gather knowledge about the system which will assist in the formulation of management policies.

In the case of Kenya, the ongoing forestry planning (KFMP 1994) has recognised the significance of drylands in both local and national development and suggested means with which a better dryland management could be achieved.

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Appendix 1. Mean density, height, diameter ($dg_{0.15}$), usable wood volume and biomass of trees above 1.3 m in height in undisturbed bushland under Typic Natrargid soil at two successive inventories (n=105). Two different species are combined in some cases when locally identified as one. Diameter and volume calculation are based on species marked with an asterisk (*) only.

Species	Acronym	Density, n ha^{-1}			Height, m		$dg_{0.15}$, cm		Usable volume, m^3 ha^{-1}		Usable biomass, kg ha^{-1}	
		> 1.3 m 1985	> 1.3 m 1994	< 1.3 m 1994	1985	1994	1985	1994	1985	1994	1985	1994
Acacia mellifera (Vahl) Denth. *	AME	1	4	1	2.1	2.4	5.7	6.0	<0.005	0.01	1	8
Acacia reficiens Wawra *	ARE	19	58	135	3.2	2.6	11.2	7.5	0.35	0.59	317	534
Acacia senegal (L.) Willd. *	ASE	3	2		2.1	4.1	3.9	7.5	<0.005	0.01	2	10
Acacia tortilis (Forssk.) Hayne *	ATO	4	11	1	4.1	3.2	15.9	9.7	1.09	1.30	809	966
Acacia zanzibarica (S.Moore) Taub. *	AZA	7	8	4	3.6	3.0	6.4	5.0	0.17	0.03	149	27
Adenium obesum (Forssk.) Roem & Schult	AOB			<0.5								
Azima tetracantha Lam.	AZT		<0.5	<0.5		1.3						
Berchemia discolor (Klotzsch) Hemsl. *	BER		3	<0.5		2.6		9.8		0.14		97
Boscia coriacea Pax / *Elaeodendron aquifolium* (Fiori) Chiov.	BCE		3	1		2.3						
Cadaba farinosa Forssk. / *Maerua triphylla* A.Rich *	MCF	4	10	3	2.3	2.5	9.6	6.5	0.03	0.02	16	14
Cadaba gillettii R.A.Graham / *Maerua macrantha* Gilg *	MCG		6	11		2.0		7.1		0.01		6
Cadaba glandulosa Forssk.	CGL		2	2		1.7						
Combretum constrictum (Benth.) Laws.	CCO	2			1.9							
Commiphora africana (A.Rich.) Engl. *	CAF	2	2		3.0	3.6	9.3	11.0	0.01	0.03	4	9
Commiphora campestris Engl. *	CCA	5	8	2	3.8	3.9	23.6	20.4	0.89	1.50	309	520
Commiphora edulis (Kl.) Engl. ssp. *boiviniana* Gillett *	CED	2	<0.5		3.6	2.5	13.3	6.3	0.03	<0.005	9	<0.5
Commiphora incisa Chiov. *	CIN	<0.5	2	<0.5	1.9	2.1	6.3	6.7	<0.005	<0.005	<0.5	<0.5
Commiphora pseudopaolii Gillett *	CPA	2	4	1	3.1	3.5	12.9	12.4	0.05	0.12	24	56

Species	Acronym	Density, n ha^{-1}			Height, m		$dg_{0.15}$, cm		Usable volume, m^3 ha^{-1}		Usable biomass, kg ha^{-1}	
		> 1.3 m 1985	> 1.3 m 1994	< 1.3 m 1994	1985	1994	1985	1994	1985	1994	1985	1994
Commiphora rostrata Engl. var. *rostrata* *	CRO	< 0.5	4		3.4	2.5	8.0	6.4	< 0.005	0.01	1	2
Commiphora sp. *	CSP	4	5	7	3.9	3.7	13.2	12.6	0.07	0.10	24	31
Cordia sinensis Lam. *	CSI	7	14	4	2.2	2.2	6.6	6.9	0.02	0.05	12	30
Dobera glabra (Forssk.) Poir. *	DOB	2	4	4	3.6	2.8	22.5	15.8	0.16	0.19	106	127
Ehretia sp.	EHR		16	9		2.1						
Givotia gosai A.R. Smith	GIV		< 0.5	< 0.5		2.0						
Grewia lilacina K.Schum.	GLI		< 0.5			2.2						
Grewia villosa Willd.	GVI	1	6	1	2.1	2.5						
Grewia tembensis Fres.	GTM		1			1.9						
Grewia tenax (Forssk.) Fiori	GTE	25	159	49	2.0	2.0						
Grewia sp.	GSP	< 0.5	< 0.5		2.2	2.2						
Lannea triphylla (A.Rich.) Engl. *	LAN		< 0.5			2.7		5.5		< 0.005		< 0.5
Maerua decumbens (Brongn.) De Wolf *	MDC	1	3	5	1.8	2.1		4.6		< 0.005		1
Maerua denhardtiorum Gilg.	MDE			< 0.5								
Meyna tetraphylla (Hiern) Robyns	MTE	< 0.5			2.4		7.5		0.01		4	
Opilia campestris Engl.	OPI		3	1		2.1						
Phyllanthus somalensis Hutch.	PHY		1	2		1.7						
Premna velutina Gürke	PVE		< 0.5			1.4						
Prosopis juliflora (Swartz) DC *	PJU		14	6		2.5		4.2		0.06		44
Salvadora persica L. *	SAL	4	10	2	3.6	3.2	13.3	7.8	0.16	0.09	101	61
Sclerocarya gillettii Kokwaro / *Ximenia americana* L.	SGX		4	1		2.4						
Sesamothamnus busseanus Engl.	SES		< 0.5			2.4						
Sterculia africana (Lour.) Fiori *	STE	< 0.5	1		2.4	2.2	16.0	13.2	0.01	0.01	2	2
Terminalia orbicularis Engl. & Diels *	TOR	1	1		4.7	6.7	33.5	20.1	0.16	0.09	105	56
Wrightia demartiniana Chiov. *	WRI	3	1	< 0.5	2.6	2.7	13.3	11.2	0.03	< 0.005	18	1
TOTAL		102	370	252					3.23	4.35	2012	2602

Appendix 2. Mean density, height, diameter ($dg_{0.15}$), usable wood volume and biomass of trees above 1.3 m in height in undisturbed bushland under Vertic Natrargid soil at two successive inventories (n=71). Two different species are combined in some cases when locally identified as one. Diameter and volume calculation are based on species marked with an asterisk (*) only.

Species	Acronym	Density, n ha^{-1}			Height, m		$dg_{0.15}$, cm		Usable volume, m^3 ha^{-1}		Usable biomass, kg ha^{-1}	
		> 1.3 m 1985	> 1.3 m 1994	< 1.3 m 1994	1985	1994	1985	1994	1985	1994	1985	1994
Acacia mellifera (Vahl) Denth. *	AME	2	6	35	2.0	3.3	8.0	10.5	0.037	0.098	27	71
Acacia paolii Chiov. *	APA	<0.5			2.3		5.0		<0.001		<0.5	
Acacia reficiens Wawra *	ARE	8	15	28	2.6	2.8	9.6	9.9	0.086	0.144	78	131
Acacia tortilis (Forssk.) Hayne *	ATO	3	3	1	4.8	4.8	21.5	14.9	1.419	0.854	1056	635
Acacia zanzibarica (S.Moore) Taub. *	AZA	834	1175	193	3.2	3.5	4.5	4.3	2.578	3.591	2235	3113
Balanites aegyptiaca (L.) Del.	BAL			<0.5								
Blepharispermum lanceolatum Chiov.	BLE		2	1		1.5						
Boscia coriacea Pax / *Elaeodendron aquifolium* (Fiori) Chiov.	BCE		13	7		2.8						
Cadaba farinosa Forssk. / *Maerua triphylla* A.Rich *	MCF	1	3	2	3.2	2.6	13.3	9.7	0.022	0.033	14	21
Cadaba gillettii R.A.Graham / *Maerua macrantha* Gilg *	MCG		3	5		2.7		4.4		0.003		2
Cadaba glandulosa Forssk.	CGL		1	1		1.7						
Cephalocroton cordofanus Hochst.	CEP	1	<0.5	1	1.4	1.8						
Combretum constrictum (Benth.) Laws.*	CCO	9			2.1		2.5		<0.001		<0.5	
Commiphora campestris Engl. *	CCA	1	1		2.6	2.9	5.5	6.9	0.001	0.001	<0.5	<0.5
Commiphora incisa Chiov. *	CIN	<0.5	4		5.0	3.1		12.1		0.096		31
Commiphora pseudopaolii Gillett *	CPA		<0.5			2.0		2.0		<0.000		<0.05
Commiphora sp. *	CSP			1								
Cordia sinensis Lam. *	CSI	9	13	7	2.3	2.7	7.1	8.8	0.036	0.141	20	81
Dobera glabra (Forssk.) Poir. *	DOB	1	2	1	2.5	2.2	12.5	9.7	0.033	0.067	22	46
Ehretia sp.	EHR		9	1		2.5						
Euphorbia sp.	EUP			1								
Givotia gosai A.R. Smith	GIV		<0.5			1.3						
Grewia tembensis Fres.	GTM		<0.5			1.3						
Grewia tenax (Forssk.) Fiori	GTE	4	49	24	1.9	2.1						
Grewia sp.	GSP		<0.5	1		2.5						
Indigofera sp.	IND		<0.5	7		1.7						
Josephinia africana Vatke	JOS		2			2.6						
Lannea triphylla (A.Rich.) Engl. *	LAN		1			2.8		9.0		0.001		<0.5
Maerua decumbens (Brongn.) De Wolf *	MDC	1	1	<0.5	1.3	1.5		6.0		0.005		3
Meyna tetraphylla (Hiern) Robyns	MTE	1			2.7							
Opilia campestris Engl.	OPI		4	1		2.6						
Phyllanthus somalensis Hutch.	PHY	1	6	2	1.8	2.0						
Prosopis juliflora (Swartz) DC *	PJU			<0.5								
Salvadora persica L.*	SAL	42	100	70	2.8	2.9	7.8	6.4	0.395	0.885	258	579
Sclerocarya gillettii Kokwaro / *Ximenia americana* L.	SGX		3	3		2.8						
Sesamothamnus busseanus Engl.	SES		<0.5			1.9						
Sterculia africana (Lour.) Fiori *	STE			1								
Tennantia sennii (Chiov.) Verde & Bridson ined.	TEN		1			3.2						
Terminalia brevipes Pampan	TVP		1			2.4						
Terminalia orbicularis Engl. & Diels *	TOR			<0.5								
TOTAL		919	1419	395					4.61	5.92	3712	4712

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