TIME	TITLE	AUTHOR/S
PLATFORM SESSION: Savanna and Woodland Dynamics I and II (Session Chair: Harry C Biggs)		
14:00-14:20	A few more trees OR a biome shift? Should we be worried about the changes in woody cover that have occurred in a mesic Zululand savanna since 1937?	***Ben J Wigley***, *William J Bond and M Timm Hoffman*
14:20-14:40	Secondary succession in the Mopani Veld of the Limpopo Valley, South Africa	***Jorrie J Jordaan***, *Dirk C J Wessels, Chris S Dannhauser and Gerrit T Rootman*
14:40-15:00	The importance of large trees in a savanna - consequences of their loss	***Rina (C) C Grant*** *and Laurence Kruger*
16:00-16:20	Can fire still control woody biomass in savannas? A preliminary look	***An van Cauter***, *Matt Waldram and William J Bond*
16:20-16:40	The effects of species and nutrient availability on tree-grass competitive interactions	***Michelle J Payne***, *Craig D Morris, Richard W Fynn and Kevin P Kirkman*
16:40-17:00	Using pre-dawn leaf water potential to explore critical limits to woodland growth	***Tony (A) R Palmer*** *and Derek Eamus*
17:00-17:20	Why are there no savanna trees in the highveld? An *Acacia* experiment	***Julia L Wakeling***, *William J Bond and Michael D Cramer*

Savanna and Woodland Dynamics I and II

SESSION CHAIR: HARRY C BIGGS

Platform Presentations

A FEW MORE TREES OR A BIOME SHIFT? SHOULD WE BE WORRIED ABOUT THE CHANGES IN WOODY COVER THAT HAVE OCCURRED IN A MESIC ZULULAND SAVANNA SINCE 1937?

Ben J Wigley*[*#], William J Bond and M Timm Hoffman

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To date bush encroachment has predominantly been attributed to changes in land use practices and management. Traditionally overgrazing and changes in burning practices are thought to play the major role in driving bush encroachment. To test whether land use practices alone can explain the changes in woody cover observed at the study sites we compared rates and extent of changes in woody cover for areas under different land use practices (communal, commercial and conservation) in the Hluhluwe area, Northern Zululand. Repeat aerial photography was used to document changes in vegetation between 1937 and 2004. We then reconstructed past management and changes in land use practices for each of the three areas using archive data and oral histories. Results showed major increases in woody cover to have occurred across all three of the land use systems in the 67-year period. The communal study site showed the least increase in tree cover. However the overall increase in tree cover from 6.2 to 25.7% (fourfold increase) is still a highly significant change. The greatest increase in tree cover was evident at the commercial study site where tree cover increased by seventeen fold. Total tree cover increased from 2.7 to 50.8% during the 67-year period. The increase in tree cover at the conservation study site was also highly significant. Tree cover increased by ~360% over the period from 14.7% in 1937 to 58.5% in 2004. Land use histories for each area were also shown to be significantly different. The results suggest that past land use practices did have significant impacts on the type of and extent of bush encroachment. However the findings suggest that land use practices alone cannot explain the widespread occurrence of bush encroachment in the area. This suggests that a widespread global driver such as increased atmospheric CO_2 concentrations or changing climate is driving the changes. The implications of these findings are considerable and will be discussed.

SECONDARY SUCCESSION IN THE MOPANI VELD OF THE LIMPOPO VALLEY, SOUTH AFRICA

Jorrie J Jordaan[1*#], Dirk C J Wessels[2*], Chris S Dannhauser[3] and Gerrit T Rootman[1]

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The long-term establishment sequences of the herbaceous and woody component of the Mopani veld of the Limpopo Valley were studied and documented. The experiment was conducted at the Messina Experimental Farm, situated in the Mopani veld (veld type 15, Acocks 1988) approximately 20 km west of Musina. The long-term mean annual rainfall of Messina Experimental Farm is 341.6 mm. During summer, temperatures often exceed 40^0C while temperatures seldom drop below freezing point during winter. During 1964, three sites were fenced off to prevent them from being grazed by livestock and game. Site 1 is situated in *Kirkia acuminata* - *Enneapogon cenchroides* short closed woodland on a shallow Hutton soil, Site 2 in riparian Mopani veld *(Blepharis diversispina* - *Combretum apiculatum* low closed woodland) on a deep Hutton soil and Site 3 in *Blepharis diversispina* - *Combretum apiculatum* low closed woodland on a very shallow, rocky Mispah soil. Experimental work at Site 3 had to be terminated in 1983 due to the erection of an electric fence as a border between South Africa and Zimbabwe, which prevented access to the site. The other two sites are currently maintained as protected sites at the Messina Experimental Farm.

Between 1964 and 1983, the grass component at the three sites was monitored annually via a 1000-point wheelpoint survey per site. Only grass species of which the rooted area of the tuft was struck by the monitor spike of the wheelpoint apparatus were recorded and results (percentage species composition) were therefore based on percentage basal cover. A similar survey was done during 2001. The woody component was surveyed at all three sites during 1969, 1982 and at the two remaining sites during 2001 by counting all woody individuals at each site. Data were expressed as tree density.

No clear fixed grass species establishment sequence was observed. The classic plant succession model of the grass component, as usually found in other veld types (for instance pioneer- through sub-climax- to climax veld that occur in various thornveld types), is absent in Mopani veld of the Limpopo Valley. In comparison to deeper soils, where perennial grasses usually dominate, shallow, nutrient poor, gravely and severely eroded soils with low moisture retention and a high runoff rate occur at the sites. They are mainly colonised by annual grasses and forbs. Perennial grasses take long to establish. This results in an unstable herbaceous component that is vulnerable to changes, which in turn depend on the erratic availability of soil moisture.

Where the woody component is concerned, eight tree species appeared to be early colonisers (*Copophospermum mopane*, *Combretum apiculatum, Boscia albitrunca, Dichrostachys cinerea, Terminalia prunioides, Commiphora* spp, A*cacia. erubescens* and *Grewia* spp.), five intermediate colonisers (*Combretum mossambicense, Fluegia virosa, Gardenia resiniflua, Ochna inermis* and *Ximenia americana* and seven late colonisers *(Acacia senegal, Euphorbia ingens, Lannea schweinfurthii, Tinnea rhodesiana, Ehretia rigida, Schlerocarya birrea* and *Sterculia rogersii).* Tree numbers increased at the sites with deeper soils, indicating that bush encroachment is a general, natural process that occurs irrespective of rest treatments applied to the herbaceous component. At Site 1, bush density increased from 948 to 1679 trees ha^{-1} between 1969 and 1982. Between 1982 and 2001, bush density decreased from 1679 to 1127 trees ha^{-1}, mainly due to a major decrease in the *D. cinerea* population. In total, bush density increased by 19% between 1969 and 2001 at site 1. The number of tree species that occurred at site 1 increased from 11 to 17. At site 2, bush density increased from 1542 to 1841 trees ha^{-1} between 1969 and 1982 and from 1841 to 2948 trees ha^{-1} between 1982 and 2001. Overall, bush density increased by 91%. The number of tree species increased from 11 to 15 between 1969 and 2001. At Site 3, bush density decreased from 712 to 548 trees ha^{-1}, totalling 23%, between 1969 and 1983. The number of tree species that occurred decreased from 15 to 10. This is probably due to the extremely shallow soil, which, together with high summer temperatures and low rainfall of the Limpopo Valley, is insufficient to maintain a proper vegetation cover. It is suggested that the bush encroachment process in the

Limpopo Valley is much more affected by soil and climate than in other veld types and climatic zones of the Limpopo Province.

THE IMPORTANCE OF LARGE TREES IN A SAVANNA - CONSEQUENCES OF THEIR LOSS

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Understanding the role of large trees in the Kruger National Park in maintaining biodiversity has become more important, as evidence of the decline in large trees over the past 50 years has mounted. The cause of this decline is not yet clear, but it is likely to be a result of a combination of the action of fire and elephant.

The removal of tall canopies seems to have little effect on bird and small mammal diversity. Research by students from the Organisation of Tropical Studies has shown that bird diversity is as high in low canopy cover along the Sabie River as in high canopy cover. Small mammal diversity is also not affected by canopy height, but there are species differences. However, in the Punda Maria area, the diversity of foliage feeding species was higher in tall canopy mopane trees than in the medium and lower height classes.

It does not seem as though species have been lost due to elephant utilization yet, but more data is required to confirm this. Certain tree species are more utilized that others and some landscapes are more heavily utilized. Along the Olifants River, a perennial river in the granites, 80% of *Diospyros mespiliformis* and 70% of *Croton megalobtris* specimens showed elephant impact. In contrast, on the Lebombo Mountains, a much richer soil type, the heaviest impact was also on *Diospyros mespiliformis* but only 45% of trees were impacted followed by *Combretum appiculatum* of which only 32% of trees showed some impact.

Large trees play an important role in nutrient cycling, and N concentrations are up to four times as high under the canopy as away from the canopy. Grazers also select strongly for grazing under these trees rather than outside the canopy cover. These nutrient hotspots are likely to be an important determinant of grazer populations.

At this stage, data is still inefficient and elephant densities probably too low too detect significant affects on biodiversity due to elephant impact, although there are some indications that a loss of structure and tall trees may have effect system function.

CAN FIRE STILL CONTROL WOODY BIOMASS IN SAVANNAS? A PRELIMINARY LOOK

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We propose that bush encroachment of savanna landscapes takes two forms: the first being a thickening up of typical savanna tree and shrub species to become closed to very dense woodland or bushveld; the other being a complete biome shift from savannah to thicket, characterized by a change in structure and species composition. This distinction is an important one when trying to manage for these sorts of changes. According to the management plan of Hluhluwe iMfolozi Park, fire is used to 'renew moribund grass and control bush encroachment'. At least in Hluhluwe there has been a failure to achieve the second goal, and bush encroachment is a major problem, especially in terms of an increase in density of woodlands. Given that there is little information available on how the two forms of bush encroachment are interrelated – if at all – this work *focuses* on the use of fire to control the thickening up of open savannas to dense woodland or bushveld and *touches* on how fire may possible affect thicket species in a dominantly savanna landscape.

To address the 'bush thickening issue, we pose two questions: 1) Can fire be used to control woody encroachment? And, 2) What is the most effective way to apply fire i.t.o. weather conditions and season to do this? We are attempting to answer these questions by monitoring tree and shrub populations in large (~5ha), replicated burning trials in the northern section of Hluhluwe iMfolozi Park. Four burn treatments have been applied - these are: a) 'Winter intense' treatment. >60 FDI. b) 'Winter mild' treatment, designed to emulate how we think the majority of burning has been done over the last century. c) 'Spring burn' which are applied in spring once the grass has greened up and *Dichrostachys cinerea* is in flower (an indicator that trees and shrubs stored carbon reserves are low, having resprouted after the dry season during which they are largely dormant). d) 'No burn' treatment. This is left unburnt. Preliminary results indicate that tree mortality rates after fires is virtually zero. Also, an increase in fire intensity, within season, produced greater top kill (i.e. trees are resprouting from rootstock), however, fires conducted in spring (low intensity) had an effect on topkill at least as large as winter intense burns. This suggests that savanna trees do not respond to fire intensity alone (in contrast to popular belief) and that season of burn may be an important factor to consider when trying to control the thickening up of savannas.

We have only started to attempt addressing the issue of biome shifts to thicket and will thus be presenting some initial results of fire trials with saplings of thicket species and some proposed ideas on testing the affects of fire on thicket bush clump dynamics. Future work can then address how the two types of encroachment are interrelated.

THE EFFECTS OF SPECIES AND NUTRIENT AVAILABILITY ON TREE-GRASS COMPETITIVE INTERACTIONS

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Savanna composition and structure is influenced to a greater or lesser degree by resource availability and disturbance. Resource availability is influenced by the physical environment, climate and competition for resources. Disturbance by various factors such as fire and herbivory impact the ability of savanna trees to utilize resources, grow and reproduce. Grasses, a major component of savanna vegetation, compete directly with trees for light, water and soil nutrient resources. Direct causes of bush encroachment are not always apparent, but are commonly ascribed to overgrazing and consequent decreased grass production. The impacts of below ground competition between trees and grasses on tree growth were identified as a critical focus area to improve understanding of the impacts of grass on trees and vice versa.

In order to investigate the competitive ability of specific tree and grass species against one another various species combinations were created and grown under both nutrient rich and nutrient poor conditions. These combinations were planted in pots with a tree species as the central phytometer with four grass seedlings planted around it. In addition the trees were grown alone and the grasses were grown in the same arrangement of four seedlings with the central phytometer absent. The tree seedlings used were *Acacia nilotica* and *Acacia karroo,* while the grass species used were *Themeda triandra, Eragrostis capensis, Hyparrhenia hirta, Aristida junciformis, Sporobolus africanis* and *Panicum maximum*. The nutrient rich treatments were watered with approximately 300 ml of 80% Hoagland's solution every 4 days while the nutrient poor treatments were given the same amount of water but no additional nutrients. Every 2 weeks plant height and number of leaves per tree and number of tillers per grass plant were measured. After 26 weeks final measurements were taken. The length of both tree and grass roots was measured. The roots were then cut at 5 cm, 10 cm, 15 cm and 25 cm lengths and each section weighed separately. Competition exerted on *A. karroo* seedlings by the surrounding grass depended to some extent on the grass species (P=0.07), with *E. capensis* reducing the above ground biomass, compared to the control, to the largest extent. The addition of nutrients (P<0.001) increased the competitive ability of all grass species considerably. Similar competitive effects were observed on the roots of *A. karroo*, with *E. capensis* being the most, and *P. maximum* the least, competitive grass species. Competition exerted on *A. nilotica* was found to be more dependent on grass species (P=0.002) than nutrient level (P=0.058) with *H. hirta* and *E. capensis* having the greatest effect on above ground biomass and *A. junciformis* and *P. maximum* having the least effect. Similar effects were seen on below ground biomass. Grass species composition and soil nutrient levels are likely to influence savanna structure and composition.

USING PRE-DAWN LEAF WATER POTENTIAL TO EXPLORE CRITICAL LIMITS TO WOODLAND GROWTH

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Pre-dawn leaf water potential provides an index of soil moisture status and is useful in quantifying the soil moisture conditions associated with variations in plant water stress. In arid and semi-arid rangelands, pre-dawn water potentials are useful in determining the critical period when plant water use and growth occurs. Pre-dawn leaf water potentials have been collected routinely in Australian Eucalyptus woodlands. However, the ecosystem leaf area index (LAI) is seldom available to permit comparison of pre-dawn water potential and LAI. LAI is a key biophysical variable controlling the exchange of energy, mass (e.g. water and CO2) and momentum between the Earth surface and atmosphere. Since 2000, the MODIS sensor has collected reflectance data which have been used to develop LAIs using a radiative transfer model. Following an analysis of pre-dawn water potentials from Australian woodland ecosystems (evergreen Eucalyptus woodland and savanna) over a three year period, we identified patterns in pre-dawn water potentials which provide a useful understanding of the role of soil moisture status in controlling LAI. In the northern savannas of the Northern Territory, the MODIS LAI product has a basal value of 0.96 during the dry season compared with a mean value of 2.5 for the wet season. This dry season value is equivalent to the LAI component of the woodland alone and corresponds well with ground-measured LAIs from other sites in the region. Pre-dawn water potentials are lowest (more negative) during the middle of the dry season (late October) at -2.5 MPa and highest (very close to 0) during the beginning of the wet season (early March). In mid-July there is a critical value of -1 MPa when the MODIS LAI displays a rapid decline from the maximum wet season value to the basal rate of approximately 1. This threshold switches off the understorey contribution to LAI and in savanna identifies the point at which understorey growth stops. This has important implications when using the MODIS LAI to model forage production.

WHY ARE THERE NO SAVANNA TREES IN THE HIGHVELD? AN *ACACIA* EXPERIMENT

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Numerous hypotheses have been proposed as to why there are no savanna trees growing on the highveld grasslands, however none appear to have been tested. We hypothesise that at higher elevations, growing seasons are too short for savanna trees to grow fast enough to escape the frequent fires. A seedling transplant experiment across a 1700m altitudinal gradient is used to test this importance of temperature on *Acacia* tree growth. Preliminary results show that trees do grow significantly slower at higher altitudes, and this, together with frequent fire, may contribute to exclusion of savanna trees from the grasslands.