Evaluation of the effects of tree clearing over time on soil properties, pasture composition and productivity

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Abstract

Tree clearing is practised for greater beef production and hence monetary gains from grazing systems of central Queensland. The high rates of clearing in the past and even recently (577, 000 ha/yr during 1999-2001) were mainly to develop land for pastures. The sustainability of cleared pasture systems over the long-term is questioned.

Three major types of tree communities i.e. *Eucalyptus populnea* F. Muell., *E. melanophloia* F. Muell. and *Acacia harpophylla* F. Muell. ex. Benth. were selected on one property in central Queensland to quantify the impacts of clearing on pasture production and composition, and soil properties. The impacts were measured over time-since-clearing (recent (<5 years), medium (11-13years) and old (>30 years)) in unreplicated cleared pastures in comparison to their replicated uncleared/intact woodland pastures of each tree community.

Measures of pasture above-ground biomass production on a single property over time-since-clearing in cleared systems showed that gains were not sustained over the long-term. The difference in response to clearing between tree communities was evident and important to support the future policy decisions. The impact of clearing on soil properties (physico-chemical and biological) was confirmed, and explained the lesser availability of nutrients with time of clearing in cleared pastures. The changes in some soil properties underscored the associated risks and changes in ecosystem functions due to clearing. Less litter was produced at cleared than uncleared pastures, but nutrient release was faster at cleared compared to uncleared systems. The overall effect of clearing in terms of pasture and litter production, and major soil parameters were analysed using multivariate analyses.

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Declaration of originality

This thesis reports the work of the author, except as otherwise stated. It has not been submitted previously for a degree at any university or other institution.

Kamaljit K. Sangha

Chapter 1. Introduction

Research problem

Tree felling, tree clearing and deforestation to develop productive agricultural systems are significant issues of concern for the global community. Worldwide, deforestation rates have increased to 14.6 million ha per annum during the last decade (FAO 2001). Australia is at sixth rank with the rate of 564,800 ha clearing per year during 1990-2000. About 75 per cent of the total clearing in Australia occurred in Queensland (State of the Environment Advisory Council 1996). Recently the Queensland Department of Natural Resources and Mines (2003) reported that the rate of clearing has increased 36 % during 1999-2001 to 577,000 ha/yr, compared to 425, 000 ha/yr in 1997-99. Interestingly, 94 per cent of the total cleared land during 1999-2001 was to develop pastures. The issue is of major concern for the environmentalists, land managers and developmentalists throughout the state.

Pastoralism is the main industry for Queensland, contributing about $3.8 billion in 2000-2001 to the total state economy (ABS 2001). Since the beginning of the last century until 1985, various governmental policies for settlement schemes and incentives encouraged clearing on a large scale to develop land for pastures and cropping (Boulter *et al.* 2000; Isbell 1962). The main reason for clearing is to increase productivity and hence the monetary benefits. In the process of developing ‘productive’ pasture systems, much of cleared area was sown to various exotic grasses especially buffel (*Cenchrus ciliaris* L.). These species performed well to capture the flux of nutrients available upon clearing. Due to large scale cultivation of a few exotic grasses, a monoculture set of pastures has been created on most of the cleared land. There is a general perception among the beef producers’ community of Queensland that clearing trees followed by sowing to exotic grasses such as *C. ciliaris* leads to greater pasture productivity. The earlier work by Burrows (1993), Burrows (1999), Burrows *et al.* (1999) and Scanlan (1991) showed enhanced pasture production with tree clearing.

The approach to achieve greater monetary gains from ‘developed’ pastures, though may be for a short term, is preferred to date. However, there are risks associated with clearing in respect of loss of biodiversity, greater soil erosion, soil acidification and salinity, and loss of ecological functions and resilience of an ecosystem (Boulter *et al.* 2000). Moreover, the interactive effects of these functions may lead to serious problems. The recent reports by

Boulter *et al.* (2000) and National Land and Water Resources Audit (2002) have raised concerns about the land use practices followed in the past. There are few data available that relate to the impact of clearing over time.

Recent government policies (Vegetation Management Act 1999), however, have introduced restrictions on clearing based on the knowledge of associated risks for clearing. This has further concerned the farming community as without providing the facts for restrictions upon tree clearing, they are expected to follow the new rules. Over the past 15-20 years this shift in policies from clearing to not clearing needs convincing supporting evidence so that such impacts, whatever they may be, can be easily made understandable for future generations. Moreover, a detailed knowledge of ecological consequences will also help to interpret the outcomes of any such future developmental activities over the long term.

Therefore, it is important for policy makers as well as land managers to recognize the consequences of tree clearing in detail. This would require the answers to questions such as: which pasture system is better - cleared or uncleared, and for cleared systems how long can the pasture production be sustained over time after clearing? Most of the work done to date (Burrows *et al.* 1990; Walker *et al.* 1972) has quantified the relationship between pasture yield and tree clearing for the most recent years after clearing (< 10 yrs) but not over a longer time period (Scanlan 2002). The initial few years after tree clearing with increased pasture yield is the most attractive phase for a producer to achieve maximum gains due to clearing. But in the absence of long-term studies for cleared pastures, it is difficult to conclude about the benefits/losses from clearing for pasture development. This would also require an understanding of the various ecological attributes (parameters) affected due to clearing and their status in natural woodlands. Moreover the responses may vary with time-since-clearing among various types of tree communities.

Aim of research

The study was set up at a property in central Queensland to quantify the effects of land use change from woodland pastures to cleared pastures and the sustainability of those cleared pastures over time. Ecological attributes quantified were pasture availability, composition, forage quality, dynamics of species change, soil status, litter production and decomposition, and nutrient return in maintenance of ecological services provided by an ecosystem (e.g. nutrient cycling, shade, diversity of flora and fauna).

The aim is to provide a better knowledge-base for landholders to help them understand the importance of ecological services provided by pasture systems and to judge the value of these ecological functions for sustainable productivity while making decisions for future production systems. The study focused on one property in central Queensland, with a number of representative natural woodlands and clearing regimes, which will provide information for a range of landholders to interpret the impacts of their own clearing.

Research approach

The present study evaluated the effects of clearing on ecological functions over varying time frames for three main types of woodland communities in central Queensland: *Eucalyptus populnea* F. Muell. (poplar box), *E. melanophloia* F. Muell. (silver-leaved ironbark) and *Acacia harpophylla* F. Muell. ex. Benth (brigalow). Of these, *A. harpophylla* grows on more fertile/alluvial soils and has been chiefly targeted for clearing in the past, while *E. populnea* and *E. melanophloia* grow on marginal country (soils) and are undergoing clearing in recent years, after most of the *A. harpophylla* (brigalow) country has been cleared.

To study the impact of time on productivity and other ecological attributes since clearing, three different time-since-clearing treatments (recent <5 year, medium 10-15 years and old >30 years) were selected for each tree community. An important aspect of the study was to include paired comparisons for each tree community for uncleared woodland and cleared sites at each time-since-clearing. As detailed in the aim of research, the following main ecological attributes were considered:

- Vegetation
- Soil
- Litter

These attributes were examined for the time-since-clearing treatments for cleared and uncleared plots. The understanding of interactions among different attributes, and their status in cleared and native woodlands will provide information about pasture and soil run-down for the central Queensland region. Details on the various research sites and experimental design are presented in Chapter 3.

Chapter 2. Literature review

Our life style has changed with the developmental activities that occurred during the 20th century and has become progressively faster, and more productive and more comfortable in material terms. Human tendency to control/manipulate the biophysical/natural processes for comfort has also become obvious in the last century. Despite all the developments in modern life, we are still very much dependent upon the availability of natural resources and their functions for the basic needs: food, fibre and shelter.

Land clearing and development to fulfil human demands often trade off development on the one hand and conservation of natural resources on the other. Availability of natural resources as well as development are vital for human living. The need is to find a balance between the development and preservation of natural resources to sustain human activity on earth.

According to the recent report by FAO (2001), 14.6 million ha of land were cleared yearly during the last decade. Australia is among the top 10 countries that account for about 50 per cent of the total clearing and followed Brazil (2,226,000 ha/yr), Indonesia (1,312,000 ha/yr), Sudan (959,000 ha/yr), Zambia (851,000 ha/yr) and Mexico (631,000 ha/yr), in the sixth position with 564,800 ha per year during 1990-2000 (FAO 2001).

In Australia, 59 per cent (456 million ha) of total land (769 million ha) is used for agriculture. Grazing for beef production is the major land use (National Land and Water Resources Audit 2002) and alone contributed about $5,051 million during 1999-2000 to the Australian economy (ABS 2003).

The high rate of land clearing to enhance agricultural production is of concern in Australia as it is the driest of the world’s inhabited continents with poor soils. The Australian State of Environment Committee (2001) reported “degradation of lands and waters remains of critical concern, especially in the intensive land use zone upon which much of Australia’s agricultural production depends.”

Modern agricultural practices by European settlers led to commercial farming systems in contrast to the subsistence-based fire practice and management by traditional owners. The use of unallocated natural resources for agricultural production has made the country self reliant in food production and other resources (National Land and Water Resources Audit 2002).

However historically, there has been little understanding about the limits to production using natural resources available on this continent. The absence of mutual understanding/exchange of knowledge between the traditional owners and the new settlers about the biophysical processes and the introduction of new species and techniques to increase production had a major impact on land use (National Land and Water Resources Audit 2002).

Details on woody vegetation clearing (referred as tree clearing in the text) in Australia especially in Queensland, and its ecological impact on grazing systems in terms of pasture production and composition, soil, litter production and decomposition are discussed below.

2.1. Tree clearing

In Queensland, the Department of Natural Resources and Mines (2003) estimated the average annual rate of tree clearing during 1999-2001 to be 577,000 ha (367, 000 ha of remnant and 210, 000 ha of non-remnant vegetation) (Fig. 2.1). This rate is 36 per cent greater than the rate in 1997-99 (425,000 ha/yr) and 41 per cent greater than in 1995-97 (340,000 ha/yr; Department of Natural Resources 2000). About 94 per cent of total cleared land was used for pasture development (Fig 2.2).

Clearing activities in Queensland started in the 1850s but gained momentum only in the 50 years following World War II, as the availability of heavy machinery made the clearing operations easier. Earlier government policies also promoted tree clearing for land development purposes in the form of pastures, agriculture, mining and infrastructure (Isbell 1962).

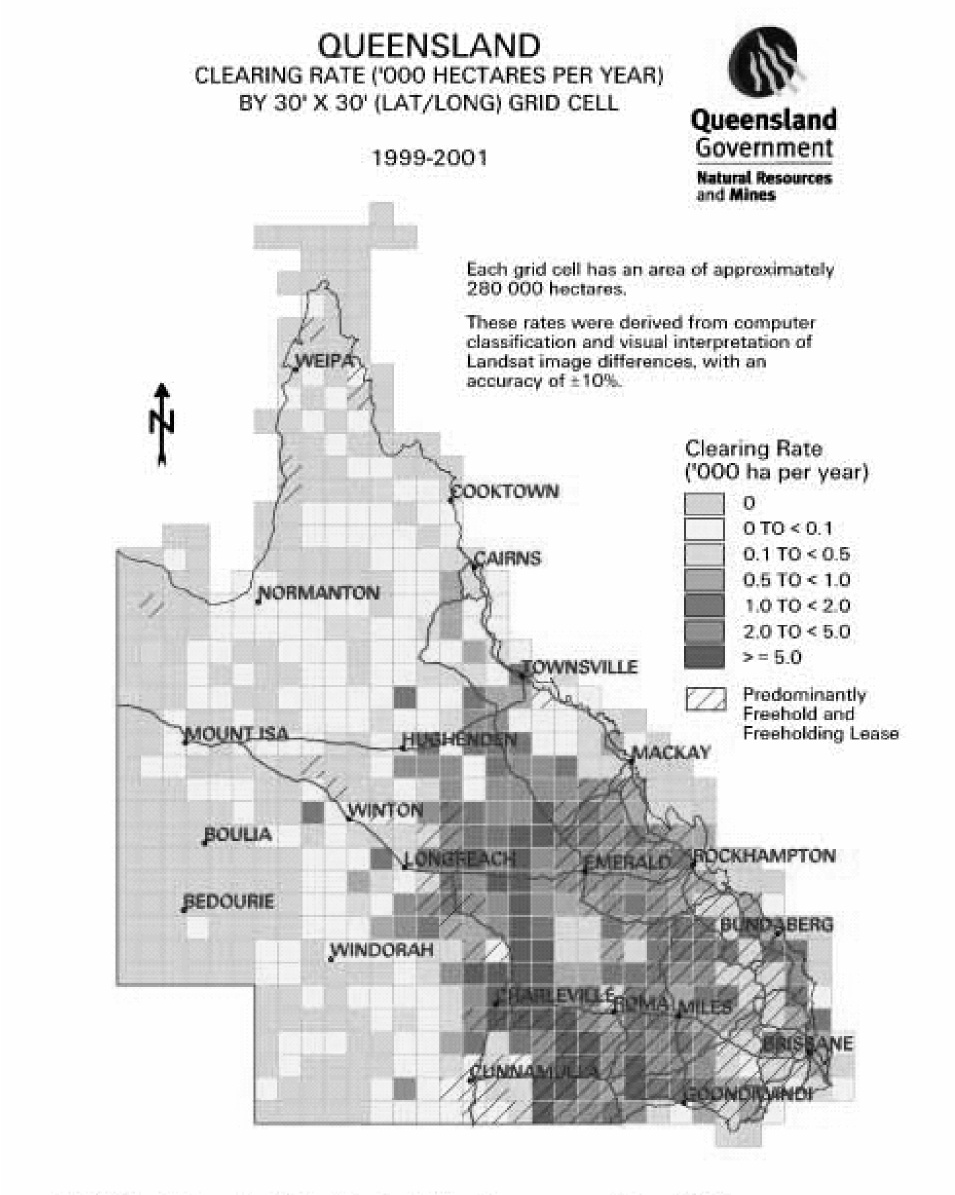

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Figure 2.2. Average annual rate of clearing for the period of 1999-2001. (Source: Queensland Department of Natural Resources and Mines 2003).

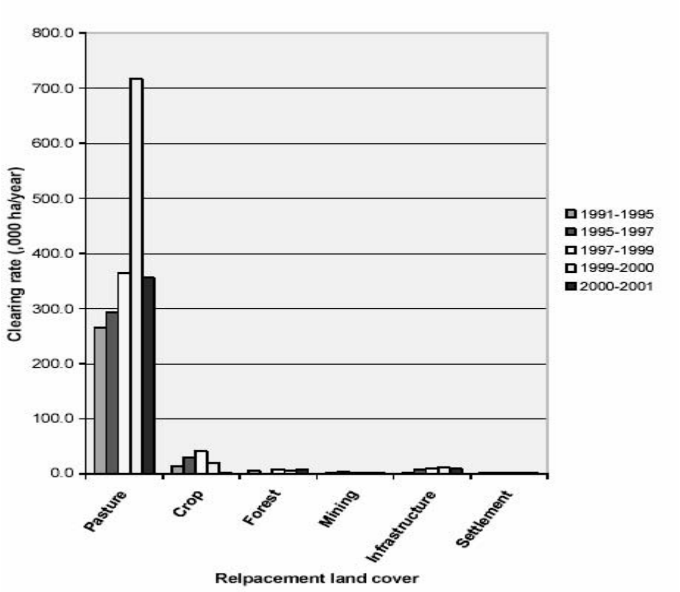


Fig 2.2 Clearing rates for different land uses (Source: Queensland Department of Natural Resources and Mines 2003).

Such high rates of vegetation clearing may have implications for loss of biodiversity, habitat fragmentation, weed invasion and land degradation especially increase in soil erosion and decline in soil fertility and soil structure (Boulter *et al.* 2000). In the past, tree clearing was considered as the key stage in the 'development'/creation of productive pastures. The main drivers for clearing are (Rolfe 2002):

- Increase in pasture production and the subsequent monetary gains
- Control of vegetation thickening and regrowth
- Easy access to the paddocks and dams
- Facilitated cattle mustering

Pasture production gains and commercial returns are the primary drivers for landholders to consider clearing option. A negative exponential relationship between pasture production and tree basal area demonstrated the greater production gains following clearing (Burrows *et al.* 1990; Scanlan and Burrows 1990).

The second important reason for clearing is to control vegetation thickening/proliferation. Vegetation thickening/proliferation of woody plants in pasture lands in most of the world's savannas occurred due to changed (lessened) fire regimes and increased grazing pressure. The latter facilitates seed dispersal, propagation of unpalatable species and reduces competition between shrubs/trees and grasses for available soil moisture in the presence of meagre grass growth (Archer 1994; Burrows *et al.* 1988; Scholes and Archer 1997). Land clearing is also practised to control vegetation thickening and to restore the original carrying capacity of land.

Schmidt and Lamble (2002) stated "regrettably, no long-term studies exist in Queensland and no studies have been carried out to encompass a wide range of ecosystem components to identify how nutrient-poor savannas function".

2.2. Tree clearing - pasture productivity and composition

Human activities are a major cause of biodiversity extinction. The changes (especially reductions) in biodiversity can alter the ecosystem processes and affect the resilience of ecosystems (Chapin III *et al.* 2000). The global biodiversity scenario for the year 2100 developed by Sala *et al.* (2000) showed that there are five major drivers: land use, atmospheric CO_2, nitrogen deposition and acid rain, climate, and biotic exchange (i.e. introduction of exotic species) for biodiversity change in ten terrestrial biomes on the earth. Among these drivers, land use change is expected to have the greatest impact on biodiversity change in all the biomes, particularly in grasslands and savannas where land use followed by atmospheric $CO_{2,}$ are the most important factors driving biodiversity change.

In Australia, large areas of land were cleared for cropping and to develop exotic pastures for grazing in the past 200 years (Fensham 1989; Prober and Thiele 1995 cited in Yates and Hobbs 1997). The conversion of woodlands to agricultural land represents one of the most significant vegetation changes in Australian history. The impacts of tree clearing on pasture production, species diversity and the relationship between these two, and cattle diet are now discussed.

2.2.1. Tree clearing and pasture production

The removal of woody plants leads to improved pasture production. Burrows (1990), Burrows *et al.* (1990), Scanlan and Burrows (1990) and Scanlan (1991) studied the allometric relationships for pasture yield and tree basal area for different tree communities growing in central Queensland. Their data showed a negative exponential relationship between pasture yield and tree basal area (Fig 2.3).

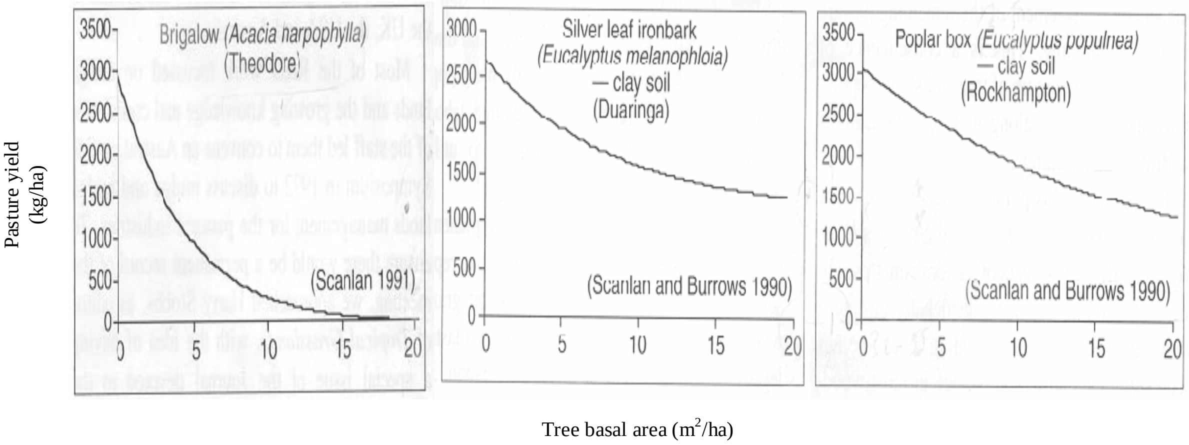


Fig 2.3. Pasture yield (kg/ha) and tree basal area (m^2/ha) relationship for *A. harpophylla, E. melanophloia* and *E. populnea* (combined source: Burrows 2002).

Similar results for increase in pasture yield with tree killing were also noticed in north Queensland (Jackson and Ash 1998; McIvor and Gardener 1995). There are also other reports (Belsky *et al.* 1993; Belsky and Amundson 1992; Belsky and Amundson 1998; Belsky 1994; Mordelet and Menaut 1995) available throughout the world showing reduced herbaceous/pasture production under trees (Table 2.1).

Table 2.1. Pasture productivity in various cleared and uncleared grazing systems.

Studies	Treatments	Tree basal area (m^2/ha)	Tree type	Pasture yield (kg/ha)	Region	Reference
1	Trees	15.00	*E. populnea*	1040	NE Queensland	Burrows *et al.* (1990)
	Cleared*	3.00		2050		
	Cleared	0.00		3000		
2 (wet site)	Trees	40.00	*E. crebra*	250-300	NE Queensland	Burrows *et al.* (1990)
	Cleared			3500		
3 (dry site)	Trees	10.00	*E. crebra*	250-300	NE Queensland	Burrows *et al.* (1990)
	Cleared			1200-1300		
4	Trees	30.20	*E. melanophloia*	1129	Central Queensland	Scanlan and Burrows (1990)
	Cleared	0.00		2581		
5	Trees	27.70	*E. populnea*	490	Central Queensland	Scanlan and Burrows (1990)
	Cleared	0.00		1816		
6	Trees	25.30	*E. crebra*	352	Central Queensland	Scanlan and Burrows (1990)
	Cleared	0.00		1900		
7	Trees	9.70	*A. harpophylla*	190	Central Queensland	Scanlan (1991)
	Cleared	0.70		2430		
8	Trees	Data not available	*E. melanophloia*	1000	Central West Queensland	Tothill (1983)
	Cleared			2300		
9	Trees - at Hillgrove	5.50	*E. crebra* and *Corymbia erythrophloia*	905	N Queensland	Jackson and Ash (1998)
	Killed trees-at Hillgrove			1805		
	Trees- at Cardigan	3.50	*E. drepanophylla*	1110		
	Killed trees - at Cardigan			1415		
10	Trees - at Hillgrove	5.50	*E. crebra* and *Corymbia erythrophloia*	1980	N Queensland	McIvor and Gardener (1995)
	Killed trees-at Hillgrove			3950		
	Trees- at Cardigan	3.50	*E. drepanophylla*	1990		
	Killed trees - at Cardigan			2540		
11	Trees	7.02	*Crossopteryx febrifuga, Bridelia ferruginea* and *Cussonia barteri*	3220	France	Mordelet and Menaut (1995)
	Cleared			6320		
12	Trees		*Acacia tortilis*	2880	East Africa	Belsky and Amundson (1992)
	Cleared			6480		

* 20 % retention of trees

The tree-grass competition for nutrients, radiation and soil moisture are considered as the main reasons for lower pasture productivity under trees (Burrows *et al.* 1988; Belsky *et al.* 1989; Belsky and Amundson 1998). In semi-arid environments such as in central Queensland, the availability of soil moisture acts as a primary limiting factor to determine plant growth (Scholes and Archer 1997). The removal of trees, thus, may enhance pasture production, but clearing for pasture development raises the questions on sustainability in production over time and suggests that ecosystem functions may differ with the change in vegetation structure.

In contrast, there are reports available on greater pasture yield in the presence of trees compared to open grasslands (Belsky *et al.* 1989 cited by Belsky and Amundson 1998; Belsky *et al.* 1993) for the main reason of improved soil nutrient status in the presence of trees.

In most of the abovementioned studies, the age of cleared pastures since clearing was not mentioned. This leads to uncertainty as to how long a cleared pasture could sustain the additional amount of pastures that became available upon clearing.

Schmidt and Lamble (2002) presented two scenarios on nutrient relations in cleared pasture systems:

- A sustainable (cleared) system with stable nutrient content where herbaceous vegetation would perform the same ecosystem functions as the trees to maintain soil stability. Pasture run-down is not expected in this system.

- An unsustainable system where trees perform different ecosystem functions to those of the herbaceous vegetation and play an important role in nutrient cycling. Thus over the medium and long term, removal of trees would result in pasture run-down.

The authors emphasised that with the present knowledge, and in the absence of any long-term studies on nutrient cycling in the Australian context, it is difficult to predict the changes that occur upon clearing.

The economic viability of clearing for pasture development was largely judged from production gains only. Burrows *et al.* (1999) calculated the economics of clearing in poplar box (*Eucalyptus populnea*) woodlands for pasture productivity. With 100 per cent tree clearing using soil absorbed arboricide - tebuthuiron at1.0 kg/ha - the net present value was \$79.5/ha whereas with 20 per cent retention of trees in the form of strips net present value was \$63.6/ha - both figures are relative to the control (intact woodland).

The economic viability of beef and pasture production calculated by Rolfe and Donaghy (2000) revealed that the net returns (which include overhead expenses, family labour and returns on capital invested) are very low or even negative for many beef producers who have fewer than 1000 cattle. Further, Rolfe (2000) pointed out that the impact of tree clearing and developments in husbandry and cattle breeeding, appears slight for beef production as more than six million hectares of land were cleared between 1977 to 1997 but cattle numbers declined over this period and total meat production increased only slightly. Later Rolfe *et al.* (2002) estimated the economic benefits of vegetation clearing for pasture development for the period 1957-8 to 1997-8. They found that clearing could have allowed a production increase of approximately 3.8 m cattle equivalents over that time period. However, after allowing for inter-state transfers and live exports, other production gains from feedlots, sowing of legumes and other improvements in pasture systems, there was only an increase of 2.8 m beef cattle equivalents over the 40 year time period. The reasons for loss (lower than expected gains) in pasture productivity may be found in decline in pasture production due to vegetation thickening, soil degradation and reductions in ecosystem services.

In the last few years, sustainability of cleared systems has been questioned since pasture run-down and soil erosion are observed in old cleared pastures in central Queensland (Partridge *et al.* 1988; Graham 1996). There may be a range of factors such as positive adaptations of native grasses/vegetation to soil properties, adaptation to the local climatic conditions, and native fauna and fire regimes that assist the native vegetation to sustain on relatively poor soils of the country. Burrows (2001) stated ‘in the past the “success” of tree clearing to promote pasture development and livestock production was judged exclusively on the perceived economic responses. It is now widely recognised that a much greater range of factors and concerns need to be taken into account before the clearing of trees from forests and savannas can be endorsed.’

2.2.2. Tree clearing and species diversity

Species diversity is important to maintain ecosystem processes and to avail the services from various ecosystem functions. Schwartz *et al.* (2000) reviewed different studies that linked biodiversity with ecosystem function and projected two types of relationship between species diversity and ecosystem functions:

- A linear relationship: Ecosystem function increases linearly with an increase in diversity, which means that even the relatively rare species contribute to an ecosystem function.
- A curvilinear relationship: Ecosystem function is effectively maximized by a relatively low proportion of total diversity.

Species diversity and stability of an ecosystem function (variously quantified by measures of plant shoot and root biomass, nitrogen use, decomposition rate, invasion susceptibility, CO_2 flux, nitrogen and potassium retention and percentage cover) showed a linear relationship in most of the studies reviewed by Schwartz *et al.* (2000).

The concerns about decline in species diversity, increase in atmospheric CO_2 levels and increase in nitrogen deposition due to human actions are highlighted by Reich *et al.* (2001). They conducted a large-scale grassland experiment with 296 plots to study the response of imposed species diversity (1, 4, 9 and 16 species per plot) to elevated CO_2 and nitrogen deposition, and found that enhanced biomass accumulation in response to elevated levels of CO_2 or nitrogen, or their combination, was less in species-poor than in species-rich assemblages. Their study raised concerns about the consequences of widespread changes in plant composition and diversity in ecosystems worldwide in response to global changes (elevated CO_2 and N deposition).

In central Queensland, few publications report on the impact of tree clearing and/or introduction of exotic pastures on function and diversity of native species. The pasture composition for native grasses (*Themeda triandra* and *Heteropogon contortus*) and native legumes differentially varied according to tree basal area in central Queensland (Scanlan and

Burrows 1990). Grasses showed a decrease with increase in tree basal area while the legumes showed the reverse, but the authors did not give a reason for this.

The impact of exotic buffel pastures (*Cenchrus ciliaris)* on plant diversity in *A. harpophylla, E. populnea, E. melanophloia* and *Acacia cambagei* vegetation types was studied by Fairfax and Fensham (2000). Each vegetation type had three treatments: uncleared woodlands (with understorey of native pasture species), cleared woodlands but with native pastures, and cleared with exotic pastures. The age for cleared treatments varied from 2-41 years. There was no significant difference in species diversity and richness between uncleared and cleared native pastures for brigalow (*A. harpophylla)* and eucalypt (*E. populnea* and *E. melanophloia*) communities while the difference was significant between all the uncleared and cleared exotic pastures. There was no significant correlation between age of clearing and species richness for any of the vegetation types. The authors reported that introduction of exotic species plays a greater role in the decline in native plant diversity than does tree clearing *per se*. However, in contrast to this, their data showed more exotic and fewer native species in native pastures than in the uncleared pastures for *A. harpophylla* and for both the *Eucalyptus* species, which suggests that the impact of clearing on change in species diversity was not apparently significant at this stage as each system takes time to equilibrate to the change (as could be the situation in cleared native pastures). The greater number of exotic species and reduction in native species in cleared native pastures suggest that open systems can be more prone to invasion by other species than the uncleared woodland pastures. Daehler and Carinop (1998) also reported the invasive behaviour of buffel grass in Hawaiian savannas. The diversity of grasses is also known to be affected by the presence and type of shrubs, especially in arid environments (Facelli and Temby 2002).

Impact of management practices on pasture composition

Pasture management practices (cultivation before sowing legumes and grasses, timber treatment, applying fertilizers and stocking rates) affected herbaceous species diversity in eucalypt woodlands of north-eastern Queensland (McIvor 1998). Among the various treatments, sowing a mixture of four legumes and grasses (*Bothriochola insculpta, Chloris gayana, C. ciliaris* and *Urochloa mosambicensis*) reduced the number of native species, as sown grasses dominated. Cultivation and application of fertilizers generally also reduced the species diversity for sown species capitalised upon those management pastures and dominated.

Interestingly, tree clearing showed no impact on total number of species at one of the sites (Cardigan), whereas at another site (Hillgrove) tree poisoning (stem injections with arboricide) led to greater species diversity (native and sown) compared to uncleared treatment. McIntyre and Martin (2001) studied the impact of biophysical (lithology, landscape and tree density) and human influences (livestock grazing, soil disturbance and habitat enrichment) on plant species richness in grasslands of south-eastern Queensland and showed that soil disturbance significantly reduced the richness of native species. Tree clearing did not result in reduction of species diversity, as also observed by McIvor (1998). In these studies the interaction of different factors (soil type, species diversity, over-sowing, grazing pressure and management) makes it hard to isolate the impact of one factor/driver on change in diversity.

Pasture species diversity shows variable response with grazing management. The interaction of these two parameters may also vary with rainfall. The decrease in species diversity with grazing was more apparent in low than in high rainfall zones (Vesk and Westoby 2001). A review of species diversity studies in pasture system throughout Australia revealed that out of the total 324 species that occurred, 77 (23%) increased consistently, 96 (30%) decreased consistently, 18 (6%) were not affected, while 133 (41%) inconsistently responded to grazing (Vesk and Westoby 2001). Fensham *et al.* (1999) also studied the change in floristics along paired roadside-paddock plots to explain the change in species diversity due to grazing disturbance. Several species such as *Parthenium hysterophorus* (a weed) increased while other grazing sensitive species, such as *Dichanthium sericeum,* declined with grazing. The native blue grass pastures (*D. sericeum*) of Queensland that are now declining due to grazing, are rated as of concern from the conservation point of view (Fensham 1999).

Effects of grazing pressure on species diversity depend on the palatability of the species. The diversity of dominant species with greater palatability increased with increase in grazing pressure whereas overgrazing reduced the species diversity if unpalatable species were dominant (McIvor 1998). Increased grazing pressure reduced the pasture yield and was predicted to change the pasture composition over time in the native pastures (*H. contortus)* of central Queensland (Orr *et al*. 2001).

Due to pasture run-down in cleared pastures, alternative practices such as sowing legumes (e.g. *Stylosanthes* spp.) are sought (Burrows 1993), which should, to some extent, increase diversity in pasture systems.

There is no exception in Queensland that the woodlands are cleared to develop pastures to produce more herbage yield, more cattle and consequently greater economic returns. It is only in the past few years that decline in biodiversity, and its consequences (Fairfax and Fensham 2000; Ludwig *et al.* 2000; McIvor 1998; McIntyre and Martin 2001), have been observed. It is of utmost importance to create a balance between productivity and sustainability of pasture systems before losing vital ecosystem functions (such as existence of various floral and faunal assemblages), which are irreplaceable.

2.2.3. Relationship between plant diversity and productivity

Plant diversity-productivity and diversity-sustainability relationships support the hypotheses that increased species diversity will lead to better production and long-term sustainability of a system (Tilman *et al.* 1996). These relationships are not well understood for Queensland pastures for there are no reports available that relate plant diversity and productivity/sustainability. However, the plant diversity-productivity relationship was estimated for grasslands in America (Tilman *et al.* 1996; Tilman 1999) and Europe (Hector *et al.* 1999). Data of Tilman *et al.* (1996; 1997) support the diversity-productivity and diversity-sustainability hypotheses based on a detailed study conducted in American grasslands. Each plot (in replicates) was planted with 1, 2, 4, 8, 16 or 32 prairie grass species selected randomly from a pool total of 32 species. Productivity in all the 289 grassland plots increased significantly with plant diversity (Fig 2.4A). Resource utility in terms of nitrate was higher in high diversity plots, and resulted in less loss of soil N (there was more leaching of nitrate in low diversity plots). The efficient use of nutrients at high diversity levels suggests that functional diversity of various species would contribute to the sustainability of nutrient cycling and productivity of such ecosystems. Tilman *et al.* (1996; 1997) showed that establishment and functioning of grassland ecosystems depends upon number of functional groups, with more diverse ecosystems being more productive (Fig 2.4B).

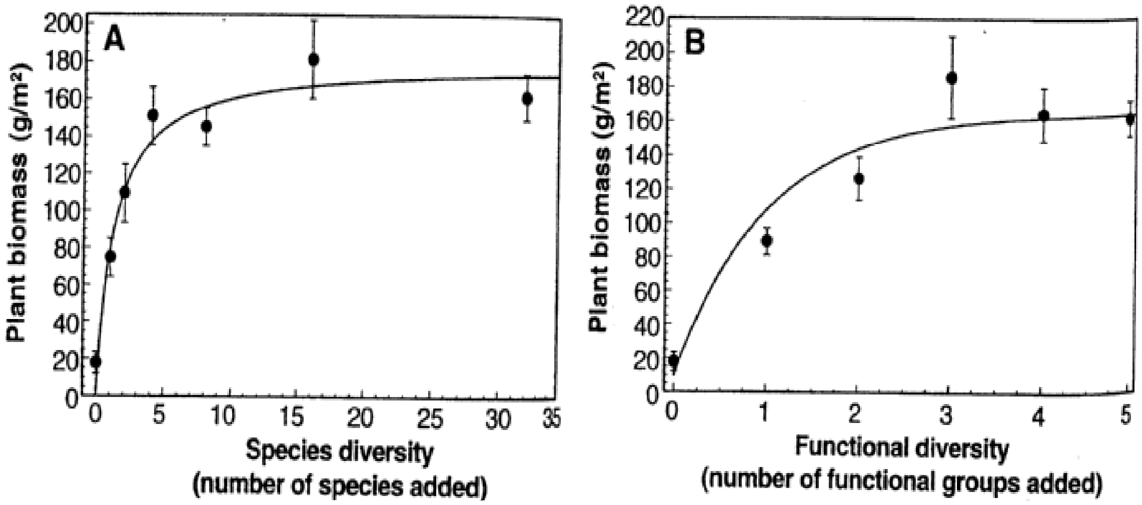


Fig. 2. 4 Productivity dependence on plant species and functional diversity (Source: Tilman *et al.* 1997):

A. Above ground biomass (mean and standard error) and number of plant species.
B. Above ground biomass (mean and standard error) and functional diversity of various species sown in 289 plots.

Another such detailed study on plant diversity and productivity (grasses and forbs) was carried out on European grasslands in seven countries: UK, Germany, Ireland, Greece, Portugal, Sweden and Switzerland, by Hector *et al.* (1999). This also revealed that species richness and functional group richness had significant effects on productivity. Communities with more functional groups (more diverse species) were more productive compared to the less diverse communities. The mechanisms for higher productivity with increase in diversity can be explained in two ways (Tilman 1999):

- Sampling effect model based on a greater probability for the presence of additional species when diversity is higher. If the productive species are better competitors (compared to less productive species) then the diverse systems are likely to be more productive because of greater chances of containing those species.

- Niche complementarity - differences among the species in resource use would allow some combination/group of species to efficiently capture and use the resources.

The niche (species) complementarity was found as the main mechanism for increase in production with increase in species diversity (Loreau and Hector 2001) and was also supported by data of Sala (2001).

Such results are in contrast to the general opinion of landholders in Queensland, where pastures with one or two exotic grass species (or with legumes) are considered to be highly productive (as mentioned in Burrows 1993; Burrows 2001).

The decline in pasture productivity over time after initial increase (Lawrence *et al.* 1988), in cleared pastures of central Queensland, may be due to loss of functional diversity in mono-culture (buffel grass) pastures. The below-ground diversity of arbuscular mycorrhizal fungi was identified as the main factor to increase diversity and productivity in European calcareous grassland (van der Heijden *et al.* 1998). Biodiversity-ecosystem function relationship between plants and the under-ground soil organisms may play an important role in determining productivity.

2.2.4. Role of species diversity and presence of trees in cattle diet

In addition to the possible enhanced functional diversity in multi-species pastures compared to exotic buffel grass dominated pastures where one species predominates, another advantage of species diversity and the presence of trees may be the increased variety for cattle diet. The interactions of over-sowing introduced grass and legume species, and of tree killing on diet quality, feed intake and live weight gain in open eucalypt woodlands of north-east Queensland were studied by Ash and McIvor (1998). Higher green leaf percentage, grass nitrogen and *in vitro* dry matter digestibility were evident in pastures (native and sown) with trees compared to the pastures without trees. A similar study in north Queensland at Hillgrove and Cardigan, by Jackson and Ash (1998), also revealed greater nitrogen concentration and dry matter digestibility in *Chrysopogon fallax* leaves growing under trees than in the inter-tree area.

Initial increase in pasture productivity from clearing may come at the cost of a decline in species diversity, forage quality, loss of ecosystem functions and stability of the pasture systems. The loss of these services/functions takes time to exhibit their impact and thus makes it hard to estimate the total cost of their loss. There is a need to look for an appropriate balance between species diversity and productivity to sustain the pasture systems over the long term.

2.3. Status of soils with land use change

2.3.1. Physico-chemical properties

Land degradation is a serious problem throughout the world. Soil salinity, erosion and depletion of nutrient content in agricultural soils are major issues of concern directly affecting productivity.

Conversion of forest/woodland soils to agriculture has chiefly affected soil organic matter and nutrient content. Soil organic matter is an important driver for nutrient availability, pH, electrical conductivity (EC) and cation exchange capacity (CEC). A greater loss of soil organic carbon due to forest clearing and cultivation (cropping and pastures) occurred in subtropical wet (60-70%) than subtropical dry (14%) soils compared to the initial pool of carbon in forest soils (Brown and Lugo 1990). Among the cropping and pasture soils, the loss was greater in cropping than in pasture soils. In the dry subtropical climate, the old pasture soils (>50 yrs) had greater C than even the mature forest soils. N mineralisation and organic matter production were greater in the warm-wet than in the dry climate. Potter *et al.* (1999) also reported greater concentrations of soil organic carbon and nitrogen in native prairie than in agricultural soils to a 120 cm depth in the tropical climates of central Taxas.

A review by Post and Kwon (2000) on changes in soil organic carbon with conversion of native vegetation/forests to pastures in different climatic zones showed an increase in organic carbon in wet tropics and loss of organic carbon in dry climates as apparent in Table 2.2. These results were contrary to those of Brown and Lugo (1990) where subtropical dry pasture soils had increased organic carbon compared to forest soils, for subtropical dry pastures were older (>50 yrs) than the pasture soils (23-25 yr) (Post and Kwon 2000).

Table 2.2. Changes in soil organic carbon with land use change (extracted from Post and Kwon 2000).

Site type	Years since conversion	Soil depth (cm)	C change ($g/m^2/yr$)	Reference
Subtropical dry forest: abandoned pasture	25	38	-13.08	Smith *et al.* (1951)
Subtropical moist forest cultivated to pasture	37	18	Out of total 10 sites: SOC increased at 8 and decreased at 2	Lugo *et al.* (1986)
Tropical dry forest cultivated to pasture	23	10	-17.4 and –13.0	Trumbore *et al.* (1995)
Tropical moist forest cleared for pasture	7-81	10-30	Out of 14 sites: increased at 11 and decreased at 3	Neil *et al.* (1997)

Soil organic matter and nutrient availability are also affected by soil texture. Soil organic matter associated with sand showed a faster rate of decomposition (short-term turnover) compared to clay and silt (Christensen 1992, cited by Guggenberger and Zech 1999). Clearing of native tropical savannas followed by introduction of deep rooted exotic grasses also increased soil organic carbon for the initial 3-6 years (Fisher *et al.* 1994). Conversion of forest land to crops and pastures initially (for 2-3 years) reduced the amount of total carbon, nitrogen and C/N ratio in the top 0-5 cm soil, but that was later restored after 12-18 years of secondary forest growth (Guggenberger and Zech 1999). Some data suggested the opposite trend. The 20-24 year old pasture (*Cynodon dactylon* L. and *Festuca arundinacea*) soils had greater potential for organic C and N build up in the top 20 cm than did the forest soils (*Pinus taeda* L.) (Franzluebbers *et al.* 2000).

From these studies, it appears that the age of cleared pastures plays an important role to determine the amount of soil organic carbon and nitrogen in pasture soils. Most of the studies on impact of land use change on soils have highlighted the soil organic carbon and/nitrogen content only. Perhaps due to global warming, the impact of land use change on the rate/amount of carbon sequestration is the major issue of concern for many scientists, but impacts of clearing on other soil nutrients and on soil physical properties should not be underestimated.

In Australia, about 50 per cent of pastoral land is affected as a consequence of human actions such as clearing. Australian soils are generally considered as being poor with low levels of available N and P (Conacher and Conacher 1996). Change in land use for various agricultural practices disturbs the natural cycling of nutrients. For intensive pastoral land use systems, the nutrients are harvested from the field/soil with little return to the system. In tropical eucalypt savannas of north-east Queensland, an adult beef animal removes 0.25 kg P/ha/year. With the available phosphorus to 1 m soil depth at about 40 kg/ha, it could take 160 years (at grazing pressure 1 beast/ha/yr) to deplete the available content of phosphorus (Burrows 1993). However, this calculation assumes that the soil conditions remains suitable for release of phosphorus over the time; this may not be so.

Cleared or disturbed areas can be transiently rich in nutrients due to enhanced mineralisation of organic matter after disturbance (Bilbao and Medina 1990). Gradually, there will be greater risks of soil erosion, salinity, decline in fertility and structure, and changes in other physical properties (pH, EC, CEC) as a consequence of clearing (Boulter *et al.* 2000). The transient soil nutrient pool leads to greater productivity in the early years of clearing which has attracted landholders to clearing. In central Queensland, Lawrence *et al.* (1988) reported an increase in pH and greater compaction, soil erosion, runoff and decline in soil fertility as a consequence of clearing *A. harpophylla* community. A flush of nutrients occurred upon pulling/burning *A. harpophylla* as available nitrogen increased seven fold, phosphorus three fold and potassium two fold in the top 10 cm of soil. Thus pastures were easy to establish and performed well after clearing. Pasture plants used the released nutrients quickly and later depended on the minerals (especially nitrogen) released by microbial decomposition. As pastures aged, organic matter as well as the microbial population increased. Lawrence *et al.* (1988) claimed that microbes convert the organic nitrogen to available nitrogen, but compete with plants to use this nitrogen. Hence, most of the available nitrogen becomes locked in microbes and roots, and as a consequence above ground parts of plants run out of available nitrogen as pastures age. Robertson *et al.* (1993; 1994) also reported similar results showing greater amounts of soil microbial biomass C and N in permanent green panic (*Panicum maximum*) compared to cropping (*Sorghum bicolour*) soils.

With the current knowledge it is difficult to predict the effects of clearing on nutrient availability after time-since-clearing (Schmidt and Lamble 2002), which vary with climate, soil and vegetation type. In a semi-arid environment, availability of soil nutrients is also limited by soil moisture content (Scholes 1993). Greater rates of mineralisation are reported (Tiessen and Shang 1998; Tiessen *et al.* 1992; Tiessen *et al.* 1994) in tropical than in temperate soils that would hasten the loss in nutrient availability for plant growth in the tropics with time since clearing. Similarly Jenkinson and Anayaba (1977) reported a four times greater rate of disappearance of labelled straw C in Nigeria than in the UK.

Nutrient cycling and their availability is of greater concern in tropical climates of Australia for it is an old continent with ancient and highly leached soils, and its savannas have a lower productivity than African and American savannas (Braithwaite 1991).

Graham *et al.* (1981) studied the impact of tree clearing on total nitrogen and organic carbon in the top 10 cm layer of *A. harpophylla* (brigalow) soils in central Queensland. They selected 31 paired (virgin (uncleared) and developed) sites over the range of 2-34 years after clearing. Cleared sites (no ploughing) with sown pastures showed significantly lower value of organic carbon than the virgin sites. The authors observed no relationship between per cent change in nitrogen and the age of clearing. The bulk density was higher at all the developed (cleared) compared to virgin sites. Their data did reveal that in cleared-pastures (without ploughing) out of a total 17 sites, 13 sites showed a decline in total nitrogen (out of these 5 had significantly ($P=0.05$) less nitrogen than the virgin sites), while only four cleared sites (out of these 3 sites were cleared less than 12 years ago and one less than 20 years ago) had higher levels of total nitrogen compared to virgin sites. Graham *et al.* (1985) also recorded a positive response of pastures to N fertilizer and concluded that pasture run-down may be due to lower availability of nitrogen rather than loss of total soil nitrogen reserves. They provided similar arguments as Lawrence *et al.* (1988) for the tying-up of minerals in the organic reserve. The increased bulk density at developed sites could be responsible for less N mineralisation. However, the impact of clearing on compositional change in microbial populations and the reasons for change in biological processes that lead to nitrogen deposition in cleared pastures, are not yet clear.

Most of the studies conducted in Queensland for pasture soil nutrient status have focussed at the top 10-20 cm depth. However, roots of the most commonly grown exotic grass *C. ciliaris* can grow to 1 m depth. Sampling of the 0-80 cm depth of soils in east-central Queensland supporting *A. harpophylla* and *Eucalyptus cambageana* showed that the pH, EC, Na, ESP (exchangeable sodium percentage), Cl and Mg concentrations increased while the amount of Ca, K, P (total and available) and total N decreased with soil depth (Webb *et al.* 1982). Land use change (clearing of woodlands to develop pastures) may affect the soil properties at deeper layers. For example, the high sodium concentrations at 60-80 cm depth as observed by Webb *et al.* (1982) raised concerns for sodicity when converting these woodlands to cropping systems. Nevertheless, little work has been done in the central Queensland region in this direction.

Impact of management practices

Various post-clearing management practices affect the soil nutrient pool. For development of pasture lands, following clearing, the raking of wooden logs/trees followed by burning is a common practice. Burning could result in a dramatic but short-term increase in inorganic N (especially NH_4-N) and soil pH in the top 0-10 cm depth (Ellingson *et al.* 2000).

Grazing pressure is another important factor that affects nutrient status in pasture soils. Its impact may also vary with soil type, rainfall, species diversity and climate. In clay soils of Oklahoma USA (annual average of 842 mm rainfall and 17 ^oC), soil organic carbon and nitrogen were less throughout 0-60 cm profile at high compared to low and moderate grazing pressure treatments (Potter *et al.* 2001). At another site in the same study, silt loam soils showed no significant impact of grazing pressure on the amount of soil organic carbon and nitrogen. Heavy grazing pressure also resulted in greater soil run-off compared to ungrazed (control) treatments (Waters 1997). Yates *et al.* (2000) reported that bulk density, soil temperature and soil erosion were greater, while the infiltration and soil cryptogram cover were lesser at heavily grazed compared to rarely/un-grazed *Eucalyptus salmonophloia* woodland soils in Western Australia (WA). The soil nutrients NO_3^--N, NH_4^+-N, and available P were also greater at heavily grazed sites, whereas quantities of organic carbon, available and exchangeable K and Ca were greater at un-grazed sites. The possible reason for high N and P content at heavily grazed sites could be the defecation and urination of animals (Yates *et al.* 2000). Loss of soil cryptogamic cover (Eldridge *et al.* 2000) and increased compaction (Bruand *et al.* 2001) in pastoral and agricultural systems also reduces the productivity. A recent report by Noble *et al.* (2002) showed that the alternatives chosen to restore pasture productivity in the past, such as introduction of legumes especially, *Stylosanthes* sp. (stylo) has accelerated soil acidification.

The impact of the abovementioned disturbances (grazing, clearing and fire) in tropical savannas on landscape functions (conserving resources and maintaining diversity) would have an adverse impact on ecological sustainability of a system (Ludwig 2000).

There is a paucity of data to draw conclusions on the impact of clearing and management practices on various physicochemical and biological properties of soils, whether for woodland-pastures and/or developed (cleared) pastures. Australian soils support quite different vegetation types and fauna to those in other continents, and coupled with climatic factors require specific data sets to assist in interpretation of the impact of land use change for pastoral and agricultural systems.

2.3.2. Biological properties

Soil microbes decompose organic matter and release nutrients into available forms for plants. They also contribute to soil formation. Microbes are sensitive indicators for change in land use (Powlson and Jenkinson 1981) and soil quality (Karlen *et al.* 1997). Sparling (1997) suggested that soil quality can be monitored based on two types of microbial indices: soil microbial biomass and soil microbial respiration.

a) Soil microbial biomass

This represents the living component of soil organic matter (excluding macrofauna and plant roots) which comprises less than 5 per cent of total organic matter (Jenkinson and Ladd 1981). It is mostly concentrated in the top 10 cm soil then declines with depth (Woods 1989 cited in Sparling 1997). The estimations of soil microbial biomass and microbial quotient (soil microbial carbon/total soil organic carbon) are very useful to estimate the health of less fertile soils (Sparling 1997).

Various methods have been proposed to measure soil microbial biomass. Dalal (1998) presented the advantages and disadvantages of both physiological and biochemical techniques used to measure soil microbial biomass. Techniques included chloroform fumigation incubation (Jenkinson and Powlson 1976), chloroform fumigation extraction (Brookes *et al.* 1985; Vance *et al.* 1987), substrate induced respiration (Anderson and Domsch 1978) and adenosine triphosphate analysis (Jenkinson *et al.* 1979; Webster *et al.* 1984). Among these, chloroform fumigation incubation and chloroform fumigation extraction are the most widely used methods.

In low fertility soils, the amounts of available nutrients are low as a high proportion of nutrients are immobilized in the living microbial biomass (Jordan 1985). A similar explanation was given by Graham *et al.* (1981) and Lawrence *et al.* (1988) in that microbial population ties up N in cleared pastures of central Queensland. Therefore, the aboveground plant parts run out of N supply with age since clearing. In grazing systems, there could be two hypotheses on soil microbial biomass relation to nutrient concentrations in soils:

Microbes serve as a source and sink for soil C and N and they mineralise soil organic matter, so a greater microbial mass available for decomposition should lead to higher return of nutrients over the longer term.

There could be greater mineralisation of soil organic matter but grass roots take up more N than the microbes for plant growth and as the aboveground plant parts are removed by grazing, the greater uptake of N by roots for plant growth make the soil deficient in N. Thus, there could be a slow rate of N return from grass root decomposition in the absence of soil disturbance (ploughing).

However no objective data are available to support these hypotheses.

Various factors such as climate, soil moisture, texture and land use change affect the amount of soil microbial biomass. Dalal (1998) and Sparling (1997) reviewed the role of these factors. Soil microbial biomass increased with increased annual precipitation, but decreased with mean annual temperature above 20 ^{o}C in a semi-arid subtropical environment (Dalal and Mayer 1987). Insam *et al.* (1989) studied the impact of precipitation and evaporation on microbial biomass and found that the microbial quotient (microbial biomass C/total soil organic C) decreased exponentially with the ratio of precipitation to evaporation (P/E), as follows:

Microbial quotient= $18.2 + 108.3^{\exp(-7P/E)}$ (no r^2 mentioned)

Seasonal variability in soil microbial biomass occurs due to changes in their substrate (quantity and quality), to temperature and to moisture content. Seasonal fluctuations in soil microbial biomass are much higher in dry than in wet tropics (Granastein *et al.* 1987; Srivastava 1992). The maximum amount of soil microbial biomass was observed at the time when roots also exhibited maximum root biomass and declined thereafter (Lynch and Panting 1982). Sarathchandra *et al.* (1989) found the higher microbial biomass C to be in late spring and autumn in sandy loam soil pastures while microbial biomass N, P and S were maximum during early spring and declined in late spring.

In fact, seasonal changes in soil moisture, temperature and substrate affect both the composition and functions of microbes (Zogg *et al.* 1997). However, no definitive conclusions can be drawn about the seasonal changes in soil microbial biomass based on present knowledge, nor can these be linked with land use management practices.

Soil texture is another important factor influencing microbial biomass. It influences the environment in which microbes act upon substrates. Fine textured soils with more clay content generally have more soil microbial biomass than coarse textured sandy soils (Spain *et al.* 1983). The observations by Powlson and Jenkinson (1981), Sorensen (1983) and Dalal and Mayer (1987) showed that microbial biomass content increases with the increase in clay content, at least until clay content comprises 50 per cent of the soil. It is hypothesised for undisturbed ecosystems that a plateau is reached in microbial biomass for soils having around 50 per cent of clay content, but no information is available on soils with clay content >50 per cent (Dalal 1998). Van Gestel *et al.* (1996) sampled soil microbial biomass across various particle size fractions of silty loam soils and found the maximum amount in soils with particle size of 0.02-0.002 mm and a lesser amount with particle size of <0.002 mm. Hassink and Whitmore (1997) suggested that the physical capacity of soils to preserve organic matter and hence the amount of soil microbial biomass, is limited. They suggested that the net rate of accumulation of organic matter also depends upon the extent to which the total capacity of soil to preserve organic matter is already occupied by organic matter. This may vary with the size and type of soil fractions.

Changes in land use also affect the soil microbial biomass. Cultivated, forest and grassland systems were estimated to contain 560, 680 and 870 mg microbial biomass carbon/kg soil, respectively (Smith and Paul 1990). The amount of microbial carbon in belah (*Casuarina cristata*) (361 mg C/kg soil) and brigalow (*A. harpophylla*) (508 mg C/kg soil) forests decreased at the rate of 4-14 mg/kg soil/year with the introduction of cultivation (Dalal and Mayer 1987). Greater amounts of microbial and total carbon were reported in native woodlands than the long term pastures located in the same area in WA (Sparling *et al.* 1994a).

However, in New Zealand, Yeates *et al.* (1991) found higher microbial carbon with lesser amount of total carbon in soils of long term pastures compared to that in native forests.

Soil microbial biomass is a useful measure to reflect changes in organic matter under various management practices, as the turnover rates of soil microbial biomass are lesser (faster) (0.5-5 years) than for soil organic matter (> 20 years) (Powlson and Jenkinson 1981 cited in Sparling 1997). But care must be taken to compare similar soil types when studying impacts of land use change, as microbial biomass varies with the soil type (Sparling 1997).

b). Soil microbial activity

Microbial respiration is an important component of soil respiration and soil CO_2-C flux, and is an important feature for monitoring decomposition (Anderson 1982). Soil respiration can provide an index of below-ground processes and of carbon cycling capacity of soil ecosystems (Raich and Schlesinger 1992).

Soil respiration includes CO_2 fluxes from live roots and associated mycorrhiza, and oxidation of root exudates, humified organic matter and plant detritus along with microbial respiration. Out of these, microbial and root respiration are the major components of the total soil respiration (Valentini *et al.* 2000). Kelting *et al.* (1998) reported that roots contribute about 32 per cent and root free soil (microbial respiration) about 48 per cent and microbes in the rhizosphere contribute the remaining 20 per cent of total soil respiration in red oak forests. In some forest ecosystems, live roots may contribute from 33 per cent (Bowden *et al.* 1993) to 47-62 per cent (Nakane *et al.* 1983; Ewel *et al.* 1987) of total soil respiration. The proportion of root respiration in total soil respiration was high in cold climates ranging from 50-93 per cent in arctic tundra (Billings *et al.* 1977), while it contributed about 17-40 per cent in grasslands of temperate zone (Kucera and Kirkham 1971; Herman 1977 cited by Raich and Tufekcioglu 2000).

Soil respiration shows fluctuations depending upon the substrate availability, soil moisture and temperature (Orchard and Cook 1983; Brookes 1995). The rates of soil respiration vary throughout the day (Blanke 1996). Mielnick and Dugas (2000) reported that mid-day measurements provide the average value of soil CO_2 flux for 24 hours in tall grass prairie (USA). Alternatively Boone *et al.* (1998) measured the soil respiration at sunrise (minimum flux) and afternoon (maximum flux) and calculated the mean for each measurement.

Change in soil CO_2-C flux with temperature is an important issue of concern due to climatic change, since soils contain about twice the amount of carbon (1500 Pg) that is in the atmosphere (Eswaran *et al.* 1993) and global soil CO_2-C flux is about 68-75 Pg CO_2-C/year (Raich and Potter 1995). Change in temperature affects the rate of CO_2 flux (Lloyd and Taylor 1994) and hence the amount of carbon sequestered in any ecosystem. Grace and Rayment (2000) predicted two possibilities for carbon sequestration based on the respiration balance of an ecosystem. First, if soil respiration responds positively to an increase in temperature, the forests' sink for carbon sequestration might disappear. Second, if soil respiration is insensitive to temperature change then the forests can be a more effective sink for atmospheric CO_2.

The response of soil CO_2-C flux with the change in temperature is a crucial issue for the scientific community. One school of thought believes that soil respiration is not influenced by change in temperature, while the other believes temperature influences soil respiration. The decomposition of organic matter, reflecting soil respiration, in forest soils from 82 sites on five continents studied by Giardiana and Ryan (2000) showed that microbial activity was not influenced by temperature. The authors suggested that organic matter decomposition by microbes is controlled by temperature only when substrate is not limited. In contrast, Valentini *et al.* (2000) studied net ecosystem carbon exchange from 15 European forests and found a significant relationship between temperature and ecosystem respiration for the sites of one location, but not for all the sites across different locations. They emphasised that sensitivity of respiration to change in temperature (Q_{10}) is much higher in colder than in warmer climates, and that an increase in temperature in a colder climate will increase the decomposition rate of soil organic matter more than in a warmer climate.

Among the various components of soil respiration, root respiration ($Q_{10} = 4.6$) is the most sensitive to change in temperature followed by soil ($Q_{10} = 3.5$) and microbial respiration ($Q_{10} = 2.5$) in temperate mixed hardwood forests (Boone *et al.* 1998). The increased production of root exudates at higher temperature may also contribute to greater soil respiration.

Soil respiration also varies with vegetation type as vegetation affects soil microclimate, quality of detritus and soil structure. The review by Raich and Tufekcioglu (2000) emphasised the correlations between vegetation and soil respiration. Respiration rates varied significantly in different biome types (Raich and Schlesinger 1992 cited by Raich and Tufekcioglu 2000). Grasslands had greater soil respiration rates than forests, though growing under similar condition, and grasslands also possessed about 25 per cent greater soil respiration rates than the croplands. The inherent capacity of grass species to allocate photosynthate belowground is mainly responsible for their greater soil respiration rates. Soil respiration also increased with litter fall in mature forest ecosystems (Raich and Nadelhoffer 1989). Since vegetation type determines the litter quality and quantity, it also influences the soil microclimate by providing habitat for microbes to act upon litter.

Soil moisture also exerts a strong influence on microbial activity, and low moisture content may reduce the microbial population (Paul and Clark 1996). Similar responses have also been observed by Kieft *et al.* (1987), West *et al.* (1988) and Van Gestel *et al.* (1992). Other factors, which potentially influence soil respiration, are the availability of C substrate for microbes, plant root densities, and soil physical and chemical properties (Raich and Tufekcioglu 2000).

Soil respiration can be expressed in term of respiratory quotient (also known as specific respiration rate) which represents the rate of soil respiration with respect to microbial biomass (g CO_2-C/h/g of microbial biomass) (Sparling 1997). Respiration quotient is useful to assess the substrate quality, soil development and response to stress (Anderson and Domsch 1990). Insam and Haselwandtel (1989) hypothesised that there is less competition for energy and less inputs for resource use in young ecosystems and that greater competition and efficiency for resource utilization occur as an ecosystem gains maturity. Consequently the microbial biomass increases and specific respiration rate decreases in established ecosystems.

However, the interpretations are not so simple for Sparling (1997) pointed out that the specific respiratory rates may be higher even in established systems (woodlands). This could be due to greater active mass of carbon in total soil carbon, and to the quality and amount of carbon substrate in different soil types. There is a complex relationship between soil microbial biomass and respiration in soil ecosystems. The major source of energy (soil organic matter) is not of uniform quality and care should be taken to compare the specific respiratory rates on uniform substrates.

2.4. Litter production and decomposition

Litter is an important component in any ecosystem to maintain nutrient and energy flow at the soil-plant interface. Plants act as a pump for the soil nutrient cycle; nutrients are taken up from soil and returned in the form of surface litter. Litter provides good habitat for various soil organisms and prevents soil erosion during rainfall through its action as a sponge and dissipator of rainfall kinetic energy.

The amount of litter production and its rate of decomposition varies with climate, season, substrate quality, type of vegetation and composition of litter. Knowledge of the total amount of litter production, and its decomposition rates, is important to estimate the C fluxes and C pool for various types of ecosystems. The amount of above-ground global litter production was estimated as 90-100 Pg dm/yr and that excludes open grasslands, shrublands and xeromorphic communities which may contribute another 5-10 Pg dm/yr (Matthews 1997).

Litter composition differs in open grasslands and native woodlands. It is chiefly small stems, leaves, fruit and seeds in the former while the latter also includes bark and twigs from tree/shrubs, and forbs.

The quantification of litter production from trees is easy through the use of litter traps, but more complicated for grasslands where ground litter is subjected to wind and more prone to other disturbances such as rainfall. Hughes (1970) and Medwecka-Kornas (1970) discussed various methods to measure litter production. Wiegert and Evans (1964) were the first to use a paired plot technique to measure ground litter production and decomposition. In grazing systems, the paired plot technique is useful to measure ground litter production, but it may not be suitable for measuring the rate of decomposition due to cattle disturbance. The litter bag technique (a certain amount of litter is stored in a netted bag to decompose over time) has been successfully used to estimate the rate of decomposition, but is also susceptible to cattle disturbance.

In southeast Queensland, Bruce and Ebersohn (1982) measured the amount of litter production, i.e. unattached herbage and litter accumulation, over the period of 112 days (at 28 day intervals) in pangola pastures (*Digitaria decumbens*) and tropical pastures (*Panicum maximum, Setaria sphacelata* and legumes). They found that the maximum amount of litter was produced in spring, and then decreased through late spring into summer with a minimum in early autumn, while litter disappearance increased through autumn, winter and spring with a maximum in summer. The amount of litter produced was greater in tropical than in pangola pastures.

Grigg and Mulligan (1999) studied the tree litter-fall (from trees and shrubs) in open woodlands of *E. populnea* and *E. cambageana* at rehabilitated coal mine sites in central Queensland and found that *E. populnea* (tree density 200 stems/ha) annually produced 1358 kg/ha total litter while *E. cambageana* (tree density 800-3200 stems/ha) produced 2318 kg/ha of total litter. Burrows and Burrows (1992) also reported similar values (920-2730 kg/ha/year rate of tree litter production) for various eucalypt communities. Both these studies showed highest litter-fall in the summer. These were also consistent with others (Hutson and Veitch 1985; Lamb 1985) for the decline in tree litter-fall with an increase in aridity or decrease in rainfall. McIvor (2001) observed that the annual quantity of tree litter fall in open woodlands of northeast Queensland was also quite similar to that seen in central Queensland. He estimated the litter fall in open woodlands of *Eucalyptus crebra* and *E. drepanophylla* to be 1270 and 720 kg/ha, respectively. The amount of tree litter-fall was correlated with tree basal area (130 kg/ha/year litter fall per 1 m^2 of basal area per hectare) and tree foliage cover (1 per cent of foliage cover produced 59 kg/ha/year litter) (McIvor 2001).

In central Queensland, most of the research on litter has emphasised quantity of production but not the process of decomposition, nor the relationship between litter mass loss and nutrient concentration, nor the release of nutrients in woodlands and/or grasslands. Therefore, the lack of information in this semi-arid grassland and woodland country makes it hard to estimate the quantities of nutrients returned to the system.

With reference to the decomposition behaviour of litter, Berg *et al.* (2000) revealed that the nutrient concentrations control the early stages (phase I) of decay while lignin exerted the dominant control at later stages (phase II). The length of each particular phase was affected by climate. Studying of the decomposition of Norway Spruce needles, Berg *et al.* (2000) found that early stages of decomposition (mass loss) were positively correlated to the Ca and K concentrations of the litter, but negatively correlated with lignin, whereas at the later stages loss of litter mass was positively correlated with Mn, Mg, K and water soluble substances. Berg (2000) developed a model for litter decomposition based on various observations in natural forests and showed that at early stages of litter decomposition, when cellulose is being degraded, nitrogen plays a positive role. But once cellulose disappears, concentration of lignin increases and the effects of nitrogen change completely; high nitrogen concentration then retards litter decomposition. This may be due to reaction of nitrogen with lignin to produce recalcitrant compounds. Alternatively, N concentration may suppress the synthesis of lignin degrading enzymes (Berg 2000). Thus knowledge of litter composition is important for the interpretation of decomposition studies.

Despite the positive role of litter in nutrient cycling, an increased amount of litter can be responsible for a decrease in plant diversity due to increased nitrogen availability. On higher fertility soils, some plants will benefit more than others and will dominate, thus reducing the species richness (Berendse 1999). Foster and Gross (1998) studied the impact of increased living biomass and litter production on the germination and establishment of plant species and found that litter removal resulted in increased species richness. They concluded that in nutrient poor conditions, an increase in both living biomass and litter reduced the germination and establishment of plant species (the former possibly due to competition for light) but that there might be some threshold value beyond which living biomass and litter do not have further negative impacts on plant diversity. Although there are few data available on the impacts of litter accumulation on species richness, it seems important to study litter production along with vegetation change to understand the dynamics and composition of species in woodland or cleared pasture systems.

2.5 Issues arising out of this literature review

Substantial data exist on tree clearing, pasture productivity and its relationship with trees for pasture systems in Queensland. However, none of the reports study the integrated impact of age since clearing on pasture production, pasture species diversity and soil properties. Only two reports, one on species diversity (Fairfax and Fensham 2000) and other on soil C and N in top 10 cm depth (Graham *et al.* 1981) studied the effects of time-since-clearing in pasture development.

Scanlan (2002) stated “There are no comprehensive studies on pasture production with time since clearing in eucalypt communities. Many studies span 1-2 years, few published data exist for studies greater than 10 years. Over these timeframes, clearing woody vegetation increased the production”.

A series of major unanswered questions arise based upon this literature review:

- What system will ecologically be more sustainable: a woodland with under-canopy pasture or an open/cleared pasture?
- How long can a cleared system be sustainable in pasture production?
- How is the soil nutrient status profile impacted with time-since-clearing in open compared to uncleared pastures?
- What are the impacts of time-since-clearing and /soil disturbance on soil microbes and their activity?
- How does the soil microbial activity affect the quantity of soil organic carbon and other nutrient reserves, and nutrient availability?
- What are the CO_2 concentrations emitted from open and woodland pastures in a semi-arid zone?
- Do the age since clearing and seasons have an impact on CO_2 emissions?
- How does the pasture composition change during different seasons in cleared and uncleared pastures and what is the role of species diversity in soil stability and cattle diet?
- What is the amount of litter production, decomposition and nutrient return in open and woodland pastures?
- How does the response vary for all the abovementioned questions for major types of tree communities growing in central Queensland?

More importantly the relationships/interactions among the various parameters, pasture production-plant diversity-cattle diet-soil status (soil nutrients-microbes-and microbial activity)-litter production, are of great importance and merit study in open and woodland pastures to provide more objective data on the impacts of clearing (Fig 2.5).

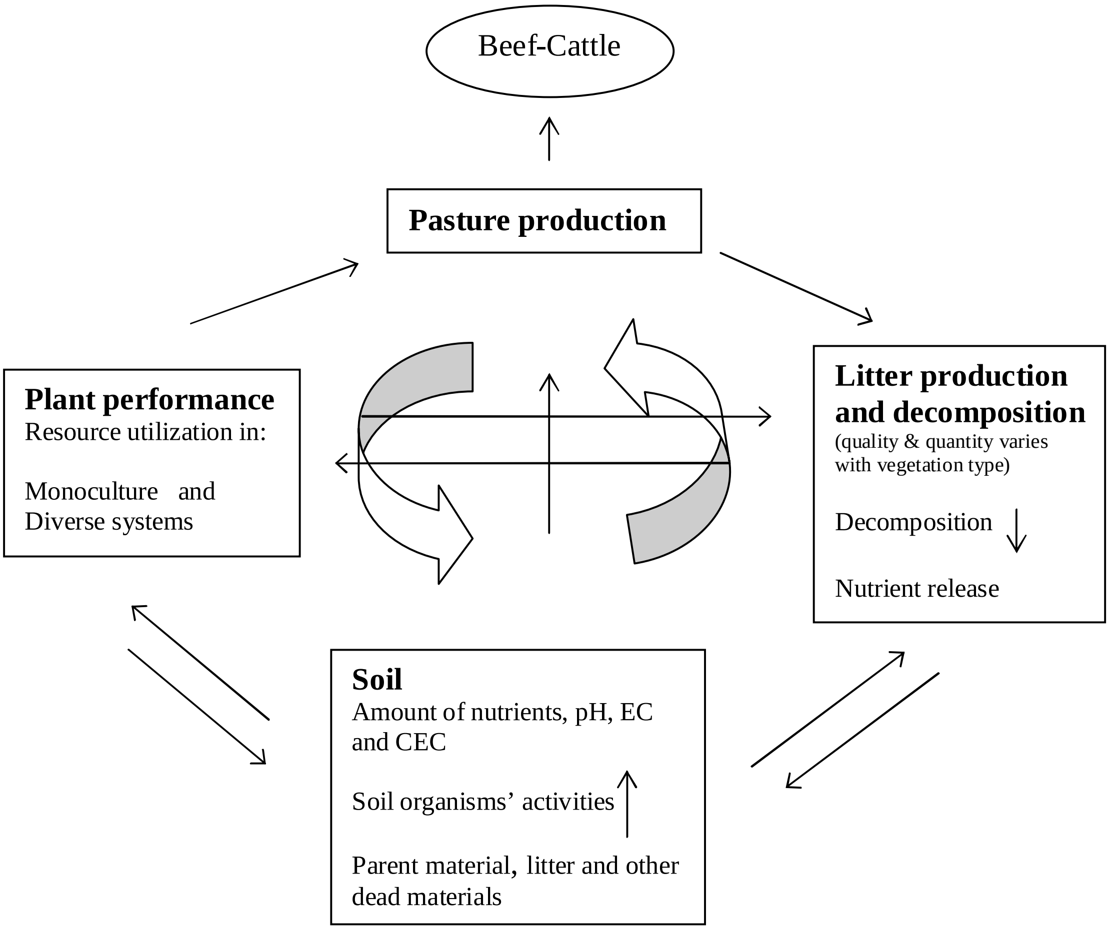


Figure 2.5. Inter-relationships between different components of a pasture system.

The present study will focus on pasture production alongside with the ecological impacts of time-since-clearing in pasture systems of central Queensland. Comparison between uncleared (intact) and cleared pastures at different times since clearing for pasture production, pasture composition change during different seasons, litter production and decomposition, and soil properties will provide the data to interpret the consequences of tree clearing in grazing systems. It will provide answers to most of the aforesaid questions, but others will need a more detailed study in the future.

Chapter 3. Research design

Introduction

As is apparent from the literature review, there are many questions to be answered about the impact of tree clearing in pasture production systems. The aim of this study is to assess the ecological implications of tree clearing in grazing systems. A particular focus is on the effect of time-since-clearing on pasture production and soil properties in cleared systems, in comparison to uncleared woodland systems. There is a little prior research on this topic. Most of the research to date has focussed on the tree density/basal area-pasture yield relationship i.e. not on soil ecosystem functions in pasture systems. The primary targets of this study are to provide data on the sustainability of cleared systems and on associated changes in ecosystem functions that accompany any measured long-term impacts on pasture production. Details of the research design are outlined in Figure 3.1. The main objectives of this research are to:

- Determine effects of clearing on the ecology of grazing systems.

- Identify and quantify the changes and trends over time-since clearing.

- Evaluate the gains in pasture production from clearing in relation to ecological losses or gains.

To address these abovementioned issues, three major types of tree communities growing in central Queensland were selected as follows:

1. *Eucalyptus populnea* F. Muell. (poplar box)
2. *E. melanophloia* F. Muell. (silver-leaved ironbark)
3. *Acacia harpophylla* F. Muell. ex. Benth. (brigalow)

Within each community, sites were chosen to represent three different ages since clearing. The age of cleared sites was considered from the start date of the study i.e. 2000:

1. Recent clearing (less than 5 years (cleared in 1996))

2. Medium age of clearing (10-13 years (cleared in 1987-90))

3. Old clearing (30-33 years (cleared in 1967))

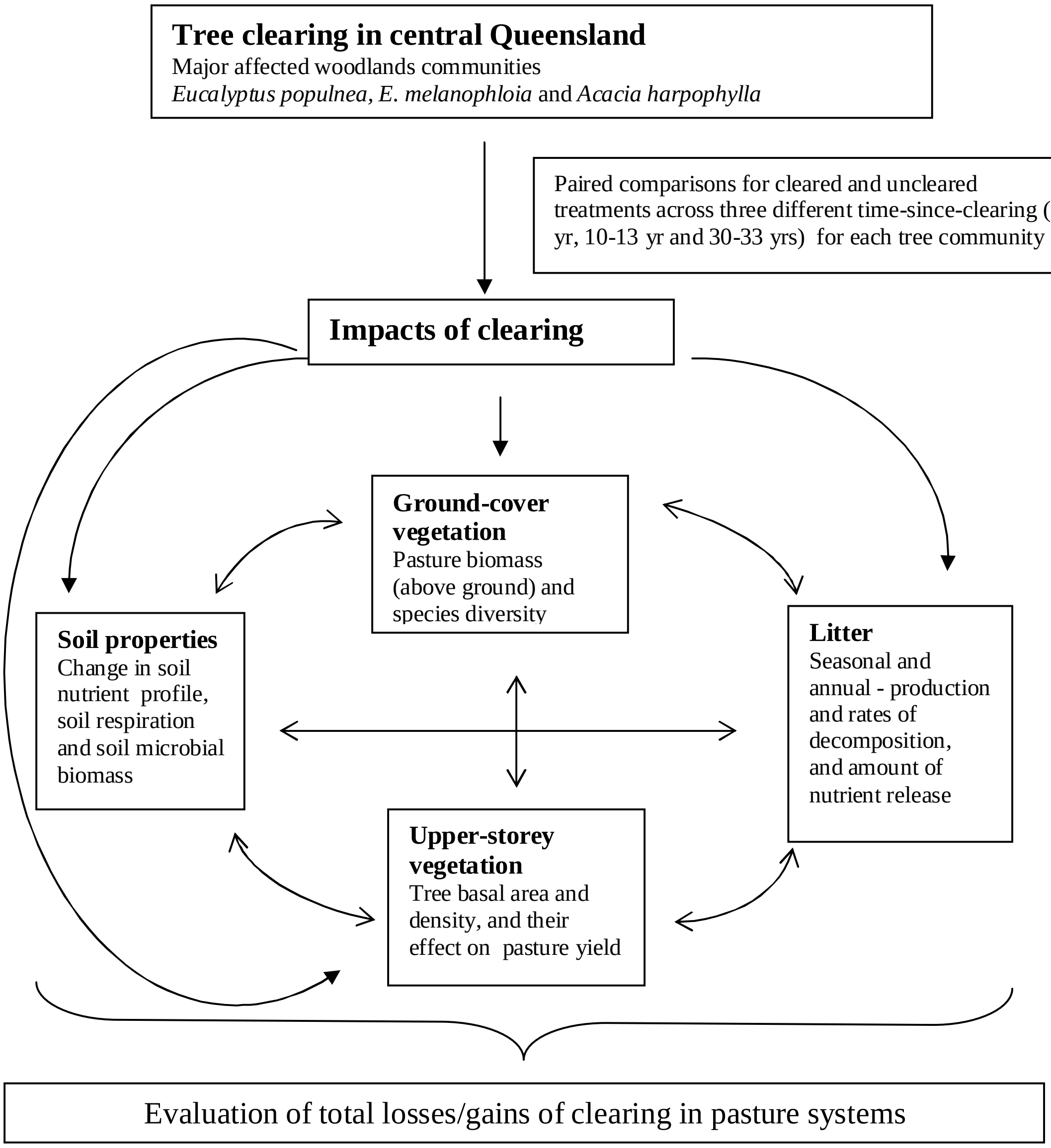


Figure 3.1. An outline of the study for main impacts of clearing and interactions of various components with each other.

There were three treatments in a factorial design of 3 (tree communities) x 3 (times since clearing) x 2 (paired cleared and uncleared (native/intact)). In total, therefore, there were 18 sites. The paired cleared and uncleared sites were selected in close proximity to have similar original biophysical characteristics (soil type, slope and vegetation) before clearing, and to some extent to minimize variation in grazing management for the same cattle grazed the cleared and their paired uncleared sites.

The cleared treatments for <5 years, 10-13 years and 30-33 years time-since-clearing will be referred in the text as recent, medium and old respectively.

Hypotheses

The present understanding of ecosystem functions in tropical woodlands as presented in the literature review indicates that different uncleared woodland communities will differ in pasture above-ground-biomass production, pasture species composition, soil properties and in the amount of litter production. An open woodland (height 10-30 m and projective foliage cover <10 %; Neldner 1993) may be as productive in terms of pasture above-ground-biomass as a cleared pasture system over a long term. The pasture above-ground-biomass production and ecosystem functions (such as floral diversity, nutrient return through litter and soil stability) may vary with time-since clearing in cleared pasture systems. A cleared system may stabilise at the same level as an uncleared system. The following hypotheses are set and tested in the present study (relevant Chapters mentioned in brackets):

If an intact woodland is cleared then, :

i. for pasture above-ground-biomass and composition (Chapter 4)

- pasture above-ground-biomass will initially increase.
- over a longer term pasture above-ground-biomass will decrease after the initial increase.
- the extent and duration for increase or decrease in pasture above-ground-biomass following clearing will be different between different cleared tree communities.
- the composition of pasture will change over time-since-clearing.
- pasture species diversity will decrease.
- pasture quality for cattle production will remain the same.

ii. for soil properties (Chapter 5 and 6)

- there will be an initial increase in available nutrients for plant growth.
- a decline in nutrient availability will occur over a longer term.
- soil organic carbon will increase with time-since-clearing.
- bulk density will increase.
- soil pH, electrical conductivity and cation exchange capacity will remain the same.
- sub-soil properties will be negatively affected.
- soil microbial respiration and soil microbial biomass will decrease.

iii. for ground litter production, decomposition and nutrient return (Chapter 7)

- litter production will decrease.
- litter decomposition rates will increase.
- the amount of nutrient release from the amount of litter produced will remain the same.

The study possessed the following main challenges in testing these hypotheses:

1. The visible gains for increase in pasture productivity upon clearing appear soon after clearing while the changes/losses in ecosystem functions take a long time to express.
2. The impact of confounding factors (grazing, rainfall and post-clearing treatment) on pasture biomass and ecological parameters (plant diversity, soils and litter) has to be minimized.

3. The inherent variability between cleared and uncleared sites for different age groups has to be minimized.

Research sites: a case study

All the cleared and uncleared sites were selected at a grazing property “Avocet” (NW Long 148.13^o Lat 23.79^o, NE Long 148.16^o Lat 23.80^o, SE Long 148.21^o Lat 23.85^o and W Long 148.12^o Lat 23.86^o) located in central Queensland (Fig 3.2). The property is located 30 km south of Emerald, in a semi-arid zone with averages for annual rainfall of 600 mm, and maximum and minimum temperature of 27-30 ^{o}C and 15-18 ^{o}C (Commonwealth of Australia Bureau of Meteorology 2003). Average (over the 136 years from 1865-2001) minimum and maximum temperatures during winter (June- Aug) are 6-8 ^{o}C and 23-25 ^{o}C, and during summer (Dec-Feb) are 22-24 ^{o}C and 33-36 ^{o}C respectively. Most rain occurs during November-February with sporadic summer storms.

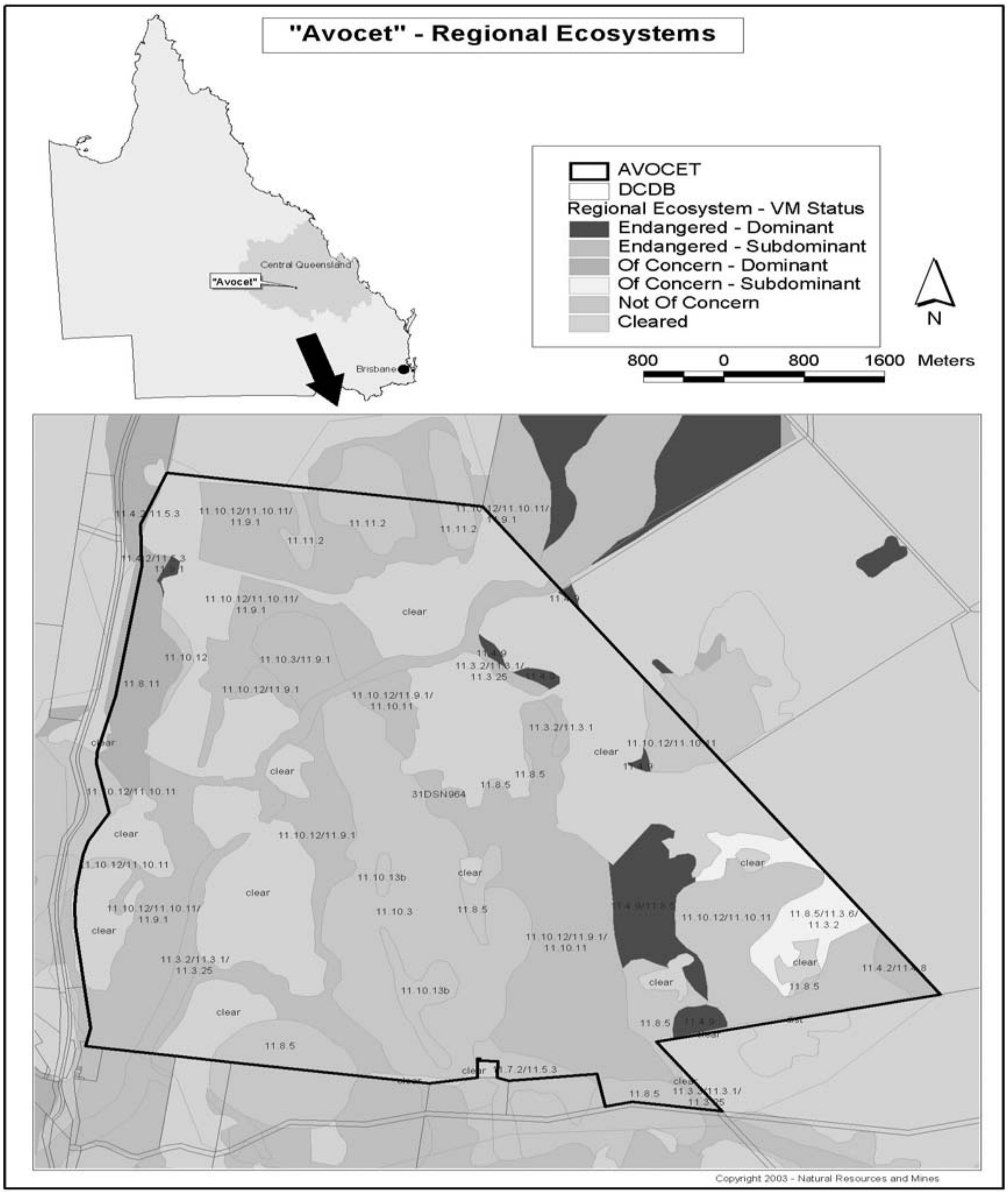


Fig 3.2. Location of the property "Avocet" and associated regional ecosystems (as per classification system used by Department of Natural Resources and Mines (NR&M)) (Source: NR&M 2003) (DCDB - Digital Cadastral Database).

The total area of the property is *c*. 5000 ha with 18 regional ecosystems represented according to the recent mapping in 2003 by the Department of Natural Resources and Mines, Queensland.

Figure 3.3 shows the distribution of various woodlands and grasslands on the property. The dominant tree communities found on ‘Avocet’ include *E. populnea*, *E. melanophloia* and *A. harpophylla*. *E. populnea* is mostly found on shallow sandy soils with sandstones underneath, *E. melanophloia* grows on loamy soils and *A. harpophylla* grows on clay and/or alluvial soils.

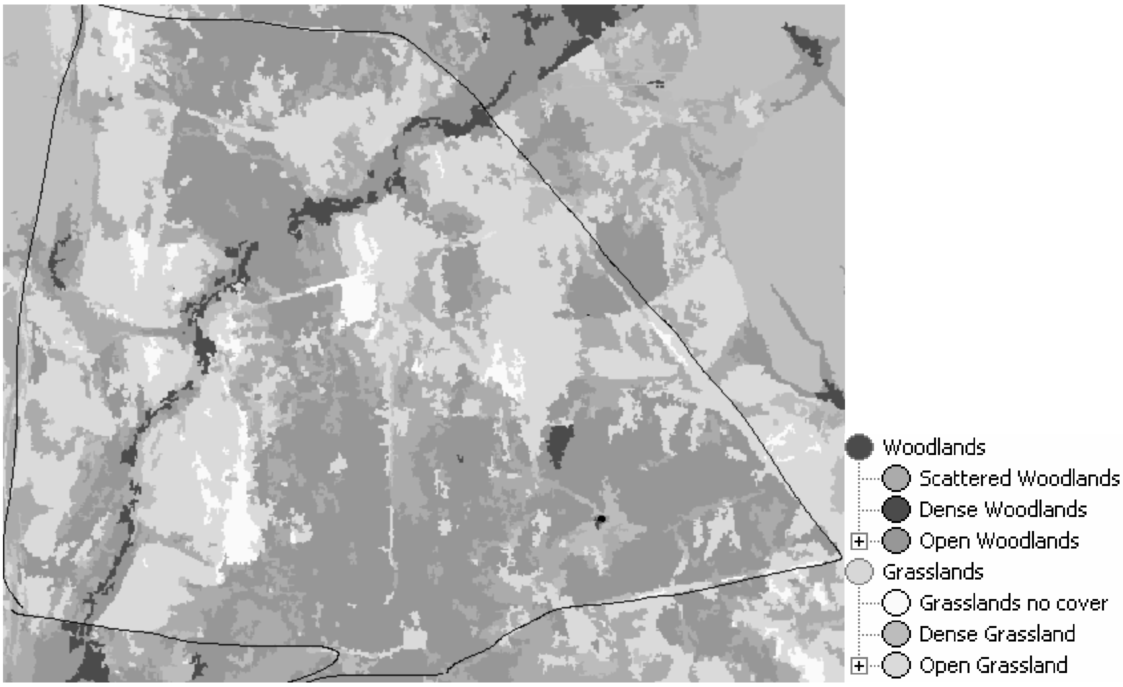


Figure 3.3. Woodland and grassland communities at ‘Avocet’ (Compact Airborne Spectroscopic Image (CASI) by Ball AIMS Group).

Research sites

The property representative of the three major tree community types of the region and for clearing treatments across different age of clearing was selected with the help and guidance of researchers at Environmental Protection Agency and at Department of Natural Resources and Mines, Emerald. Moreover, exploratory visits to other properties during site selection also suggested that the sites on the chosen property were typical of the region.

A total of 18 sites (three types of tree communities at three different times-since-clearing and paired cleared and uncleared plots for each treatment) were selected on the property. The uncleared sites were intact (never cleared) and were mostly uniform in vegetation structure across the three replicates of each type of tree community. Cleared sites had no regrowth. Blade ploughing at old cleared sites would have contributed to control regrowth (Table 3.1).

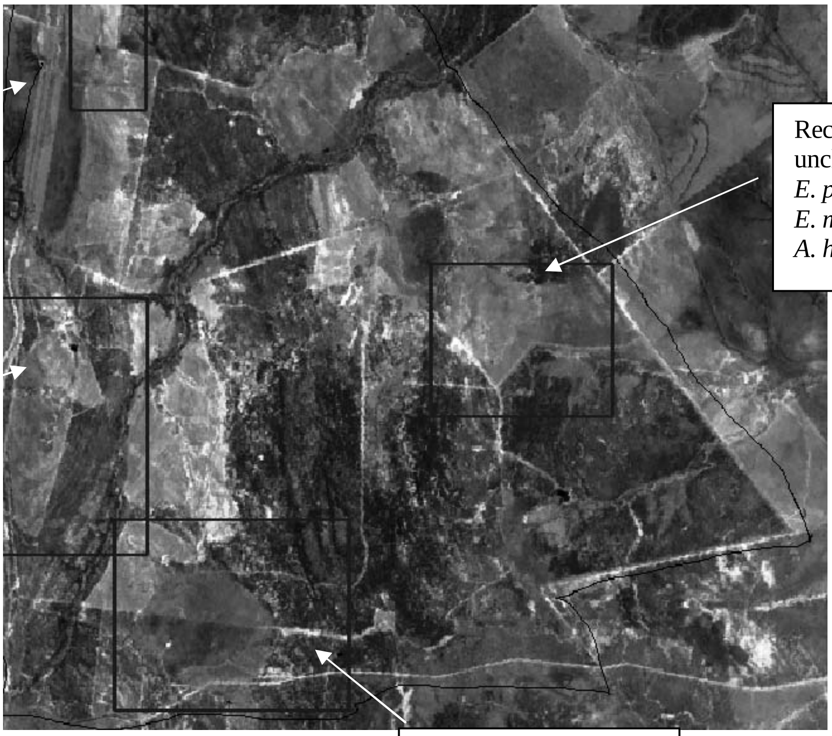
No other sites were treated for regrowth. The cleared land was sown to buffle grass (*Cenchrus ciliaris*) within 12 month of clearing except for recent cleared *E. melanophloia* where kangaroo grass (*Themeda triandra*) predominated and no sowing took place. The details of each site are outlined in Table 3.1.

Table 3.1. Details of clearing (time, methods and post-clearing treatment) and annual average stocking rate (SR) at cleared and uncleared sites for *E. populnea*, *E. melanophloia* and *A. harpophylla* communities.

Tree community	**Cleared treatments**				**Uncleared (intact) treatments**	
	Time of clearing	Method of clearing	Post-clearing treatment	GP (cattle/ha)	Uncl-eared	SR (cattle/ha)
E. populnea						
Recent clearing	May 1996	Chain pulled	Burnt in Oct 1999	1/5	Intact	1/5
Medium clearing	Dec 1987	Chain pulled, raked in 1988	Blade ploughed in 1994	1/3	Intact	1/4.8
Old clearing	July 1967	Chain pulled	Blade ploughed in 1987	1/6	Intact	1/4.8
E. melanophloia						
Recent clearing	May 1996	Chain pulled	Burnt in Oct 1999	1/5	Intact	1/5
Medium clearing	Oct 1990	Chain pulled and raked in Oct 1991	Blade ploughed in 1994	1/3	Intact	1/3
Old clearing	July 1967	Chain pulled	Blade ploughed in 1987	1/6	Intact	1/6
A. harpophylla						
Recent clearing	May 1996	Chain pulled	Burnt Oct 1999	1/5	Intact	1/11
Medium clearing	Dec 1987	Chain pulled and raked in 1988	Blade ploughed in 1994	1/3	Intact	1/4.8
Old clearing	July 1967	Chain pulled	Blade ploughed in 1987	1/6	Intact	1/4.8

Each cleared and uncleared (intact) treatment comprised a minimum area of five hectares. After the detailed survey, a representative area of 1 ha out of the total area available for a treatment was marked at each site. Within the marked 1 ha, an area of 10 m x 10 m was fenced at the central location at each site to exclude grazing.

All the paired cleared and uncleared sites were located at a distance of 50 m on either side of a fence line except for the 5 yr cleared and uncleared *E. melanophloia* where no fence existed. For each age group of clearing, cleared and paired uncleared sites for all the three communities were selected at the same location with the exception of medium age cleared and uncleared *E. melanophloia* (Fig 3.4).



Medium cleared and uncleared sites for:
E. melanophloia

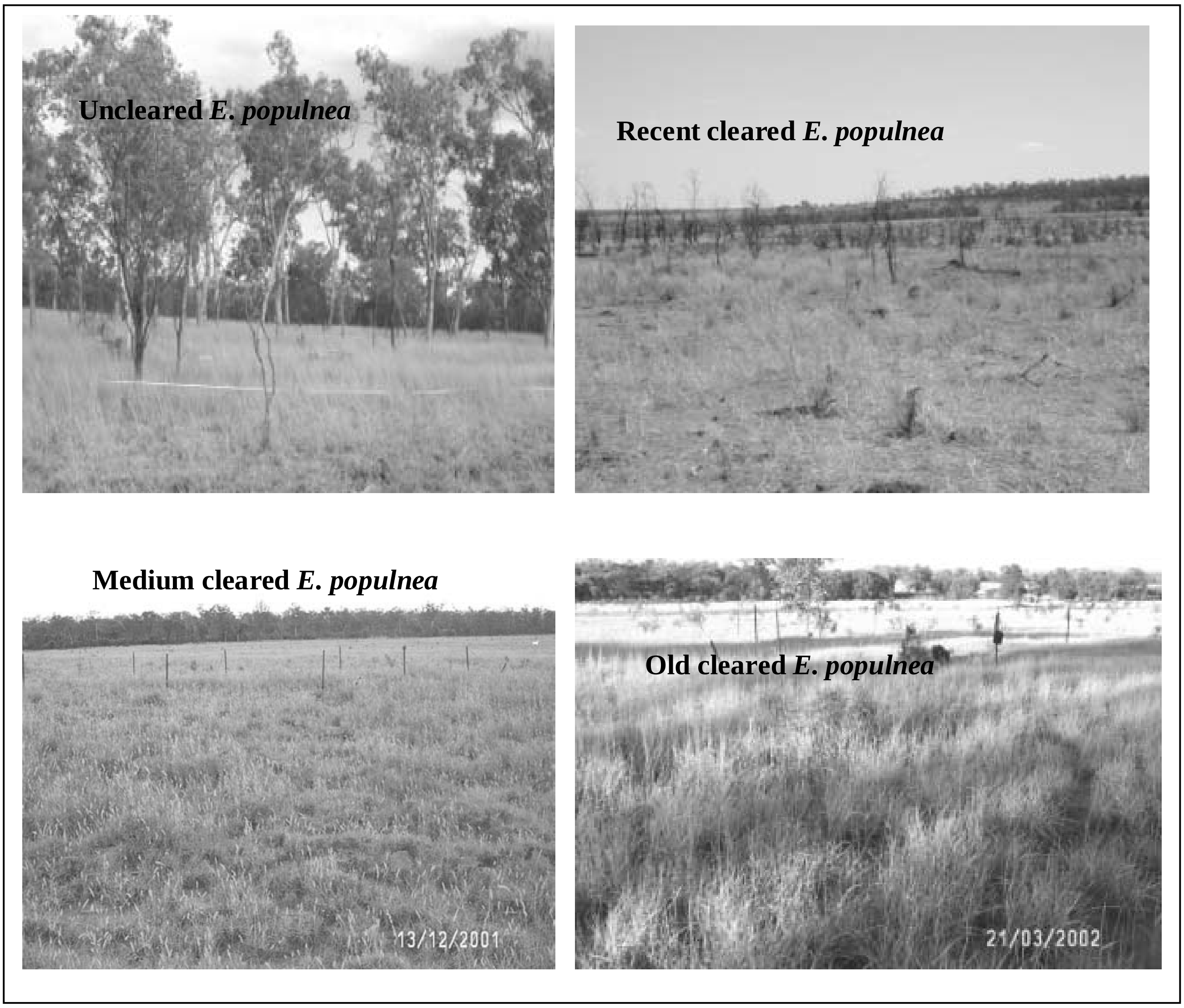
Recent cleared and uncleared sites for:
E. populnea,
E. melanophloia and
A. harpophylla

Medium cleared and uncleared sites for:
E. populnea and
A. harpophylla

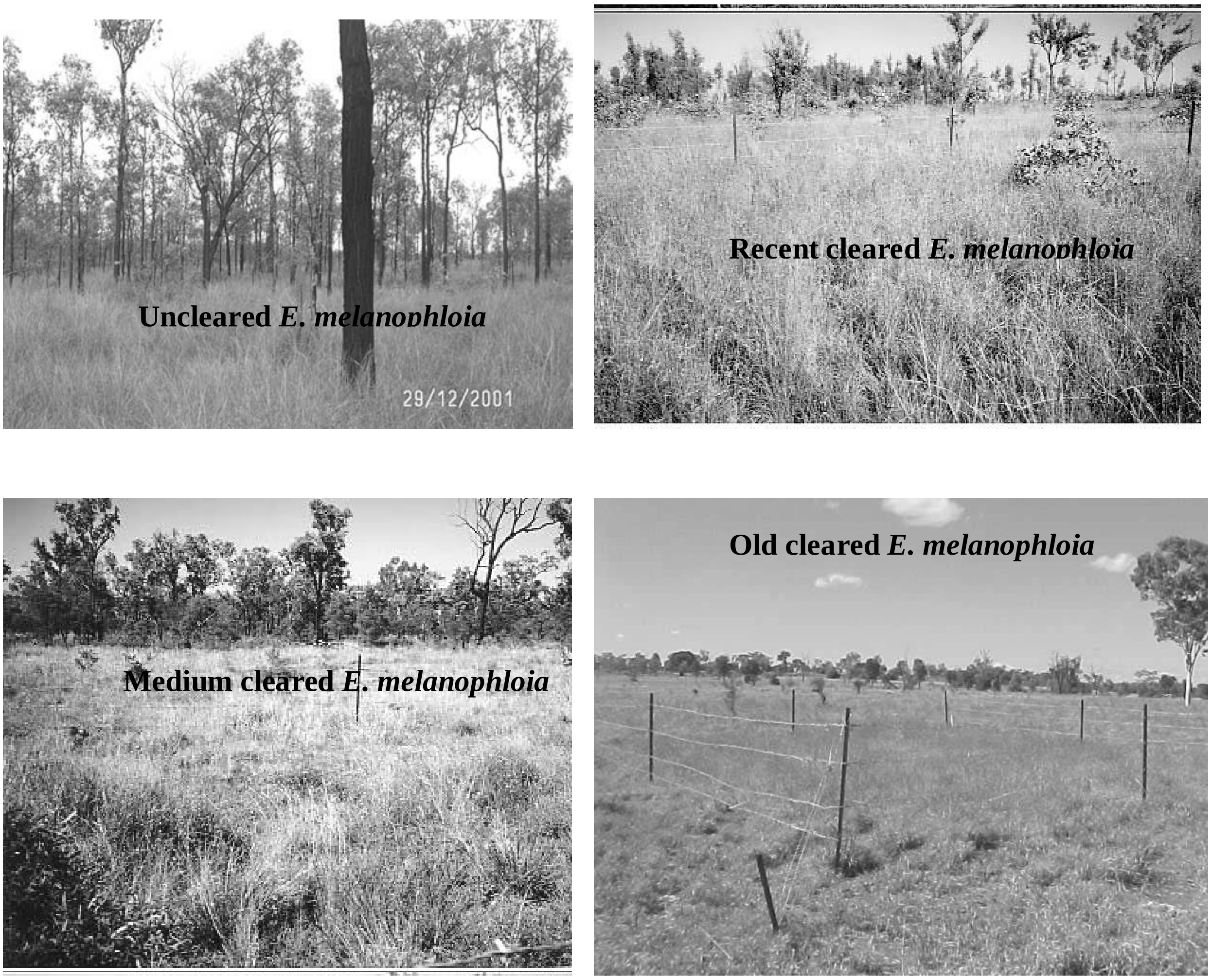
Old cleared and uncleared sites for:
E. populnea,
E. melanophloia and
A. harpophylla

Figure 3.4. Location of research sites at ‘Avocet’.

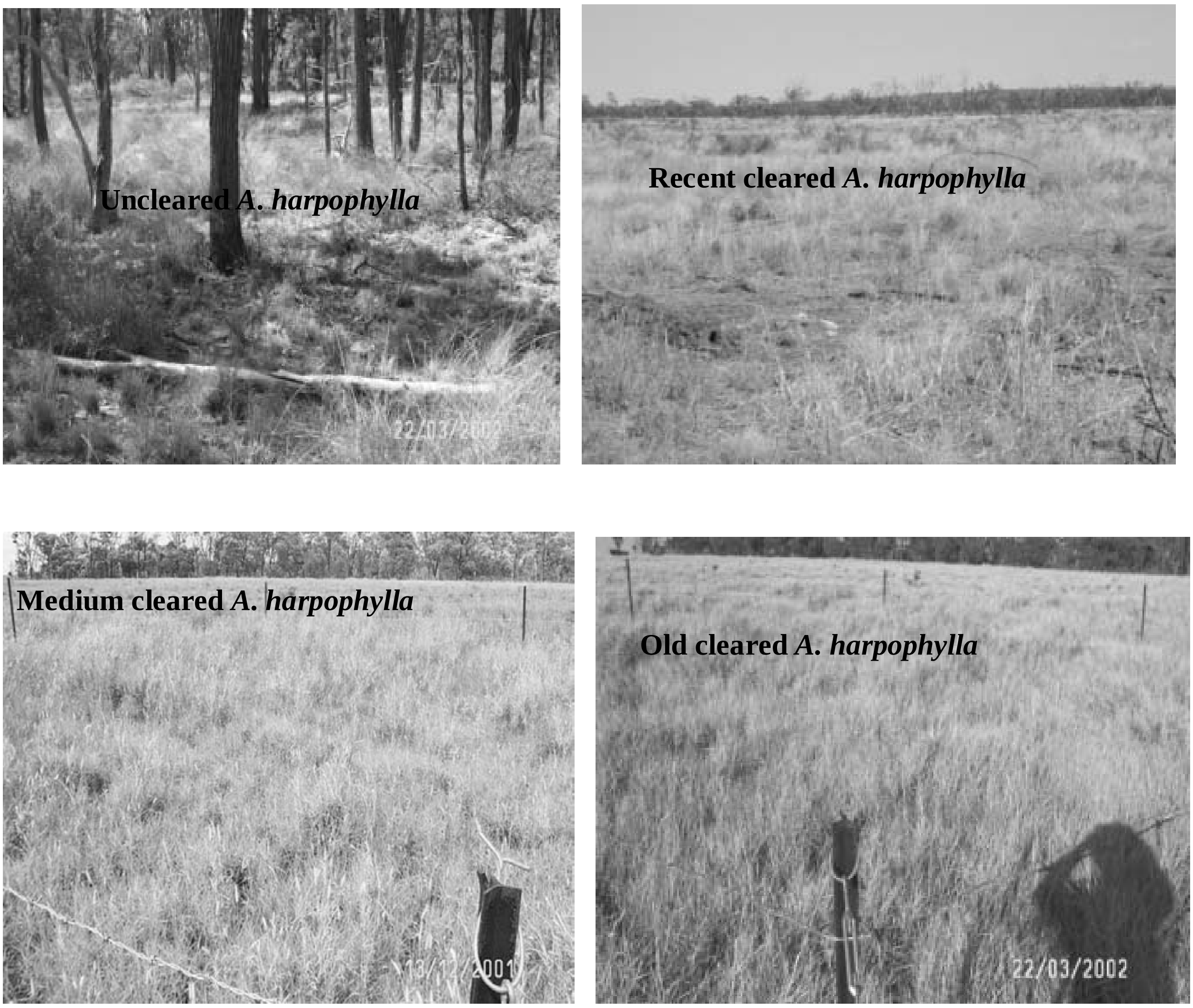
Pictorial images for the various cleared and uncleared sites are presented in Figure 3.5.



a) Uncleared and cleared sites for *E. populnea*.



b) Uncleared and cleared sites for *E. melanophloia.*



c) Uncleared and cleared sites for *A. harpophylla*.

Fig 3.5. Photographs for cleared and uncleared sites for a) *E. populnea,* b) *E. melanophloia* and c) *A. harpophylla* communities. Note the fenced areas in some photographs.

At each of the 18 sites the following ecological attributes were monitored:

i. Vegetation

a) Pasture above-ground biomass and species composition studied in both unfenced and fenced plots for four times a year (starting from Nov 2000 to Nov 2001 at unfenced and from Mar 2001 to Mar 2002 at fenced plots, each at 4 month intervals). Pasture quality was measured once in Aug 2002 during the study (since this was the most suitable time for the land manager to execute this experiment).

b) Tree and shrub: density and basal area measured once during the study.

ii. Soil

a) Physico-chemical properties analysed once in Jan 2002 for each of 0-5, 5-10, 10-20, 20-30 and 30-60 cm soil depth in terms of pH, electrical conductivity, bulk density, and soil organic carbon, total C, total and available content of N and P, and exchangeable K, Ca, Mg, Na and micro-elements (Zn, Mn, Cu and Fe). Soil temperature was measured four times a year (Aug and Nov 2001, Mar and July 2002).

b) Biological properties: Soil respiration measured four times a year (Aug and Nov 2001, Mar and July 2002) and soil microbial biomass was quantified once in March 2002.

iii. Ground litter production, decomposition and nutrient return

a) Litter production was quantified periodically in Mar 2001, July 2001, Nov 2001, and Mar 2002 at cleared and uncleared sites.

b) Seasonal litter decomposition was studied three times per year i.e. starting in Apr 2001 to Aug 2001, from Aug 2001 to Dec 2001, Dec 2001 to Apr 2002, and annual litter decomposition studied from Apr 2001 to Apr 2002.

c) C, N and P release through annual litter decomposition.

Full details of the methods utilized are presented in the following Chapters.

Pasture biomass along with identified ecological parameters (e.g. pasture plant diversity, soil properties, litter production, rates of decomposition and nutrients release through litter) studied for both cleared and uncleared (intact) systems provided detailed perspective for the impact of clearing not only on pasture yield but also on associated changes in ecological parameters. The measurements for a range of 5 to 33 year old cleared pastures illustrated trends in different parameters over time-since-clearing occurred. This comprehensive approach provided an understanding of the interactions/relationships among the various parameters i.e. pasture species diversity and quality, pasture yield, soil physico-chemical properties, litter production and nutrient return as well their individual influence on pasture biomass.

Arguments for a case study approach and its limitations

i. Arguments

All the research was carried out at one property. In the scoping stage of research, many properties were surveyed to find appropriate sites to allow for testing of the various hypotheses. The criteria for site selection i.e. dominant vegetation communities of central Queensland, times-since-clearing, and each tree community with paired comparisons for cleared and uncleared sites, restricted the number of suitable properties. It was difficult to find paired comparisons of cleared and suitably sized uncleared sites for a particular tree community, as only isolated remnants of uncleared vegetation communities exist on many properties.

During the exploratory visits and talks with landholders, it was evident that marked variation existed between properties in management, methods of clearing and post-clearing treatments. Additionally, rainfall is also very localised in the region (Commonwealth of Australia Bureau of Meterology 2003). Therefore, by keeping all the research sites at one property, the impact of such confounding factors would be minimized. Of all the visited properties, only Avocet contained the three required tree communities as paired comparisons for intact and clearing treatments, and for the three different age groups of clearing. However, this approach limited the estimation of impacts of tree clearing for different age of clearing within a tree community, because there was no replication for time-since-clearing treatments for a particular type of tree community.

The alternative to the chosen approach to select cleared treatments at different age groups of clearing was to implement a longitudinal study. A longitudinal approach would mean addressing the issues by monitoring the cleared treatments over a continuous period of time for recent, medium and old period of clearing. This was not practical because of the time constraints in the present research.

Therefore, a case study approach was preferred to maintain the consistency in data for different parameters measured over time with minimal effect of other factors (management, clearing methods and climate). The presence of all sites at one property made logistics easy to monitor the changes in pasture composition, soil respiration, and litter production and decomposition in different seasons.

In addition working at one property allowed closer interaction with the landholder to learn about the grazing management practices. Co-operation with the landholder helped to minimize the differences in grazing practices between sites during the study period. This ensured consistency in the data set.

ii. Limitations

In this research, there were no replicates for time-since-clearing treatments within a tree community. However, cleared treatments for a particular age group in each of the three tree communities had been subjected to almost similar management treatments. Each uncleared (intact) site had three replicates because uncleared treatments were chosen in pairs with each time-since-clearing treatment. It was difficult to find other cleared sites with similar management practices i.e. clearing methods, post-clearing practices, soil type for the range of chosen age groups of clearing. However, there were still some differences between the chosen cleared sites in terms of grazing pressure and post-clearing treatments; such as blade ploughing, which were not possible to prevent.

Within the given time-frame and resources, the present approach to work on various parameters in depth for different time-since-clearing treatments at one property was judged to be the most appropriate. This detailed study conducted on a single property will provide sufficient data to interpret the impacts of time-since-clearing for three dominant tree communities on Avocet.

Statistical methods

Data were analysed using GenStat® ver. 6.0 (GenStat Committee 2002). Pasture availability, soil properties, and litter production and decomposition were analysed using residual maximum likelihood method (REML; Patterson and Thompson 1971). The residual variation was based on the three replicates of each of the virgin communities (i.e. 3 replicates x 3 communities = 9 ‘plots’ representing 3 treatments) allowing comparisons among these and the singularly replicated cleared community ‘plots’ (1 replicate x 3 communities x 3 clearing treatments = 9 ‘plots’ representing 9 treatments). Thus there were a total of 18 ‘plots’ assigned to 12 treatments. The REML model included the fixed effects of community (*E. populnea*, *E. melanophloia* and *A. harpophylla*), clearing treatment (‘cleared-uncleared’, i.e. uncleared, recent, medium and old) and their interaction. In the presence of a significant ($P<0.05$) interaction, results for the interaction were presented while if the interaction was not significant ($P>0.05$), it was removed from the model allowing the main effects to be tested and presented.

Pairwise comparisons of means were conducted using the protected least significant difference (LSD) test at $P=0.05$. As treatment means were based on a varying number of replicates, LSDs differed for different comparisons. The exact LSD values were used to test the treatment differences but only the average LSD values are presented in results in the following Chapters.

It should be noted that each of the clearing treatments for each community are confounded with site, and the clearing treatments also slightly differed in post-clearing management practices.

Relationships between key parameters were investigated with correlation and regression analyses. Multivariate techniques such as cluster analysis and canonical variates analysis were used to assess the overall impact of clearing on key parameters considered in the study (discussed in detail in later Chapters).

An example for REML analysis

Abbreviations used in Table 3.2:

PB: Poplar box (*E. populnea*)
SLIB: Silver leaved-iron bark (*E. melanophloia*)

BG: Brigalow (*A. harpophylla*)
Rep: Replicates for paired treatments according to time-since-clearing
Clr_Uncl: Various time-since-clearing and uncleared treatments

Table 3.2. Data for above-ground pasture biomass (average per yr) at fenced plots.

Community	Rep (3 age gps)	Clr-Uncl	Plot	Quadrat	Avg per yr
PB	1	Recent	1	Q1	1825.18
PB	1	Recent	1	Q2	2193.95
PB	1	Recent	1	Q3	2430.92
PB	1	Recent	1	Q4	1433.57
PB	1	Recent	1	Q5	1391.67
PB	1	Uncleared	2	Q1	1101.57
PB	1	Uncleared	2	Q2	1065.65
PB	1	Uncleared	2	Q3	1707.37
PB	1	Uncleared	2	Q4	699.32
PB	1	Uncleared	2	Q5	1310.72
SLIB	1	Recent	3	Q1	4818.77
SLIB	1	Recent	3	Q2	5615.97
SLIB	1	Recent	3	Q3	3915.70
SLIB	1	Recent	3	Q4	3095.83
SLIB	1	Recent	3	Q5	3421.30
SLIB	1	Uncleared	4	Q1	3234.58
SLIB	1	Uncleared	4	Q2	2859.68
SLIB	1	Uncleared	4	Q3	2713.82
SLIB	1	Uncleared	4	Q4	3678.11
SLIB	1	Uncleared	4	Q5	3011.25
BG	1	Recent	5	Q1	2672.55
BG	1	Recent	5	Q2	2288.58
BG	1	Recent	5	Q3	2138.52
BG	1	Recent	5	Q4	3213.07
BG	1	Recent	5	Q5	1977.60
BG	1	Uncleared	6	Q1	493.90
BG	1	Uncleared	6	Q2	727.87
BG	1	Uncleared	6	Q3	336.88
BG	1	Uncleared	6	Q4	567.93
BG	1	Uncleared	6	Q5	549.48
PB	2	Medium	1	Q1	3801.17
PB	2	Medium	1	Q2	3570.13
PB	2	Medium	1	Q3	3896.61
PB	2	Medium	1	Q4	4526.77

PB	2	Medium	1	Q5	4299.00
PB	2	Uncleared	2	Q1	1417.17
PB	2	Uncleared	2	Q2	2071.77
PB	2	Uncleared	2	Q3	1797.37
PB	2	Uncleared	2	Q4	1514.23
PB	2	Uncleared	2	Q5	1604.52
SLIB	2	Medium	3	Q1	2480.29
SLIB	2	Medium	3	Q2	2281.05
SLIB	2	Medium	3	Q3	3103.57
SLIB	2	Medium	3	Q4	2333.55

Community	Rep (3 age gps)	Clr-Uncl	Plot	Quadrat	Avg per yr
SLIB	2	Uncleared	4	Q1	2938.20
SLIB	2	Uncleared	4	Q2	3122.98
SLIB	2	Uncleared	4	Q3	2250.97
SLIB	2	Uncleared	4	Q4	2066.93
SLIB	2	Uncleared	4	Q5	1932.83
BG	2	Medium	5	Q1	4506.07
BG	2	Medium	5	Q2	5247.67
BG	2	Medium	5	Q3	4433.60
BG	2	Medium	5	Q4	4211.63
BG	2	Medium	5	Q5	5099.95
BG	2	Uncleared	6	Q1	1980.67
BG	2	Uncleared	6	Q2	1692.65
BG	2	Uncleared	6	Q3	1952.52
BG	2	Uncleared	6	Q4	2373.77
BG	2	Uncleared	6	Q5	1449.98
PB	3	Old	1	Q1	2908.90
PB	3	Old	1	Q2	2777.87
PB	3	Old	1	Q3	2687.85
PB	3	Old	1	Q4	2846.82
PB	3	Old	1	Q5	3648.32
PB	3	Uncleared	2	Q1	1107.75
PB	3	Uncleared	2	Q2	632.05
PB	3	Uncleared	2	Q3	808.00
PB	3	Uncleared	2	Q4	888.50
PB	3	Uncleared	2	Q5	609.28
SLIB	3	Old	3	Q1	2256.17
SLIB	3	Old	3	Q2	2231.98
SLIB	3	Old	3	Q3	2195.78
SLIB	3	Old	3	Q4	1905.12
SLIB	3	Old	3	Q5	2568.17
SLIB	3	Uncleared	4	Q1	2898.38
SLIB	3	Uncleared	4	Q2	2650.35
SLIB	3	Uncleared	4	Q3	2450.30
SLIB	3	Uncleared	4	Q4	3358.83
SLIB	3	Uncleared	4	Q5	4024.50
BG	3	Old	5	Q1	3812.17

BG	3	Old	5	Q2	3164.96
BG	3	Old	5	Q3	2718.37
BG	3	Old	5	Q4	3566.40
BG	3	Old	5	Q5	3206.90
BG	3	Uncleared	6	Q1	469.17
BG	3	Uncleared	6	Q2	802.55
BG	3	Uncleared	6	Q3	839.58
BG	3	Uncleared	6	Q4	631.20
BG	3	Uncleared	6	Q5	660.92

Genstat output

```
***** REML Variance Components Analysis *****
```

In REML, variance was estimated from both the fixed and random models. The fixed model included tree community and cleared and uncleared treatments as the main components whereas the random model included age of clearing as the major component to determine variance. The quadrats were included in the random model at sub-plot level only for precision in variance estimation of plots. The final values of variance were estimated through an iterative process of comparing fixed and random models for their maximum likelihood.

```
Response Variate: Avg_pasture biomass per yr

Fixed model: Constant+Community+Cl_Uncl+Community.Cl_Uncl
Random model: Rep_agegp+Rep_agegp.Plot+Rep_agegp.Plot.Quadrat

Number of units: 90

* Rep_agegp.Plot.Quadrat used as residual term
 * Sparse algorithm with AI optimisation

 *** Estimated Variance Components ***

Random term           Component       S.e.
Rep_agegp                  0.      BOUND
Rep_agegp.Plot        251757.    168585.


*** Residual variance model ***

Term             Factor       Model(order)  Parameter       Estimate     S.e.

Rep_agegp.Plot.Quadrat        Identity      Sigma2          200036.   33339.


*** Estimated Variance matrix for Variance Components ***
                    Rep_agegp   1       0
```

```
          Rep_agegp.Plot   2       -1 28420899203
  Rep_agegp.Plot.Quadrat   3        6 -222302237 1111509002
                                    1       2                    3

*** Wald tests for fixed effects ***

   Fixed term             Wald statistic     d.f.    Wald/d.f.    Chi-sq prob

* Sequentially adding terms to fixed model

  Community                   8.11            2          4.06             0.017
  Cl_Uncl                    37.00            3         12.33            <0.001
  Community.Cl_Uncl   33.56            6          5.59            <0.001

* Dropping individual terms from full fixed model

  Community.Cl_Uncl   33.56            6          5.59            <0.001
```

(If Chi-sq probability for Community.Cl_Uncl interaction was >0.05 then the interaction was removed from the fixed model and the main effects were tested for community and clearing_uncleared treatments as: Fixed model: Constant+Community+Cl_Uncl
Random model: Rep_agegp+Rep_agegp.Plot+Rep_agegp.Plot.Quadrat)

```
* Message: chi-square distribution for Wald tests is an asymptotic approximation
  (i.e. for large samples) and underestimates the probabilities in other cases.

 *** Table of predicted means for Constant ***

          2780.    Standard error:   142.3

 *** Table of predicted means for Community ***

Community      BG       PB       SLIB
               2872     2518      2951

Standard errors of differences between pairs

Community    BG   1       *
Community    PB   2    348.7       *
Community   SLIB 3    348.7    348.7       *
                        1           2       3

Standard error of differences:  348.7

*** Table of predicted means for Cl_Uncl ***
```

```
    Cl_Uncl      Medium        Old      Recent   Uncleared
                   3746        2833       2829        1712
```

Standard errors of differences between pairs

```
Cl_Uncl    Medium  1       *
Cl_Uncl       Old  2   441.0      *
Cl_Uncl    Recent  3   441.0   441.0      *
Cl_Uncl Uncleared  4   360.1   360.1   360.1      *
                           1       2     3        4
```

Standard error of differences: Average= 400.6, Maximum= 441.0, Minimum=360.1

*** **Table of predicted means for Community.Cl_Uncl** ***

```
    Cl_Uncl       Medium    Old        Recent    Uncleared
  Community
        BG         4700     3294        2458        1035
        PB         4019      2974        1855        1222
      SLIB         2519      2231        4174        2879
```

REML output provides standard errors of differences for all treatments given the input log file with command for PSE= all differences (Table 3.3).

Table 3.3. Standard errors of differences between various treatments (obtained from REML).

BG.Cl_Uncl	Medium	1	*											
BG.Cl_Uncl	Old	2	763.9	*										
BG.Cl_Uncl	Recent	3	763.9	763.9	*									
BG.Cl_Uncl	Uncleared	4	623.7	623.7	623.7	*								
PB.Cl_Uncl	Medium	5	763.9	763.9	763.9	623.7	*							
PB.Cl_Uncl	Old	6	763.9	763.9	763.9	623.7	763.9	*						
PB.Cl_Uncl	Recent	7	763.9	763.9	763.9	623.7	763.9	763.9	*					
PB.Cl_Uncl	Uncleared	8	623.7	623.7	623.7	441	623.7	623.7	623.7	*				
SLIB.Cl_Uncl	Medium	9	763.9	763.9	763.9	623.7	763.9	763.9	763.9	623.7	*			
SLIB.Cl_Uncl	Old	10	763.9	763.9	763.9	623.7	763.9	763.9	763.9	623.7	763.9	*		
SLIB.Cl_Uncl	Recent	11	763.9	763.9	763.9	623.7	763.9	763.9	763.9	623.7	763.9	763.9	*	
SLIB.Cl_Uncl	Uncleared	12	623.7	623.7	623.7	441	623.7	623.7	623.7	441	623.7	623.7	623.7	*
			1	2	3	4	5	6	7	8	9	10	11	12

Standard error of differences: Average = 691.9
Maximum = 763.9
Minimum = 441.0

Table 3.4. Means obtained from REML analysis (arranged in a matrix).

Medium	BG	4700											
Old	BG	3294	1406										
Recent	BG	2458	2242	836									
Uncleared	BG	1035	3665	2259	1423								
Medium	PB	4019	681	725	1561	2984							
Old	PB	2974	1726	320	516	1939	1045						
Recent	PB	1855	2845	1439	603	820	2164	1119					
Uncleared	PB	1222	3478	2072	1236	187	2797	1752	633				
Medium	SLIB	2519	2181	775	61	1484	1500	455	664	1297			
Old	SLIB	2231	2469	1063	227	1196	1788	743	376	1009	288		
Recent	SLIB	4174	526	880	1716	3139	155	1200	2319	2952	1655	1943	
Uncleared	SLIB	2879	1821	415	421	1844	1140	95	1024	1657	360	648	1295

The difference in means between any two treatments obtained from REML analysis (Table 3.4) were tested for least significant difference (LSD test) according to the formula:

LSD = t value for given degrees of freedom (df) * standard error of difference between any two treatments

t value for 6 df = 2.45 (in the presence of a significant interaction between cleared_uncleared* tree community treatments)
t value for 12 df = 2.18 (in the absence of a significant interaction between cleared_uncleared* tree community treatments)

If the mean differences between any two treatments were greater than their calculated LSD value, than the treatment differences were significant, otherwise there was no significant difference between treatments. The significant difference between any two treatments was calculated as: difference in means (Table 3.4) > standard error of difference (Table 3.3)* 2.45, and presented as ‘Diff’ for significant difference between any two treatments and ‘not’ for no significance difference between treatments in Table 3.5.

Table 3.5. Treatment differences calculated according to LSD.

Medium	BG												
Old	BG	not											
Recent	BG	Diff	not										
Uncleared	BG	Diff	Diff	not									
Medium	PB	not	not	not	Diff								
Old	PB	not	not	not	Diff	not							
Recent	PB	Diff	not	not	not	Diff	not						
Uncleared	PB	Diff	Diff	not	not	Diff	Diff	not					
Medium	SLIB	Diff	not	not	not	not	not	not	not				
Old	SLIB	Diff	not	not	not	not	not	not	not	not			
Recent	SLIB	not	not	not	Diff	not	not	Diff	Diff	not	Diff		
Uncleared	SLIB	Diff	not	not	Diff	not	not	not	Diff	not	not	not	
		Medium BG	Old BG	Recent BG	Uncleared BG	Medium PB	Old PB	Recent PB	Uncleared PB	Medium SLIB	Old SLIB	Recent SLIB	Uncleared SLIB

Testing of hypotheses

The pasture production upon clearing is a major tangible gain from clearing. The thesis begins with evaluation of the impact of clearing on pasture production between cleared and uncleared treatments with the question such as - Is there an increase in pasture yield upon clearing? If so, then how sustainable can these gains be? The study then unveils the impact of clearing on underlying soil physico-chemical and biological properties. The clearing impacts on associated changes such as litter production and nutrient release at cleared and uncleared sites are presented and discussed in relation to other parameters. The final chapter concludes on the overall impact of clearing for key parameters in all the three communities.

Chapter 4. Pasture above-ground-biomass

Introduction

The availability of pastures in tropical open and woodland savannas is of importance for land managers for increased cattle production and hence monetary gain. In Queensland, various practices are followed to enhance pasture production. Clearing/felling of trees followed by raking/burning and then sowing to exotic grasses is the most common practice. Generally the cleared land is sown to exotic "productive" grass species such as *Cenchrus ciliaris* L. for it performs well in disturbed soils and hence increases the pasture production. The earlier reports by Burrows *et al.* (1990), Burrows (2002) and Scanlan and Burrows (1990) have demonstrated an increase in pasture production with land clearing.

Greater pasture production with subsequent monetary gains from developed pastures led to high rates of tree clearing in Queensland. During 1999-2001, a total of 577,000 ha (about 367, 000 ha remnant and 210, 000 ha non-remnant woody vegetation) were cleared per annum and about 94 per cent of the total clearing was to develop land for pastures (Department of Natural Resources and Mines 2003). Clearing trees for pasture and agricultural land has historically been favoured under different government policies until 1985 (Boulter *et al.* 2000).

Tree clearing has helped to develop land for pastoral and agricultural systems since European settlement. Cattle production has contributed considerably to state development. There are increasing concerns that most of the development that occurred in the past involved little understanding of the inherent potential of natural resources available on this continent. For example, there could be differential responses in pasture production with clearing depending on vegetation, soil and climate type. Moreover, there are associated risks for pasture run-down, land degradation and loss of biodiversity that have been largely overlooked in the past. The short-term approach to obtain maximum benefits, without taking into account associated ecological risks, may lead to over-estimation of the potential of clearing for pasture production. Indeed, in the absence of long-term studies, it is difficult to draw conclusions about the net gains and losses due to tree clearing in pasture production systems. Unfortunately most of the work reported to date has highlighted pasture yield only up to about 10 years after clearing (Scanlan 2002) and without adequately accounting for the associated changes in various ecological functions.

With this in mind, research was undertaken to quantify the impact of clearing on pasture production, for various tree communities, for clearing durations longer than 10 years. Three major types of tree communities: *Eucalyptus populnea* F. Muell. (poplar box), *E. melanophloia* F. Muell. (silver-leaved ironbark) and *Acacia harpophylla* F. Muell. ex. Benth. (brigalow) were selected in central Queensland to quantify the above-ground pasture biomass at cleared and uncleared (intact) pasture systems, and the species distribution in both settings. The chemical composition of pasture plants for all cleared and uncleared sites, and pasture quality in terms of digestibility, dietary crude protein, non-grass component and faecal N for one set of cleared and uncleared sites, were also studied. The selection of three vegetation communities at various times since clearing should provide information about their specific response over time for pasture production post-clearing. This will assist in assessing whether cleared or uncleared pasture systems perform better and in examining the sustainability of a cleared system over time for a particular type of tree community.

Materials and methods

Study area and design

Full details were presented in Chapter 3.

Paired plots of cleared and intact/uncleared woodlands for *E. populnea*, *E. melanophloia* and *A. harpophylla* communities across three age groups of clearing i.e. recent (5 yr), medium (11-13 yr) and old (33 yr) were selected. The study represents a 3 (types of tree communities) x 3 (time since clearing) x 2 (cleared v/s intact) factorial design (Table 4.1).

Table 4.1. Paired treatments for each of *E. populnea*, *E. melanophloia* and *A. harpophylla*.

Treatments	**Time since clearing**
Cleared	**Recent (5 years)**
Uncleared/intact	Never cleared
Cleared	**Medium (11-13 years)**
Uncleared/intact	Never cleared
Cleared	**Old (33 years)**
Uncleared/intact	Never cleared

Pasture above-ground-biomass was studied under normal grazing practices (referred to as "unfenced" in the text) as well as for exclusion of grazing (referred to as "fenced" in the text).

i. Pasture above-ground-biomass

At each site an area of 1 ha as a representative of the total area (unfenced) and a fenced plot of 10 m x 10 m located centrally at each site (as mentioned in Chapter 3 - Fig. 3.4 and 3.5) were selected to quantify pasture biomass. The quadrat method was used to estimate the amount of pasture biomass (Kent and Coker 1992). Measurements were taken for different seasons in a year in Nov 2000, March 2001, July 2001 and Nov 2001 for the unfenced plots. Since fencing was done in Dec 2000 for the 10 m x 10 m plots, the first set of measurements started from March 2001 and continued four-month intervals until March 2002. In March 2002, the recently cleared site for *E. melanophloia* was accidently burnt, so measurements for that site were not available.

Total aboveground biomass was quantified from five randomly assigned quadrats of 1 m x 1 m area in each of the fenced and unfenced sites (size of quadrat was determined based upon species number for determination (see later) of species diversity). Quadrats were located at different positions at each sampling date. Plant samples from each quadrat were harvested just above ground level, taken to the laboratory and dried at 60 ^{o}C for 48 hours to determine their biomass.

The average quantity of pasture above-ground-biomass for grazing was calculated over a year from the seasonal measurements. For repeated measurements i.e. Nov 2000 and Nov 2001 at unfenced sites, their average was considered with other seasonal measurements (Mar 2001 and July 2001) to calculate the overall average amount of pasture biomass for the year. Similarly for fenced plots, the average was taken for the repeated readings (March 2001 and March 2002) to calculate the total average amount of pasture biomass per year.

The pasture above-ground-biomass at each unfenced site was also computed in relation to the amount of pasture consumed by cattle per year, to overcome the confounding factor of varied grazing pressure at unfenced sites. The average amount of pasture consumed per year by cattle was taken as 3139 kg/beast given that this grazing property supports mostly breeder cattle of 400 kg/beast for which per beast dry matter consumption is about 8.6 kg/day (Minson and McDonald 1987). There is likely to be some variation in different sites according to type of plant species available, but these estimations provides some idea to assess the pasture production at unfenced sites.

Data were analysed using Genstat ver 6.0 (2002). The residual maximum likelihood (REML) (Patterson and Thompson 1971) method was used. For this, all the uncleared treatments of a tree community were considered as replicates while the cleared treatments for each age group were taken as such with no replication. Pasture availability data for each season and the average per year were compared across all cleared and uncleared treatments.

The main effects for type of tree community and uncleared-cleared (recent, medium and old) treatments within each tree community were analysed. Models included the fixed effects of community, clearing treatments plus their interaction (community*cleared-uncleared), and the random effects of age since clearing and uncleared treatments within a community. If the interaction between community and cleared-uncleared treatments was not significant ($P>0.05$) then it was removed from the fixed model to test the main effects. The means from REML analysis were used in presenting the results for all the parameters, both those showing significant treatment interaction as well as in its absence, to maintain consistency.

The variance matrix derived from REML analysis was used to calculate approximate LSDs (least significant differences of means) at $P <0.05$. The significance levels were considered with respect to the exact LSD value for a treatment, but only the average values for approximate LSD for all the treatments were mentioned in the results.

ii. Pasture species diversity and pasture quality

a) Pasture species diversity

At the same time as collection of biomass data, the total number of herbaceous plant species was counted in each quadrat. To estimate the dominant species at a site, the total number of individuals for all species that occurred at each site was quantified, and a common number was considered for all the sites i.e. a species was considered dominant with >250 individuals/ha/year. Data were analysed for Shannon Wiener's index of diversity, using species diversity and richness software (Henderson and Seaby 1998) (PISCES Conservation Ltd).

Distribution of species at different sites in relation to environmental variables (pasture production, litter production, soil pH_w, available P and NO_3-N, microbial biomass in terms of C and N in top soil, details on litter production are presented in Chapter 7, details on soil data collected are presented in Chapter 5 and on microbial biomass in Chapter 6) was analysed using Canoco 4 (Braak and Smilauer 1998). Canonical Correspondence Analysis (CCA) was applied and ordination diagrams (biplots) were obtained for corresponding sites and species. CCA (a generalization of CVA (Canonical Variates Analysis)) explains the distribution of species in a particular sample space (sites) using Eigen values for different canonical axes. It follows the similarity rule as a species close to a site (sample) is more likely to be most abundant at that site compared to a far away site. CCA also contrasts sites in ordination place by environmental variables. These ordination diagrams i.e. biplots herein, explained relationship between species and sites in relation to environmental variables. Only the most abundant species are presented in biplots.

The ordination results are presented for unfenced plots only, as the data on litter and soil properties were available only for those not for the fenced plots.

b) Pasture quality

It was hypothesised that species diversity would impact on pasture quality. To test this, a small experiment was conducted at the medium age cleared and uncleared sites for *A. harpophylla* only. An area of approximately two hectares was temporarily fenced in both the cleared and uncleared sites. At each site 7-8 cattle were enclosed for 14 days. The same cattle breed and age group were used. After 13 days of grazing at the sites, dung samples were collected from the two groups of cattle and stored in a cold box for cooling during transport. Dung samples were pooled from 7-8 cattle and dried at 48 ^{o}C for 48 hours and then sent for NIR (Near Infra Red) analysis at Davies Laboratory, CSIRO, Townsville, Qld. The pasture quality was assessed from faecal N (%), non-grass component (%), digestibility and crude protein (%).

The pasture plants collected over different seasons using quadrat sampling (for biomass estimation) were bulked per site for each season and analysed for C, N and P content. Data were analysed for MANOVA using the interaction term for tree community*cleared-uncleared treatments for measurements in different seasons.

iii. Tree basal area and density

At intact woodland sites, basal area and number of trees per unit area were quantified using the transect method (Back *et al.* 1997). At each site three transects of 50 m x 4 m area were laid. All the woody trees and shrubs greater than 1.5 m height were included for measurements. Tree basal area was estimated from the circumference taken with a measuring tape at 30 cm above the ground. Averages were calculated for all the trees and shrubs that occurred in three transects and expressed as square metres basal area per hectare for each tree community. The numbers of trees in each transect were also counted. Data were analysed for variance to estimate the difference between the communities.

Results

i. Pasture above-ground-biomass

The amount of pasture biomass was quantified in response to three different times of clearing: recent (5 year), medium (11-13 year) and old (33 year) for *E. populnea*, *E. melanophloia* and *A. harpophylla* for different seasons in a year.

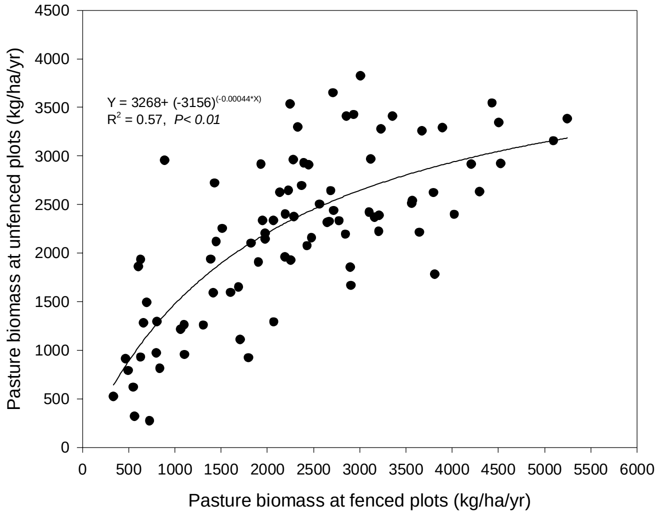


Fig 4.1. Pasture biomass at unfenced and fenced plots for all the treatments (cleared and uncleared) at *E. populnea*, *E. melanophloia* and *A. harpophylla* communities.

Pasture biomass at unfenced sites was positively related with that at fenced sites ($r^2 = 0.57$, $P<0.01$) at all sites regardless of age-since-clearing (Fig 4.1). The results are presented separately for fenced and unfenced plots.

REML analysis quantifies the level of interaction between tree type and time-since-clearing for all the cleared and uncleared treatments. A significant interaction between the communities and time-since-clearing treatments revealed a specific response of each tree community. Such results with significant interaction at $P<0.05$ are mentioned with respect to each community. However, in the absence of significant interaction, results were mentioned only for the main effects of clearing and tree community.

1) Fenced plots

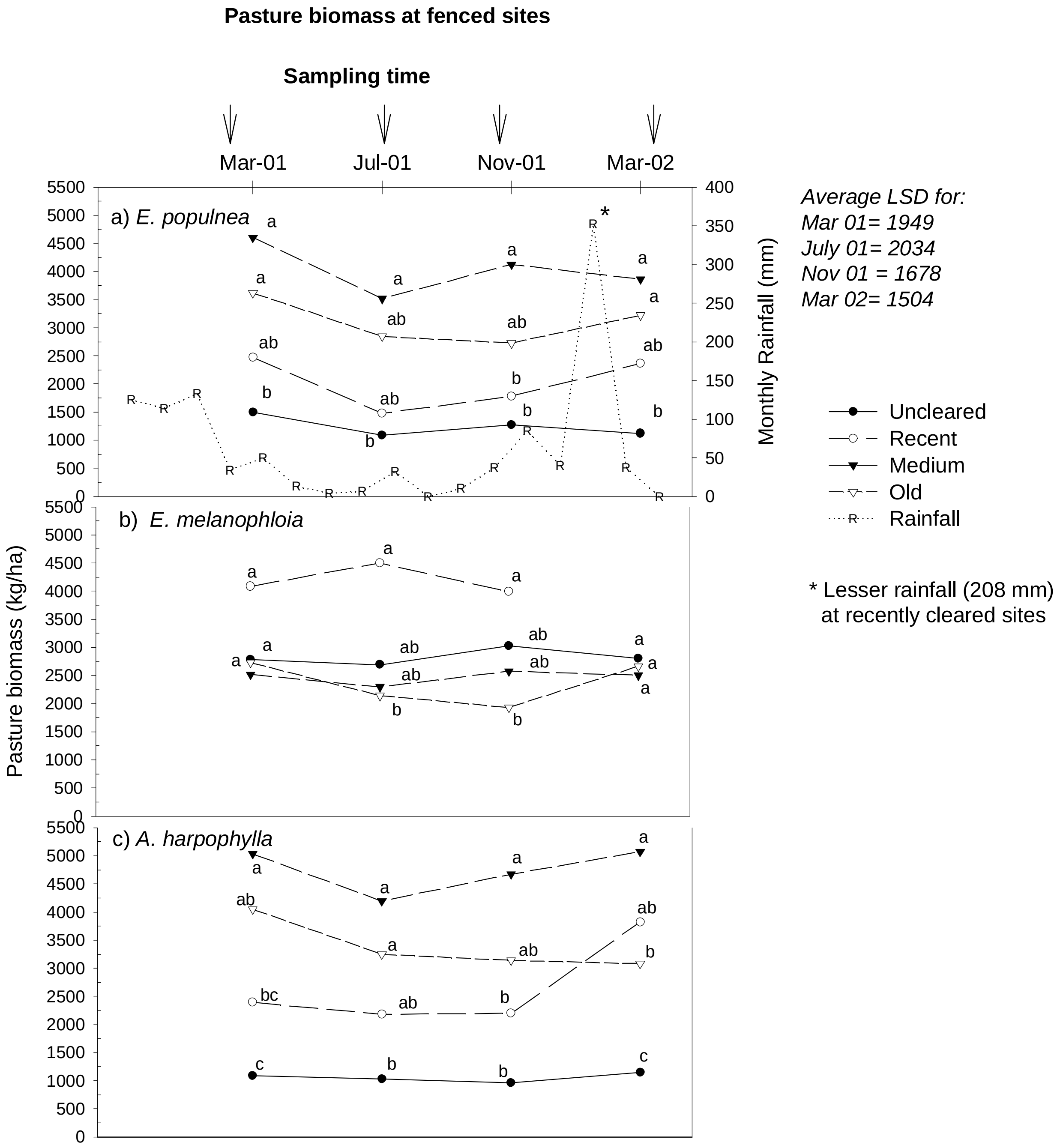
Data revealed a significant ($P <0.05$) interaction among the tree communities for cleared and uncleared treatments. Therefore, the quantity of pasture biomass in cleared and uncleared treatments is mentioned with reference to each tree community.

E. populnea

Clearing enhanced the amount of pasture available for grazing given that most cleared sites showed greater pasture biomass than the uncleared sites during all the seasonal readings (Fig 4.2a). The medium age of clearing (after 13 years) showed a highly significant increase in pasture biomass compared to uncleared site for all the seasonal measurements. However increase in pasture yield was not consistent over time of clearing. The oldest cleared treatment i.e. 33 years had the same pasture quantity as the uncleared site for July and Nov 2001, while the pasture biomass in Mar 2001 and Mar 2002 was greater at the old clearing than at the uncleared site (Fig 4.2a).

The amount of pasture biomass tended to increase from recent to medium age of clearing, but the differences were significant only in Nov 2001 at $P < 0.05$ (Fig 4.2a).

The average amount of pasture biomass (kg/ha/yr) throughout the year was in line with that seen during different seasons (Table 4.2 and Fig 4.2a). Clearing significantly increased the average annual amount of pasture biomass at medium age of clearing (4019 kg/ha/yr) compared to the uncleared (1222 kg/ha/yr) site. This was followed with a decline at old age clearing (2974 kg/ha/yr) but still with significantly greater pasture biomass than at uncleared sites (Table 4.2).



Average LSD for:
Mar 01= 1949
July 01= 2034
Nov 01 = 1678
Mar 02= 1504

* Lesser rainfall (208 mm) at recently cleared sites

Fig 4.2. Pasture biomass (kg/ha/yr) and monthly rainfall (mm) at fenced plots for various cleared and uncleared treatments for a) *E. populnea,* b) *E. melanophloia* and c) *A. harpophylla* communities (similar rainfall as at *E. populnea*, at *E. melanophloia* and *A. harpophylla* sites).

Any two treatments with different letters on any one sampling date within a tree community denote significant difference at *P <0.05 (Average LSD denotes the least significant difference of means between any two treatments for a season).*

E. melanophloia

Pasture biomass was different following clearing of *E. melanophloia* compared that to following clearing of *E. populnea*. A substantial increase in pasture yield with clearing at the recently cleared site was followed by a decline through medium to old age of clearing (Fig 4.3b). Interestingly, there was no significant effect of clearing on pasture biomass for different age-since-clearing compared to uncleared sites, in any of the seasons. Between the cleared sites, pasture biomass was significantly greater at recent compared to old cleared sites for July and Nov 2001 (Fig 4.2b).

The average amount of pasture biomass for the year (kg/ha) showed no significant difference between the ages of clearing and uncleared sites (Table 4.2). However, the decline in pasture biomass with age since clearing was apparent at the oldest compared to recent age-since-clearing.

A. harpophylla

Clearing *A. harpophylla* greatly enhanced the pasture yield even after a long time post-clearing. The amount of pasture was greater in the medium and old cleared compared to the uncleared sites for all the seasons except for Nov 2001 in old cleared and uncleared sites (Fig 4.2c). Only in Mar 2002 did the recent clearing yield more than the uncleared site (Fig 4.2c).

Within the cleared sites, clearing led to a greater increment in pasture biomass for medium (13 years) age of clearing compared to recent age of clearing. A similar but smaller increase over the recent clearing was evident for the oldest age-since-clearing treatments (Fig 4.2c).

The average amount of pasture per year also revealed similar trends to the seasonal data, with significant increase at medium (4700 kg/ha/yr) and old (3294 kg/ha/yr) age of clearing compared to the uncleared site (1035 kg/ha/yr) (Table 4.2). Between uncleared sites, *E. melanophloia* supported the maximum pasture yield (2879 kg/ha/yr) compared to *E. populnea* (1222 kg/ha/yr) and *A. harpophylla* (1035 kg/ha/yr) communities (Table 4.2).

Table 4.2 Average amount of pasture biomass (kg/ha/year) at fenced plots for uncleared and cleared (recent, medium and old age of clearing) treatments for *E. populnea, E. melanophloia* and *A. harpophylla* communities.

Tree community	Uncleared	Recent	Medium	Old
*E. populnea**	${}_{b}1222^{c}$	${}_{b}1855^{bc}$	${}_{ab}4019^{a}$	${}_{a}2974^{ab}$
E. melanophloia	${}_{a}2879^{ab}$	${}_{a}4174^{a}$	${}_{b}2519^{ab}$	${}_{a}2231^{b}$
A. harpophylla	${}_{b}1035^{b}$	${}_{ab}2458^{ab}$	${}_{a}4700^{a}$	${}_{a}3294^{a}$

*different superscripts in a row and subscripts in a column represent the significant levels of difference at $P<0.05$ between treatments.

Pasture biomass was related to the distribution of rainfall; the maximum yield at most sites was measured during Mar 2001 and 2002 after summer rainfall (Nov- Feb) compared to that in July or Nov 2001 (Fig 4.2).

2) Unfenced plots

REML analysis for pasture biomass at cleared and uncleared treatments in unfenced plots revealed significant and non-significant interactions between the main effects of community and time-since clearing. The interactions were significant for Nov 2000 and July 2001 (presented in Fig 4.3) but non-significant for Mar and Nov 2001 data (presented for main effects of time-since-clearing across all three tree communities in Table 4.3).

In general, there was a trend that clearing led to an increase in pasture biomass during all the seasons in *E. populnea* and *A. harpophylla* (Fig 4.3a and c) while *E.melanophoila* showed such a response only at recent age of clearing (Fig 4.3b). The significant effect was apparent during all the seasons at recent clearing in *E. melanophloia* and *A. harpophylla* except for July 2001 (Fig 4.3a and c).

Pasture biomass showed a common response to increase at recent clearing in Mar and Nov 2001 in all the treatments, in the absence of any interactions between communities and age since clearing effects (Table 4.3).

At medium age of clearing (11-13 years), pasture biomass was particularly greater for *A. harpophylla* compared to its uncleared control (Fig 4.3c). The trend persisted across all seasons. No other tree communities exhibited significant difference between medium age cleared and uncleared treatments.

Among the time-since-clearing treatments, there was no notable effect on pasture biomass between recent, medium or old age of clearing for *E. populnea* and *A. harpophylla*. However, the recent clearing for *E. melanophloia* had significantly greater pasture biomass than medium and old age treatments (Fig 4.3b).

The average quantity of pasture per year was a maximum at medium, followed by recent and old clearing, and only medium had a significant increase compared to uncleared sites (Table 4.3).

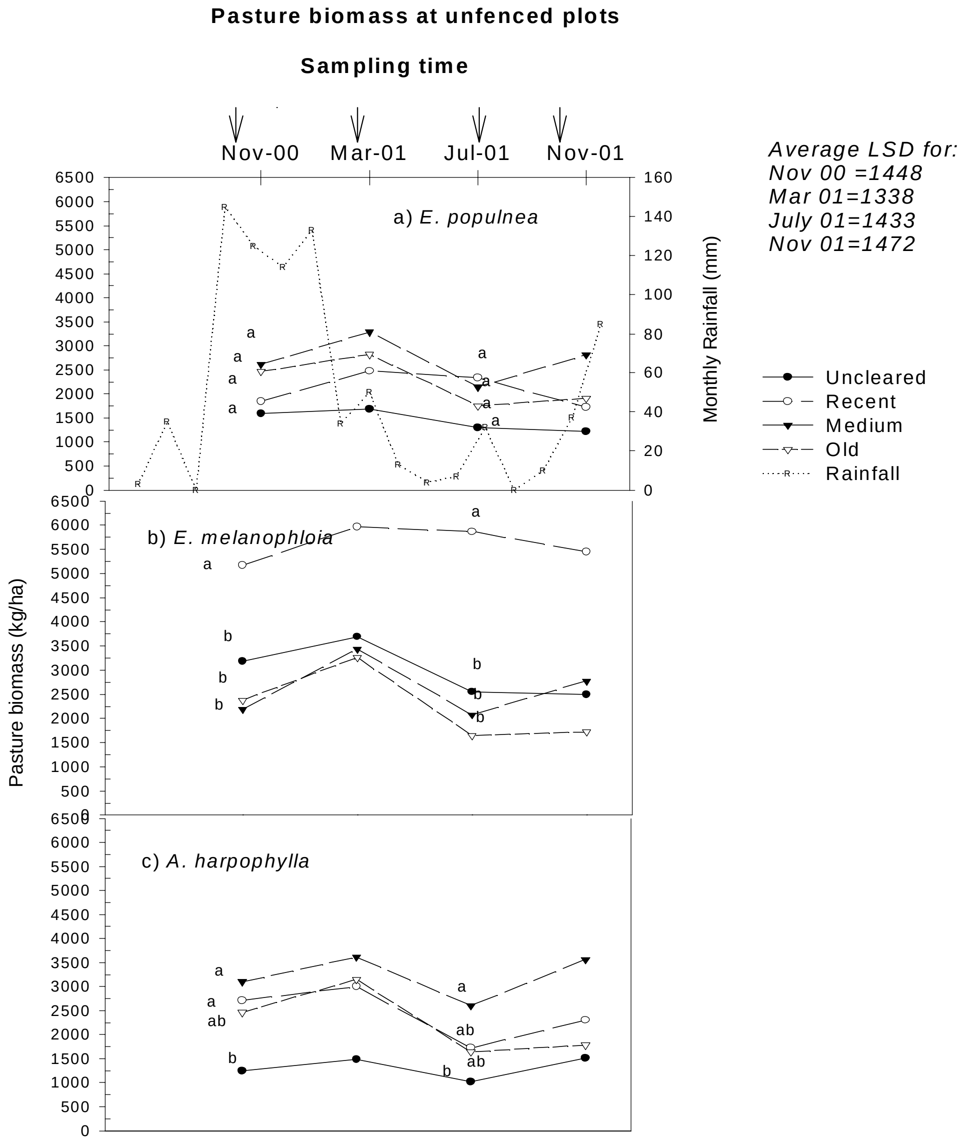


Fig 4.3. Pasture biomass (kg/ha) and monthly rainfall (mm) at unfenced plots for various cleared and uncleared treatments for a) *E. populnea,* b) *E. melanophloia* and c) *A. harpophylla* communities (similar rainfall as at *E. populnea,* at *E. melanophloia* and *A. harpophylla* sites).

Any two treatments with different letters on any one sampling date within a tree community denote significant difference at *P <0.05 (Average LSD denotes the least significant difference of means between any two treatments for a season).*

Table 4.3. Pasture biomass (kg/ha) in Mar 2001 and Nov 2001 and average annual amount of pasture (kg/ha/yr) at uncleared and recent, medium and old age of clearing.

	Uncleared	Recent	Medium	Old
Mar 01*	2289^{b}	3813^{a}	3443^{ab}	3077^{ab}
Nov 01	1738^{b}	3157^{a}	3049^{ab}	1803^{ab}
Average	2104^{b}	3565^{ab}	4573^{a}	2030^{b}

*different superscripts in a row represent the significant levels of difference at *P<0.05* between cleared and uncleared treatments.

Grazing management was likely to affect pasture biomass at different time-since-clearing treatments (grazing time-Appendix 1a, and grazing pressure-Table 4.4). When average pasture biomass per year calculated in relation to cattle consumption, was compared across sites, pasture biomass was significantly greater at medium clearing than uncleared sites in *E. populnea* and *A. harpophylla*, and at recent clearing compared to uncleared site for *E. melanophloia* (Table 4.4.). These results were in line with pasture biomass at fenced treatments (Table 4.2). Between uncleared communities, *E. melanophloia* supported the maximum average pasture biomass over a year (3761 kg/ha/yr) compared to *E. populnea* (2107 kg/ha/yr) and *A. harpophylla* (1823 kg/ha/yr) (Table 4.4).

The pasture biomass varied during different seasons in unfenced plots. Rainfall seemed to be an important factor along with grazing management that contributed to seasonal variation at cleared and uncleared sites. In general, pasture biomass was more during Mar 2001 after major rainfall events in Dec and Jan 2001 compared to that in other seasons (Fig 4.3).

Table 4.4. Average pasture biomass (kg/ha/yr) adjusted in relation to cattle consumption, and grazing pressure (cattle/ha) at cleared (recent, medium and old) and uncleared sites for *E. populnea, E. melanophloia* and *A. harpophylla* communities.

Tree community		Uncleared	Recent	Medium	Old
E. populnea	Average pasture biomass*	${}_{b}2107^{b}$	${}_{b}2817^{ab}$	${}_{a}3792^{a}$	${}_{a}2748^{ab}$
	Grazing pressure	1/5	1/5	1/3	1/6
E. melanophloia	Average pasture biomass*	${}_{a}3761^{b}$	${}_{a}6335^{a}$	${}_{a}3743^{b}$	${}_{a}2816^{b}$
	Grazing pressure	1/5	1/5	1/3	1/6
A. harpophylla	Average pasture biomass*	${}_{b}1823^{b}$	${}_{b}3030^{ab}$	${}_{a}4259^{a}$	${}_{a}2803^{ab}$
	Grazing pressure	1/6	1/5	1/3	1/6

*different superscripts in a row and subscripts in a column represent the significant difference at $P<0.05$ between treatments.

ii. Pasture plant diversity and pasture quality

a) Pasture species diversity

1) Fenced plots

The sown grass species *C. ciliaris* was dominant at all cleared sites for *E. populnea, A. harpophylla* and at the oldest cleared site for *E. melanophloia* communities (Table 4.5). Uncleared sites had mostly native grass species i.e *Aristida* spp. at *E. populnea; Themeda* *triandra* and *Heteropogon contortus* at *E. melanophloia;* and *Eragrostis, Paspalidium* and *Enteropogon* species at *A. harpophylla,* along with some legumes and *C. ciliaris* (Table 4.5).

Table 4.5. Pasture composition (dominant species (>250 individuals of a species/ha/yr)) at recent, medium, old age cleared and uncleared fenced sites for *E. populnea, E. melanophloia* and *A. harpophylla* communities.

Tree community	**Uncleared**	**Recent**	**Medium**	**Old**
E. populnea	*Ancistrachne uncinulata* *Aristida benthamii* *Aristida calycina* *Cenchrus ciliaris* *Enteropogon acicularis* *Eragrostis lacunaria* *Eriochloa pseudoacrotricha* *Fimbristylis dichotoma* *Hybanthus enneaspermum* *Melinis repens* *Perotis rara* *Sida filiformis* *Sida subspicata*	*Aristida calycina* *Aristida caputmedusae* *Cenchrus ciliaris* *Chloris gayana* *Eragrostis lacunaria* *Eriochloa pseudoacrotricha* *Fimbristylis dichotoma* *Melinis repens* *Spermacoce sp.*	*Cenchrus ciliaris* *Eriochloa pseudoacrotricha*	*Cenchrus ciliaris* *Evolvulus alsinoides* *Hybanthus enneaspermum* *Melinis repens* *Portulaca filifolia* *Stylosanthes scabra* *Vernonia cinerea* *Wahlenbergia gracilis*
E. melanophloia	*Aristida calycina* *Bothriochloa bladhii* *Cenchrus ciliaris* *Heteropogon contortus* *Rhynchosia minima* *Stylosanthes scabra* var. *seca* *Themeda triandra*	*Heretopogon contortus* *Melinis repens* *Peripleura hispidula* *Phyllanthus virgatus* *Themeda triandra*	*Cenchrus ciliaris* *Enteropogon acicularis* *Heteropogon contortus* *Melinis repens* *Stylosanthes scabra* var. *seca*	*Cenchrus ciliaris* *Fimbristylis dichotoma* *Stylosanthes scabra* var. *seca*
A. harpophylla	*Brachiaria piligera* *Cenchrus ciliaris* *Enteropogon acicularis* *Eragrostis lacaunaria* *Paspalidium caespitosum*	*Aristida calycina* *Aristida caputmedusae* *Cenchrus ciliaris* *Enteropgon acicularis* *Eragrostis lacunaria* *Leptochloa decipiens* *Melinis repens* *Sida filiformis*	*Cenchrus ciliaris* *Stylosanthes scabra* var. *seca*	*Cenchrus ciliaris* *Vernonia cinerea* *Wahelnbergia gracilis*

All uncleared sites with the exception of *A. harpophylla* exhibited greater number of species than the cleared sites. For *A. harpophylla*, recent clearing showed greater species diversity than the uncleared site. The most prominent effects occurred at medium age since clearing, for which the species diversity was the least, but pasture biomass was at a maximum (Table 4.6).

Table 4.6. Species number (ha/yr), Shannon Wiener's index of species diversity and average pasture biomass (kg/ha/yr) at uncleared, and at recent, medium and old age of clearing at fenced sites for *E. populnea, E. melanophloia* and *A. harpophylla* communities.

Tree type	Treatments	Species number	Shannon Wiener's index*	Pasture biomass*
E. populnea	Uncleared	32	2.5201^{a}	1222^{c}
	Recent	21	2.2478^{b}	1855^{bc}
	Medium	13	0.3217^{d}	4019^{a}
	Old	20	0.7941^{c}	2974^{ab}
E. melanophloia	Uncleared	37	2.3881^{a}	2879^{ab}
	Recent	24	1.6142^{b}	4174^{a}
	Medium	17	1.2675^{c}	2519^{ab}
	Old	19	1.1236^{d}	2231^{b}
A. harpophylla	Uncleared	34	2.4751^{b}	1035^{b}
	Recent	35	2.6729^{a}	2458^{ab}
	Medium	9	0.2068^{d}	4700^{a}
	Old	10	0.6318^{c}	3294^{a}

*different superscripts in a column represent significant difference at $P<0.05$ for cleared and uncleared treatments within a tree community.

Between cleared treatments, species diversity (Shannon Wiener's index) was greater at recent followed by old and medium age since clearing for *E. populnea* (Table 4.6). In *E. melanophloia*, Shannon Wiener's diversity index showed continuous decrease in diversity with age since clearing. A slightly different trend was seen for *A. harpophylla* with maximum species diversity at recent clearing followed by old and medium cleared treatments.

There was some increase in species diversity over time of clearing as the old cleared treatment had greater diversity than the medium for *E. populnea* and *A. harpophylla* (Table 4.6).

Species diversity (Shannon Wiener's index) was negatively correlated with pasture biomass in all tree communities ($r = -0.72$ at $P < 0.05$). Species number for each seasonal measurement and for the average over a year were also negatively correlated with the corresponding seasonal and average pasture biomass measurements (Mar 2001 r = -0.43, July 2001 r = -0.39, Nov 2001 r = -0.40, Mar 2002 r = -0.51, and for the average over a year r = -0.65, all significant at $P < 0.05$).

2) Unfenced sites

Species composition differed between cleared and uncleared sites. *C. ciliaris* was dominant at most of the cleared sites for *E. populnea* and *A. harpophylla*, whereas native grasses (*T. triandra*, *H. contortus* and *Bothriochloa* sp.) were mainly present at *E. melanophloia* sites (Table 4.7).

Table 4.7. Pasture composition (dominant species (>250 individuals of a species/ha/yr)) at uncleared and recent, medium, old age of cleared unfenced sites for *E. populnea*, *E. melanophloia* and *A. harpophylla* communities.

Tree community	**Uncleared**	**Recent**	**Medium**	**Old**
E. populnea	*Alloteropsis semialata*	*Aristida calycina*	*Cenchrus ciliaris*	*Cenchrus ciliaris*
	Ancistrachne uncinulata	*Aristida caputmedusae*		*Melinis repens*
	Aristida benthamii	*Calotis cuneifolia*		
	Aristida calycina	*Cenchrus ciliaris*		
	Cenchrus ciliaris	*Enteropogon acicularis*		
	Enneapogon lindleyanus	*Eragrostis lacunaria*		
	Eragrostis lacunaria	*Eriochloa pseudoacrotricha*		
	Eragrostis sororia			
	Eriochloa pseudoacrotricha			
	Evolvulus alsinoides			
	Fimbristylis dichotoma			
	Hybanthus enneaspermum			
	Indigofera colutea			
	Melinis repens			
	Perotis rara			
	Sida filiformis			
E. melanophloia	*Aristida calycina*	*Aristida calycina*	*Bothriochloa sp.*	*Bothriochloa sp.*
	Aristida leptopoda	*Fimbristylis dichotoma*	*Cenchrus ciliaris*	*Cenchrus ciliaris*
	Aristida ramosa	*Heteropogon contortus*	*Enteropogon acicularis*	*Eragrostis lacunaria*
	Bothriochloa pertusa	*Melinis repens*	*Heteropogon contortus*	*Fimbristylis dichotoma*
	Calotis cuneifolia	*Peripleura hispidula*	*Indigofera brevidens*	*Heteropogon contortus*
	Cenchrus ciliaris	*Phyllanthus virgatus*	*Melinis repens*	*Portulaca oleracea*
	Eragrostis lacunaria	*Themeda triandra*	*Parsonsia lanceolata*	*Stylosanthes scabra* var. *seca*
	Evolovulus alsinoides		*Stylosanthes scabra* var. *seca*	*Wahlenbergia gracilis*
	Heteropogon contortus			
	Melinis repens			
	Peripleura hispidula			
	Stylosanthes scabra var. *seca*			
	Themeda triandra			
A. harpophylla	*Aristida calycina*	*Aristida caputmedusae*	*Cenchrus ciliaris*	*Cenchrus ciliaris*
	Brachiaria piligera	*Aristida ramose*	*Melinis repens*	*Stylosanthes scabra* var. *seca*
	Cenchrus ciliaris	*Brachyachne convergens*	*Stylosanthes scabra* var. *seca*	*Vernonia cinerarea*
	Enneapogon lindleyanus	*Cenchrus ciliaris*		*Wahlenbergia gracilis*
	Enteropogon acicularis	*Chloris gayana*		
	Enteropogon unispiceus	*Dicanthium sericeum*		
	Eragrostis lacunaria	*Enteropogon acicularis*		
	Maireana microphylla	*Eragrostis lacunaria*		
	Paspalidium caespitosum	*Eriochloa pseudoacrotricha*		
	Portulaca oleracea	*Melinis repens*		

The number of species was greater at uncleared compared to cleared sites for all tree communities. Shannon Wiener's index of diversity was significantly greater at uncleared compared to all the cleared treatments in all tree communities (Table 4.8). The index of diversity did not differ between medium and old age of clearing (Table 4.8).

Species diversity (Shannon Wiener's index) was negatively correlated with pasture biomass (r = -0.41 at $P < 0.05$). There was no significant correlation for various seasonal and the average per year species number and the corresponding pasture biomass across the three tree communities.

Table 4.8. Species number (ha/yr), Shannon Wiener's index of species diversity and average pasture biomass (kg/ha/yr calculated in relation to cattle consumption) at uncleared, and at recent, medium and old age cleared unfenced treatments for *E. populnea*, *E. melanophloia* and *A. harpophylla* communities.

Tree type	Treatments	Species number	Shannon Wiener's index*	Pasture biomass
E. populnea	Uncleared	41	2.7432^{a}	2107^{b}
	Recent	30	2.3279^{b}	2817^{ab}
	Medium	16	0.5573^{c}	3792^{a}
	Old	17	0.8206^{c}	2748^{ab}
E. melanophloia	Uncleared	42	2.4821^{a}	3761^{b}
	Recent	38	2.0245^{b}	6335^{a}
	Medium	23	1.5606^{c}	3743^{b}
	Old	22	1.4825^{c}	2816^{b}
A. harpophylla	Uncleared	43	2.8643^{a}	1823^{b}
	Recent	26	2.4925^{b}	3030^{ab}
	Medium	14	0.8606^{c}	4259^{a}
	Old	14	0.7744^{c}	2803^{ab}

*different superscripts in a column represent significant difference at $P<0.05$ for cleared and uncleared treatments within a tree community.

The distribution of pasture species in the biplot (obtained from CCA) for *E. populnea* showed that most species were distributed close to the uncleared site followed by fewer at the recent, and very few species were scattered near the medium and old cleared sites (Fig 4.4). The species distribution at uncleared was mostly determined by litter production. At recent cleared sites, pasture species showed a tendency to grow on soils with more NO_3-N, available P and microbial biomass (C and N). However, medium and old cleared sites had only few scattered species and these species showed more biomass compared to that at uncleared and recently cleared sites (Fig 4.4).

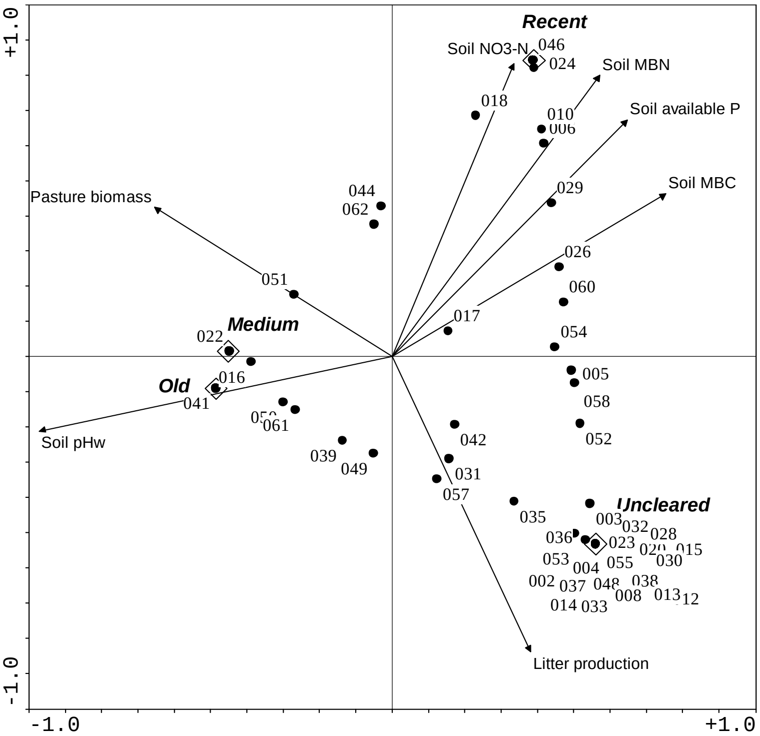


Fig 4.4. Biplot for species distribution (●) at uncleared, recent, medium and old cleared unfenced sites (◊) of *E. populnea* in relation to environmental variables (pasture biomass, litter production, soil NO_3-N, available P, soil microbial biomass for C (Soil MBC) and N (Soil MBN) and soil pH_w). Only most abundant species are shown. Numbers from 001-063 represent the name of species: *Achyranthes aspera* (001), *Alloteropsis semialata* (002), *Ancistrachne uncinulata* (003), *Aristida benthamii* (004), *Aristida calycina*(005), *Aristida caputmedusae* (006), *Boerhavia dominii* (007), *Brachiaria piligera* (008), *Brunoniella australis* (009), *Calotis cuneifolia* (010), *Calotis hispidula* (011),*Canthium cuneifolium* (012),*Canthium oleifolium* (013), *Capparis lasiantha* (014),*Cassinia laevis* (015),*Cenchrus ciliaris* (016), *Cheilanthes distans* (017), *Chloris gayana* (018), *Cyperus dietrichiae* (019), *Cyperus exaltatus* (020), *Einadia nutans* (021), *Enneapogon lindleyanus* (022), *Enteropogon acicularis* (023), *Epaltes australis* (024), *Eragrostis lacunaria* (025), *Eragrostis leptocarpa* (026), *Eragrostis sororia* (027), *Eriochloa pseudoacrotricha* (028), *Euphorbia drummondii* (029), *Evolvulus alsinoides* (030), *Fimbristylis dichotoma* (031), *Goodenia strangfordii* (032), *Grewia retusifolia* (033), *Heteropogon contortus* (034), *Hybanthus enneaspermum* (035), *Indigofera brevidens* (036), *Indigofera colutea* (037), *Leptochloa decipiens* (038), *Maireana microphylla* (039), *Melhania oblongifolia* (040), *Melinis repens* (041), *Oldenlandia corymbose* (042), *Parsonsia lanceolata* (043), *Parthenium hysterophorus* (044), *Paspalidium caespitosum* (045), *Peripleura hispidula* (046), *Perotis rara* (047), *Phyllanthus virgatus* (048), *Portulaca filifolia* (049), *Portulaca oleracea* (050), *Pseuderanthemum variabile* (051),*Setaria surgens* (052), *Sida filiformis* (053), *Sida rohlenea* (054), *Sida* sp. (055), *Sida subspicata* (056), *Sida trichopoda* (057), *Solanum ellipticum* (058), *Spermacoce* sp. (059), *Stylosanthes scabra* var. *seca* (060), *Trianthema triquetra* (061), *Wahlenbergia gracilis* (062) and *Zornia muriculata* (063).

In *E. melanophloia*, the maximum number of pasture species were again scattered near uncleared and recently cleared sites while a few species were distributed at the medium and old cleared sites (Fig 4.5). The species at uncleared and recently cleared sites showed the tendency to produce greater pasture biomass and grew on soils with higher pH_w. Medium and old cleared sites had lesser species and these species grew on soils with more NO_3-N and available P at medium, and on soils with more soil microbial biomass for N at old cleared sites (Fig 4.5).

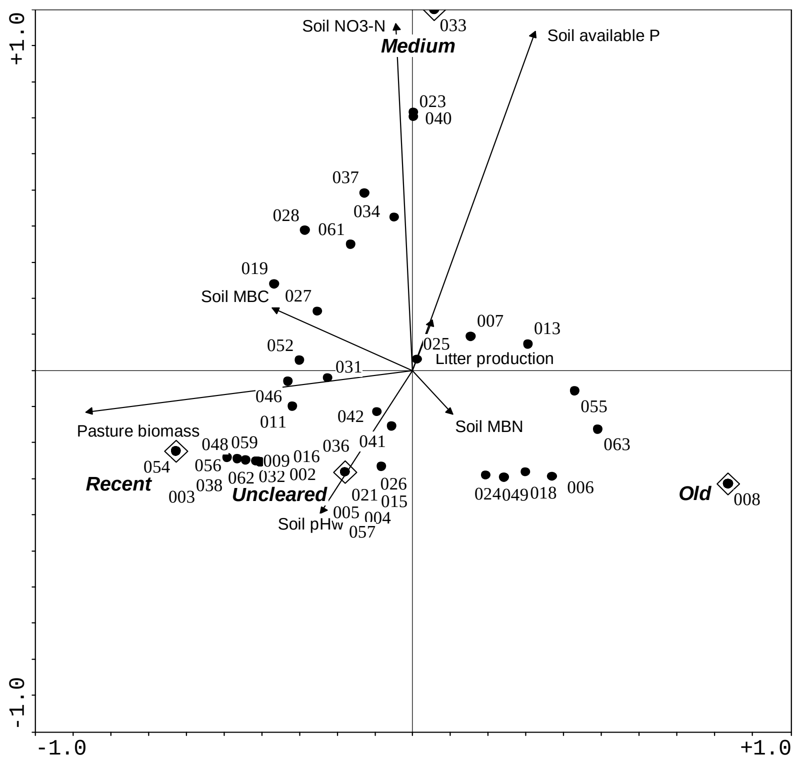


Fig 4.5. Biplot for species distribution (●) at uncleared, recent, medium and old cleared unfenced sites (◊) of *E. melanophloia* in relation to environmental variables (pasture biomass, litter production, soil NO_3-N, available P, soil microbial biomass for C (Soil MBC) and N (Soil MBN) and soil pH_w). Only most abundant species are shown. Numbers from 001-064 represent the name of species: *Alloteropsis semialata* (001), *Aristida benthamii* ((002), *Aristida calycina* (003), *Aristida leptopoda* (004), *Aristida ramosa* (005), *Bothriochloa bladhii* (006), *Bothriochloa pertusa*, (007), *Brachiaria piligera* (008), *Brunoniella australis* (009), *Calotis cuneata* (010), *Calotis cuneifolia* (011), *Carissa ovata* (012), *Cenchrus ciliaris* (013), *Chloris gayana* (014), *Chrysocephalum apiculatum* (015), *Chrysopogon fallax* (016), *Citriobatus spinescens* (017), *Cyperus dietrichiae* (018), *Cyperus gracilis* (019), *Dianella longifolia* (020), *Enneapogon gracilis* (021), *Enneapogon lindleyanus* (022), *Enteropogon acicularis* (023), *Eragrostis lacunaria* (024), *Evolovulus alsinoides* (025), *Fimbristylis dichotoma* (026), *Glycine tabacina* (027), *Glycine tomentella* (028), *Gomphorena celosiodes* (029), *Grewia retusifolia* (028), *Heteropogon contortus* (029), *Hybanthus enneaspermum* (030), *Indigofera brevidens* (033), *Indigofera linifolia* (034), *Leptochloa decipiens* (035), *Melhania oblongifolia* (036), *Melinis repens* (037), *Oxalis corniculata* (038), *Panicum effusum* (039), *Parsonsia lanceolata* (040), *Peripleura hispidula* (041), *Phyllanthus virgatus* (042), *Portulaca filifolia* (043), *Portulaca oleracea* (044), *Pterocaulon serrulatum* (045), *Rhynchosia minima* (046), *Rostellularia adscendens* (047), *Scleria mackaviensis* (048), *Sida filiformis* (049), *Sida rhombifolia* (050), *Sida rohlenae* (051), *Sida subspicata* (052), *Sida trichopoda* (053), *Spermacoce* sp. (054), *Stylosanthes scabra* var. *seca* (055), *Tephrosia brachyodon* (056), *Tephrosia supina* (057), *Teucrium corymbosum* (058), *Themeda triandra* (059), *Trianthema triquetra* (060), *Vernonia cinerarea* (061), *Vigna lanceolata* (062), *Wahlenbergia gracilis* (063) and *Zornia muriculata* (064).

Uncleared *A. harpophylla* site had the maximum spread of species compared to its cleared sites (Fig 4.6). The pasture species at uncleared *A. harpophylla* site showed the tendency to grow on soils with greater microbial biomass and litter production. Species present at recently cleared sites had a tendency to grow on soils with more NO_3-N content. However, medium and old cleared sites had similar distribution of species that showed greater pasture biomass, and the tendency to grow on soils rich in available P and with high pH_w (Fig 4.6).

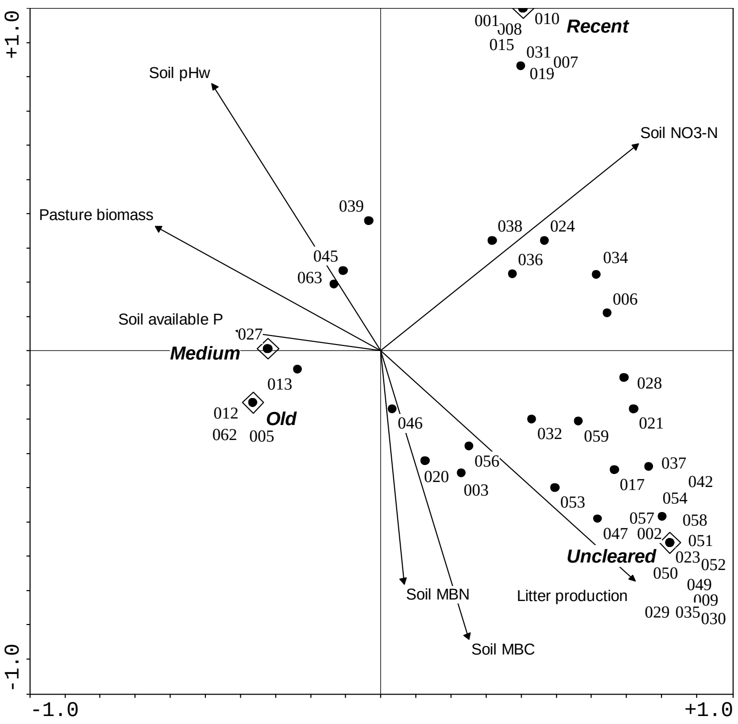


Fig 4.6. Biplot for species distribution (●) at uncleared, recent, medium and old cleared unfenced sites (◊) of *A. harpophylla* in relation to environmental variables (pasture biomass, litter production, soil NO_3-N, available P, soil microbial biomass for C (Soil MBC) and N (Soil MBN) and soil pH_w). Only most abundant species are shown. Numbers from 001-063 represent the name of species: *Acacia harpophylla* (001), *Abutilon oxycarpum* (002), *Achyranthes aspera* (003), *Ancistrache uncinulata* (004), *Aristida benthamii* (005), *Aristida calycina* (006), *Aristida caputmedusae* (007), *Aristida ramose* (008), *Brachiaria piligera* (009), *Brachyachne convergens* (010), *Brunoniella australis* (011), *Capparis lasiantha* (012), *Cenchrus ciliaris* (013), *Cheilanthes distans* (014), *Chloris gayana* (015), *Commelina arvensis* (016), *Cyperus dietrichiae* (017), *Cyperus gracilis* (018), *Dicanthium sericeum* (019), *Einadia nutans* (020), *Enchylaena tomentose* (021), *Enneapogon gracilis* (022), *Enneapogon lindleyanus* (023), *Enteropogon acicularis* (024), *Enteropogon ramosus* (025), *Enteropogon unispiceus* (026), *Epaltes australis* (027), *Eragrostis lacunaria* (028), *Eragrostis sororia* (029), *Eragrostis tenuifolia* (030), *Eriochloa pseudoacrotricha* (031), *Evolvulus alsinoides* (032), *Heteropogon contortus* (033), *Hibiscus strutii* (034), *Leptochloa ciliolate* (035), *Leptochloa decipiens* (036), *Maireana microphylla* (037), *Melhania oblongifolia* (038), *Melinis repens* (039), *Parsonsia lanceolata* (040), *Paspalidium albovillosum* (041), *Paspalidium caespitosum* (042), *Peripleura hispidula* (043), *Perotis rara* (044), *Phyllanthus virgatus* (045), *Portulaca filifolia* (046), *Portulaca oleracea* (047), *Pseudoranthemum variabile* (048), *Salsola kali* (049), *Sarcostemma viminale* (050), *Sclerolaena muricata* (051), *Sida filiformis* (052), *Sida rhombifolia* (053), *Sida rohlenea* (054), *Sida spinosa* (055), *Sida trichopoda* (056), *Spermacoce* sp. (057), *Sporobolus caroli* (058), *Stylosanthes scabra* var. *seca* (059), *Tetragonia tetragonioides* (060), *Trianthema triquetra* (061), *Vernonia cinerarea* (062) and *Wahlenbergia gracilis* (063).

b) Pasture quality

To determine the role of species diversity in pasture quality, parameters of pasture quality using NIR analysis were determined at unfenced sites for medium age cleared and uncleared *A. harpophylla*. The amount of faecal N (%) and non-grass component in pasture diet were greater at the uncleared compared to that at the cleared site (Table 4.9). However, digestibility (%) and crude protein content (%) were greater at the cleared. The data on dietary crude protein, although indicative of benefit at the cleared site, should be viewed with caution as the method used measures to an accuracy of ⌖ 1%.

Table 4.9 Pasture quality* in terms of dietary crude protein (%), faecal nitrogen (%), digestibility (%) and non-grass component (%) and accuracy of measurements (%) for each parameter (values obtained from Davies Laboratory, Qld), at medium age cleared and uncleared treatments for *A. harpophylla*.

Treatments	Dietary crude protein (%)	Faecal N (%)	Digestibility (%)	Non-Grass component (%)
Cleared	7.8	1.43	54	19
Uncleared	7.1	1.62	50	25
Accuracy	1	0.07	2	7

*Bulked analysis of 7-8 dung samples.

An alternative approach to NIR analysis of dung samples for pasture quality was to study chemical composition of pasture plants. Chemical analysis of all the pasture plants (collected for biomass measurements) in different seasons for all the cleared and uncleared sites showed greater C and N concentration at uncleared compared to cleared sites (Table 4.10). Pasture plants at cleared sites had greater P concentration than uncleared sites in both fenced and unfenced plots of all three communities except for Mar 2001, July 2001 and Mar 2002 at unfenced plots for *E. melanophloia* (Table 4.10).

There were no notable differences in C, N and P concentrations in pasture plants across different seasons. The average C, N and P concentrations over all seasons (over a year) in pasture plants of unfenced plots for all the tree communities reflected those of different seasons except that cleared and uncleared *E. melanophloia* showed no difference in average P concentration (Table 4.10).

Table 4.10 Chemical composition in terms of C, N and P concentration (%) of cleared and uncleared pasture plants available at *E. populnea*, *E. melanophloia* and *A. harpophylla* tree communities.

		E. populnea		*E. melanophloia*		*A. harpophylla*		
		Cleared	Uncleared	Cleared	Uncleared	Cleared	Uncleared	*s.e.d*
	a) Fenced plots							
C	Mar-02	39.52	43.52	38.81	40.79	40.57	43.28	*0.501*
N	Mar-02	0.58	0.77	0.69	0.52	0.54	1.08	*0.071*
P	Mar-02	0.11	0.07	0.11	0.09	0.10	0.06	*0.017*
	b) Unfenced plots							
C	Mar-01	40.17	43.17	40.75	41.12	40.56	42.07	*0.405*
	Jul-01	41.32	43.60	40.87	41.16	41.74	42.87	*0.374*
	Nov-01	41.46	43.05	40.75	39.66	41.25	43.07	*0.448*
	Mar-02	39.41	44.65	41.50	42.78	41.06	43.65	*1.553*
N	Mar-01	0.46	0.54	0.50	0.54	0.50	0.98	*0.042*
	Jul-01	0.43	0.58	0.39	0.37	0.42	0.98	*0.066*
	Nov-01	1.01	0.62	0.61	0.46	0.92	1.21	*0.133*
	Mar-02	0.50	0.55	0.51	0.55	0.57	0.90	*0.073*
P	Mar-01	0.13	0.07	0.12	0.13	0.12	0.08	*0.019*
	Jul-01	0.09	0.06	0.08	0.07	0.07	0.07	*0.009*
	Nov-01	0.13	0.06	0.11	0.05	0.13	0.06	*0.014*
	Mar-02	0.08	0.04	0.09	0.08	0.09	0.04	*0.009*
	Average for all seasons (unfenced plots)							
C		40.59	43.69	37.35	37.55	41.15	42.91	*1.763*
N		0.60	0.58	0.41	0.41	0.60	1.02	*0.062*
P		0.11	0.06	0.08	0.07	0.10	0.06	*0.013*

s.e.d. - standard error of difference of means at *P= 0.05* obtained from MANOVA (interaction tree community*cleared-uncleared treatments, total degree of freedom (df) = 17, residual df =12).

iii. Tree basal area and pasture availability at uncleared sites

Pasture availability in uncleared sites varied according to tree community. However, data were insufficient (only 9 data points; Appendix 1b) to show that this was due to the differences in basal area and tree density (data not presented here), and therefore variation between pasture biomass across uncleared sites must have been due to other factors, as discussed later.

Discussion

The argument for increase in pasture production with clearing has been generalised for a number of tree communities of central Queensland. As a result, marginal tree communities such as *E. melanophloia* have also been targeted for clearing to enhance pasture yield. The present study has indicated the specific response for pasture biomass due to clearing in each of the three communities studied. *E. populnea* and *A. harpophylla* responded with an increase in pasture biomass, with the maximum after 13 years, followed by a decline at 33 years of clearing (Fig 4.2 and 4.3). However, in *E. melanophloia* the maximum pasture biomass occurred at 5-6 years after clearing and then followed a decline from 11 years through to 33 years of clearing (Fig 4.2 and 4.3). These trends were evident in both fenced and unfenced plots. Although pasture biomass at unfenced treatments was confounded by grazing pressure, the trends in pasture biomass with time-since-clearing at unfenced sites were paralleled by data from fenced plots.

An attempt to compute average pasture biomass per year according to the amount of dry matter consumed by cattle over a year (conservative average) showed that pasture biomass did not differ between recent or old cleared and uncleared sites in *E. populnea* and *A. harpophylla*, and between medium or old cleared and uncleared sites for *E. melanophloia* (Table 4.4). The benefits of clearing for increase in quantity of pasture between cleared and uncleared treatments narrowed with increased age-since-clearing over medium to old clearing in all the tree communities. If the timeframe of the study were >33 years of time-since-clearing, and the differences for pasture yield between cleared and uncleared treatments continued to narrow, then the sustainability of cleared pastures may be questioned.

Pasture biomass showed seasonal variation with the maximum in March after summer rain (Oct-March) and the minimum in July, with very little rainfall during winter (March-July) (Fig 4.2 and 4.3). The variation in rainfall would notably affect the pasture production in a semi-arid region such as central Queensland. The period for the present study i.e. from Nov 2000 to Mar 2002, experienced normal rainfall patterns for the region (average annual rainfall at the property was 455 mm in 1997, 480 mm in 1998, 374 mm in 1999, 422 mm in 2000, 445 mm in 2001 and 367 mm in 2002). The average values obtained for pasture yield in the present study are believed to be typical of the sites, in part due to the rainfall, and in part because stocking rates were maintained constant over the past 3-10 years, without loss of liveweight gain.

The data support the earlier reported increases in pasture biomass after tree clearing, but furthermore show that the increase holds only for the initial years (a decade) after clearing. The increase in pasture with clearing has also been intimated from pasture yield and tree basal area relationships developed for various eucalypt and acacia woodlands by Burrows *et al.* (1990), Burrows (1993), Burrows (2002), Scanlan (1991), Walker *et al.* (1972) and (1986).

Walker *et al.* (1986) showed a linear relationship for increase in pasture yield with decrease in tree density in *E. crebra* woodlands. Nevertheless, other reports from tropical savannas suggest a positive role of trees in terms of shade and nutrients that enhance pasture production (Belsky 1994; Weltzin and Coughenour 1990).

Among uncleared sites, *E. melanophloia* had the maximum average amount of pasture over a year at both fenced and unfenced treatments compared to uncleared *E. populnea* and *A. harpophylla* communities (Table 4.2 and 4.4). This may be due to lesser tree density and tree basal area at *E. melanophloia* (density 1216 trees/ha, basal area 21 m^2/ha) compared to *E. populnea* (1583 trees/ha, 37 m^2/ha) and *A. harpophylla* (3622 trees/ha, 21 m^2/ha). Burrows (2002) also showed that for a given basal area, *E. melanophloia* and *E. populnea* supported more pasture yield than *A. harpophylla*. Although earlier reports by Burrows *et al.* (1988) and Burrows (1993) revealed that lesser trees reduce the trees-grass competition for soil nutrients and moisture, this conclusion was for individual tree species/communities. There was no definitive conclusion in the present study on pasture biomass and tree density/basal area relationship due to small number of data. The present data for *E. populnea* and *E. melanophloia* fitted closely the pasture yield-tree basal area relationships developed by Burrows (2002). However, this was not true for *A. harpophylla*. According to Burrows (2002), pasture biomass of uncleared *A. harpophylla* was predicted to be zero at 21 m^2/ha basal area, whereas in the present study uncleared *A. harpophylla* (with 21 m^2/ha basal area) supported *c.* 1160 kg/ha/yr pasture biomass.

The question remains - for how long can the enhanced pasture production be sustained over time after clearing? Most of the work to date has quantified the relationship between pasture yield and tree clearing for only a few years after clearing, but not over a longer period (Scanlan 2002). Short-term benefits manifested as increase in pasture production following clearing were seen in the present study. The first few years post-tree clearing, with increased pasture yield, is the most attractive phase for a producer to achieve maximum economic gains due to clearing. Release of soil nutrients upon clearing (Lawrence *et al.* 1988), absence of competition among trees and grasses for nutrients, soil moisture and light (Burrows *et al.* 1988; Belsky *et al.* 1989; Belsky 1994; Mordelet and Menaut 1995; Obot 1988) facilitated grass species establishment and capture of the newly released nutrients for growth.

However, the scenario changed after the maximum productivity obtained at 11-13 years age of clearing, as seen in Fig 4.2 and 4.3; the increased production was not maintained over time. This duration of benefit was very short for *E. melanophloia*. Assessment for benefits of clearing based on initial gains in pasture availability could be inappropriate when computed over the long term as they overestimate the benefits. Indeed, the ‘stable stage’ of pasture production over time after clearing should be considered as a standard against which to compare productivity rather than the initial phase of very high production (Scanlan 2002).

The short-term gains in pasture production due to clearing may be offset by the loss of many long-term ecological services. One such loss could be pasture species diversity. However, this was negatively correlated with pasture biomass in the present study (Fig 4.4-4.6). Functional diversity of different plant species in pasture systems may play an important role to sustain ecosystem functions (e.g. diversity of fauna, soil stability), but this study did not provide evidence for the same. Possibly if the plots had have been cleared for even longer, such an association may be evident.

No reports are available that have explored the relationship between native plant species diversity and pasture production in Australia. The ecosystem functions of diverse plant species in fact may be important in poor and old Australian soils to deliver ecological services over the longer period. The role of species diversity in pasture production studied in American and European grasslands revealed that species diversity led to a significant increase in pasture production with greater efficiency in resource use (Tilman *et al.* 1996; Hector *et al.* 1999). In the present study, native pastures with greater species diversity at uncleared *E. melanophloia* performed similarly in terms of production compared to its medium age cleared pastures. A detailed study on native species diversity and productivity in pasture systems could provide the answer for the relationship between the two in the Australian context.

The relationship between pasture plant diversity and pasture quality was examined in a grazing experiment performed at medium and uncleared sites of *A. harpophylla*. The non-grass component in cattle diet was greater at the uncleared site, whereas digestibility was greater at cleared sites that could support more beef production. Chemical composition of pasture plants indicated better diet at uncleared sites in terms of N which is essential for protein synthesis, but not the same for P which was greater in pasture plants at cleared sites (Table 4.10). The present results do not provide a definitive answer on pasture quality and species diversity relationship. A few reports are available that show higher grass nitrogen and *in vitro* dry matter digestibility in pastures with trees compared to without trees (Ash and McIvor 1998; Jackson and Ash 2001) and specifically in grasses growing under the tree canopy compared to inter-tree canopy areas (Jackson and Ash 1998).

Tree clearing for pasture production may also adversely affect the soil properties and therefore, by implication, the growth of pasture species. These issues will be discussed in detail in Chapter 5.

The gains from increase in pasture production with clearing may be present over a longer or shorter term depending upon the potential of a system, but they do not persist over time-since-clearing and may even raise the risks associated with decline in ecosystem services i.e. soil degradation, loss of faunal and floral species diversity. The benefits of tree clearing in pasture systems occur with a compromise for loss of some ecosystem services as decline in pasture plant diversity presented herein. A total value assessment for gains in pasture production and for negative impacts of clearing on soils, litter production and nutrient recycling, and other ecosystem services could provide a better understanding on benefits or loss due to tree clearing in pasture development. This will be addressed in later Chapters.

Chapter 5. Soil physico-chemical properties

Introduction

The soil is a major component of life supporting system on earth. With increasing population and the application of new techniques in agriculture to enhance agricultural produce for economic and social gains, soils have been considered as an asset to exploit. As a consequence, soil degradation has become a serious global problem.

In Australia, land degradation began with the onset of massive tree clearing during the 1950s (Conacher and Conacher 1996). Large areas of land were cleared for agriculture and pastoral development. There was greater emphasis to maximise rather than to optimise production. The National Land and Water Resources Audit (2002) stated that the inherent potential of Australian soils, that are older as well as poorer compared to the other productive soils found elsewhere in the world, has been largely misinterpreted in the past.

For the development of pasture systems in central Queensland, land is cleared and commonly sown to introduced grass species (such as *Cenchrus ciliaris* L.) that exploit the flux of nutrients released upon clearing (Partridge *et al.* 1988). This led to greater production and hence favourable economic gains for producers. Later, pasture run down with time-since-clearing was observed as a major problem in cleared/developed pastures (Lawrence *et al.* 1988; Graham 1996). The lesser availability of soil nutrients, particularly N and P, after the initial few years of clearing were believed to be the main cause for decline in pasture production (Lawrence *et al.* 1988; Graham *et al.* 1981). It has been reported that conversion of woodland pastures to cleared/developed pastures has not affected the total amount of nutrients (N and P) in the soil but has reduced their availability (Lawrence *et al.* 1988; Robertson *et al.* 1993). Woodland (*Eucalyptus populnea*) soils also did not help to increase productivity compared to that of cleared pastures (Scanlan and Burrows 1990), possibly due to tree-grass competition for nutrients and soil moisture.

The greater economic returns from increased pasture production upon clearing is a main driver for the continued clearing of land, which is still at a high rate (577,000 ha/yr during 1999-2001 (Department of Natural Resources and Mines 2003)). Clearly the producer's emphasis is on productivity rather than sustainability. As a result, associated risks for salinity, sodicity, soil erosion and structural degradation and changes in soil processes due to clearing remain largely undetermined. It is, therefore, difficult to quantify the longer-term losses/benefits of tree clearing in pasture development until these associated risks accounted for. The research presented in this Chapter thus emphasised:

- Comparative soil status between cleared and uncleared (native woodland) pastures to quantify the long-term consequences of clearing in developed pasture systems
- Trends in soil properties over time since clearing
- Search for factors (availability of nutrients, physical properties or the interaction of both) that are responsible for decline in pasture production with time-since-clearing
- Variation in soil status with depth

The impact of clearing could vary according to soil and vegetation type. In the present study, various physico-chemical properties of soils were studied for cleared and uncleared pasture systems. The cleared pastures were compared with uncleared controls in three major types of native tree communities of central Queensland at three different scales of age since clearing. The selection of paired sites for cleared and uncleared treatments for each tree community minimized variation in soil type. The nutrient status was studied down the soil profile, contrasting with previous studies (Graham *et al.* 1981 and 1985; Lawrence *et al.* 1988) where only the top (0-10 cm mostly) depth was sampled. It was important to assess the variation in soil status with depth as changes in deep layers may have greater potential to alter soil functions (Brown and Lugo 1990). A recent report by Page *et al.* (2002) showed an accumulation of ammonium-nitrogen at deeper layers (1.2-3 m) in old cleared cropping soils (70 years) compared to minimal accumulation in native *Acacia harpophylla* stands.

Different time-since-clearing treatments provided information to interpret the long-term consequences of clearing, if they exist. The inferences drawn from different tree communities for changes in soil properties will be useful to predict the specific response for each community and could play a role in supporting future decisions on tree clearing. They will also help to estimate whether the gains from increased productivity upon clearing are offset by losses of soil properties.

Materials and methods

Study sites

Three types of tree dominant communities, each at three different times-since-clearing with their respective uncleared controls, were selected in central Queensland (Table 5.1).

Table 5.1. Paired treatments for each of *E. populnea*, *E. melanophloia* and *A. harpophylla*.

Treatments	**Time since clearing**
Cleared	**Recent (5 years)**
Uncleared/intact	Never cleared
Cleared	**Medium (11-13 years)**
Uncleared/intact	Never cleared
Cleared	**Old (33 years)**
Uncleared/intact	Never cleared

Each tree community was associated with a specific soil type - *E. populnea* on sandy, *E. melanophloia* on loamy and *A. harpophylla* on alluvial clay soils. Full details are presented in Chapter 3.

At each site an area 100 m x 100 m was marked for data collection. All the cleared and uncleared treatments were selected as pairs of sites, at *c*. 50 m distance from each other. This approach helped to minimize variability in soil type between pairs. After clearing, however the cleared sites experienced different treatments (fire, sowing and blade ploughing) from their paired uncleared controls. Post-clearing and grazing management practices, including grazing pressure, would have confounded the impact of clearing on soil properties. The details for various post-clearing treatments and grazing pressure at each site are given in Chapter 3 - Table 3.1.

Soil sampling and processing

Preliminary soil samples were collected to determine soil variability to calculate the number of samples required at each site. Initially nine samples were taken for 0-10 cm depth and analysed for total carbon concentration. Based on the calculations:

$N = 4\ \sigma /E^2$ where N= number of samples, σ = standard deviation, E = acceptable error (taken as 10 per cent)

eight samples were required per site. Eight was taken as a standard number for all parameters at each depth.

Soil sampling using a hydraulic soil corer, was carried out in Jan 2002 randomly at different distances and directions from a central point in each plot. Each sample was targeted for a meter deep core (4 cm diameter) but most were restricted between 60-80 cm depth due to hard soil profile at deeper layers that was variable according to vegetation type. Each core was segmented into 0-5, 5-10, 10-20, 20-30, 30-60 and 60-100 cm and stored in plastic containers under normal conditions. Samples were dried at 40^{o} C and prepared according to the normal standards for soil analysis. All eight samples from a site were bulked for each depth. A sub-sample from bulked sample for each site was dried at 105^{o} C to determine bulk density. Visible roots and pebbles were removed during processing.

In the present study, samples to 60 cm depth were analysed, as the availability of 60-100 cm samples was not consistent across sites. The samples were analysed for pH in $CaCl_2$ (referred as pH_{Ca}), pH in water extract (referred as pH_w), EC (electrical conductivity), CEC (cation exchange capacity), bulk density, exchangeable cations (Ca, K, Mg and Na), macro- (available and total N and P, organic carbon) and micro- (Cu, Zn, Fe and Mn) nutrients. The details of methods used in soil analysis are mentioned in Appendix 2.

Statistical analysis

The main effects (fixed terms) of tree community (*E. populnea*, *E. melanophloia* and *A. harpophylla*), cleared (recent, medium, old age of clearing) and uncleared treatments, and soil depth (each for 0-5, 5-10, 10-20, 20-30 and 30-60 cm depth) plus their interactions were analysed for each variable applying residual maximum likelihood test (REML - Repeated Measures) using Genstat ver. 6.0. For this analysis, all the uncleared treatments of a tree community were treated as replicates, while cleared treatment for each age group of a tree community was taken as such without any replication. The correlation within a treatment across different depths was analysed for all the main effects by applying different variance models (Antedependence order 1 (AD 1), Antedependence order 2 (AD 2), Power (with and without allowing for heterogeneity) or Unstructured). The adequacy of a variance model was assessed from deviance for a given degrees of freedom. AD 2 (Antedependence order 2) was the most appropriate model for all the variables. Hence, the data were analysed for main effects (tree type, cleared and uncleared, and soil depth) plus their interactions using AD 2 variance model for correlation within a treatment across the soil depth from 0 to 60 cm.

REML analysis for soil variables provided various two- and three-way levels of interactions but only significant interactions ($P < 0.05$) were considered. Non-significant interactions were dropped from the fixed term, and the main effects for each individual term (tree type, or cleared and uncleared, or soil depth) were analysed. The data revealed significant ($P < 0.05$) three-way interactions (tree community*cleared-uncleared*depth) for main effects in all the variables except EC where only two-way interactions (tree community*cleared-uncleared and tree community*depth) were significant.

For each variable, a sed (standard error of differences) matrix (obtained from REML analysis) for all the treatment effects was used to calculate LSDs (least significant differences of means) to compare cleared (recent, medium, old) and uncleared treatments for each specific depth in a particular tree type. All the cleared (recent, medium and old) and uncleared comparisons were made according to an exact LSD value for each treatment. However, for convenience in interpretation only the average values of LSDs between any cleared and uncleared, any two cleared or any two uncleared treatments are presented. The means from REML analysis were used to present the results.

Results

Various time-since-clearing (recent, medium and old age of clearing) and uncleared treatments were compared to evaluate the impact of tree clearing for a given time frame on soil physico-chemical properties in *E. populnea, E. melanophloia* and *A. harpophylla*. Soil status was studied with profile for different depths (0-5, 5-10, 10-20, 20-30 and 30-60 cm). Cleared and uncleared treatments of a tree community were compared for each depth. REML analysis revealed three way significant interactions for the main effects of tree community, time-since-clearing and depth for all parameters except EC for which only two-way interactions were significant.

Physico-chemical properties

Bulk density increased with depth in all the tree communities (Fig 5.1.1a, b and c). The old cleared sites were more compacted at the shallow (0-5 cm) depth than the recent and medium cleared, and uncleared sites. Among all the cleared treatments, soil compaction was relatively less at recently cleared than the medium or old cleared sites with few exceptions. The degree of compaction in *E. melanophloia* was less at recent clearing compared to the uncleared site (Fig 5.1.1b).

Soil pH_{Ca} increased with depth from 30 to 60 cm in all the tree types (Fig 5.1.2a, b and c). The top 0-5 cm showed significantly greater pH_{Ca} in all the cleared compared to uncleared sites for *E. populnea* and *A. harpophylla* (Fig 5.1.2a and c). A significant effect of clearing for increase in pH_{Ca} at 30-60 cm depth occurred at old cleared compared to uncleared sites for all the tree types. There was no definite trend with age since clearing except in *A. harpophylla* where pH_{Ca} increased with age since clearing at the 30-60 cm depth (Fig 5.1.2c). Increased pH (>7.5) at medium clearing for *E. populnea* and old clearing at *E. melanophloia* and *A. harpophylla,* could result from greater concentrations of Na, Ca and Mg and may lead to micronutrient deficiencies. This will be discussed later.

There was a common response for change in electrical conductivity with time-since-clearing across different depths in each tree type (Fig. 5.1.3) (the interaction between time-since-clearing and soil depth was not significant). Clearing significantly increased EC at recent, but not at medium and old cleared sites in *E. populnea* and *A. harpophylla* (Fig 5.1.3). The response of different tree types with depth (in the absence of a significant interaction with time-since-clearing) showed greater EC from 20 to 60 cm depth in *A. harpophylla,* followed by *E. populnea* and *E. melanophloia* (Fig 5.1.4).

Fig 5.1.1 Bulk density

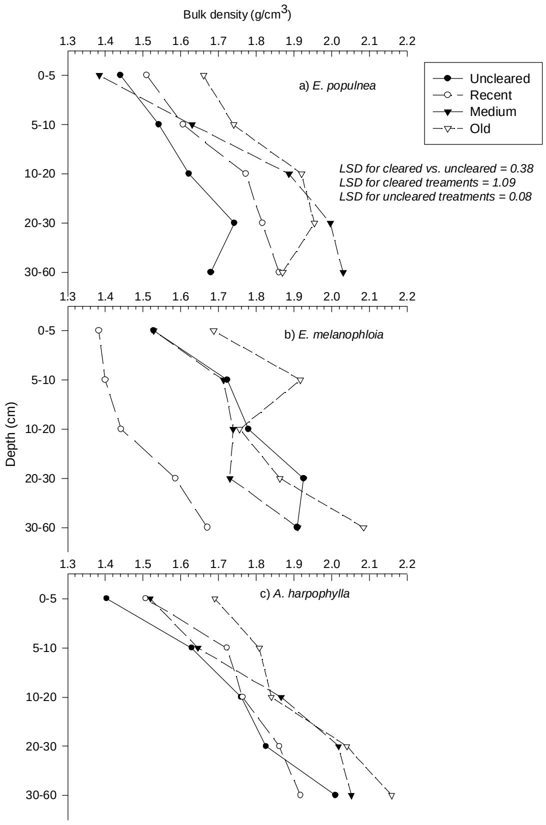


Fig 5.1.2. Soil pH$_{Ca}$

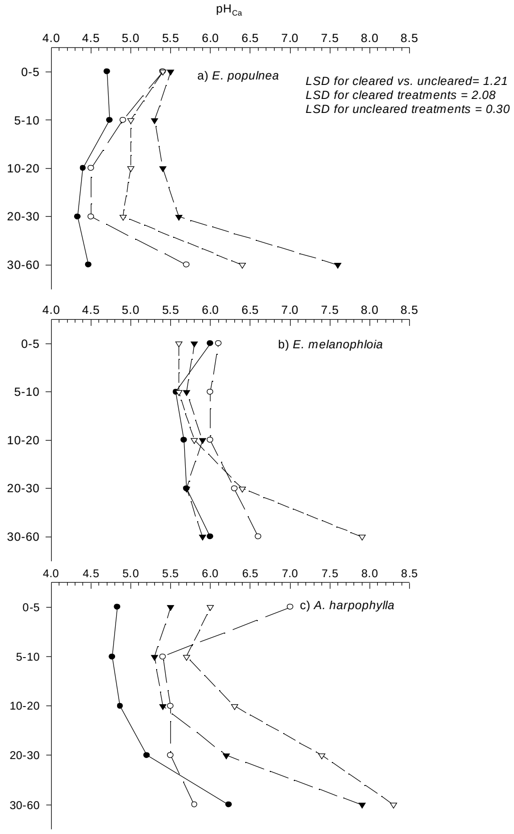


Fig. 5.1.3. EC for cleared and uncleared treatments

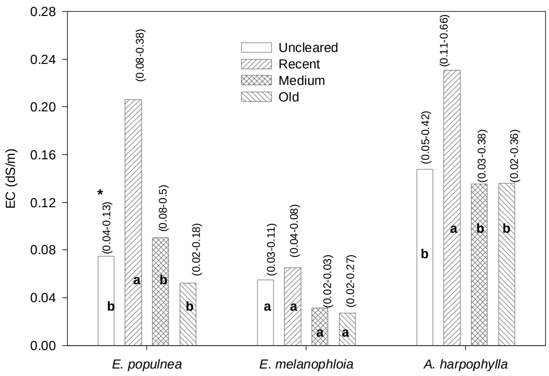


Fig. 5.1.4. EC for tree communities in relation to depth

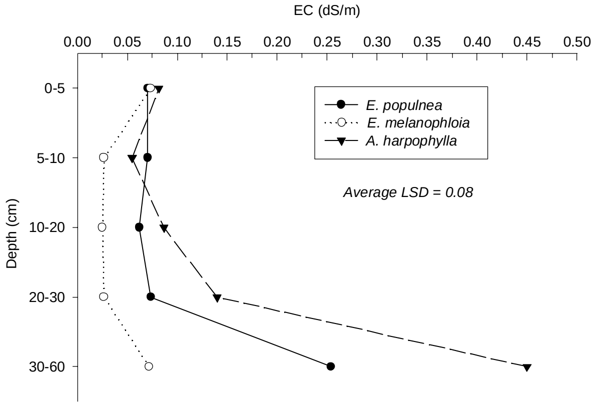


Fig 5.1. Soil bulk density (1), pH$_{Ca}$ (2) for a) *E. populnea*, b) *E. melanophloia* and c) *A. harpophylla* communities, and electrical conductivity (EC-for cleared and uncleared treatments in relation to age of clearing (3) and for different tree communities in relation to depth (4) (in absence of a significant three-way interaction)) for various cleared and uncleared treatments of three tree communities.

LSD denotes the least significant difference of means between any cleared and. uncleared or any two cleared or any two uncleared treatments at P<0.05.

*In Fig 5.1.3. - Figures in parentheses denote the range of EC for a treatment and different letters on bars denote significant difference between cleared and uncleared treatments within a tree community at P<0.05.

Macronutrients

Tree clearing did not exhibit a significant effect on soil organic carbon except at 30-60 cm depth where clearing at recent age had a greater concentration than the uncleared in *E. populnea* and than the medium (11-13 years) clearing in *A. harpophylla* (Fig 5.2.1a, b and c). The soil organic carbon concentration declined sharply with depth at all sites. A similar effect of clearing was seen on total and available amount of P with no significant difference between cleared and uncleared sites for *E. populnea* and *A. harpophylla* (Fig 5.2.2 and 5.2.3- a and c), whereas in *E. melanophloia,* the total and available P was exceptionally high at medium cleared than the recent and old cleared treatments (Fig 5.2.2 and 5.2.3 b). This was due to the presence of basalt in the parent rock. The large amount of total and available P in uncleared *E. melanophloia* soil was also due to basalt in the parent rock (uncleared site paired for medium age of clearing had a basaltic parent rock and thus increased the average for uncleared *E. melanophloia*).

Total N also declined with increase in soil depth in line with soil organic carbon, without a significant impact of clearing other than a greater concentration at recent age since clearing compared to medium clearing for top 0-5 and bottom 30-60 cm depth in *A. harpophylla,* and to uncleared *E. populnea* for 10-30 cm depth (Fig 5.2.4a, b and c). The available N in the form of nitrate (NO_3-N) was highly variable in the top 0-10 cm depth. It increased due to clearing at recently cleared sites in *E. populnea* and *A. harpophylla,* and was less at old cleared sites (Fig 5.2.5a and c), whereas, the medium cleared site had greater NO_3-N than the uncleared *E. melanophloia* for 0-5 cm depth (Fig 5.2.5b). With clearing, a significant increase occurred in ammonium-N concentration in the top 0-5 cm and bottom 30-60 cm soil depth at recent age of clearing in *E. populnea* and *A. harpophylla* (Fig. 5.2.6a and c).

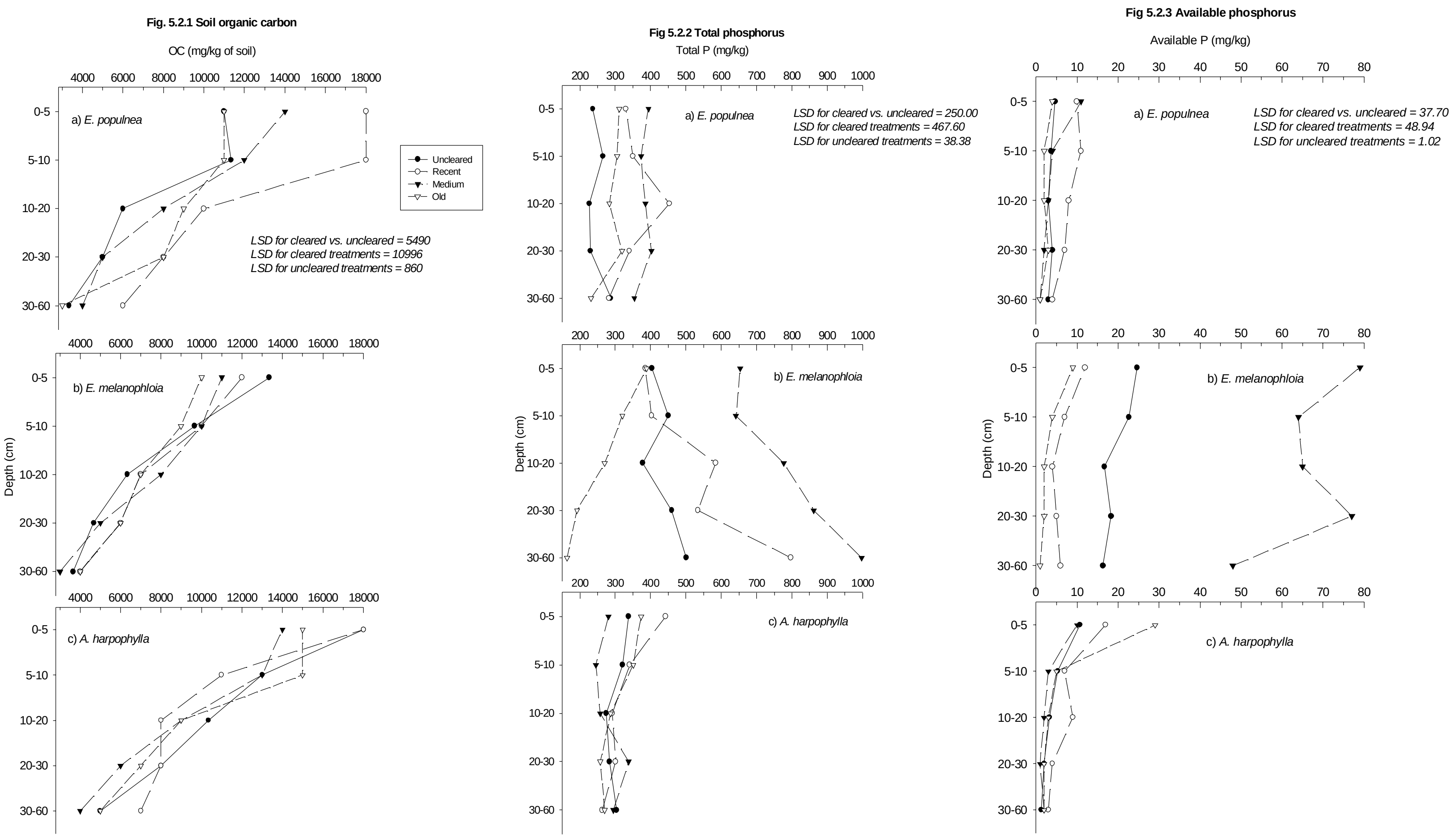


Fig 5.2. Soil organic carbon, and total and available - P at various cleared and uncleared treatments for a) *E. populnea*, b) *E. melanophloia* and c) *A. harpophylla* communities. *…contd.*

LSD denotes the least significant difference of means between any cleared and. uncleared or any two cleared or any two uncleared treatments at P<0.05.

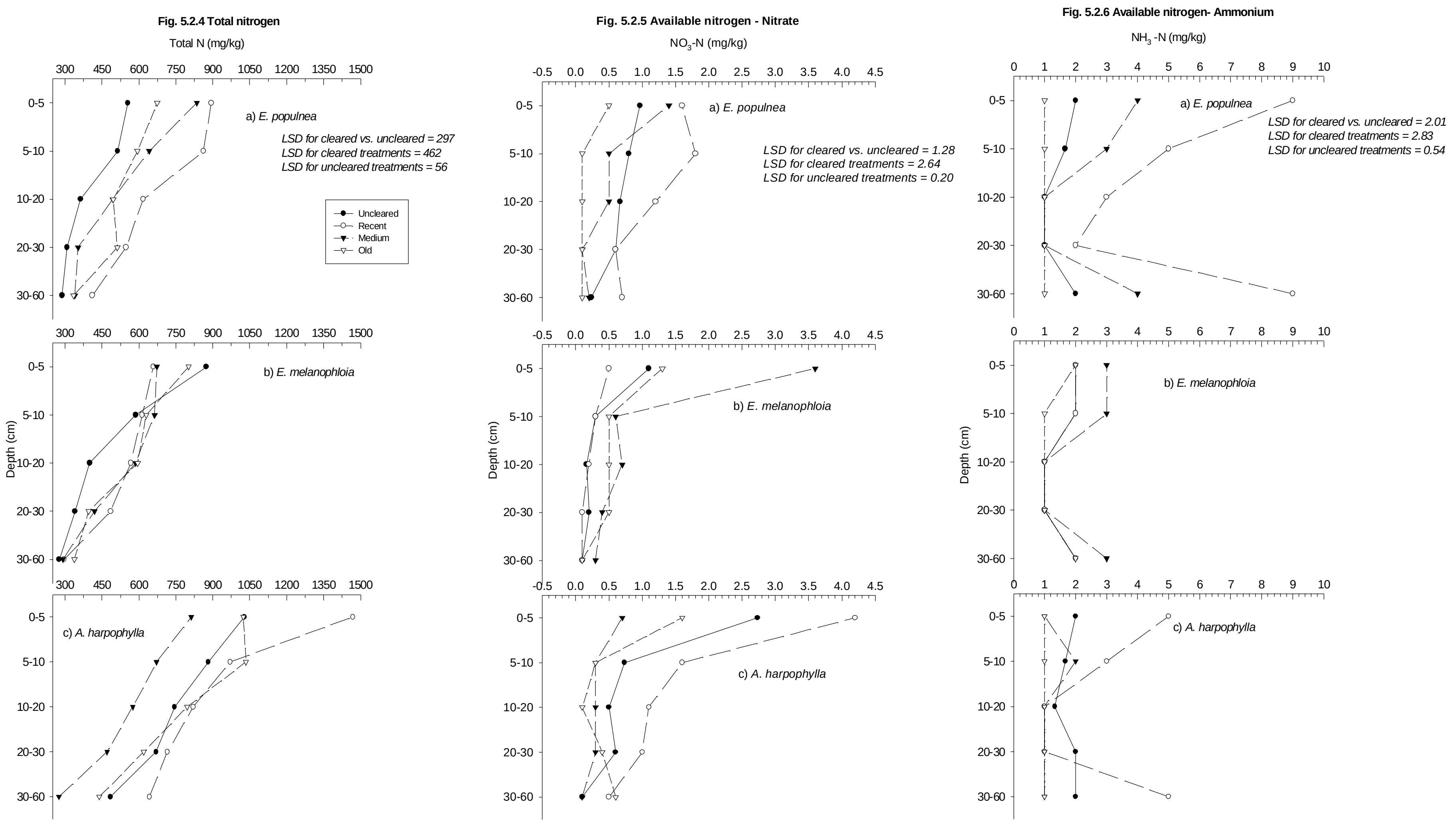


Fig. 5.2.4 Total nitrogen

Fig. 5.2.5 Available nitrogen - Nitrate

Fig. 5.2.6 Available nitrogen- Ammonium

LSD for cleared vs. uncleared = 297
LSD for cleared treatments = 462
LSD for uncleared treatments = 56

LSD for cleared vs. uncleared = 1.28
LSD for cleared treatments = 2.64
LSD for uncleared treatments = 0.20

LSD for cleared vs. uncleared = 2.01
LSD for cleared treatments = 2.83
LSD for uncleared treatments = 0.54

Fig 5.2. Soil total and available - N at various cleared and uncleared treatments for a) *E. populnea*, b) *E. melanophloia* and c) *A. harpophylla* communities.

LSD denotes the least significant difference of means between any cleared and. uncleared or any two cleared or any two uncleared treatments at P<0.05.

Micronutrients

There were few conspicuous differences due to clearing or time-since-clearing in Cu, Fe, Zn and Mn concentrations in *E. populnea*, *E. melanophloia* and *A. harpophylla* soils (Fig. 5.3). The amount of copper was at a maximum at old age of clearing in all the three tree types, but significant impact was evident only at 20-30 and 30-60 cm in *E. populnea* (Fig 5.3.1a, b and c). Clearing increased Zn concentrations in soils at 20 to 60 cm depth at recent and medium in *E. populnea* and at recent cleared sites in *A. harpophylla*, compared to their uncleared sites (Fig 5.3.2a and c). For 0-5 cm, Fe concentration increased at recent age of clearing in *E. populnea* while it decreased in the oldest cleared *A. harpophylla* compared to their uncleared treatments (Fig 5.3.3a, b and c). Clearing led to an increase in Mn concentrations in *E. populnea* as evident in Fig. 5.3.4a. There was no definitive trend for change in micronutrient concentrations with soil depth except for 30-60 cm where the amount of Cu, Fe and Mn decreased and Zn increased in all the tree types with only a few exceptions (Fig 5.3).

Fig. 5.3.1 Cu

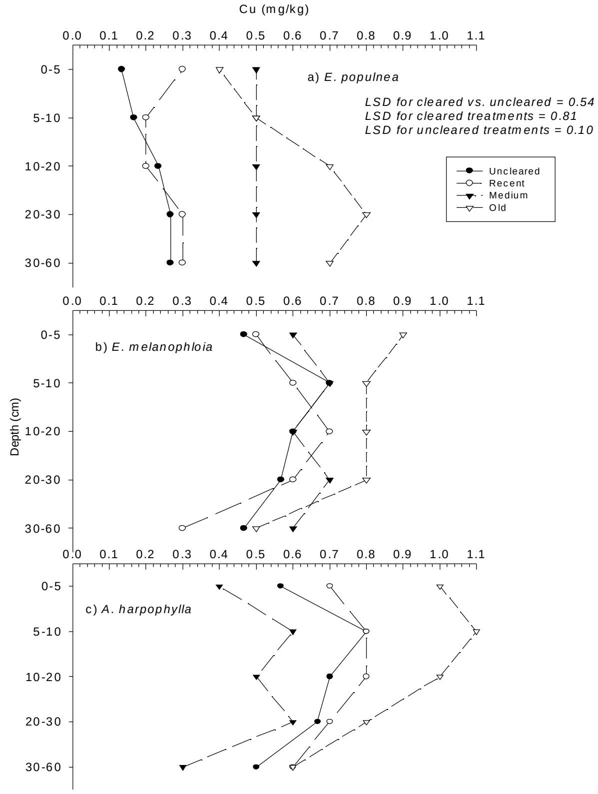


Fig 5.3.2. Zn

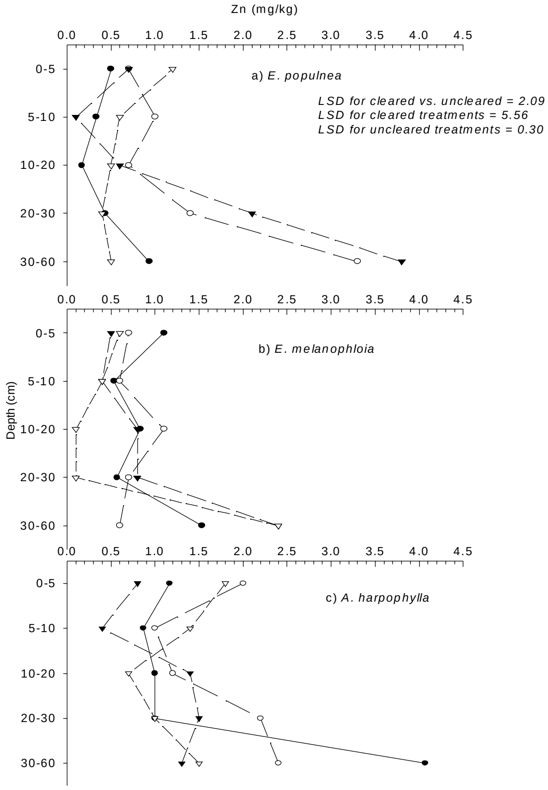


Fig. 5.3.3. Fe

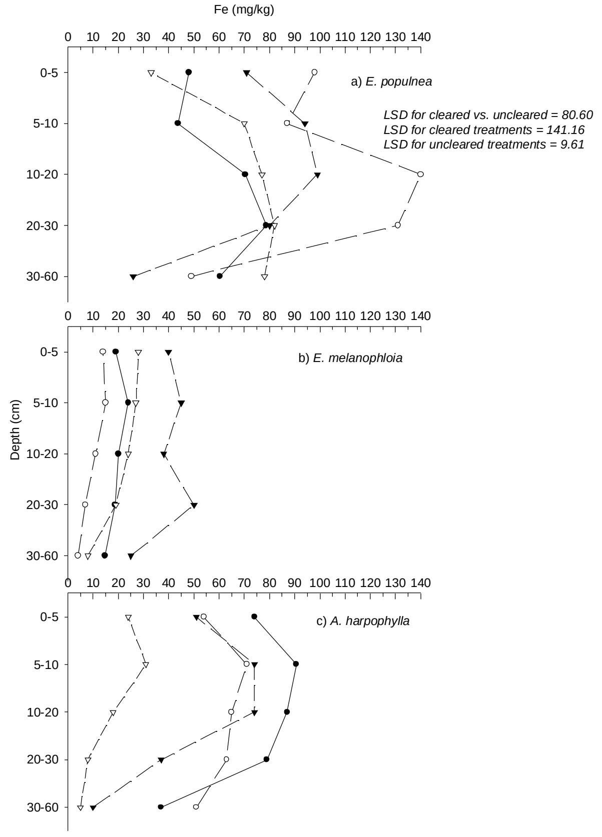


Fig. 5.3.4. Mn

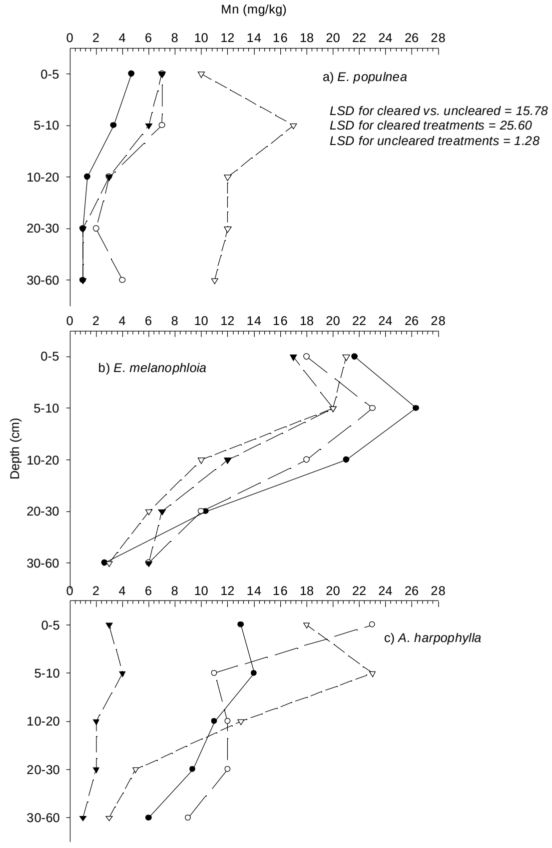


Fig 5.3. Soil micronutrients Cu, Zn, Fe and Mn at various cleared and uncleared treatments for a) *E. populnea*, b) *E. melanophloia* and c) *A. harpophylla* communities.

LSD denotes the least significant difference of means between any cleared and. uncleared or any two cleared or any two uncleared treatments at P<0.05

Exchangeable cations

Exchangeable Ca and Mg concentrations increased with clearing in *E. populnea* and *E. melanophloia* (Fig 5.4.1 and 5.4.2 - a and b). At 10 to 60 cm, Ca concentrations increased significantly at old cleared compared to uncleared sites for *E. populnea, A. harpophylla,* and at recent cleared compared to uncleared sites for *E. melanophloia*. Mg showed an increase with depth at all sites (Fig 5.4.2 a, b and c). It increased at recent age of clearing at 30-60 cm depth compared to uncleared sites in *E. populnea*. Ca/Mg ratio showed a decline with depth due to the greater Mg concentration present at depth (Fig 5.4.3a, b and c) at all the sites.

There was a profile wide increase in exchangeable K concentration with clearing at recently cleared sites for *E. populnea* and *E. melanophloia* (Fig 5.4.4a, and b). But such an increase was evident only for 0-5 cm depth in *A. harpophylla*. The trend for increase with soil depth as seen in Mg was also apparent for exchangeable Na (Fig 5.4.5a, b and c). At 30-60 cm, Na concentration was greater at all the cleared sites in *E. populnea*, and at old clearing in *E. melanophloia*, compared to their respective uncleared sites. The greater than double concentrations of Na at depth indicated that clearing may tend to cause sodicity in *E. populnea* and *E. melanophloia* communities. This effect was also apparent with estimations of exchangeable sodium percentage (ESP). The oldest clearing in *E. melanophloia* had significantly greater ESP compared to the uncleared site at 30-60 cm depth (Fig 5.4.6b). ESP showed a general trend for increase with depth at all the sites. The analysis also showed that uncleared *E. populnea* and *A. harpophylla* soils had inherently large values of ESP in the subsoil (30-60 cm depth) whereas *E. melanophloia* had inherently lower values of ESP but increase in ESP at old clearing perhaps was the result of clearing (Fig 5.4.6a, b and c).

Cation exchange capacity (CEC) was greater at all the cleared compared to uncleared sites with significant differences (at $P<0.05$) at 30-60 cm depth in all the tree types except medium clearing for *A. harpophylla* (Fig 5.4.7a, b and c). All the cleared and uncleared sites showed an increase in CEC from 30 cm to 60 cm depth except for recently cleared *A. harpophylla*.

Fig. 5.4.1 Calcium

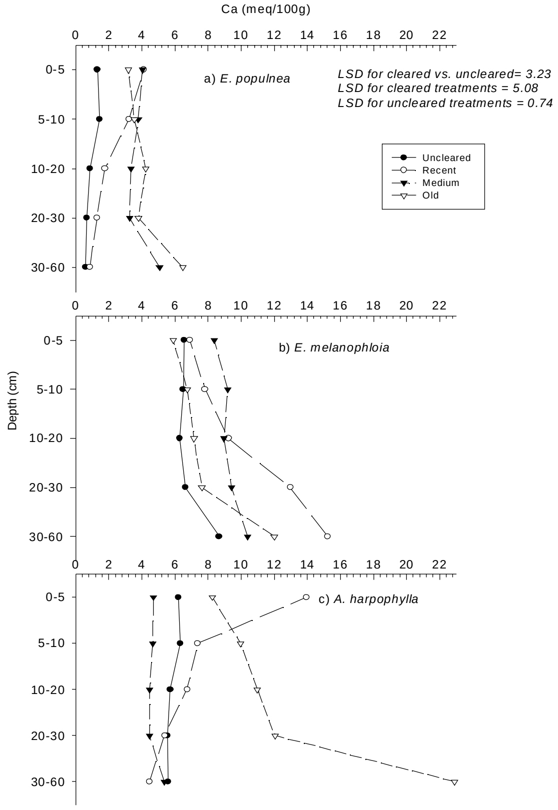


Fig. 5.4.2 Magnesium

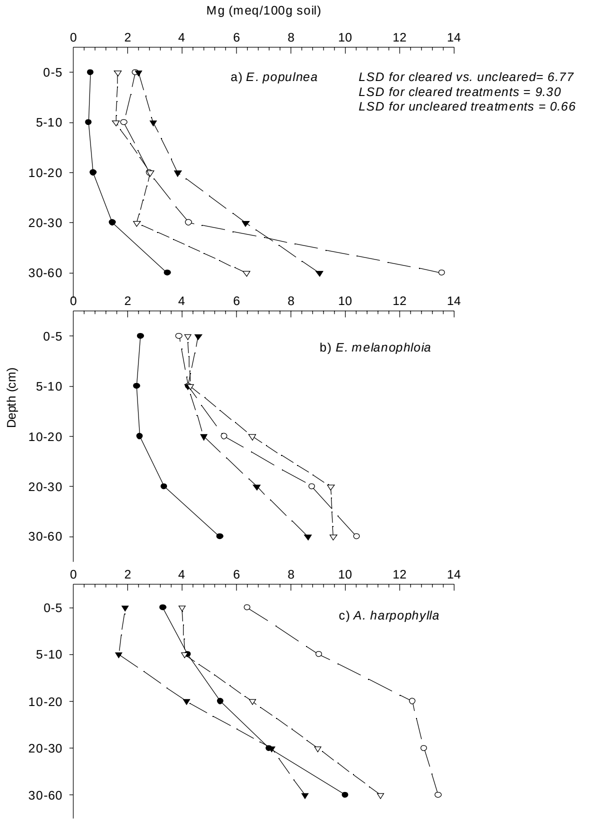


Fig. 5.4.3 Calcium / Magnesium ratio

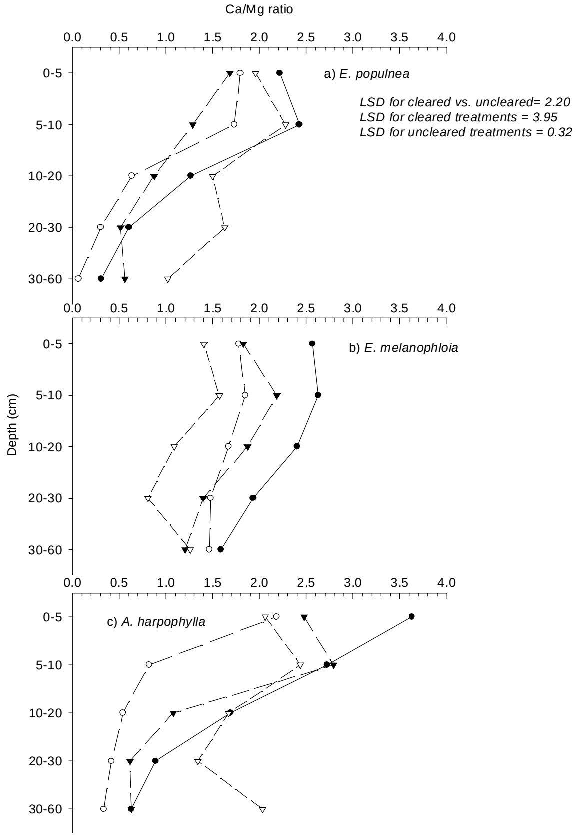


Fig. 5.4.4 Potassium

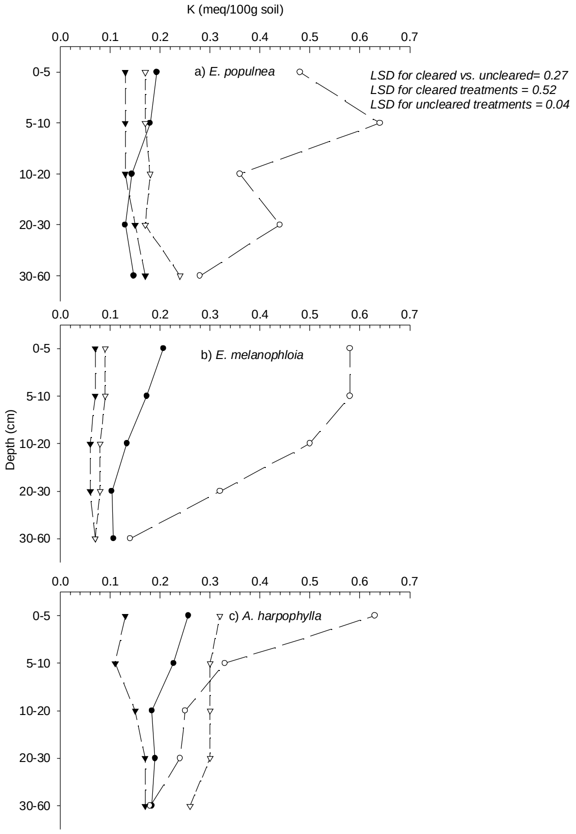


Fig 5.4. Soil exchangeable cations Ca and Mg, Ca/Mg ratio and K at various cleared and uncleared treatments for a) *E. populnea*, b) *E. melanophloia* and c) *A. harpophylla* communities. ....contd.

LSD denotes the least significant difference of means between any cleared and. uncleared or any two cleared or any two uncleared treatments at P<0.05

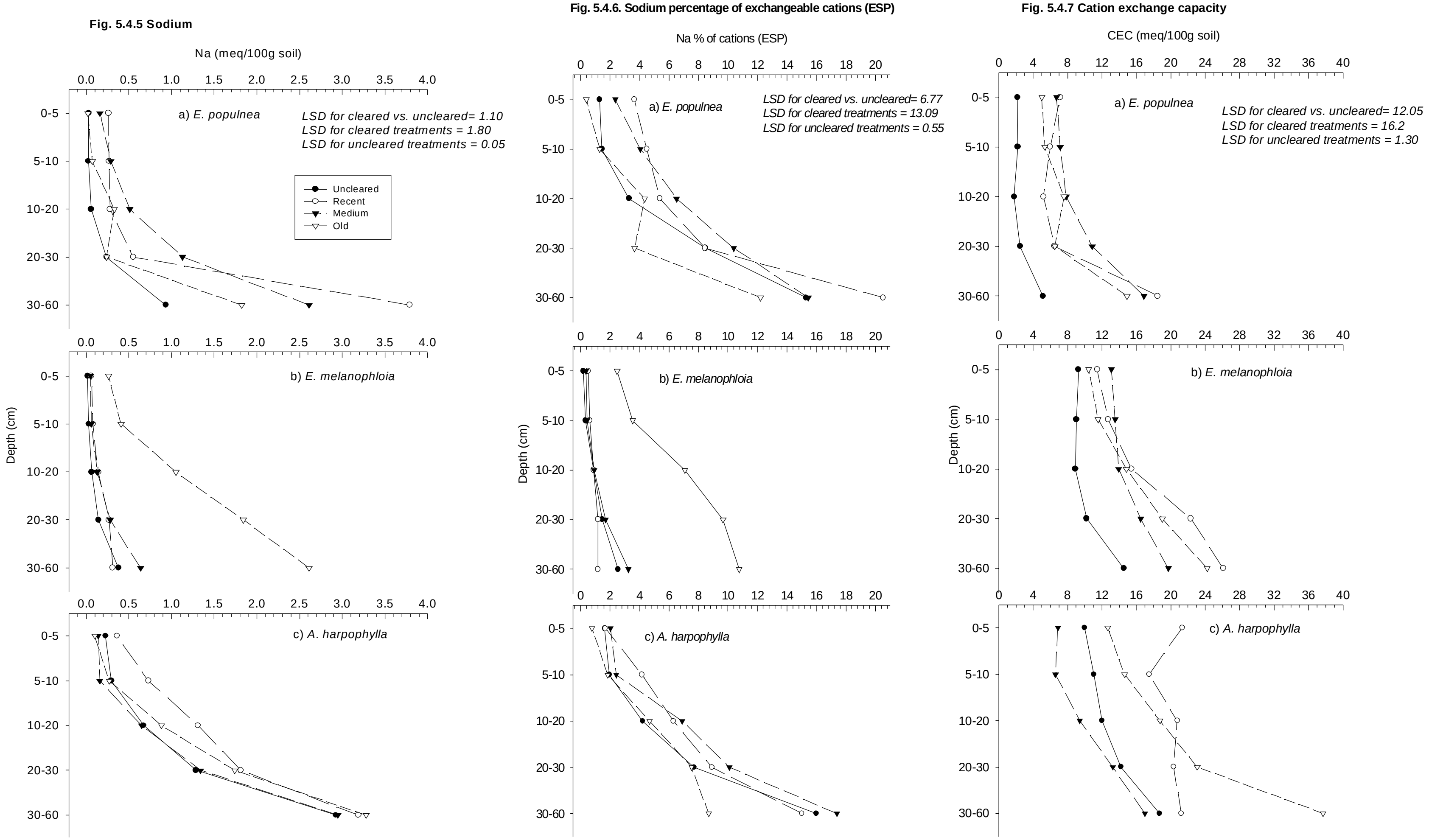


Fig 5.4. Soil exchangeable cation Na, sodium percentage of exchangeable cations (ESP) and CEC at various cleared and uncleared treatments for a) *E. populnea*, b) *E. melanophloia* and c) *A. harpophylla* communities.

LSD denotes the least significant difference of means between any cleared and. uncleared or any two cleared or any two uncleared treatments at P<0.05

Discussion

The concentrations of nutrients in the soil are of interest, but their interactions among themselves as well as with the surrounding environment are of great significance for plant growth. Nutrient availability and soil moisture are the major factors that control primary production in grasslands of semi-arid environments (Scholes 1993). In these, water acts as a primary limiting factor to determine plant growth. In the presence of sufficient soil moisture, availability of soil nutrients acts as the secondary limiting factor that controls growth rate. Thus, both the soil moisture and the nutrient availability regulate duration and rate of growth. In the present study, soil moisture was not measured due to difficulty in inserting the delicate Hydrosense probes in hard soils. However, Yates *et al.* (2000) reported for pasture systems in WA that grazing led to an increase in soil compaction and reduced infiltration which further affected the pasture growth.

Soil nutrient availability for pasture growth is an important concern in pasture systems of central Queensland and is discussed in detail in this chapter. Within a year of clearing a dramatic increase in available N (700%), P (225%) and K (70%) was evident compared to uncleared sites in *A. harpophylla* (Partridge *et al.* 1988). Fisher *et al.* (1994) and Neil *et al.* (1997) (cited by Post and Kwon 2000) also reported that clearing native tropical savannas or forests for pasture development led to increase in soil organic carbon.

The results of the present study also showed either non-significant or greater amounts of macro- and micro-nutrients at cleared compared to uncleared sites for *E. populnea*, *E. melanophloia* and *A. harpophylla* for the top soil layer, with only a few exceptions (Fig.5.1, 5.2, 5.3 and 5.4). The nutrient release would have been greater at the time of clearing, as some has already been taken up by plant growth with age of clearing. The newly sown grass species such as *C. ciliaris* have very fine roots and hence are efficient at capturing soil nutrients released upon clearing (Converys 1999). This explains the greater amount of pasture above-ground biomass at cleared compared to uncleared sites (pasture biomass data presented in Figures and Tables 4.2 and 4.3 in Chapter 4). Removal of trees may eliminate tree-grass competition for soil nutrients and help enhance pasture yield.

The competition between trees and grasses for soil moisture and available nutrients at uncleared sites appears to contribute to lesser pasture yield compared to cleared sites. The higher root:shoot and root length:root biomass ratios in grass species provide competitive advantage over tree to acquire nutrients (Wilson 1998 cited in Schmidt and Lamble 2002), especially in the upper soil layers. Most of the native grasses (species of *Themeda, Aristida*, *Paspalidium, Eragrostirs* and *Dicanthium*) growing under tree canopies use C4 pathway for photosynthesis, hence tree-plant competition for light is likely to occur but it could not be severe due to the open woodland structure. Soil moisture seems to be a major limiting factor in semi-arid climates, such as central Queensland (Scholes, 1993; Burrows, B. personal communication on 20th May 2003). Hence, the competition for soil moisture and soil nutrients in addition to solar radiation between grass species and trees, limits pasture production under natural tree stands. Similar arguments are given by Burrows *et al.* (1988).

The soil nutrients declined with increase in depth at all the sites with a few exceptions. Similar results were also reported by Graham (1978), and Brown and Lugo (1990). Most of the previous studies conducted in the region (Graham *et al.* 1981; Lawrence *et al.* 1993; Robertson *et al.* 1993) presented data for the top 0-10 cm depth and showed increased amount of nutrients with clearing, which was not evident at deeper layers (Fig 5.1-5.4).

In the present study, at 30-60 cm, most of the cleared (old and medium) sites had lesser concentrations of nutrients especially soil organic carbon, total P, available P and ammonium-N compared to uncleared sites in each tree community (Fig 5.2). However, to some extent the effects of clearing on nutrient concentrations were influenced by grazing and post-clearing treatments. Since grazing took place at all the sites within a smaller range of grazing pressure (1 head/3 ha/yr - 1 head/5ha/year) and almost similar rates at paired sites, its effect could be considered less important than post-clearing practices that were superimposed upon recent, medium and old cleared sites. The scenario would have changed as a result of different management practices e.g. blade ploughing at medium and old cleared sites. It was reported by Graham *et al.* (1985) that blade ploughing help to renovate pastures. Hence blade ploughing at old and medium clearing would have contributed to the improvement of nutrient status at these sites. Since the soil status in medium and old clearing (increased pH, Na, and no notable increase in available N and P) was not better than that of uncleared plots, yet those cleared treatments were blade ploughed, it is likely that without blade ploughing soils would have been even poorer.

Increased nutrient concentration (in the top soil layer) with clearing provides an important option to increase pasture production, and for this reason clearing is practised in the region. But the lesser amount of pasture biomass at the oldest cleared compared to medium (*E. populnea* and *A. harpophylla*), and recent (*E. melanophloia*) sites (Figures and Tables: 4.2 and 4.3 in Chapter 4) may occur due to diminished availability of nutrients (due to greater extraction by pastures) and/or change in ecosystem functions.

Partridge *et al.* (1988) reported that even the fertile cleared soils of *A. harpophylla* showed decline in nutrient availability 5-7 years after clearing. Most studies (Lawrence *et al.* 1988; Graham 1996; Burrows 1993) explained pasture run-down as being due to non-availability of nitrogen and phosphorus through immobilisation of these nutrients in soil microbial biomass, and not because of their reduced amounts in the soil. Later Robertson *et al.* (1997) related the reduced availability of nitrogen in pasture soils to immobilization largely in soil organic matter, clay, plant material and to a lesser extent in soil microbial biomass. According to these hypotheses, a large amount of soil organic reserve would be expected at old cleared pastures. In contrast, the present work showed that soil organic carbon was lesser or the same at old cleared pastures compared to other treatments (Fig 5.2.1a, b and c). Similar declines in soil microbial biomass will be discussed in Chapter 6. There seem to be other reasons for soil and pasture run down that could include:

- Associated changes in soil physical properties that could affect nutrient availability
- Trends in these changes over time-since-clearing

The availability of a nutrient with respect to recommended pH_w range (Truog 1948) was assessed for a common sample date at all the sites (data for pH_w not for pH_{Ca} was used since the availability range for a particular nutrient exists only for pH_w, and pH_w and $pH_{Ca,}$were strongly correlated with each other (r = 0.94, $P<0.05$). There are other factors (soil moisture, temperature, interactions among various nutrients) besides pH that affect the availability of nutrients in soils (Peverill *et al.* 1999). However, the pH_w was taken as an indicator since the data showed significant differences in pH_w with time-since-clearing. The availability of nutrients (not the levels) in relation to pH_w at cleared and uncleared sites of each tree community are discussed in detail in the following paragraphs.

In *E. populnea,* macro- and micro-nutrients, and exchangeable cations were the most favourable for plant growth at medium cleared site at pH_w 6.3 (Table 5.2), that also supported the maximum pasture productivity. At 30-60 cm, pH_w increased with time-since-clearing thus lessening the nutrient availability, which would have reduced pasture biomass at the oldest clearing. The system may take some time to equilibrate with the changes before having an impact on pasture yield. The higher concentration of Fe at 30-60 cm at old cleared site indicated ferrolysis (a process that could lead to structural degradation that occurs due to native vegetation clearing in Australia (Peverill *et al.* 1999)), and as a consequence may contribute to pasture run down.

Table 5.2 Availability of various macro- and micro-nutrients (data from Fig 5.1-5.4) according to the range prescribed by Truog (1948), and rating levels of EC (electrical conductivity), ESP (exchangeable sodium percentage) and CEC (cation exchange capacity) for 0-10 cm (mean of 0-5 and 5-10 cm depths) and 30-60 cm soil depths, and pasture biomass - at uncleared, and cleared (recent, medium and old) treatments for *E. populnea*$^{\$}$.

	0-10 cm soil depth				30-60 cm soil depth			
Parameters	Uncleared	Recent	Medium	Old	Uncleared	Recent	Medium	Old
pH_w	5.78^{c}	5.70^{c}	6.30^{b}	6.45^{a}	5.60^{b}	6.60^{ab}	8.10^{a}	7.20^{ab}
EC (dS/m)#	0.04^{b} (VL)	0.14^{a} (VL)	0.08^{b} (VL)	0.02^{b} (VL)	0.13^{b} (VL)	0.38^{a} (L)	0.50^{a} (M)	0.18^{b} (L)
NH_3-N (mg/kg)	1.83^{c}	7.00^{a}	$\mathbf{3.50^{b}}$	$\mathbf{1.00^{c}}$	2.00^{c}	$\mathbf{9.00^{a}}$	$\mathbf{4.00^{b}}$	$\mathbf{1.00^{c}}$
NO_3-N (mg/kg)	0.88^{a}	1.70^{a}	$\mathbf{0.95^{a}}$	$\mathbf{0.30^{a}}$	0.23^{b}	$\mathbf{0.70^{a}}$	$\mathbf{0.20^{b}}$	$\mathbf{0.10^{b}}$
Available P (mg/kg)	$\underline{4.17^{a}}$	$\underline{10.50^{a}}$	$\mathbf{7.50^{a}}$	$\mathbf{3.00^{a}}$	$\underline{3.00^{a}}$	$\mathbf{4.00^{a}}$	1.00^{a}	$\mathbf{1.00^{a}}$
Cu (mg/kg)	$\mathbf{0.15^{a}}$	$\mathbf{0.25^{a}}$	$\mathbf{0.50^{a}}$	$\mathbf{0.45^{a}}$	$\mathbf{0.27^{b}}$	0.30^{b}	$\underline{0.50^{ab}}$	0.70^{a}
Fe (mg/kg)	$\mathbf{45.84^{b}}$	$\mathbf{92.50^{a}}$	$\mathbf{82.50^{ab}}$	51.50^{b}	$\mathbf{60.33^{a}}$	49.00^{a}	$\underline{26.00^{a}}$	$\underline{78.00^{a}}$
Zn (mg/kg)	$\mathbf{0.42^{a}}$	$\mathbf{0.85^{a}}$	$\mathbf{0.40^{a}}$	$\mathbf{0.90^{a}}$	$\mathbf{0.93^{b}}$	3.30^{a}	$\underline{3.80^{a}}$	0.50^{b}
Mn (mg/kg)	$\mathbf{4.00^{a}}$	$\mathbf{7.00^{a}}$	$\mathbf{6.50^{a}}$	$\mathbf{13.50^{a}}$	$\mathbf{1.00^{b}}$	4.00^{b}	$\underline{1.00^{b}}$	11.00^{a}
Ca (meq/100 soil)	$\underline{1.38^{b}}$	$\underline{3.67^{a}}$	3.91^{a}	3.37^{ab}	$\underline{0.61^{b}}$	0.87^{b}	$\mathbf{5.07^{a}}$	$\mathbf{6.48^{a}}$
K (meq/100g soi)	0.19^{a}	0.56^{a}	$\mathbf{0.13^{a}}$	$\mathbf{0.17^{a}}$	0.15^{b}	$\mathbf{0.28^{a}}$	$\mathbf{0.17^{ab}}$	$\mathbf{0.24^{ab}}$
Mg (meq/100g soil)	$\underline{0.59^{a}}$	$\underline{2.08^{a}}$	2.67^{a}	1.60^{a}	3.46^{b}	13.54^{a}	$\mathbf{9.04^{ab}}$	$\mathbf{6.36^{b}}$
ESP (%)	1.39^{ab}	4.07^{a}	3.22^{ab}	0.85^{b}	15.28^{a}*	20.51^{a}*	15.45^{a}*	12.22^{a}*
CEC (meq /100 g soil) #	2.19^{a} (L)	6.57^{a} (M)	6.94^{a} (M)	5.18^{a} (M)	5.15^{b} (M)	18.48^{a} (H)	16.89^{ab} (H)	14.90^{ab} (H)
Pasture biomass (kg/ha/yr)	2107^{b}	2817^{ab}	3792^{a}	2748^{ab}				

\# ( ) letters in a row represent rating (Bruce and Rayment 1982): VL: very low, L: low, M: medium, H: high.
Bold figures represent availability of a nutrient at optimum pH range recommended by Truog (1948).
Non-bold figures represent availability at wider range of pH_w where availability is less than at optimum pH_w range.
Underlined figures represent non-availability of a nutrient at given pH (beyond the availability range recommended by Truog (1948)).
* represents levels of ESP (Exchangeable Sodium Percentage of cations), >6 indicates sodicity.
$^{\$}$ different superscripts in a row represent the significance levels of difference at $P < 0.05$ after REML analysis.

In *E. melanophloia*, pH_w and nutrient availability did not differ much between cleared and uncleared sites for 0-10 cm (Table 5.3). The nutrient availability was at a maximum at the recent cleared site that also had greatest pasture biomass compared to other sites. At 30-60 cm, the increased pH_w at the oldest cleared site could adversely affect the plant uptake of micro-nutrients, that leads to decline in pasture. Clearing led to a significant increase in ESP

(Exchangeable Sodium Percentage of cations) at the oldest site which indicated sodicity and that as a result deteriorates soil physical conditions and affects plant growth.

Table 5.3 Availability of various macro- and micro-nutrients (data from Fig 5.1-5.4) according to the range prescribed by Truog (1948), and rating levels of EC (electrical conductivity), ESP (exchangeable sodium percentage) and CEC (cation exchange capacity) for 0-10 cm (mean of 0-5 and 5-10 cm depths) and 30-60 cm soil depths, and pasture biomass - at uncleared, and cleared (recent, medium and old) treatments for *E. melanophloia*$^{\$}$.

	0-10 cm soil depth				30-60 cm soil depth			
Parameters	Uncleared	Recent	Medium	Old	Uncleared	Recent	Medium	Old
pH_w	6.68^{a}	7.10^{a}	6.70^{a}	6.90^{a}	7.10^{b}	7.40^{ab}	7.20^{ab}	8.80^{a}
EC (dS/m)#	0.07^{a} (L)	0.07^{a} (L)	0.03^{a} (L)	0.02^{a} (L)	0.03^{a} (VL)	0.05^{a} (VL)	0.03^{a} (VL)	0.27^{a} (L)
NH_3-N (mg/kg)	**2.00^{a}**	**2.00^{a}**	**3.00^{a}**	**1.50^{a}**	**2.00^{a}**	**2.00^{a}**	**3.00^{a}**	2.00^{a}
NO_3-N (mg/kg)	**0.70^{b}**	**0.40^{b}**	**2.10^{a}**	**0.90^{ab}**	**0.10^{a}**	**0.10^{a}**	**0.30^{a}**	0.10^{a}
Available P (mg/kg)	**23.67^{b}**	**9.50^{b}**	**71.50^{a}**	**6.50^{b}**	**16.33^{ab}**	**6.00^{b}**	**48.00^{a}**	**1.00^{b}**
Cu (mg/kg)	0.58^{a}	0.55^{a}	0.65^{a}	0.85^{a}	0.47^{a}	0.30^{a}	0.60^{a}	0.50^{a}
Fe (mg/kg)	21.50^{a}	14.50^{a}	42.50^{a}	27.50^{a}	14.67^{a}	4.00^{a}	25.00^{a}	8.00^{a}
Zn (mg/kg)	0.82^{a}	0.65^{a}	0.45^{a}	0.50^{a}	1.53^{a}	0.60^{a}	2.40^{a}	2.40^{a}
Mn (mg/kg)	24.00^{a}	20.50^{a}	18.50^{a}	20.50^{a}	2.67^{a}	6.00^{a}	6.00^{a}	3.00^{a}
Ca (meq/100 soil)	6.53^{a}	**7.37^{a}**	8.79^{a}	6.33^{a}	**8.67^{b}**	**15.23^{a}**	**10.39^{b}**	**12.01^{a}**
K (meq/100g soi)	**0.19^{b}**	**0.58^{a}**	**0.07^{b}**	**0.09^{b}**	**0.11^{b}**	**0.14^{a}**	**0.07^{b}**	**0.07^{b}**
Mg (meq/100g soil)	2.40^{a}	**4.06^{a}**	4.40^{a}	4.25^{a}	**5.39^{a}**	**10.41^{a}**	**8.62^{a}**	**9.55^{a}**
ESP (%)	0.26^{b}	0.58^{b}	041^{b}	3.02^{a}	2.55^{b}	1.19^{b}	3.25^{b}	10.77^{a}*
CEC (meq/100 g soil) #	9.15^{a} (M)	12.08^{a} (M)	13.32^{a} (M)	11.00^{a} (M)	14.55^{b} (M)	26.09^{a} (H)	19.72^{ab} (M)	24.24^{a} (H)
Pasture biomass (kg/ha/yr)	3761^{b}	6335^{a}	3743^{b}	2816^{b}				

( ) letters in a row represent rating (Bruce and Rayment 1982): VL: very low, L: low, M: medium, H: high.
Bold figures represent availability of a nutrient at optimum pH range recommended by Truog (1948).
Non-bold figures represent availability at wider range of pH_w where availability is less than at optimum pH_w range.
Underlined figures represent non-availability of a nutrient at given pH (beyond the availability range recommended by Truog (1948)).
* represents levels of ESP (Exchangeable Sodium Percentage of cations), >6 indicates sodicity.
$^{\$}$ different superscripts in a row represent the significance levels of difference at $P < 0.05$ after REML analysis.

The nutrient availability for cleared and uncleared *A. harpophylla* was maximum at medium age of clearing which was matched by the greater pasture growth than at recent and old cleared, and uncleared sites (Table 5.4). At 30-60 cm, the pH_w increased from recent to old cleared sites and reduced the nutrient availability, and as a result lesser pasture productivity was evident at old cleared site.

Table 5.4 Availability of various macro- and micro-nutrients (data from Fig 5.1-5.4) according to the range prescribed by Truog (1948), and rating levels of EC (electrical conductivity), ESP (exchangeable sodium percentage) and CEC (cation exchange capacity) for 0-10 cm (mean of 0-5 and 5-10 cm depths) and 30-60 cm soil depths, and pasture biomass - at uncleared, and cleared (recent, medium and old) treatments for *A. harpophylla*$^{\$}$.

	0-10 cm soil depth				30-60 cm soil depth			
Parameters	Uncleared	Recent	Medium	Old	Uncleared	Recent	Medium	Old
pH$_w$	5.80^{c}	7.30^{a}	6.50^{b}	6.90^{ab}	6.93^{bc}	6.20^{c}	8.60^{ab}	9.10^{a}
EC (dS/m) #	0.05^{b} (VL)	0.17^{a} (L)	0.03^{b} (VL)	0.03^{b} (VL)	0.42^{b} (L)	0.66^{a} (M)	0.38^{b} (L)	0.36^{b} (L)
NH_3-N (mg/kg)	1.83^{b}	$\mathbf{4.00^{a}}$	$\mathbf{1.50^{b}}$	$\mathbf{1.00^{b}}$	$\mathbf{2.00^{b}}$	$\mathbf{5.00^{a}}$	1.00^{b}	1.00^{b}
NO_3-N (mg/kg)	1.73^{ab}	$\mathbf{2.90^{a}}$	$\mathbf{0.50^{b}}$	$\mathbf{0.95^{ab}}$	$\mathbf{0.10^{b}}$	$\mathbf{0.50^{a}}$	0.10^{b}	0.60^{a}
Available P (mg/kg)	8.00^{a}	$\mathbf{12.00^{a}}$	$\mathbf{6.50^{a}}$	$\mathbf{17.00^{a}}$	$\mathbf{1.33^{a}}$	3.00^{a}	$\mathbf{2.00^{a}}$	$\mathbf{2.00^{a}}$
Cu (mg/kg)	$\mathbf{0.68^{a}}$	0.75^{a}	$\mathbf{0.50^{a}}$	1.05^{a}	0.50^{a}	$\mathbf{0.60^{a}}$	0.30^{a}	0.60^{a}
Fe (mg/kg)	$\mathbf{82.34^{a}}$	62.50^{ab}	62.50^{ab}	27.50^{b}	37.00^{a}	$\mathbf{51.00^{a}}$	10.00^{a}	5.00^{a}
Zn (mg/kg)	$\mathbf{1.02^{ab}}$	1.50^{ab}	$\mathbf{0.60^{b}}$	1.60^{a}	4.07^{a}	$\mathbf{2.40^{a}}$	1.30^{a}	1.50^{a}
Mn (mg/kg)	$\mathbf{13.50^{a}}$	17.00^{a}	$\mathbf{3.50^{a}}$	20.50^{a}	6.00^{ab}	$\mathbf{9.00^{a}}$	1.00^{b}	3.00^{b}
Ca (meq/100 soil)	6.27^{ab}	10.67^{a}	4.68^{b}	9.11^{ab}	5.59^{b}	4.46^{b}	5.35^{b}	22.90^{a}
K (meq/100g soil)	0.24^{a}	$\mathbf{0.48^{a}}$	$\mathbf{0.12^{a}}$	$\mathbf{0.31^{a}}$	$\mathbf{0.18^{a}}$	$\mathbf{0.18^{a}}$	$\mathbf{0.17^{a}}$	$\mathbf{0.26^{a}}$
Mg (meq/100g soil)	3.75^{a}	7.72^{a}	1.79^{a}	4.05^{a}	9.99^{a}	13.41^{a}	8.50^{a}	11.28^{a}
ESP (%)	1.80^{a}	2.93^{a}	2.23^{a}	1.32^{a}	16.00^{a}**	15.02^{a}**	17.38^{a}**	8.70^{b}*
CEC (meq/100 g soil) #	10.53^{ab} (M)	19.41^{a} (H)	6.74^{b} (L)	13.65^{a} (H)	18.69^{b} (H)	21.24^{b} (H)	16.97^{b} (H)	37.72^{a} (H)
Pasture biomass (kg/ha/yr)	1823^{b}	3030^{ab}	4259^{a}	2803^{ab}				

( ) letters in a row represent rating (Bruce and Rayment 1982): VL: very low, L: low, M: medium, H: high.

Bold figures represent availability of a nutrient at optimum pH range recommended by Truog (1948).

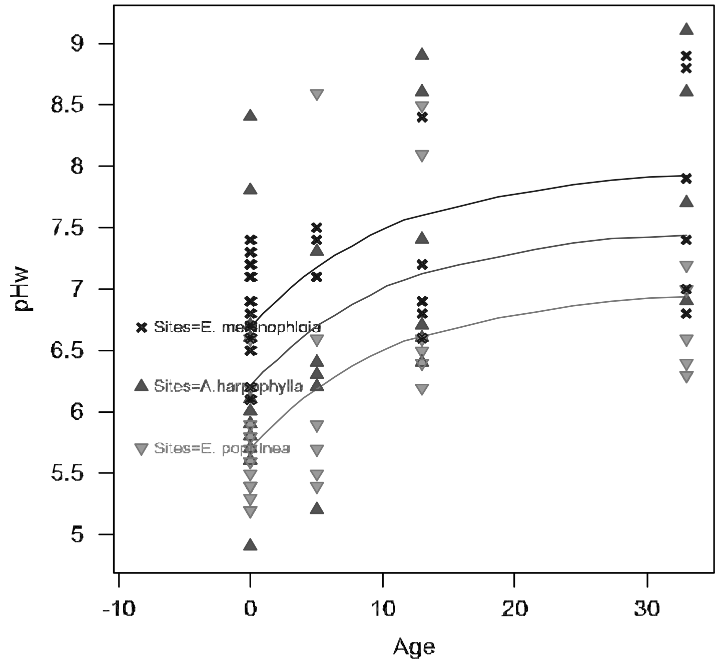
Non-bold figures represent availability at wider range of pH$_w$ where availability is less than at optimum pH$_w$ range.

Underlined figures represent non-availability of a nutrient at given pH (beyond the availability range recommended by Truog (1948)).

*represents levels of ESP (Exchangeable Sodium Percentage of cations), (<6 - non sodic, and >14 strongly sodic).

$^{\$}$ different superscripts in a row represent the significance levels of difference at $P < 0.05$ after REML analysis.

There was a remarkable increase in pH_w with time-since clearing for all tree types (Fig 5.5) that affected the nutrient availability as discussed above, and that would have contributed to pasture run-down over time. Similar results for either increase or no change in soil pH with clearing in pasture systems were also observed over Queensland in a soil carbon project (Harms, B. personal communication on 12th May 2003). The change in pH was strongly correlated (at $P<0.05$) with Ca (r = 0.72), Mg (r = 0.53) and CEC (r = 0.74). An increase in pH of cropping soils was also reported by Dalal and Mayer (1986) to be highly correlated with Na and Mg concentrations. The increased pH as a consequence of clearing in Amazonian forests occurred in association with an increase in concentrations of base cations such as Ca, Mg and Na (McGrath *et al.* 2001).



For all data:
$Y = a+(-1.29)^{(-0.041*\text{time-since-clearing})}$
$R^2 = 0.42$ at $P < 0.01$

Where intercepts (a) for:
E. populnea = 6.990
E. melanophloia = 7.976
A. harpophylla = 7.494

The intercepts showed significant difference at $P<0.01$ but not the slopes, between tree communities.

Fig 5.5 Soil pH_w at uncleared (0) and recent (<5 yrs), medium (11-13 yrs) and old (>30 yrs) cleared sites for *E. populnea*, *E. melanophloia* and *A. harpophylla* (graph reproduced from Genstat).

Changes in pH are due to reactions that occur in soil solution, in which various biotic and abiotic factors play an important role. Soil microclimate, microbial activity, interaction among various ions and physical factors exert a direct significant impact on nutrient availability for plant growth and through their effect on soil pH. This intricate network is responsible for the soil stability, nutrient availability and subsequently the plant productivity. The question is whether the intricate network can be maintained by herbaceous vegetation in the absence of trees or whether trees play a different role from herbaceous vegetation in maintaining various soil functions. In Queensland, Schmidt and Lamble (2002) presented two scenarios on nutrient relation in pasture systems:

- A sustainable (cleared) system with stable nutrient content where herbaceous vegetation would perform the same ecosystem functions as trees, to maintain soil stability. Pasture run-down is not expected in this system.

- An unsustainable system where trees perform different ecosystem functions to the herbaceous vegetation and play an important role in nutrient cycling. Thus over the medium- and long term, removal of trees would result in nutrient and pasture run-down.

Based on literature, evidence suggests that trees provide a stable environment (shade, litter, stable hydrological cycle, recycling of nutrients, substrate for various soil microbial activities) that results in improved physical and chemical conditions of soil (Vetaas 1992). Williams *et al.* (1993) suggested that tree clearing influences and disturbs the equilibrium of soil processes (nutrient recycling, decomposition), and results in changes in various soil functions as the system adjusts to the change. The impact of clearing on soil biological properties and litter production and nutrient release will be discussed in the later Chapters.

It is concluded from the present study that concentrations of nutrients in soils increase with clearing but they become less available (necessarily not the nutrient levels, but their availability) with age of clearing. This suggests that trees perform different functions to the herbaceous vegetation, in support of the second scenario (above) for an unsustainable system, as proposed by Schmidt and Lamble (2002). The changes in soil characteristics at depth (such as increased pH and sodium concentrations in old cleared *E. melanophloia*; increased pH in medium and old cleared, and ammonium at recent cleared *E. populnea* and *A. harpophylla*) indicate the long-term consequences of clearing. Tree clearing also led to increase in concentrations of cations (Ca, Mg, K and Na - Fig 5.4) due to disturbance that disintegrates soil particles, and hence exchangeable cations become free in soil solution and could be more exposed to discharge (Wild 1989). The free cations may react with H_2O and release H^+ which could further reduce Fe^{3+} (ferric ion) to increase Fe^{2+} concentration (as occurred at cleared sites for *E. populnea* and *E. melanophloia*) and OH^- ions that could lead to an increase in pH. These reactions are complex but increased pH could be a result of such changes in soil solution. The increased NH_4^+ concentrations but not the NO_3-N at depth in *E. populnea* and *A. harpophylla* showed that mineralisation of NH_4^+ could be a problem due to increased pH as Robertson (1989) explained the strong negative effect of high pH on N mineralisation. A recent study in southern Queensland by Page *et al.* (2002) also reported similar results for accumulation of ammonium at depth due to the absence of nitrifying bacteria, which are unable to colonise because of change in pH. All the changes in soil solution that affect soil processes, are complicated but the present results have demonstrated the impact of such changes, and their effect on the soil environment.

The changes in ecosystem functions due to change in vegetation with clearing (such as soil microclimate, environment for soil microbes, nutrient recycling, litter composition) are most likely to impact negatively on soil processes and lead to pasture run-down with time-since-clearing.

Chapter 6. Soil biological properties (respiration and microbial biomass)

Introduction

An effect of any disturbance on soils is rapidly manifested as a change in soil biological properties which represent the living component of soil, mainly soil micro- and macro-organisms and roots. Soil respiration and microbial biomass are the common measures for soil biological activity.

Soil respiration represents the amount of CO_2 evolved from soil microbes and roots, and to a lesser extent by oxidation of root exudates, plant detritus and humified organic matter (Raich and Schlesinger 1992). Microbial biomass typically constitutes <5 percent of soil organic matter (Jenkinson and Ladd 1981) and is a major contributor to total soil respiration. Microbial activity and /or soil respiration depends upon quantity and quality of organic matter (Giardiana and Ryan 2000). Thus vegetation plays an important role in total soil and microbial respiration since it is one determinant of microbial composition and biomass (Kutsch and Dilly 1999).

Any change in vegetation may affect soil biological properties by influencing soil microclimate, the quality and quantity of litter or detritus material, soil organisms and availability of their substrate, and the root structure (Raich and Tufekcioglu 2000).

At the global scale, the net effect of land use change (mostly conversion of forests to agricultural land) contributed about 2.1 Pg C during the year 2000 to the atmosphere which includes estimates for vegetation removal and its decomposition, regrowth and changes in soil carbon (Houghton and Hackler 2002). The annual flux of carbon from soils to the atmosphere is estimated at 76.5 Pg C per yr (about 10 per cent of total CO_2 emissions to atmosphere) (Raich and Potter 1995). Soil contains 1500 Pg of C, about twice the amount that is in the atmosphere (Eswaran *et al.* 1993). Thus any change in land use that affects soil respiration is of major concern for global climate warming (Raich and Tufekcioglu 2000).

Soil respiration rate differs with climate, soil and vegetation type, but temperature and vegetation type are the two major factors. On a global scale, each year, forests and savannas alone contribute about 42 Pg of C while temperate grasslands, tundra, desert, cultivated and other ecosystems only the 18 Pg C to the total emissions of about 60 Pg C of respiration (both of vegetation and of microbial decomposition of organic matter) (global estimates vary from 60-75 Pg C) (Grace and Rayment 2000). Soil respiration is about 20 percent greater in grasslands than forest (Raich and Tufekcioglu 2000), and about 1.42 times greater in the tropical (629 g C/m^2/yr) than in the temperate grasslands (442 g C/m^2/yr) (Raich and Schlesinger 1992). Therefore, clearing tropical woodlands is likely to result in a net increase in CO_2 emissions. Most of the detailed recent studies (Boone *et al.* 1998, Giardiana and Ryan 2000, Valentini *et al.* 2000), and the previous studies as Raich and Schlesinger (1992) stated, are conducted in temperate zones, with little knowledge of soil respiration rate and its sensitivity to temperature in the semi-arid and tropical ecosystems. No such reports exist for Queensland.

Conversion of woodlands to open/cleared pastures is a common practice in Queensland. Changes in soil nutrient status due to clearing have been reported (Lawrence *et al.* 1993; Graham *et al.* 1981) but the effects of clearing on soil biological properties, such as soil respiration and soil microbial biomass, largely remain unclear.

To address this, the first part of the present study was conducted in the field. It was set up to estimate total soil respiration and the contribution of root and microbial respiration to total soil respiration, and to quantify microbial biomass, as affected by change in vegetation structure from woodland (trees, some shrubs with herbaceous understorey) to open grassland. The study also focussed on change in soil respiration during different seasons in response to associated change in temperature. The location was a semi-arid zone in central Queensland. Three major types of tree woodland communities i.e. *Eucalyptus populnea*, *E. melanophloia* and *Acacia harpophylla* were selected for paired comparison of uncleared and cleared (grasslands) treatments. Tree felling followed by sowing to exotic grasses alters vegetation structure and function, and was predicted to change soil biological properties. The effect of clearing on soil biological properties was studied at different scales of time-since-clearing in cleared grasslands to interpret the long-term effects of clearing.

Since root respiration is a major component in total soil respiration (varying from 10-90 percent depending on vegetation type and season of the year - Hanson *et al.* 2000), the second part of the study investigated the contribution of each of root, microbial and rhizosphere respiration to total soil respiration and the impact of simulated grazing on soil respiration under polyhouse conditions. During this experiment (conducted in the polyhouse), soil respiration response to change in soil temperature and moisture, and plant growth was studied in detail.

Materials and methods

a) Field study

i. Research sites and design

The research sites were located in a semi-arid zone in central Queensland. Full details are presented in Chapter 3.

ii. Measurements

Soil respiration

Soil respiration was measured with a soil respiration chamber (10 cm diameter x 24 cm height) connected to an infrared gas analyser (Environmental Gas Monitor (EGM-3), PP Systems, UK). Preliminary trials were conducted to measure rate of soil respiration with change in soil temperature (during different times of a day) and moisture. Based on these observations and to keep consistency in data, the field measurements were taken on normal (non-rainy) days for the morning hours (6:00-9:00 am) within a soil temperature range of 5 ^{o}C.

The data were collected over the span of a year, in August and November 2001, and March and July 2002. Six readings were taken randomly at an approximate 15 m distance from each other within a 100 m x 100 m of total area at each site in each season. Measurements were taken between plants (inter-plant) in the open in cleared plots and at random but not closer than 1 m from tree trunks in the uncleared plots. The readings did not differ much with distance from a tree trunk, as grass roots are sporadic in a woodland system. Due to accidental patchy burning, no readings were taken at the recent cleared site for *E. melanophloia* in March and July 2002. Soil temperature at 5 cm depth was also measured during respiration readings with a probe attached to the EGM-3.

The litter layer (if any) was removed before placing the soil respiration chamber on the soil surface. The chamber was placed onto bare soil, avoiding grass stumps and exposed roots and data were collected from the EGM-3 that measured CO_2 fluxes at every 8 seconds interval over 2 minutes period. Soil temperature probes were inserted into the soil near to the chamber. The measurements were taken on clear mornings firstly in the cleared (because of its open exposure to sun) and subsequently at the paired uncleared sites to minimize the effect of soil warming. On cloudy days, either the cleared or uncleared site of a pair was taken first, at random. Only one pair of cleared and uncleared treatments were monitored on any one day. Measurements were consecutively over 9 days for all sites.

To study the response of soil respiration to soil moisture and soil temperature measurements were taken in Dec 2001 at the oldest set of all the cleared and uncleared sites before (1-3 days) and after rain (starting from 2nd till 6th day after rain). Soil moisture content was measured to 12 cm depth with probes (Hydrosense Soil Moisture Measurement System, Campbell Scientific Australia) on each day.

Root respiration (for herbaceous plants)

Eight soil cores to 60 cm depth (4 cm diameter) were taken at each site during January 2002 using a hydraulic soil corer. The cores were taken from space between plants. Soil samples were dried and bulked per site. The visible roots (>1 mm diameter) were taken out by hand following sieving, dried at 60 °C and weighed. This method would not have removed the very fine roots, hence total root biomass values may have been underestimated.

The rate of root respiration per unit root biomass was determined in the polyhouse experiment with the most commonly grown grass species *Cenchrus ciliaris* (discussed later). The average rate of root respiration per unit root biomass (g) obtained from polyhouse experiment (0.08 g $CO_2/m^2/hr$ per g root biomass) was used to compute in-field root respiration based on in-field root biomass determination.

Microbial respiration

Soil microbial respiration was computed from average soil respiration per year minus computed root respiration based upon the estimate of root biomass, for each site.

Soil microbial biomass

Samples of soil from the top 0-5 cm were collected in March 2002 at each site. Soil microbial biomass for carbon (MBC) and nitrogen (MBN) were analysed using the chloroform fumigation extraction method (Vance *et al.* 1987) at the Natural Resource Sciences Laboratories (Department of Natural Resources and Mines, Brisbane, Qld).

Soil microbial biomass and soil respiratory quotients were computed as (Sparling 1997):

$$\text{Soil microbial biomass quotient (\%)} = \frac{\text{Soil microbial biomass carbon}}{\text{Soil organic carbon}} \times 100$$

$$\text{Soil respiratory quotient} = \frac{\text{Total soil respiration (g } CO_2\text{/hr)}}{\text{Soil microbial carbon (g)}}$$

iii. Statistical analysis

The data were analysed using residual maximum likelihood analysis (REML) in Genstat ver 6.0 (2002). For this, all the uncleared sites for a tree type at each group of clearing were taken as replicates, whereas the paired cleared sites for each age group were considered as such with no replication within a community but replicated across the communities. The data are therefore presented for uncleared, recent, medium and old age since clearing treatments in each tree community.

Soil respiration

The main effects (fixed term) of tree community*cleared-uncleared and random effects of time-since-clearing for recent, medium and old clearing of paired cleared and uncleared treatments for all the tree communities were analysed using REML.

For the main effects, if the interaction between a tree community*cleared-uncleared was significant at $P < 0.05$, then LSDs (least significant difference of means) were computed for each treatment. In the absence of any significant interaction, the individual effects for a tree community and cleared-uncleared treatments were computed. The means obtained from REML analysis were used in presenting the results to maintain consistency for both the significant and non-significant levels of interaction.

Soil, microbial and root respiration, root biomass and soil microbial biomass for C and N

REML was applied as for soil respiration. Paired t-tests were also applied to compare means for all the cleared and uncleared treatments.

Soil respiration response to soil temperature and soil moisture

Soil respiration relationships to temperature and to moisture were analysed using regression analysis. Soil respiration response to temperature over a year was computed from the temperature quotient $Q_{10} = e^{\beta_1 T}$ (represents the sensitivity of respiration for every 10 ^{o}C change in temperature) obtained from an exponential regression $y = \beta_0 e^{\beta_1 T}$ (where y = soil respiration, β_0 and β_1 = fitted constants and T = temperature (^{o}C)) fitted to the data. An ANOVA was applied for the data collected before and after rain at all cleared and uncleared treatments.

b) Root respiration experiment in the polyhouse

i Experimental design and measurements

Root respiration was difficult to measure in the field at each site in different seasons because core digging was a tough task in hard soils. However, soil samples were taken once at each site to quantify soil chemical properties and to determine root biomass. To complement these, an experiment was conducted to estimate the contribution of root respiration to total soil respiration. The estimated specific root respiration rate (i.e. in relation to root biomass) was used to determine in-field root respiration based on root biomass (in-field).

The common pasture grass *C. ciliaris* was grown in pots (34 cm diameter, 35 cm height) in two sets to estimate root respiration in the absence (set I) and presence of defoliation (simulated grazing i.e. removing leaves and stems to the half of plant height- set II) (Table 6.1). For defoliation, plants were successively defoliated once, twice and thrice in defoliation treatments D1, D2 and D3 respectively (Table 6.1). Two types of controls were maintained: C1 with soil but no grass to measure root-free soil (microbial) respiration, and C2 with grass but without defoliation (Table 6.1).

Table 6.1. Various treatments to estimate root respiration.

	Treatment	Abbreviations used in text	No. of pots	Growth period (months)
Control 1	No grass, soil only	C1	5	-
Control 2	No defoliation	C2	3	9
I set	No defoliation	D0	5	4
II set	Defoliation - once	D1	3	6
	Defoliation - twice	D2	3	7.5
	Defoliation - thrice	D3	3	9

Seeds were sown on 4th April 2002 to sandy soil in each pot, and supplied with hydroponic nutrient solution (Manutec Pty Ltd, SA) during the growth phase. The pots were watered to 80 % of their field capacity and the next irrigation event was followed when the volumetric water content fell to 30-35 % of field capacity. Sandy soil was chosen as an appropriate medium to minimize root loss during extraction.

The experiment was set up in the polyhouse (temperature 7-32 °C and relative humidity 14-48 per cent), in a randomised block design with random distribution of all the grass and control pots in four blocks (in the beginning of the experiment, the same treatments as for grass were also maintained for a legume (*Stylosanthes scabra* secca), and all the pots (grass, legume and controls for both) were distributed randomly in 4 blocks. However, the legume plants were very slow growing and those data were not reported herein).

The soil respiration chamber connected to the EGM-3 was placed on soil in the centre of each pot with the soil moisture and soil temperature probes, at the time of data collection. Soil respiration, soil temperature and soil moisture were monitored from when the plants were two months old (approximate height >30 cm) until their uprooting for root respiration measurements. The measurements were taken over consecutive days following each irrigation event. Measurements on control C2 pots continued (without any defoliation) from second month of growth onward over consecutive days after each irrigation event until the whole experiment was terminated i.e. 9 months after sowing. Data collected from the control C2 (grass without defoliation) on soil respiration, temperature and moisture were compared with their corresponding sampling data taken for defoliation treatments D1 (once), D2 (twice) and D3 (thrice).

To determine the contribution of root respiration to total soil respiration, total soil respiration was recorded for all pots just before uprooting. The plants D0 were uprooted after 4 months of growth, whereas D1, D2, D3 and C2 were uprooted after 6, 7.5 and 9 months of growth respectively. The roots were extracted by emptying a pot onto a plastic sheet to take the plants out of the sand. Sand and shoot parts were then removed before measuring the root respiration. A special PVC chamber the size of the soil respiration chamber was constructed. The roots were kept in the PVC chamber and the soil respiration chamber was placed vertically on this to measure respiration (R_{root}). The pots were uprooted one at a time with measurements completed within a minimum time gap and without drying. After respiration measurements, roots were washed to completely remove the sand particles (if any), dried at 60 ^{o}C for 48 hours and weighed.

The pots were refilled with the same soil once the plants were removed, and monitored at 3-4 day intervals until respiration stabilised (normally about one month). The stabilised rate of respiration was taken as root-free soil respiration/microbial respiration (R_{rfs}) as per the basal respiration method outlined by Kelting *et al.* (1998). R_{rfs} was then used to estimate rhizosphere respiration (R_{rhizo}) (i.e. respiration attributable to microbial activities and decomposition of root exudates in the rhizosphere), as follows (Kelting *et al.* 1998):

Total soil respiration $R_s = R_{root} + R_{rfs} + R_{rhizo}$ (i)

Where each component is measured as:

R_s - using soil respiration chamber

R_{root} - by extraction + soil respiration chamber

R_{rfs} - using basal method

Rearranging equation (i) solving for R_{rhizo}:

$R_{rhizo} = R_s - (R_{rfs} + R_{root})$ (ii)

The above-mentioned procedure, a combination of two methods i.e. the root extraction and the basal, was followed to determine root, microbial and rhizosphere respiration and to avoid overestimation of any of these three components of soil respiration. Use of one method either extraction method or basal method overestimates the root and microbial components of total respiration (Kelting *et al.* 1998). These are explained as below:

Excising roots by extraction to measure R_{root} overestimates R_{rfs} (microbial respiration) since rhizosphere respiration is not isolated: $R_{rfs} = R_s - R_{root}$ (R_{rfs} includes rhizosphere respiration).

The basal method measures R_s and R_{rfs}, and then R_{root} is calculated as: $R_{root} = R_s - R_{rfs}$, but it overestimates root respiration since the rhizosphere component is not isolated (R_{root} includes rhizosphere respiration).

ii Statistical analysis

The soil, root, microbial/root-free soil, and rhizosphere respiration, and root biomass data were analysed using ANOVA for comparisons between treatments. Simple regression analysis was used for R_{root} relationship to root biomass. R_s response to temperature and moisture was analysed using multiple regression for the interactive, and simple regression for individual effects of temperature and moisture for each treatment (C1, C2, D0, D1, D2 and D3).

Results

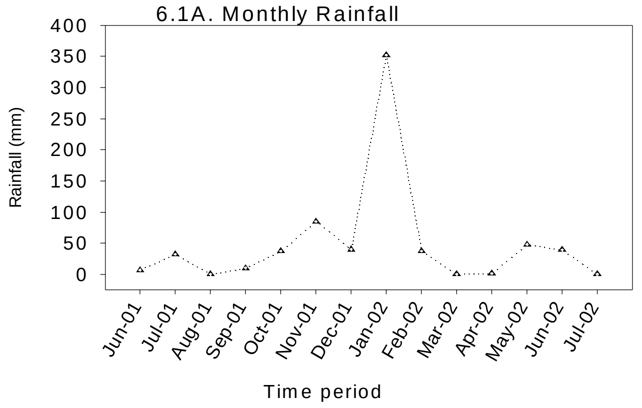
a) Field study

i. Soil respiration

There was no significant difference in rate of soil respiration between cleared and uncleared sites, nor with time-since-clearing for any of the tree communities (Fig 6.1B (a, b and c)) with the exception that the recent clearing had a greater rate than that of uncleared, medium and old cleared sites for *E. populnea* in Aug 2001 (Fig 6.1B).

Soil respiration rate showed seasonal variation, and was notably greater during the wet (Nov 2001 and Mar 2002) compared to the dry season (Aug 2001 and July 2002) (Fig 6.1A and 6.1B). The respiration rates differed between wet (0.14 ± 0.004 g $CO_2/m^2/hr$ (mean ± standard error of means)) and dry (0.09 ± 0.003) seasons for all treatments irrespective of whether cleared or uncleared, or of tree type.

The increase in soil respiration in the wet compared to the dry season did, however express itself more in the uncleared (0.10 ± 0.004 g $CO_2/m^2/hr$) and medium age of clearing (0.10 ± 0.001 g $CO_2/m^2/hr$ increase) than in the recent (an increase of 0.06± 0.005) and the oldest (0.05 ± 0.001), regardless of tree community.



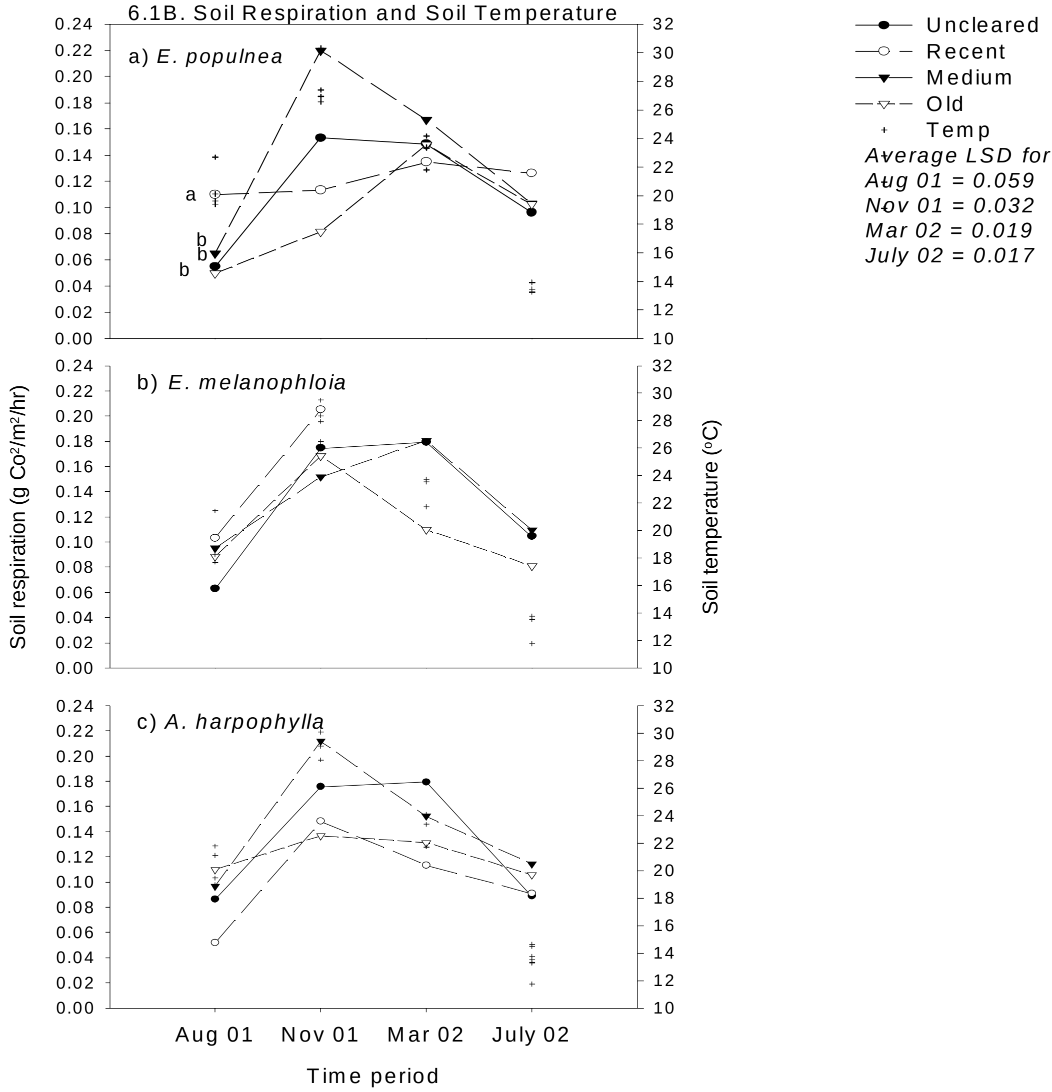


Fig. 6.1A. Monthly rainfall at all sites during soil respiration measurements.

Fig. 6.1B. Soil respiration (g $CO_2/m^2/hr$) and soil temperature (^oC) at cleared (recent, medium and old) and uncleared treatments for a) *E. populnea*, b) *E. melanophloia* and c) *A. harpophylla* communities.

A significant difference (*P<0.05)* between cleared and uncleared treatments existed only for a) *E. populnea* in Aug 2001 where any two treatments with different letters differed from each other.
Average LSD denotes the least significant difference of means of soil respiration between any two treatments for a season.

ii. Microbial and root respiration

Average annual total rate of soil respiration (sum of root and microbial respiration, Fig 6.2) showed no significant difference between cleared and uncleared sites. Microbial respiration was greater at uncleared compared to the oldest clearing for *E. populnea*, and to all cleared sites for *A. harpophylla* (Fig 6.2 a and c), while it showed no significant difference at cleared and uncleared sites for *E. melanophloia* (Fig 6.2 b). Root respiration (computed according to root biomass of herbaceous plants) was greater at recent and medium age of clearing compared to uncleared sites in all the tree types other than the medium cleared site of *E. melanophloia* (Fig 6.2).

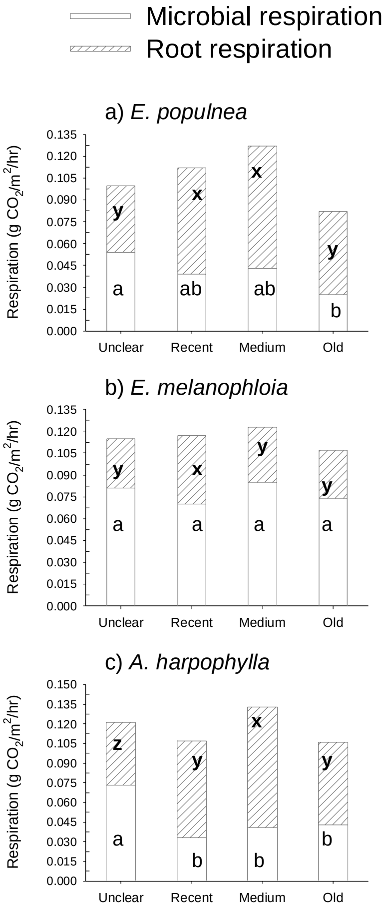


Fig. 6.2. Soil microbial (empty bars)- and root (lined bars)- respiration at cleared and uncleared treatments for a) *E. populnea*, b) *E. melanophloia* and c) *A. harpophylla* communities.

Different letters on lined (root respiration) and empty (microbial respiration) bars represent significant difference at P<0.05 between treatments within a tree community corresponding to similar bars.

Root respiration was computed in relation to root biomass at each site. Root biomass of fibrous roots for 0-60 cm depth was greater at all cleared than uncleared treatments for *E. populnea* and *A. harpophylla,* and at recent clearing than uncleared site at *E. melanophloia* (Table 6.2).

Soil microbial respiration was positively but weakly related to MBC (r = 0.20 at *P=0.05*) and MBN (r = 0.27) across all the sites. In *E. populnea,* uncleared, medium and recent clearing had greater MBC compared to the oldest clearing, but did not significantly differ in terms of MBN (Table 6.2). In *E. melanophloia,* MBC did not show difference between any of the cleared and uncleared sites, but MBN was greater at uncleared than recent clearing. MBC and MBN were greater at uncleared *A. harpophylla* than the cleared treatments except for the oldest clearing. The cleared and uncleared treatments for all the tree types did not differ significantly in soil organic carbon with the exception that the recent clearing had greater amount than the uncleared site in *E. populnea* (Table 6.2).

Microbial quotient (represents the ratio of MBC to soil organic carbon) did not show any notable difference between cleared and uncleared treatments. Respiratory quotient (ratio of soil respiration to MBC) was greater at recent and the oldest clearing for *E. melanophloia,* and at all cleared sites for *A. harpophylla* compared to their uncleared sites, but did not differ between cleared and uncleared treatments at *E. populnea* (Table 6.2).

Table 6.2 Root biomass for 0-60 cm depth, microbial biomass carbon (MBC) and nitrogen, and soil organic carbon for 0-5 cm depth, microbial quotient (%), and respiratory quotient for cleared (recent, medium and old) and uncleared treatments at *E. populnea, E. melanophloia* and *A. harpophylla*.

Parameters	Tree community	Uncleared*	Recent*	Medium*	Old*	LSD#
Root biomass (fibrous roots) (g/0.005 m^3)	*E. populnea*	3.56^{d}	5.71^{b}	6.59^{a}	4.41^{c}	0.52
	E. melanophloia	2.64^{b}	3.67^{a}	2.96^{b}	2.61^{b}	
	A. harpophylla	3.74^{d}	5.74^{b}	7.18^{a}	4.95^{c}	
Microbial biomass carbon (mg/kg)	*E. populnea*	349.2^{ab}	447.7^{a}	339.1^{ab}	178.4^{b}	240.45
	E. melanophloia	315.5^{a}	181.0^{a}	250.0^{a}	147.1^{a}	
	A. harpophylla	492.9^{a}	148.1^{b}	204.9^{b}	386.5^{ab}	
Microbial biomass nitrogen (mg/kg)	*E. populnea*	32.1^{a}	45.4^{a}	28.4^{a}	30.2^{a}	23.48
	E. melanophloia	40.6^{a}	21.1^{b}	28.7^{ab}	28.0^{ab}	
	A. harpophylla	47.8^{a}	20.1^{b}	19.2^{b}	47.8^{a}	
Soil organic carbon (g/kg)	*E. populnea*	11^{b}	18^{a}	14^{ab}	11^{b}	4.97
	E. melanophloia	13^{a}	12^{a}	11^{a}	10^{a}	
	A. harpophylla	18^{a}	18^{a}	14^{a}	15^{a}	
Microbial quotient (%)	*E. populnea*	3.26^{a}	2.49^{a}	2.42^{a}	1.62^{a}	2.716
	E. melanophloia	2.38^{a}	1.51^{a}	2.27^{a}	1.47^{a}	
	A. harpophylla	2.93^{a}	0.82^{a}	1.46^{a}	2.58^{a}	
Respiratory quotient (g CO_2/hr per g MBC)	*E. populnea*	0.10^{a}	0.08^{a}	0.14^{a}	0.14^{a}	0.061
	E. melanophloia	0.12^{b}	0.23^{a}	0.16^{b}	0.22^{ab}	
	A. harpophylla	0.09^{b}	0.24^{a}	0.21^{a}	0.28^{a}	

*different superscripts in a row represent significant difference at $P<0.05$
#Average values for least significant difference (LSD) of means at $P <0.05$

Comparison between cleared and uncleared treatments regardless of time-since clearing and tree type showed that uncleared sites had significantly greater microbial respiration compared to cleared sites (Table 6.3). But the root respiration was greater at cleared (0.06 g CO_2/m^2/hr) than uncleared (0.04 g CO_2/m^2/hr) sites. The microbial respiration contributed 45 per cent at cleared and 62 per cent at uncleared sites to total soil respiration. Root respiration was 38 per cent of total soil respiration at uncleared sites. At cleared sites, root biomass was significantly higher compared to uncleared sites (Table 6.3), and consequently the respiration of roots contributed 55 per cent of total soil respiration.

Table 6.3. The mean values (± standard error of mean) for microbial and root respiration, root biomass, and soil microbial biomass for carbon and nitrogen for all the cleared and the uncleared treatments.

	Uncleared	Cleared
Microbial respiration ($g\ CO_2/m^2/hr$)	0.07(±0.004)*	0.05 (±0.006)
Root respiration ($g\ CO_2/m^2/hr$)	0.04 (±0.003)*	0.06 (±0.007)
Root biomass ($g/0.005\ m^3$)	3.31 (±0.21)*	4.87 (±0.53)
Microbial biomass - carbon (mg/kg)	385.9 (±37)*	253.6 (±37)
Microbial biomass - nitrogen (mg/kg)	40.17 (±3.29)*	29.87 (±3.45)

t value at 8 df =2.306 at $P = 0.05$,
*represents significant difference in a row after applying t-test

Factors affecting soil respiration

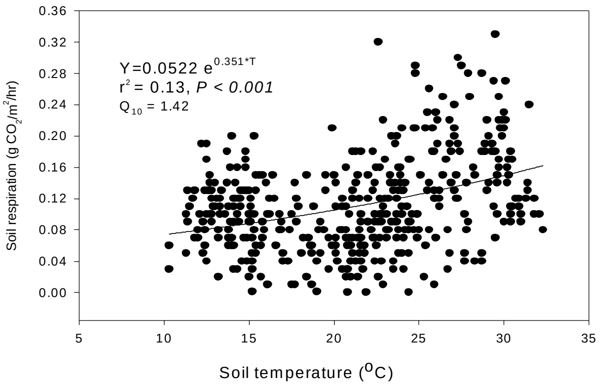


Fig. 6.3. Soil respiration as a function of soil temperature, data collected throughout a year at all cleared and uncleared sites for *E. populnea*, *E. melanophloia* and *A. harpophylla* communities.

Soil respiration increased with rise in soil temperature (Fig 6.3) although there was background variation within the data set. The Q_{10} value of 1.42 was calculated for exponential change in soil respiration with each 10 °C rise in temperature over the range 10-32 °C.

A significant increase in soil respiration occurred after rainfall (Table 6.4). The data were collected in Dec 2001 in the middle of the rainy season (40 mm rainfall in Dec, after 85 mm in Nov and 37 mm Oct). It was not, therefore, the first response of a sudden rise in soil respiration in response to rain after the dry season.

Table 6.4. Soil respiration (g $CO_2/m^2/hr$), soil moisture (SM - % volumetric water content) and temperature (ST- °C) before (-3 to-1 day) and after (2-6 day) rainfall.

Days after rain	-3 to -1 (*before*)	2	3	4	5	6	F value
Soil respiration*	0.1508^c	0.3081^a	0.2888^a	0.2847^a	0.2911^a	0.245^b	0.001
ST*	27.52^a	26.05^a	25.51^a	26.38^a	26.19^a	28.07^a	0.065
SM*	4.0^b	8.6^a	8.7^a	7.1^a	5.1^{ab}	4.0^b	0.001
STx SM							0.276

df = 191

*different superscripts in a row represent significant difference at $P<0.05$

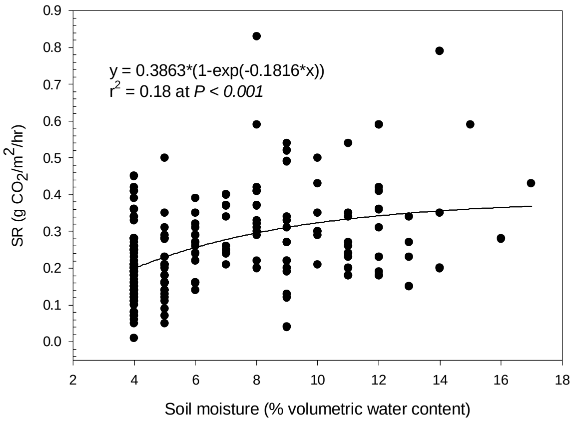


Fig. 6.4. Soil respiration as a function of soil moisture (before and after rainfall event) at the oldest cleared and uncleared sites for *E. populnea*, *E. melanophloia* and *A. harpophylla* communities.

Measured daily before (13-15th of Dec 2001) and over 6 days (18th- 23rd of Dec 2001) after a rainfall event on 16-17th of Dec 2001, soil moisture accounted for 18 per cent of the variation in soil respiration (Fig 6.4).

b) Root respiration experiment in the polyhouse

An experiment was conducted i.e. without defoliation and with defoliation, on *C. ciliaris* to quantify the different components of root, microbial and rhizosphere respiration in soil respiration.

The total soil respiration (R_s) and root respiration (R_{root} per unit chamber volume and pot) increased over the period of growth from four to nine months. The R_s of defoliation treatments D2 and D3 did not differ from the un-defoliated control C2; defoliation *per se*, therefore, had no effect on R_s (Table 6.5).

Root-free soil R_{rfs} (microbial) respiration and R_{root} per unit root biomass showed no difference between any of the treatments (Table 6.5). Rhizosphere respiration (R_{rhizo}) was at a maximum in D2, was less in D3 and was least in D0 (no defoliation) (Table 6.5).

Table 6.5.Total soil (R_s), microbial (root-free soil) (R_{rfs}), rhizosphere (R_{rhizo}) and root (R_{root}) respiration (g $CO_2/m^2/hr$) per soil respiration chamber volume (1885.71 cm^3), and root respiration (R_{root}) per pot (volume 20757 cm^3), per unit root biomass (g), and root biomass (RB) (g) per pot for controls C1 (soil but with no grass) and C2 (with grass, no defoliation), and for plants before (D0) and after successive stages of defoliation (D1-once, D2-twice and D3-thrice) treatments.

Treatments -defoliation (growth period)	R_s^*	R_{rfs}^*	R_{rhizo}^*	R_{root}^*	R_{root}/ pot*	R_{root} /RB*	RB/pot*
C1	0.02^d	0.02^a					
C2-none (9 month)	1.13^a	0.02^a	0.39^b	0.72^a	7.92^a	0.07^a	129.4^a
D0-none (4 month)	0.19^c	0.01^a	0.08^c	0.09^c	1.05^c	0.08^a	11.1^b
D1-once (6 month)	0.76^b	0.01^a	0.38^b	0.37^b	4.07^b	0.09^a	41.3^b
D2-twice (7.5 month)	1.37^a	0.02^a	0.93^a	0.42^b	4.64^b	0.07^a	69.0^b
D3-thrice (9 month)	1.22^a	0.02^a	0.31^b	0.89^a	9.77^a	0.08^a	121.2^a
LSD	*0.268*	*0.025*	*0.178*	*0.200*	*0.021*	*0.02*	*41.96*

[*] Different superscripts in a column represent significant difference according to least significant difference of means (LSD) at $P = 0.05$

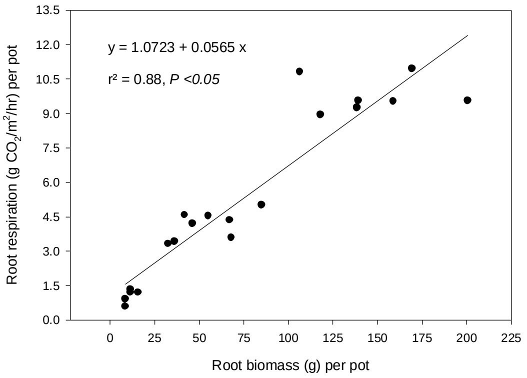


Fig. 6.5. Root respiration and root biomass relationship for *C. ciliaris.*

R_{root} per unit root biomass was similar across all the treatments. But R_{root} per pot or per unit volume of soil respiration chamber increased with plant growth (Table 6.5). R_{root} was clearly related to root biomass (Fig 6.5). The increase in root biomass contributed to increased rate of R_{root} with the growth of plant, and consequently increase of R_s.

Factors affecting R_s (polyhouse experiment)

Soil temperature accounted for significant variation in R_s (r^2= 0.64 at $P<0.05$) in all defoliation treatments D1, D2, D3 and their control C2. The R_s response to change in temperature was very small in C1 (control without plants; r^2 = 0.07 at $P<0.05$) for a range of 16.6 °C-25.4 °C temperature and in D0 (4 months growth but no defoliation; r^2 =0.13 at

P<0.05) for a range of 10.8 ^{o}C-25.9 ^{o}C temperature. There was a significant response (only of intercept) of R_s to temperature in defoliation treatments, probably due to plant growth.

R_s response to temperature in D3 was significantly different from its control C2 and from D1 or D2 at *P<0.05* (Fig 6.6 and Table 6.6), whereas differences between D1, D2 and C2 were not significant.

Further analysis of R_s response to temperature for defoliation treatments D1, D2 and D3 in comparison to their respective controls C2-1, C2-2 and C2-3 (there were 3 sets of measurements for C2: C2-1 against D1, C2-2 against D2, C2-3 against D3) demonstrated a significant response (for both intercept and slope, at *P<0.05*) for D3 compared to its control C2-3. For D1 and D2, there was no significant response of R_s to temperature when compared with their respective controls C2-1 and C2-2. At D3 (third stage of defoliation), the greater R_s response to temperature showed that defoliation increased R_s sensitivity to temperature.

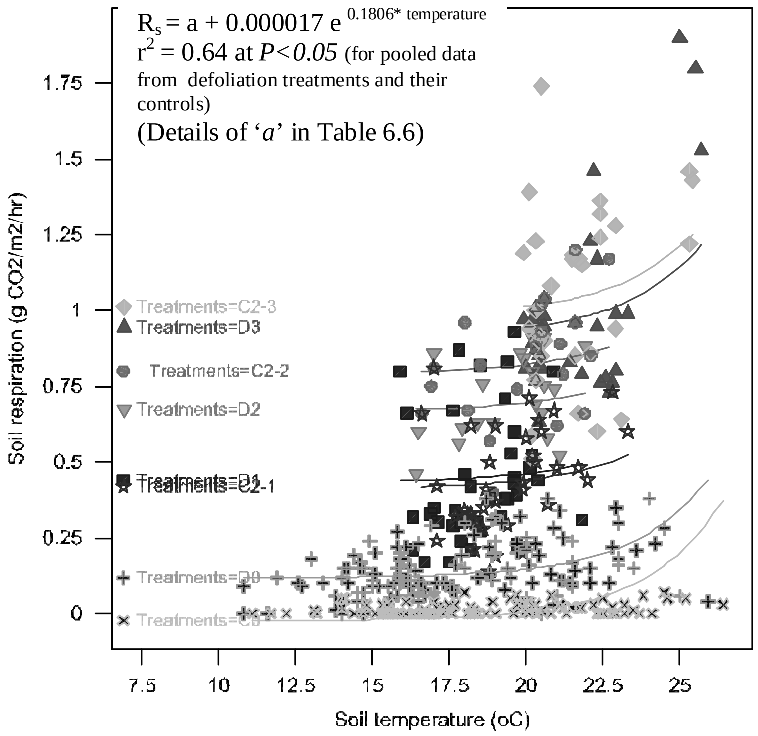


Fig 6.6 Soil respiration response to soil temperature for defoliation treatments and their corresponding controls (graph reproduced from Genstat):
D1 (defoliated once) and control C2-1 (treatment labels are overlapping in plot)
D2 (defoliated twice) and control C2-2
D3 (defoliated thrice) and control C2-3
D0 (no defoliation)
C0 (control with no grass) (treatment labels C0 and D0 are overlapping in plot)

Table 6.6 Regression analysis results for exponential relationship ($R_s = a + b\ e^{(-k*temperature)}$) between soil respiration and soil temperature for various treatments.

Treatment	a (Intercept)	b (Slope)[#]	k	r^2 *
D1 (Defoliated once)	0.4065	0.00001749	-0.1806	0.64
D2 (Defoliated twice)	0.6278			
D3 (Defoliated thrice)	0.8055			
C2-1 (Control against D1)	0.3711			
C2-2 (Control against D2)	0.7421			
C2-3 (Control against D3)	0.7584			
D0 (no defoliation)	n.s.			
C0 (Control with no plant)	n.s.			

[#] Slopes were not significantly (at $P<0.05$) different across various treatments hence all data had a common slope, meaning that the lines were parallel but with different intercepts.
* for pooled data from defoliation treatments D1, D2, D3 and their controls C2-1, C2-2 and C2-3.
n.s. - no significant response of soil respiration to soil temperature at $P<0.05$.

Soil moisture accounted for only four per cent of the variation in soil respiration in all the treatments (Fig 6.7), therefore, temperature proved to be the main factor that accounted for variation in soil respiration in the pot experiment.

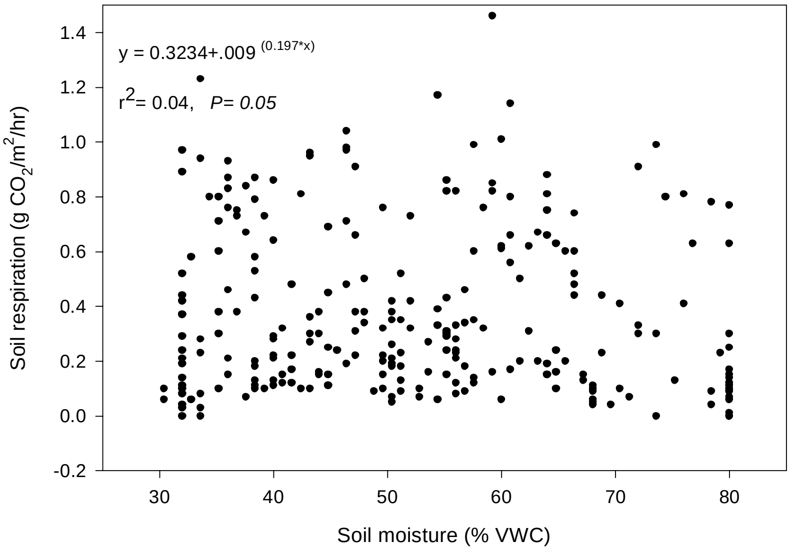


Fig 6.7 Soil respiration response to soil moisture (% volumetric water content) in all treatments (C1, C2, D0 and defoliation treatments D1, D2 and D3).

Discussion

Cleared grasslands and native woodlands had similar average annual rates of soil respiration, equivalent to 964 g $CO_2/m^2/yr$ (263 g $C/m^2/yr$) with measurements taken from inter-plant areas in the open in cleared and at random but at least at a distance of 1 m from tree trunks in woodlands. The field measurements indicated that the rate of soil respiration varied between different seasons of the year with a greater rate in the rainy (Nov-Mar) than the dry season (July and Aug). This was most likely due to increased activity of soil microbes during the wet season as illustrated by the response of soil respiration to rain in Table 6.4, and the cessation of their activities in the dry season (Grace and Rayment 2000).

The average rate of soil respiration reported herein for the semi-arid zone (263 g $C/m^2/yr$) was less than that reported for semi-arid woodlands of north Queensland (380 g $C/m^2/yr$ (Holt *et al.* 1990, cited by Raich and Schlesinger 1992)), tropical savannas and grassland (629 g C $/m^2/yr$), temperate grassland (442 g $C/m^2/yr$) (Raich and Schlesinger 1992), and much less than the 1.71 kg CO_2-$C/m^2/yr$ reported for tall grass prairie (Mielnick and Dugas 2000). The reasons may be the differences in climate, chiefly rainfall and temperature, and may be in part due to the vegetation communities.

In the semi arid climate of central Queensland, the conversion of woodlands to open pastures (grasslands) did not increase the rate of total soil respiration in the present study. Hence, it is unlikely that clearing for pasture development (except for clearing operations e.g. pulling, fire and initial decomposition of vegetation left after clearing) in this region contributes to increased levels of CO_2 and consequently to global warming. The established cleared pastures (after 5, 11-13 or 33 yrs of development) did not show any difference in the rate of total soil respiration to uncleared pastures, in contrast to other studies (Raich and Tufekcioglu 2000; Batjes and Sombroek 1997) where increased CO_2 emissions accompanied land use change.

Soil respiration differs in response to temperature and moisture, but the degree of response may vary in different climatic zones. The Q_{10} (sensitivity for change in soil respiration with temperature, $Q_{10} = e^{\beta_1 T}$) value of 1.42 suggested only little response to temperature in field conditions of central Queensland. The average Q_{10} value for various ecosystems reported is 2.4 with a range of 1.3-3.3 (Raich and Schlesinger 1992). The temperate zones are more sensitive to increase in soil respiration with increase in temperature (Valentini *et al.* 2000) and may have Q_{10} = 3.5 (Boone *et al.* 1998). Soil moisture (considered a primary factor that limits growth in semi-arid woodlands (Scholes 1993)) and soil temperature or their interaction with other environmental factors, may affect the rates of soil respiration in the semi-arid climates.

In the present study, soil moisture was not measured in the field in the dry season due to difficulty in inserting the delicate soil moisture probes in dry hard soils. Measurements were taken during the rainy season where soil respiration showed a small response to soil moisture (r^2 = 0.18) for the successive 2 to 6 day after rain. This suggested that soil respiration was not strongly related to soil moisture. Under polyhouse experimental conditions, soil respiration response to soil moisture was minimal (r^2 = 0.04) for a range of soil moisture content (35-80 per cent volumetric water content), but this was in the absence of any wet or dry periods which would have occurred under natural conditions as seen for in-field soil respiration response to rain. Both the polyhouse and in-field experiments indicated a very small response of soil respiration to soil moisture. Maybe there are other limiting factors such as soil nutrients or soil carbon source that regulate root and microbial activity.

The contribution of root respiration (herbaceous plants) calculated according to root biomass (for 0-60 cm depth) to total soil respiration was larger (55 %) in cleared pastures than that in woodlands (36 %). Similar results for 17-40 per cent root respiration of total soil respiration in grasslands were reported by Raich and Tufekcioglu (2000). In the polyhouse experiment for *C. ciliaris*, respiration from roots during defoliation (D0, D1, D2 and D3) contributed about 31-73 percent, rhizosphere about 25-68 percent and root-free soil (microbial) respiration 1.4-5 percent of total soil respiration. The defoliation/simulated grazing had no significant impact on soil respiration. The root biomass and root respiration relationship from the polyhouse experiment suggested that for in-field measurements the differences in root biomass between cleared and uncleared sites (Tables 6.2 and 6.3) were largely responsible for minimizing the differences between treatments in total soil respiration.

The sensitivity of soil respiration to temperature was greater for *C. ciliaris* in polyhouse experiment (Fig 6.6, r^2 = 0.64, Q_{10} = 3.31) compared to in-field response (r^2 = 0.13, Q_{10} = 1.42, Fig 6.3). In the field, *C. ciliaris* was predominantly growing in association with other plant species at cleared sites, but native grass species dominated at uncleared sites. The average root biomass per unit volume was greater in the polyhouse experiment (6 mg/cm^3) compared to in-field conditions (1 mg/cm^3). Such a response of soil respiration to temperature in polyhouse conditions was mainly due to the greater root respiration (root respiration was strongly related to root biomass ($r^2 = 0.88$)). This was concluded based on the evidence that compared to grass pots, the control C1 with no grass showed no significant soil respiration response to temperature ($r^2 = 0.07$). Hence, it is suggested that the soil respiration response to temperature was in the main due to root biomass and therefore root respiration response to temperature. Greater sensitivity of root respiration than soil respiration to change in temperature has also been reported before (Boone *et al.* 1998). If the root respiration were sensitive to temperature, then the greater rate of root respiration could be of concern in cleared pastures for increase in CO_2 emissions with change in temperature (global warming).

Conversion of woodlands to cleared pastures presents a different scenario for microbial activities and for microbial carbon and nitrogen biomass than for total soil respiration. Overall, the cleared pastures had lesser rate of microbial activity and lesser microbial biomass for C and N than did the uncleared pastures (Table 6.3). Similar results were also seen by Sparling *et al.* (1994b). For established woodland systems, the lesser microbial activity in relation to their biomass (respiratory quotient) in *E. melanophloia* and *A. harpophylla* than that at cleared pastures indicated the potential of these woodlands to optimise the use of soil resources whereas higher respiratory quotient values in woodlands indicate stress response and 'poor soil health' (Sparling 1997).

The soil microbial biomass results were contrary to the earlier claims by Graham *et al.* (1981) and Lawrence *et al.* (1988). They suggested that with age cleared pastures decline in soil fertility due to an increase in soil microbial biomass consequently tying up nutrients in microbial biomass as it competes with plants at cleared pastures in Queensland. According to their argument, the oldest cleared pastures in the present study should have had the maximum amount of microbial biomass. However, the results herein showed a trend for lesser microbial biomass and activity at the oldest and medium clearing than the uncleared sites (Fig 6.2, and Tables 6.2). But, there was an increase in soil microbial biomass for C and N at the oldest clearing, after initial decline at recent and medium clearing in *A. harpophylla* which may be due to season, root turnover or presence of cattle dung, but reasons are not clear.

Overall results for cleared and uncleared sites indicated a decrease in microbial biomass and microbial respiration for cleared pastures, which is most likely due to change in vegetation structure from woodlands to open grasslands. Changed pasture composition from a multi-species system in native woodlands to monocultures of *C. ciliaris* (predominantly, and with some other species) seems responsible for altered soil biological properties. The reduced species diversity may affect the use of resources in cleared systems as the diverse systems possess better potential for use of resources due to greater functional diversity, compared to the less diverse systems in grasslands (Tilman *et al.* 1996).

The composition and eco-physiological traits of plant species in an ecosystem are also important determinants for size, activity and structure of microbial communities (Wardle *et al.* 1998, cited in Bardgett *et al.* 1999). They affect soil biological properties through availability and quality of root exudates which are an important nutrient source for microbes (Klein *et al.* 1988 cited by Bardgett *et al.* 1999), and through alterations in nutrient competition (Bardgett *et al.* 1999). Introduction of exotic grass species such as *C. ciliaris*, and clearing of native vegetation disturb the plant-soil relationship in a particular environment. The different species harbour different microbial communities that affect the ecosystem functions (Bardgett *et al.* 1999). Substrate quantity and quality, soil microclimate and adaptability of microbes change with the type of vegetation (Kutsch and Dilly 1999).

Tree clearing resulted in changes at the macroscopic (vegetation communities - changed upper-storey and under-storey (composition discussed in Chapter 4)) and at the microscopic level (microbial biomass (Tables 6.2 and 6.3) and microbial respiration (Fig 6.2)). These changes may alter the ecosystem functions e.g. nutrient availability at cleared and uncleared sites (discussed in Chapter 5), quantity and quality of substrate available for microbial decomposition and nutrient return in a system. The quality and quantity of substrate available for decomposition and release of nutrients will be discussed in the next Chapter.

In the present study, the decline in soil biological properties in cleared treatments with few exceptions, is likely to be responsible for pasture run-down with time-since-clearing.

Chapter 7. Litter production, decomposition and nutrient release

Introduction

Litter is an important component in an ecosystem that maintains nutrient and energy flow at the soil-plant interface, provides habitat for various soil organisms and acts as a sponge to protect soil from erosion. Agren and Bosatta (1996) described litter as “the bridge between plant and soil”.

The amount of litter production and its decomposition varies with climate, season, substrate quality and type of vegetation. Litter is an important abiotic component of soils that influences biological and physiochemical properties of top soil (Heal and Dighton 1986, cited by Kutsch and Dilly 1999). Chemical composition of litter (type of vegetation) influences structure and activity of microbial communities inhabiting soils (Kutsch and Dilly 1999).

Conversion of woodlands to grasslands for greater pasture production leads to change in the vegetation structure. In central Queensland, large areas were cleared in the past and clearing continues today (577, 000 ha per annum during 1999-2001; Department of Natural Resources and Mines 2003) to develop pastures. The influence of change in vegetation structure from native woodlands (trees with understorey grasses and some shrubs) to open grasslands (largely mono-culture of a grass species - *Cenchrus ciliaris* L.), on litter production, composition, nutrient release, and any associated change in stability of soil properties is not yet clear.

The present study was set up to test whether the amount of litter production, composition and release of nutrients from decomposing litter differs with vegetation change from uncleared (woodland) to cleared (grassland) pastures. The time-since-clearing of cleared pastures was also taken into account to study the change in litter production, decomposition and nutrient release over time. The role of soil properties i.e. microbial biomass for C and N, organic carbon and pH, cation exchange capacity (CEC) in litter decomposition was also examined. To address these, three types of major tree communities i.e. *Eucalyptus populnea*, *E. melanophloia* and *Acacia harpophylla* were selected in a semi-arid region of central Queensland.

Materials and methods

Research sites and design

The sites were selected as pairs for uncleared and cleared treatments at different i.e. recent, medium and old age of clearing for each of three tree communities (*E. populnea*, *E. melanophloia* and *A. harpophylla*), in a 3 (tree communities) x 3 (times-since-clearing) x 2 (paired plots for cleared and uncleared) design. Full details are presented in Chapter 3.

Woodland communities of *E. populnea*, *E. melanophloia* and *A. harpophylla* supported mostly native grass species (*Aristida* species, *Themeda triandra*, *Dicanthium sericeum*) coupled with some shrubs and legumes (*Rhyncosia*, *Indigofera* species) on their floors, while cleared sites were dominated by the grass species *C. ciliaris.*

Since the tree density and herbaceous biomass influence litter production, data on these parameters are presented in Table 7.1. Tree density at uncleared sites was maximum at *A. harpophylla* followed by *E. populnea* and *E. melanophloia*. Overall, cleared sites had greater above-ground biomass of herbaceous plants compared to uncleared sites. Between cleared sites, the annual above-ground biomass of herbaceous plants was the most at medium clearing for *E. populnea* and *A. harpophylla,* and at recent clearing for *E. melanophloia* (Table 7.1).

Table 7.1. Tree density, tree basal area and annual average aboveground biomass for herbaceous plants at uncleared, and aboveground biomass at cleared (recent, medium and old) sites for *E. populnea, E. melanophloia* and *A. harpophylla* communities.

Tree community	Parameter	Uncleared	Recent	Medium	Old
E. *populnea*	Tree density (number/ha)	1583	-	-	-
	Tree basal area (m^2/ha)	37	-	-	-
	Above ground biomass for herbaceous plants (kg/ha/yr)*	2107^{b}	2817^{ab}	3792^{a}	2748^{ab}
E. *melanophloia*	Tree density (number/ha)	1216	-	-	-
	Tree basal area (m^2/ha)	21	-	-	-
	Above ground biomass for herbaceous plants (kg/ha/yr)*	3761^{b}	6335^{a}	3743^{b}	2816^{b}
A. *harpophylla*	Tree density (number/ha)	3622	-	-	-
	Tree basal area (m^2/ha)	21	-	-	-
	Above ground biomass for herbaceous plants (kg/ha/yr)*	1823^{b}	3030^{ab}	4259^{a}	2803^{ab}

*different superscripts in a row represent significant differences at $P<0.05$ between treatments within a tree community.

Measurements

Ground litter (mostly leaves, branches and twigs <1 cm in circumference) production was studied during different seasons regularly at four month intervals in a year. Decomposition was studied seasonally from litter collected during different seasons, and annually from litter collected at the start of study. Seasonal decomposition data for one date (Apr 2002) were lost for recent clearing in *E. melanophloia* due to accidental mild and patchy burning.

The paired-plot technique (Wiegert and Evans 1964) was followed to measure the amount of litter produced on the ground during different seasons. In Mar 2001, three quadrats of 1 m x 1 m were selected randomly in three different directions and marked (1, 2 and 3 as shown in Fig 7.1) at each site. The standing green and dry herbage was removed and dead litter on the ground was then collected from each quadrat. The litter samples were air dried and screened to exclude large sticks of circumference >1 cm. For the next sampling date in July 2001, litter was collected from adjacent plots paired to the marked plots used in Mar 2001 (1', 2' and 3' in Fig 7.1) and processed in the same way as in Mar 2001. The same procedure was followed in Nov 2001 and Mar 2002 measurements.

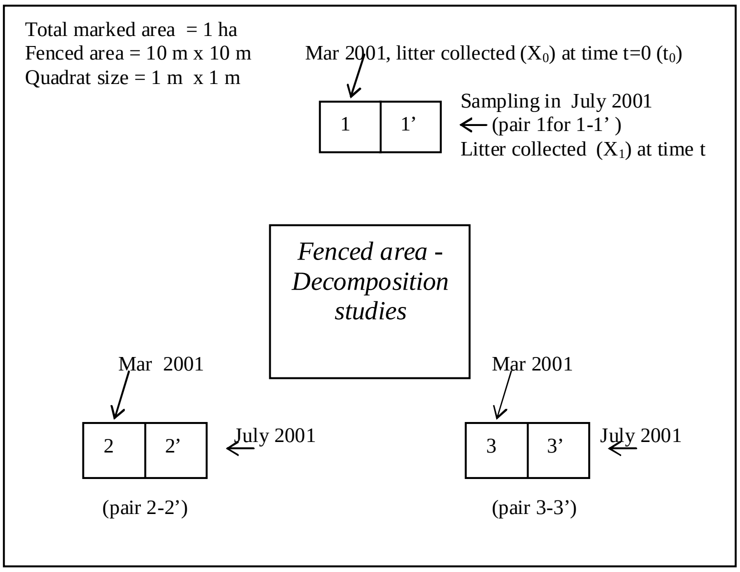


Fig 7.1. Paired plot design for litter production measurements.

The litter study involved measurements for:

- Seasonal decomposition.
- The amount of litter production during a season.
- Nutrient release during different seasons in a year.

Litter bags (technique proposed by Wiegert and Evans 1964) made of nylon mesh (mesh size 1 mm x 1.5 mm) were used for seasonal and annual decomposition. The size of a litter bag and the weight of litter per bag were calculated based on the amount of litter distributed per unit area in the field. An average value of 5 g in each 15 cm x 15 cm bag was used for all the sites. Litter collected in Mar 2001 was kept for seasonal decomposition from Apr 2001 till Aug 2001 (Table 7.2). For Aug 2001 to Dec 2001 decomposition, litter collected in July 2001 was used for seasonal decomposition over that period, and so on for Dec 2001 to Apr 2002. Longer term decomposition was studied on litter collected in Mar 2001, from Apr 2001 until Apr 2002, regularly at four (Aug 2001), eight (Dec 2001) and 12 (Apr 2002) month intervals (Table 7.2).

Table 7.2. Sampling time for litter production, and for seasonal and longer term decomposition.

Litter production	Seasonal litter decomposition with litter bags-			Longer term litter decomposition with litter bags-		
	left on		*collected on*	*left on*		*collected on*
18-24th Mar 2001	1st Apr 2001	→	12th Aug 2001	1st Apr 01	→	12th Aug 2001
24-30th July 2001	12th Aug 2001	→	13th Dec 2001		↘	13th Dec 2001
26-30th Nov 2001	13th Dec 2001	→	11th Apr 2002		↘	11th Apr 2002
24-30th Mar 2002						

Litter bags were placed on the soil surface for decomposition in a fenced area of 10 m x 10 m located centrally at each site to avoid any disturbance due to grazing (Fig 7.1). Three litter bags per site were collected at the end of time period, cleaned of soil, and dried to a constant weight. Likewise three bags were removed at four month intervals for the longer term decomposition trial. The coefficient of variation of litter production and litter decomposition measurements for all data over a year and for data within each sampling date ranged from 17-35 %.

The rate of litter decomposition (R) for time period t (months) was calculated as:

$$R = \frac{X_0\,(g) - X\,(g)}{X_0 \times t}$$

where X_0 = original weight of litter left for decomposition

X = weight of litter left undecomposed at time period t
t = time period

The decay constant (k) for annual rate of decomposition was calculated from $X/X_0 = e^{-kt}$, where t = time in years, and k = the annual decay constant (Olsen 1963 cited by Blair 1988).

Based on seasonal decomposition rates (R1, R2, R3 for different seasons as in Table 7.3) and amount of litter collected during different seasons, litter production per season was computed as in Table 7.3. The method assumes that the amount of litter produced at a particular time of sampling at paired plots is similar. For each sampling date, litter was collected from one of the paired plots and left uncollected in the adjacent paired plot for a particular time (t_0). For the next sampling date (t time), the amount of litter added to the plot that was left uncollected at t_0 represents the amount of litter produced during that period (t_0-t) (e.g. in Fig 7.1- Quadrats 1 and 1' had X_0 amount of litter at time t_0, after t time quadrat 1' had X_1, so the amount of litter produced for time t_0-t = X_1-X_0). Since the previous amount of litter (X_0) underwent decomposition during time t_0-t, the seasonal rates of decomposition were of necessity computed and used to account for decomposed litter where precisely calculating the total amount of litter produced in a season (Table 7.3).

Table 7.3. Calculations for litter production during different seasons.

Amount of litter collected	Seasonal decomposition rate	Amount of litter produced during different seasons
Mar 2001 = X_0		
July 2001= X_1	Apr 2001-Aug 2001= R_1	Mar 2001-July01= X_1-X_0+R_1
Nov 2001= X_2	Aug 2001-Dec 2001= R_2	July 2001-Nov01= X_2-X_1+R_2
Mar 2002 = X_3	Dec 2001-Apr 2002 = R_3	Nov 2001-Mar02 = X_3-X_2+R_3

After each of four, eight and 12 months of litter decomposition, the samples from three litter bags for each sampling date were separately dried, manually cleaned to remove soil particles, bulked and ground for C, N and P analysis. Litter samples collected in Mar 2001 (without decomposition) from each site were also thoroughly mixed, ground, and analysed for C, N and P.

Percent nutrient remaining in undecomposed litter at t time was computed according to Blair (1988) as:

% Litter mass remaining x nutrient concentration in the remaining mass (%)

= ---

Initial concentration of a nutrient

The total amount of nutrient released in relation to total amount of litter produced at each site was calculated from the amount of litter decomposed during the time from t_0 to t and nutrient content associated with decomposed litter at t_0 and t time, as follows:

Amount of nutrients at t_0 in litter (Y_0) = Amount of litter (X_0) x Initial nutrient concentration

At t, amount of litter decomposed = X_t,

Thus nutrient content at t in remaining litter (Y_1) = Amount of litter left undecomposed (X_0-X_t) x Nutrient concentration at t.

The amount of nutrients released the time from t_0 to t = Y_0-Y_1

Statistical analysis

Amount of litter production, decomposition, nutrient composition and nutrient release were analysed using residual maximum likelihood technique (REML) (Genstat 2002). In REML analysis, all the uncleared treatments of a tree community were considered as replicates while cleared treatments for each age group were taken as such with no replication within a tree type, but replicated across different tree types. The results are reported for uncleared and at recent, medium and old age of clearing for each tree community.

The main effects of community*cleared-uncleared and random effects of time-since-clearing for recent, medium, old cleared and uncleared treatments were analysed for three types of tree communities. Where interactions were significant relevant tables/graphs are presented. The differences between treatments were computed from LSDs (least significant differences of means) based on an exact value for each treatment, but only the average LSD values are presented for convenience. The means from REML analysis were used to present the results.

Results

Litter production

The amount of litter production varied during different seasons for different tree communities and their cleared treatments (Table 7.4). Uncleared sites generally produced more litter than did the cleared sites.

E. melanophloia and *A. harpophylla* woodlands produced the maximum amounts of litter during Nov 2001 to Mar 2002, while *E. populnea* did the same in July to Nov 2001 (Table 7.4). There was no significant difference (at $P<0.05$) in the amount of litter produced between cleared and uncleared sites of *E. melanophloia*, irrespective of season. Uncleared *A. harpophylla* produced a significantly (at $P<0.05$) greater amount of litter compared to cleared sites for all the seasons, as was so for *E. populnea* in July to Nov 2001 (Table 7.4). The total amount of litter produced per annum (kg/ha/yr) was at a maximum for uncleared and least for recently cleared sites for all the tree communities. Within the cleared treatments, medium clearing had notably greater amounts of litter than the recent and the oldest clearing (Table 7.4).

Table 7.4. Litter production (kg/ha) at uncleared and cleared (recent, medium and old age) sites for *E. populnea*, *E. melanophloia* and *A. harpophylla* communities.

		E. populnea	*E. melanophloia*	*A. harpophylla*	*LSD**
Mar - July 2001	Uncleared	448	523	883	*379.9*
	Recent	211	234	365	
	Medium	482	351	322	
	Old	189	313	231	
July -Nov 2001	Uncleared	786	583	774	*380.2*
	Recent	383	401	259	

	Medium	265	520	497	
	Old	401	416	516	
Nov 2001-Mar 2002	Uncleared	498	842	939	*427.9*
	Recent	272	472	722	
	Medium	552	645	372	
	Old	360	496	336	
Total (kg/ha/yr)	Uncleared	1732	1948	2596	*873.3*
	Recent	755	996	917	
	Medium	1369	1586	1261	
	Old	990	1266	1124	

*LSD represents average value of least significant differences of means ($P<0.05$) for all three communities at any one sampling date.

Litter decomposition

Seasonal decomposition

Litter decomposed at a greater rate during Aug-Dec in *E. melanophloia*, while the rate did not vary much across different seasons in A. *harpophylla* and *E. populnea* (Fig 7.2). Recently cleared sites for all the tree communities supported significantly ($P<0.05$) greater rates of litter decomposition than the uncleared sites with few exceptions. There were no notable differences in decomposition of litter at uncleared and medium or old cleared sites in *E. populnea* and *E. melanophloia*. Uncleared *A. harpophylla* had lower rates of decomposition than old cleared sites during Aug to Dec 2001 and Dec 2001 to Apr 2002 (Fig 7.2).

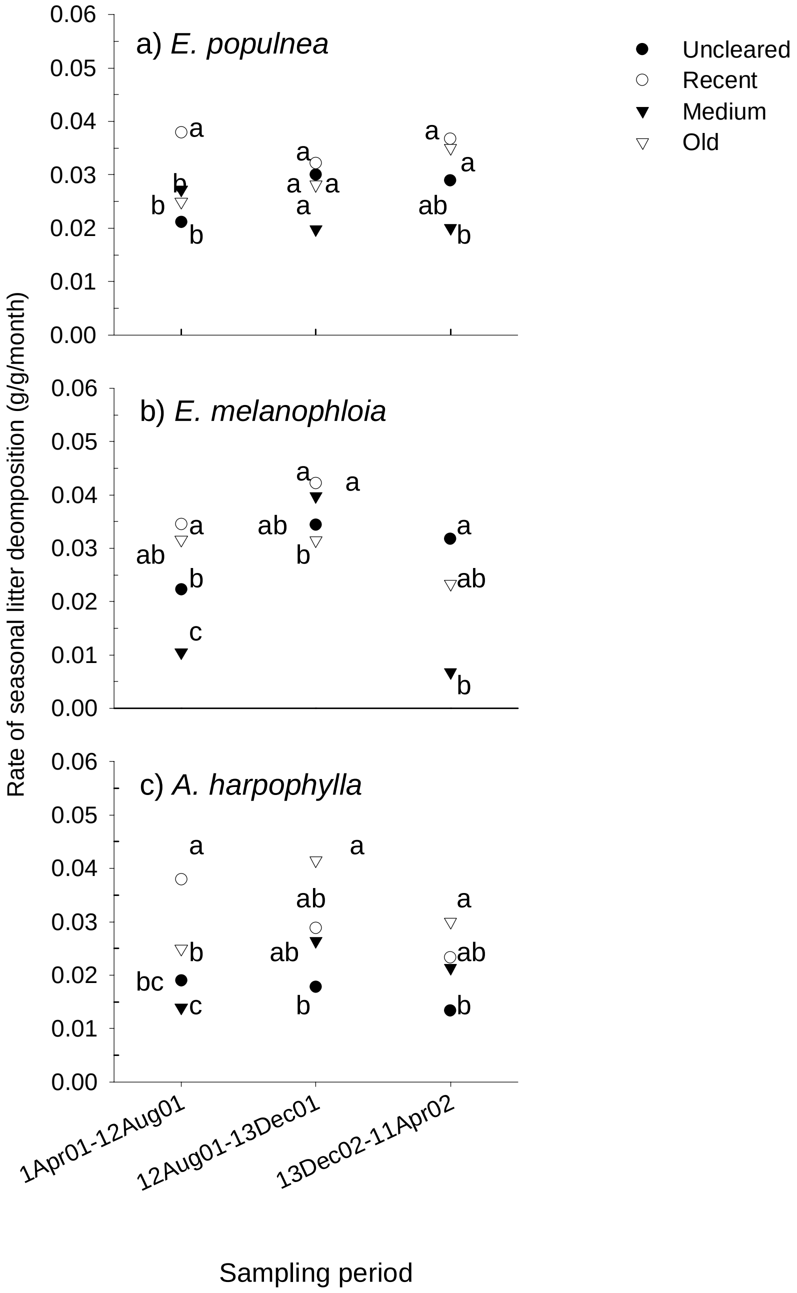


Fig. 7.2 Seasonal decomposition rate for litter produced at cleared and uncleared sites for a) *E. populnea*, b) *E. melanophloia* and c) *A. harpophylla* communities.

Any two treatments with different letters on any one sampling date within a tree community denote significant difference at P<0.05.

Longer term decomposition

The proportion of litter remaining after one year of decomposition was greatest at the uncleared sites (i.e. litter decomposition proceeded slower than at other sites). The average litter loss over a year at uncleared sites was 15-20 per cent of original mass while cleared sites had 25-35 per cent mass loss (Fig 7.3).

The maximum proportion of litter was decomposed at recently cleared sites for all tree types, followed by the oldest clearing in *E. melanophloia* and *A. harpophylla* (Fig 7.3).

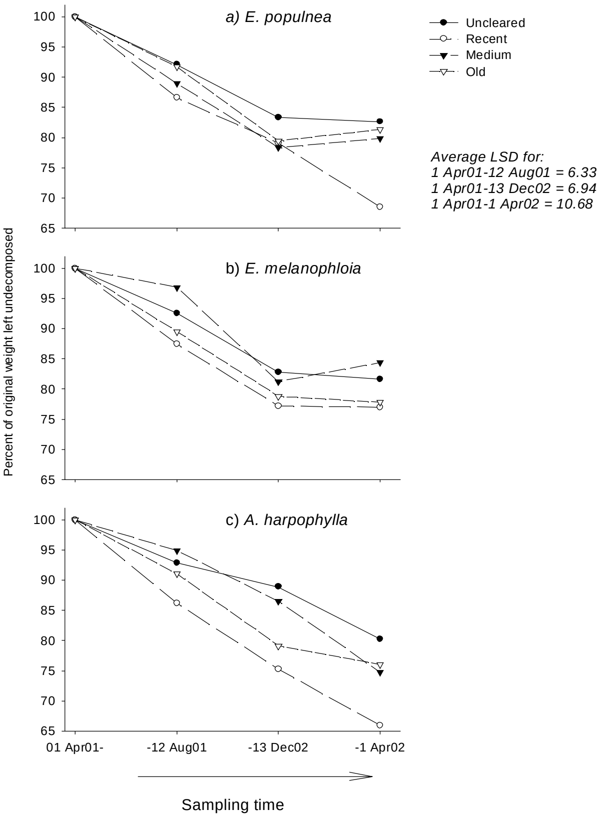


Fig. 7.3 Decomposition of litter at cleared and uncleared sites over one year for a) *E. populnea*, b) *E. melanophloia* and c) *A. harpophylla* communities.
Average LSD at any one sampling time denotes average value of least significant difference of means at P<0.05.

The decay constant (k) reflected the faster rate of litter decomposition at recently cleared sites with k values of -0.25 to -0.38 and half lives of 1.8 to 2.6 years, compared to k values of -0.19 to -0.20 and half lives of 3-4 years of uncleared sites (Table 7.5).

Table 7.5. Decay constants (k) and half lives calculated over one year period of decomposition of litter at cleared and uncleared sites for *E. populnea*, *E. melanophloia* and *A. harpophylla* communities.

Tree Community	Treatment	k	Half life (yrs)
E. populnea	Uncleared	-0.19	3.64
	Recent	-0.38	1.83
	Medium	-0.22	3.08
	Old	-0.21	3.36
E. melanophloia	Uncleared	-0.20	3.41
	Recent	-0.26	2.65
	Medium	-0.17	3.09
	Old	-0.25	2.76
A. harpophylla	Uncleared	-0.20	3.41
	Recent	-0.26	2.65
	Medium	-0.17	3.09
	Old	-0.25	2.76

C, N and P concentrations in litter during longer term decomposition

At the beginning of the experiment (Mar 2001) C and N concentrations were greater in litter from uncleared than those from cleared sites at *E. populnea* (except N concentrations at the recent cleared site) and *A. harpophylla*, but not for P. In contrast, *E. melanophloia* showed no notable difference for C, N and P concentrations between cleared and uncleared sites (Fig 7.4.1, 7.4.2 and 7.4.3).

At uncleared sites for all the tree communities, C concentrations changed only slightly over time of decomposition (Fig 7.4.1). In contrast, C was lost as decomposition proceeded at recent and medium clearing for *E. populnea,* and medium and old clearing for *A. harpophylla,* while no notable change occurred in *E. melanophloia* (Fig 7.4.1).

As decomposition continued, concentrations of N in litter increased after a slight initial decline in Aug 2001 at all sites other than the recent and medium clearing for *E. populnea* and *E. melanophloia* respectively (Fig 7.4.2a,b and c).

P concentration compared to its initial concentration in undecomposed litter declined as decomposition proceeded for all the cleared sites other than the oldest clearing for *E. melanophloia*. The decline over time was not continuous. At uncleared sites, P accumulated with decomposition over time in *E. populnea* in all the seasons, and after a little loss in Aug 2001 in *A. harpophylla*, while no notable difference was evident over time of decomposition in uncleared *E. melanophloia* (Fig 7.4.3a, b and c).

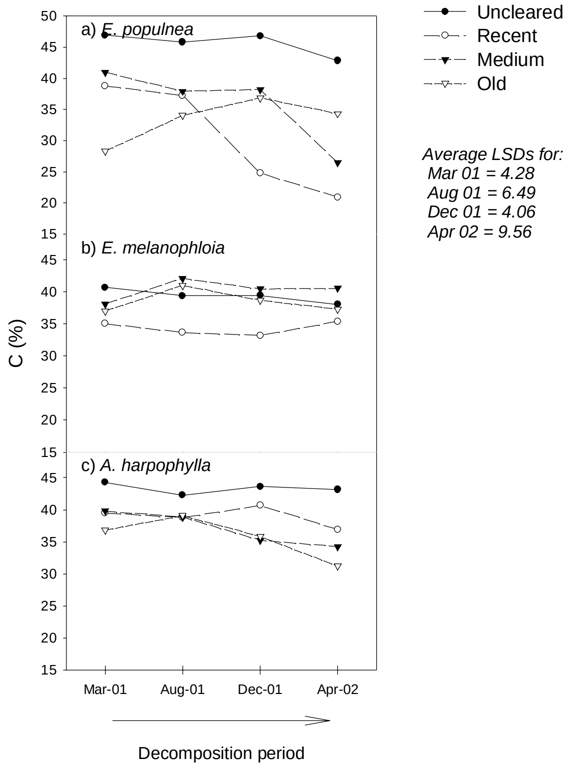


Fig 7.4.1. Carbon concentration

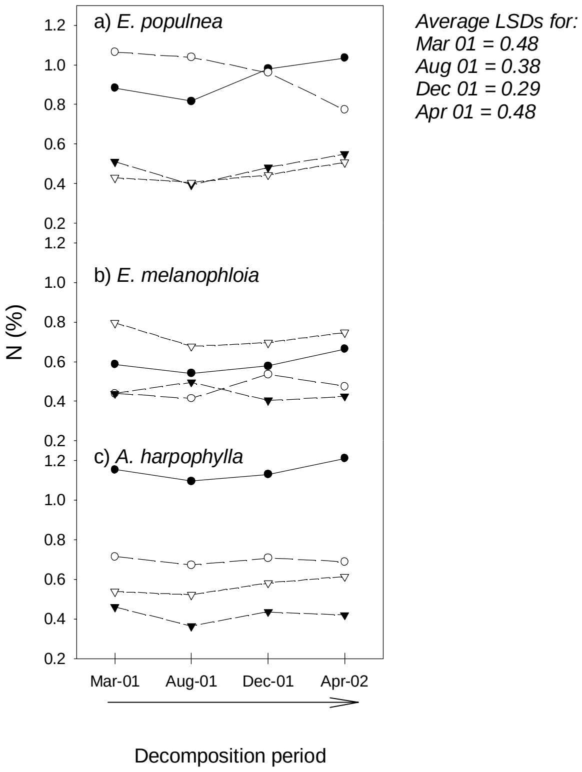


Fig 7.4.2. Nitrogen concentration

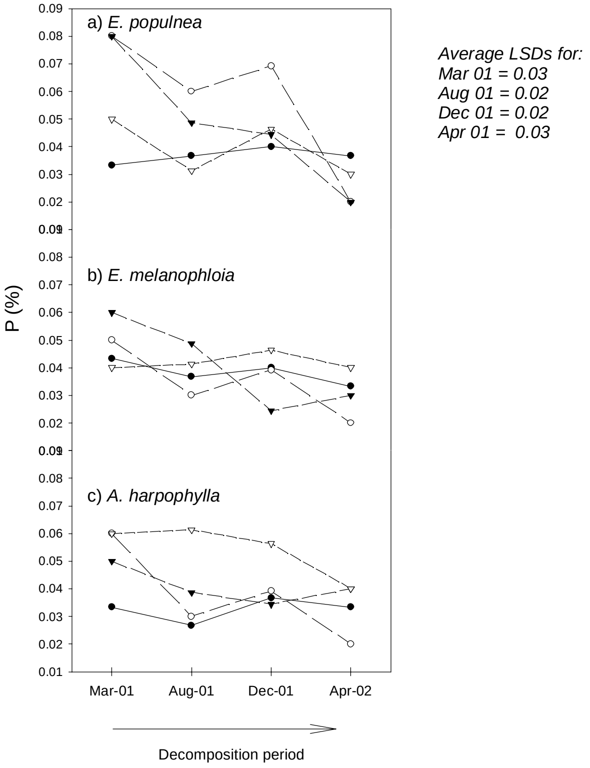


Fig 7.4.3. Phosphorus concentration

Fig 7.4. (1, 2 and 3) Carbon, nitrogen and phosphorus concentration in litter during a one year period of decomposition at cleared and uncleared sites for a) *E. populnea*, b) *E. melanophloia* and c) *A. harpophylla* communities.

Average LSD represents the least significant difference of means at P<0.05 for any one decomposition period.

Based upon the amount of nutrients remaining in the litter mass at any sampling date of decomposition, it was evident that C, N and P were released as decomposition proceeded at all sites, with few exceptions (Fig 7.5). There was no release of C at the oldest clearing, and of N except a little initial decline at uncleared sites for *E. populnea* (Fig 7.5.1a and 7.5.2a). P was released at all the sites except at uncleared *E. populnea* and *A. harpophylla* (Fig 7.5.3a and c). The smaller release of P and N than expected based upon the litter mass at uncleared compared to cleared sites occurred because the rate of incorporation of these nutrients into decaying litter exceeded the loss of litter mass over time of decomposition (Fig 7.5).

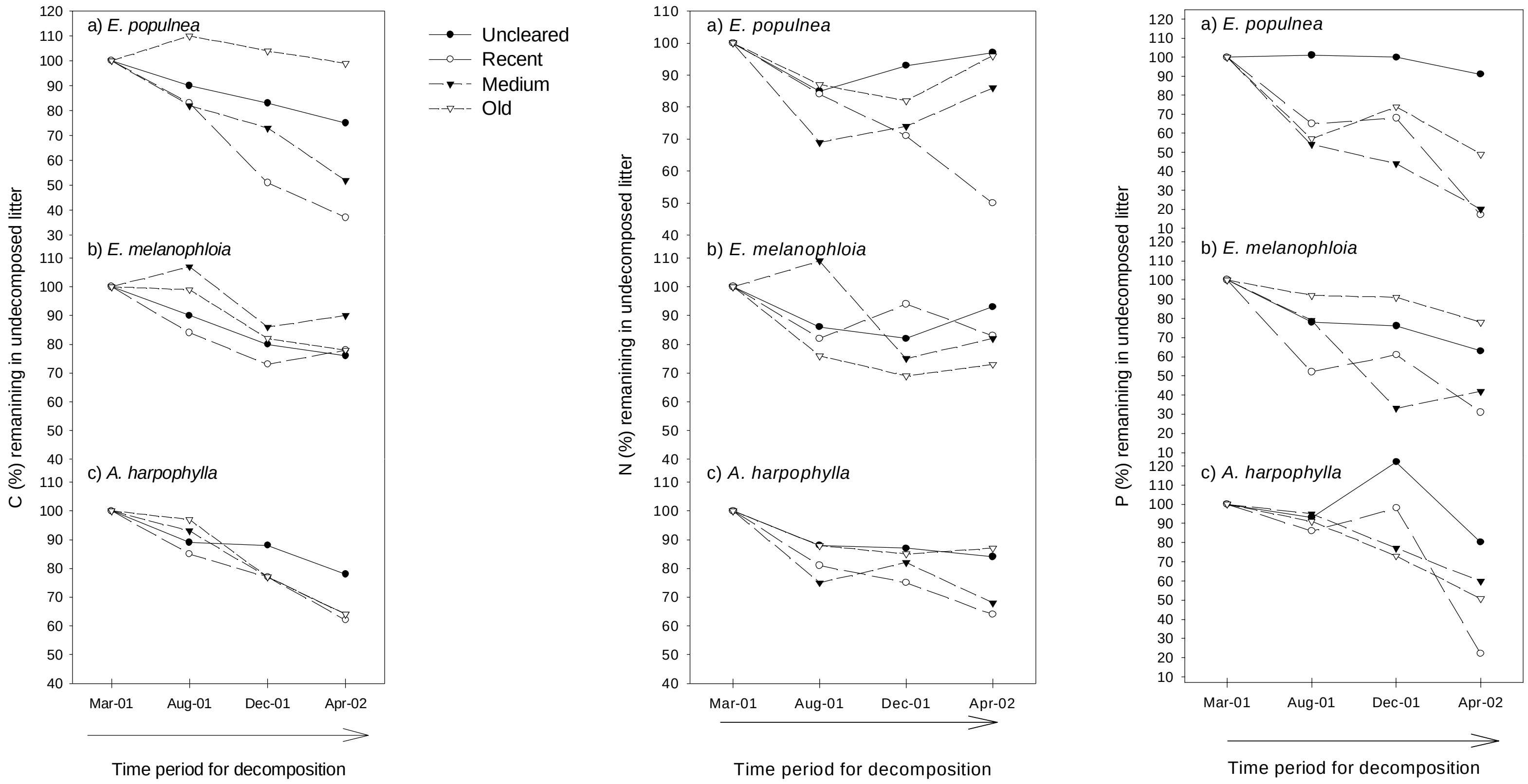


Fig 7.5.1. Carbon

Fig 7.5.2 Nitrogen

Fig 7.5.3 Phosphorus

Fig 7.5. (1, 2 and 3) The percent content of carbon, nitrogen and phosphorus in undecomposed litter at any sampling date during one year period of decomposition at cleared and uncleared sites for a) *E. populnea*, b) *E. melanophloia* and c) *A. harpophylla* communities.

The content of nutrients released (%) after one year period of decomposition compared to their original content in undecomposed litter showed the greater loss of C, N and P at the recent cleared than the uncleared sites (Table 7.6). Between cleared sites, the nutrient loss was at a maximum at the recent clearing compared to the medium or the oldest clearing after one year of decomposition (Table 7.6).

Table 7.6. The content of nutrients released after 12 months decomposition at cleared (recent, medium and old) and uncleared sites for *E. populnea*, *E. melanophloia* and *A. harpophylla* communities.

Nutrient	Tree community	Uncleared	Recent	Medium	Old	*LSD**
C (%)	*E. populnea*	24	64	46	1	*20.60*
	E. melanophloia	23	23	7	23	
	A. harpophylla	22	39	33	37	
N (%)	*E. populnea*	3	51	14	4	*42.56*
	E. melanophloia	7	17	18	29	
	A. harpophylla	15	36	32	13	
P (%)	*E. populnea*	9	83	80	51	*67.18*
	E. melanophloia	37	69	58	22	
	A. harpophylla	20	78	40	49	

** LSD represents the average value of least significant difference of means at $P<0.05$*

Discussion

Uncleared *E. populnea*, *E. melanophloia* and *A. harpophylla* produced greater amounts of litter per annum (range of 1732-1948 kg/ha/yr for eucalypt and 2596 kg/ha/yr for acacia communities) compared to their cleared or open grassland (755-1586 kg/ha/yr). Similar values over the range of 900-2700 kg/ha/yr for tree litter in eucalypt and corymbia communities of central Queensland were reported by Burrows and Burrows (1992). However, the values reported here are greater than litter production (720-1270 kg/ha/yr) in eucalypt communities (*Eucalyptus crebra, E. drepanophylla, C. erythrophloia*) of northern Queensland (McIvor 2001). There has been no detailed study in the region on litter decomposition and release of nutrients in cleared and uncleared systems. The present study in this semi-arid region fills this gap, and reveals decomposition behaviour and release of nutrients from ground litter in open grassland and woodland pastures.

The annual ground litter production was greater in woodlands than the open grasslands for all the tree communities, since the former had litter from trees as well as from understorey vegetation (Table 7.4). However, the decomposition of litter was somewhat faster at cleared especially at recent cleared sites compared to uncleared sites (Fig 7.3). The litter produced at cleared sites from grass and other herbaceous plants was less lignified compared to some of the litter (herbaceous and woody plants) obtained from vegetation at uncleared sites and that resulted in faster decomposition at cleared sites. Another important factor governing litter decomposition is the C:N ratio. Robertson *et al.* (1994) showed that the greater C:N ratio (50-75) led to a greater microbial activity in pasture systems compared to the cropping systems with C:N ratio of 36-46 (sorghum). In the present study, the C:N ratio was greater in litter produced at cleared than uncleared sites (Table 7.7) but only significantly so for the leguminous tree community, *A. harpophylla*. Litter from *E. populnea* and *E. melanophloia* woodlands showed comparable rates of decomposition to that at cleared sites (except the recent clearing) (Fig 7.2 and 7.3), probably because of the similarity of their C:N ratio (Table 7.7), given that greater concentrations of cellulose promote decomposition (Berg 2000). Litter at uncleared *A. harpophylla* had a very low C:N ratio (Table 7.7), and decomposed slowly (Fig 7.2 and 7.3).

Table 7.7. Mean (± standard error of mean) C: N ratio in undecomposed litter at cleared (average for all the three time-since-clearing treatments) and uncleared sites for *E. populnea*, *E. melanophloia* and *A. harpophylla*.

	Cleared	Uncleared
E. populnea	61 (± 13.8)	55 (± 6.51)
E. melanophloia	72 (± 13.2)	70 (± 5.24)
A. harpophylla	70 (± 9.39)	38 (± 1.76)

Release of C, N and P (calculated according to loss of litter mass) was generally faster at cleared sites than in woodlands, with few exceptions (Fig 7.5). Over one year period of decomposition, litter in the recent clearing lost most nutrients stored therein. This was followed by medium and then old age of clearing (Table 7.6). More importantly, the N and P release through litter decomposition declined with age of clearing, especially from recent to medium or old clearing (Table 7.6).

There was no significant difference between cleared and uncleared systems in the amounts of C, N and P released from the total litter production per annum at each site (Table 7.8). Within cleared pastures, the amount of N and P loss from the total litter produced in a year showed a tendency to decline with age of clearing (Table 7.8). In particular return of C, hence the organic matter from litter may not be sustained following clearing.

Table 7.8 Release of C, N and P (kg/ha/yr) according to total amount of litter produced at recent, medium and old cleared, and uncleared sites for *E. populnea*, *E. melanophloia* and *A. harpophylla*.

	Uncleared	Recent	Medium	Old	LSD*
C	214	95.8	173	104.6	*248.74*
N	2.14	2.22	1.11	1.41	*5.567*
P	0.19	0.42	0.56	0.23	*0.399*

* LSD denotes the least significant difference of means in a row at $P < 0.05$, *in the absence of significant interaction between tree community and time-since-clearing. The values are presented for a common response among all the communities.*

The amount of C, N and P release from annual litter production over a year was in fact underestimated in Table 7.8, due to their incorporation in decomposed litter (Fig 7.4), given that the decomposition process was not yet complete. The release of nutrients from total litter production at a site may have been greater if the litter had been completely decomposed. In one year of the present decomposition study, P largely showed a loss over time compared to the initial concentration in undecomposed litter while N and C did not show much loss of concentration, they indeed accumulated at a few sites, especially uncleared ones (Fig 7.4). Release of N in beech litter (*Fagus sylvatica* L.) over three years decomposition was studied in detail by Zeller *et al.* (2000). They explained that incorporation of N in decomposed litter occurs for the initial years of decomposition and is derived from external sources such as throughfall (rain/mist that trickles through the tree canopy), soil fauna, fungi and bacteria.

Though the nitrogen concentration was greater in litter produced at uncleared *A. harpophylla*, *E. populnea* and *E. melanophloia* than their cleared sites with few exceptions, there was minimal or no release of N during decomposition especially at uncleared sites (Fig 7.4.2). The greater N concentrations may be responsible for slow decomposition of litter at uncleared sites since N content in litter has a negative linear relationship with decomposition once cellulose is decomposed (Berg 2000). During decomposition in the present study, C concentrations decreased while N increased or changed only a little. Moreover, eucalypt leaves have a waxy coating, and are hard to decompose (Bernhard-Reversat 1999).

Decomposition of litter not only depends upon litter composition but also upon soil type, microbial communities, soil physical properties and the climate. All these factors influence the release of nutrients (Kutsch and Dilly 1999). In the present study, soil microbial biomass, organic carbon, pH and cation exchange capacity (CEC) were studied to determine their relationship with litter decomposition but sampling was done only once between Jan to Mar 2001, in contrast to sampling for litter decomposition which was done each season. Perhaps the effects of soil properties on litter decomposition and microbial activities responsible for litter decomposition, vary with seasons.

The annual decomposition rate (over a year) was positively correlated with soil pH (correlation coefficient $r = 0.26$ at $P=0.05$) and CEC ($r = 0.50$ at $P=0.05$) (Fig 7.6).

For seasonal decomposition rates, the summer phase of decomposition (Dec-Apr) was correlated with the amount of soil microbial biomass of carbon ($r = 0.30$ at $P=0.05$) and nitrogen ($r = 0.56$ at $P=0.05$), and soil organic carbon content ($r = 0.30$ at $P=0.05$) (Fig 7.6).

The variable relationship between decomposition and soil properties for seasonal and annual rates could be due to change in microbial activities in different seasons. The positive relation between decomposition rate in summer (Dec-Apr) and soil microbial biomass, and soil organic carbon was most likely due to rainfall as decomposition rate for Dec-Apr was significantly correlated with rainfall ($r = 0.51$).

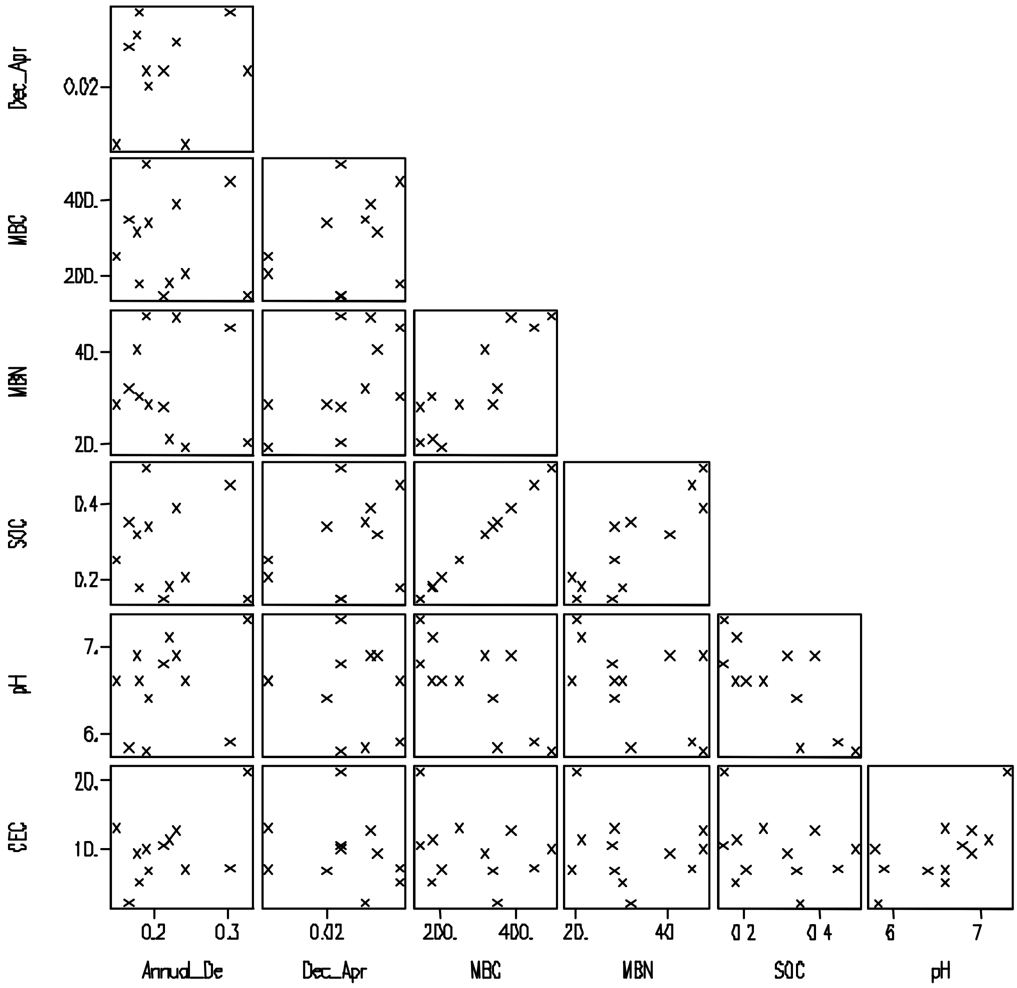


Fig 7.6. Relationship between different soil parameters (for 0-5 cm depth) and litter decomposition:

Annual_De - Annual rate of decomposition of litter (g/year)
Dec-Apr - Seasonal decomposition during summer (Dec 01-Apr01) (g/month)
MBC - Soil microbial biomass carbon (mg/kg of soil)
MBN - Soil microbial nitrogen (mg/kg of soil)
SOC - Soil organic carbon (g/kg of soil)
pH
CEC - Cation exchange capacity (meq/100g soil)

The slow rates of decomposition at uncleared sites may be an adaptation of eucalypt and acacia woodlands in Australian soils. From an ecological perspective, Grubb (1989) explained with examples from different ecosystems that poor soils support vegetation communities, which are adapted to poor nutrient status. There is a two-way relationship between structure or type of vegetation communities and soils, and it is still not clear which plays a greater role to determine the other (Grubb 1989). Probably, the sclerophyllous nature of eucalypt and acacia communities growing in central Queensland region is an adaptation to nutrient poor soils and to the semi-arid climate. Slow but continuous release of nutrients from litter could be a result of this adaptation. It may also benefit the system from leaching of nutrients (a problem in many ecosystems (Whitmore 1989)).

The nutrient content released upon clearing and to a lesser extent through the faster rate of litter decomposition in grasslands than uncleared communities, may contribute to greater pasture production compared to that in woodland pastures (pasture production discussed in Chapter 4; Figures and Tables 4.2 and 4.3). However, the content of C and N stored in litter produced at uncleared sites (though it decomposed slowly) was more than at cleared sites (Table 7.9). With clearing, the steady return of nutrients as seen in uncleared sites would have been markedly reduced (Table 7.9) for the reason that the content of nutrients contained in litter would be less with lesser amount of litter produced at cleared compared to uncleared sites.

Table 7.9. The potential content of nutrients (kg/ha) stored in litter produced at uncleared, and recent, medium and old cleared sites for *E. populnea*, *E. melanophloia* and *A. harpophylla* communities.

Site		Uncleared	Recent	Medium	Old
E. populnea	C	812.43	296.45	542.28	288.53
	N	15.30	8.04	6.63	4.49
	P	0.58	0.60	1.10	0.50
E. melanophloia	C	791.94	353.70	582.72	478.84
	N	11.40	4.38	6.56	10.39
	P	0.84	0.50	0.95	0.51
A. harpophylla	C	1149.44	366.72	484.64	423.26
	N	29.97	6.55	5.49	6.32
	P	0.87	0.55	0.63	0.67

In woodlands, nutrient cycling is regular but slow, thus the pasture productivity remains more or less consistent. In eucalypt woodlands, allelochemicals (Bernhard-Reversat 1999), or antimicrobial activities of their essential oils (Dellacassa *et al.* 1989), or the toxic effect of allelochemicals on soil microbes (Chander *et al.* 1995) contribute to slow nutrient cycling from litter. In addition to this, tree-grass competition for soil nutrients, moisture and radiation (Burrows *et al.* 1988; Scholes and Archer 1997) reduces the supply of nutrients for understorey grasses and this leads to lesser productivity. Unfortunately, there are no data available on the effect of allelochemicals released from eucalypts species in their native soils.

Most of the results on allelopathy (as mentioned above) in eucalypts have been confirmed from laboratory experiments and/or in-field studies conducted in other countries.

It would be interesting to study how the microbial communities decompose litter from these trees in their natural environment and how this relates with their soils; perhaps both have adapted over time to poor nutrient status and compromised with a slow but stable return of nutrients.

Chapter 8. Synthesis

Questions about long term sustainability of cleared pastures compared to that of uncleared (intact) woodland pastures, the effects of tree clearing on pasture yield, soil nutrient profile, soil microbial biomass and nutrient release through litter were addressed individually in the previous Chapters. The comparisons of recent (5 yrs), medium (11-13 yrs) and old (33 yrs) times-since-clearing and uncleared treatments for *Eucalyptus populnea, E. melanophloia* and *Acacia harpophylla* communities conducted on one property in central Queensland provided tangible data to demonstrate the effects of clearing over time. The sites were selected as typical of the region with guidance from researchers at EPA and NR&M, Emerald. The vegetation structure was almost uniform at all the marked plots of a particular type of tree community, however there was some variation in soils at paired medium cleared and uncleared *E. melanophloia* (basaltic parent rock with high P levels) compared to the paired plots for recent and old cleared and uncleared *E. melanophloia* (non-basaltic soils with normal P levels).

To assess the overall impact of clearing in pasture systems, the interactive effect of various parameters such as pasture biomass, pasture species composition, soil properties and litter production are presented in this Chapter. This should provide integrated information on losses and benefits from clearing to develop pastures in the studied three tree communities.

Materials and methods

Data were analysed for all the parameters undertaken for the study using multivariate analysis techniques such as cluster analysis and canonical variates analysis (CVA) in Genstat ver. 6.0 to evaluate the overall impact of clearing in all tree communities. Based on preliminary analyses (data not presented), it was concluded that pasture biomass (aboveground) and species diversity at fenced and unfenced plots, annual litter production, soil NO_3-N, soil organic carbon, soil pH_w, available P in soil, and soil microbial biomass for C and N were the primary variables to manifest the impact of clearing. The selected soil parameters are also important predictors of productivity as determined from earlier correlation and graphical analysis (Chapters 4, 5, 6 and 7 and preliminary analyses). Approximately equal numbers of soil and plant variables were used in the multivariate analyses to avoid the dominance of either soil or plant related parameters.

Cluster analysis was performed to determine whether it was possible to group cleared and uncleared sites based upon the selected parameters. For this, abovementioned plant and soil attributes were standardized. The similarities were generated using a Euclidean metric $(1-\{(x_i - x_j) / \text{range}\}^2)$ and the group average criterion (minimum distance between cluster means) were used to form clusters. The analysis investigated similarities among sites based on the primary variables already identified above. Closely-related sites cluster together while less related sites form distinct clusters based on their similarity index.

Further, these primary data were subjected to CVA to determine the parameters that would best lead to differentiation between clearing treatments and between tree communities. The CVA analysis finds linear combinations of the original variables that maximize the ratio of between-group to within-group variation where groups are either clearing treatments or tree communities. Two canonical variates were considered to explain variation between clearing treatments and between tree communities.

Results and discussion

Cluster analysis was used to group sites (cleared or uncleared) based on their similarity for pasture biomass and species diversity at fenced and unfenced plots, annual litter production, soil NO_3-N, soil organic carbon, soil pH_w, available P in soil, and soil microbial biomass for C and N. The impact of clearing was apparent for *E. populnea* and *A. harpophylla*, as most of the uncleared and the recently cleared sites clustered together (at 86 per cent similarity level) whereas the medium and the oldest cleared sites formed separate groups (Fig 8.1). In contrast, *E. melanophloia* did not show any effect of clearing at 84 per cent level of similarity as recent and medium cleared, and uncleared sites grouped together; and the oldest cleared and uncleared site clustered together in association with old clearing for *E. populnea* and *A. harpophylla* (Fig 8.1).

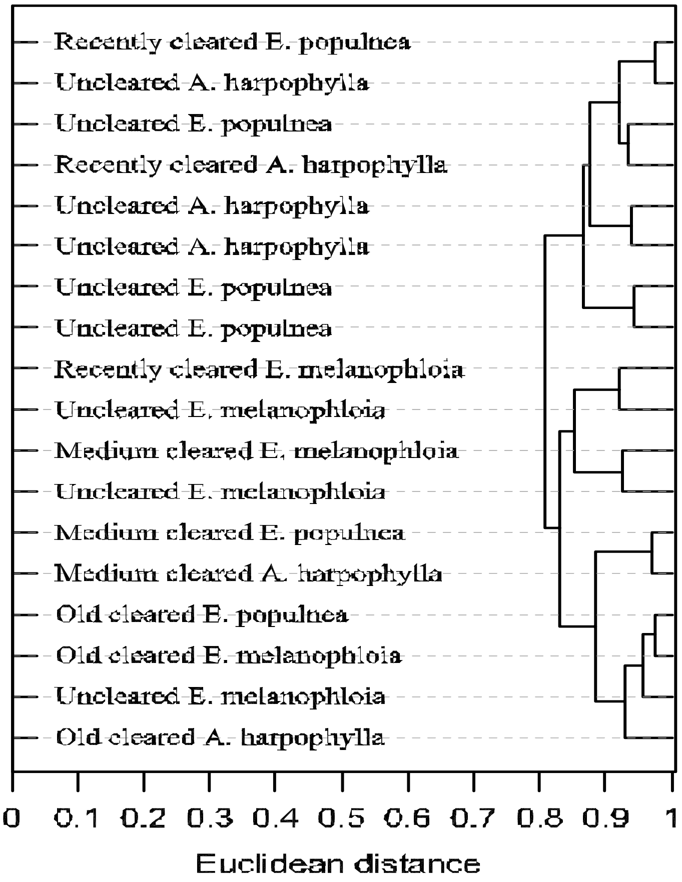
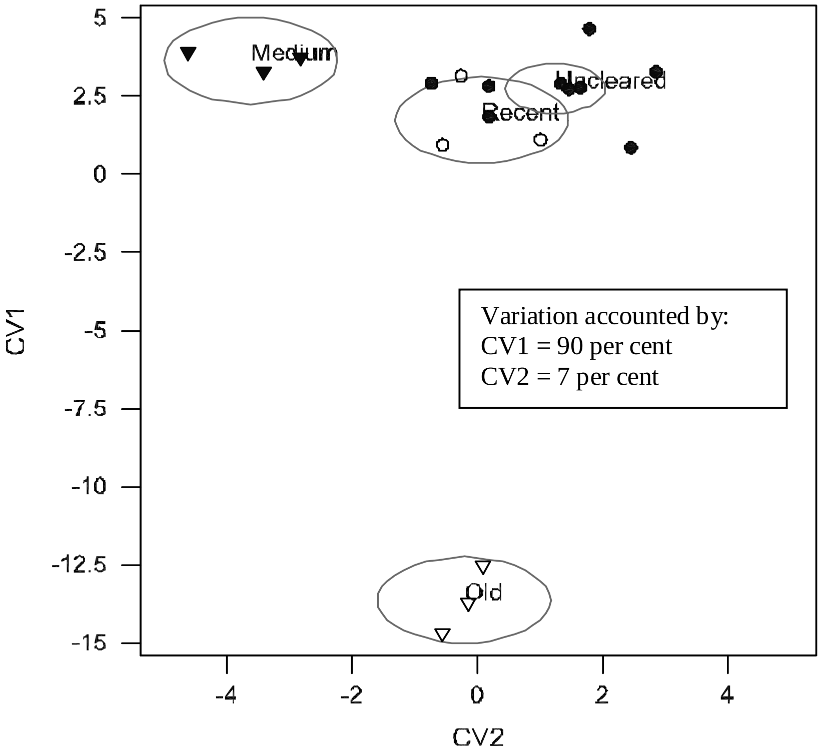


Fig 8.1. Cluster analysis for various cleared and uncleared treatments in *E. populnea*, *E. melanophloia* and *A. harpophylla*.

Clearing effects varied with time-since-clearing for the analysed parameters. In *E. populnea* and *A. harpophylla*, each of recent, medium and old cleared treatments formed separate clusters, however these clusters included cleared and uncleared sites for *E. melanophloia*. In *E. melanophloia*, both recent and medium cleared treatments grouped together, but were separate from the oldest clearing; and at the same time each cluster also contained uncleared *E. melanophloia*. The oldest clearings for all the tree communities associated together at 92 per cent level of similarity along with uncleared *E. melanophloia* (Fig 8.1).

All the treatments were clustered into two groups at 82 per cent similarity: uncleared *E. populnea* and *A. harpophylla* with their recent clearing, and cleared treatments along with uncleared *E. melanophloia* (Fig 8.1). Uncleared *E. melanophloia* was different at 82 percent similarity to uncleared *E. populnea* and uncleared *A. harpophylla*. The clearing effects were more clearly manifested in *E. populnea* and *A. harpophylla* but not in *E. melanophloia*.

In CVA, the first canonical variate (CV1) explained 90 per cent of variation among clearing treatments and distinguished the oldest cleared treatment from medium and recent cleared, and uncleared treatments (Fig 8.2). CV1 was mostly influenced by soil NO_3-N, pasture biomass, litter production, species diversity and soil pH_w. Comparatively greater values of +ve loadings for these parameters than the others showed that recent and medium cleared, and uncleared treatments had better soil status with greater pasture biomass and litter production than that at old cleared treatments (Table 8.1). The maximum value of negative loading for pH_w demonstrated that pH_w was greater at the oldest clearing than at medium and recent cleared, and uncleared treatments (Table 8.1).



Variation accounted by:
CV1 = 90 per cent
CV2 = 7 per cent

Fig 8.2. Relationship between first and second canonical variates for cleared (recent, medium and old) and uncleared treatments (with 95 per cent confidence regions around means).

A further 7 % of the variation among clearing treatments was accounted by the second canonical variate (CV2). CV2 showed that the medium cleared treatment was different to the recent and old cleared, and uncleared treatments (Fig 8.2). The species diversity and soil microbial biomass for C and N had a greater relative influence on CV2 than did other parameters (Table 8.1). The greater values of -ve loadings for pasture yield, than for example litter production and soil NO_3-N, explained the lesser pasture biomass at old and recent clearing, and uncleared treatments compared to medium clearing (Table 8.1).

Table 8.1 Loading values for various variables from the canonical variate analysis for time-since-clearing and uncleared treatments.

Variables	Loading values for CV1	Loading values for CV2
Pasture biomass (fenced plots)	3.60	-0.47
Pasture biomass (unfenced plots)	3.60	-0.47
Species diversity (fenced plots)	2.54	0.86
Species diversity (unfenced plots)	2.03	0.75
Litter production	3.40	-0.39
Soil organic carbon	-1.80	-0.07
Soil pH_w	-3.24	0.09
Soil NO_3-N	3.82	-0.27
Available P in soil	-1.74	-0.30
Soil microbial biomass-C	1.86	0.25
Soil microbial biomass-N	0.72	0.55

For CVA between the communities, 60 per cent of the variation was explained by CV1. *E. melanophloia* and *E. populnea* were similar and different to *A. harpophylla* (Fig 8.3) confirming the difference between the eucalypt and acacia communities. This was explained by greater values of +ve loadings for species diversity at fenced plots than the than other parameters (e.g. soil microbial biomass C and N, and available P) (Table 8.2). The -ve loadings for litter production, soil organic carbon and NO_3-N indicated that *A. harpophylla* had greater litter production, soil organic carbon and NO_3-N and was different to *E. melanophloia* and *E. populnea* (Table 8.2).

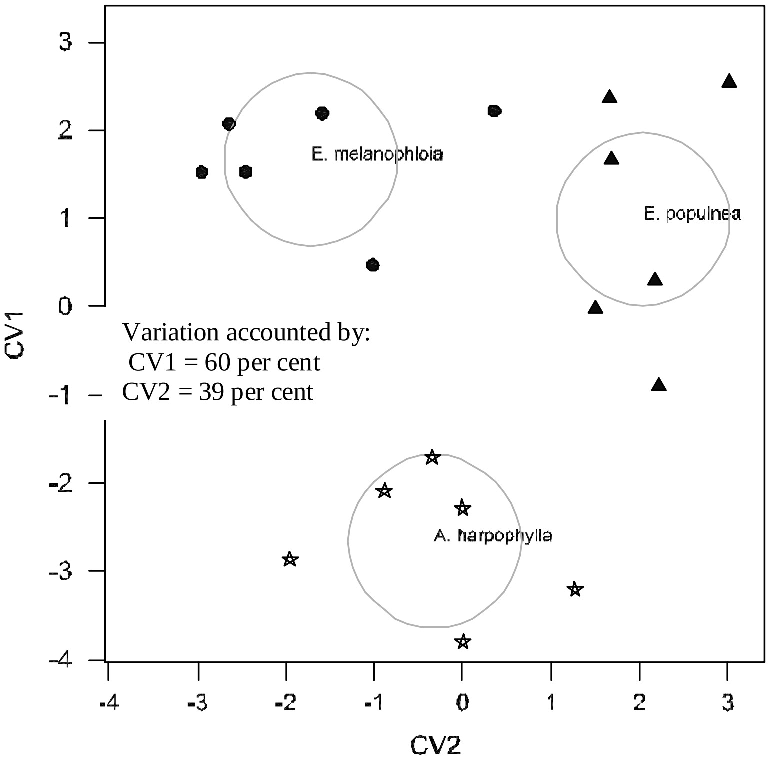


Fig 8.3. Relationship between first and second canonical variates for *E. populnea*, *E. melanophloia* and *A. harpophylla* communities (with 95 per cent confidence regions around means).

Almost all the remaining variation was explained by the second canonical variate (CV2 - 39 per cent). *E. populnea* community differed from *A. harpophylla* and *E. melanophloia*. These results were indicated by relatively higher values of -ve loadings for litter production, species diversity, soil NO_3-N, soil pH_w and pasture biomass than the other parameters such as available P or soil organic C (Table 8.2). This illustrated that the pasture yield, litter production, soil NO_3-N and species diversity were greater in the *E. melanophloia* and *A. harpophylla* compared to the *E. populnea* tree communities (Fig 8.3).

Table 8.2 Loading values for various variables from the canonical variate analysis for *E. populnea*, *E. melanophloia* and *A. harpophylla* tree communities.

Variables	Loading values for CV1	Loading values for CV2
Pasture biomass (fenced plots)	0.28	-0.10
Pasture biomass (unfenced plots)	0.28	-0.10
Species diversity (fenced plots)	1.85	-0.45
Species diversity (unfenced plots)	0.89	-1.11
Litter production	-2.07	-1.21
Soil organic carbon	-1.91	0.00
Soil pH_w	-0.72	-1.49
Soil NO_3-N	-0.93	-0.75
Available P in soil	0.54	-0.37
Soil microbial biomass-C	0.77	0.82
Soil microbial biomass-N	0.95	0.05

This analysis as well as results from previous Chapters (4, 5, 6 and 7) provided substantial evidence on the impact of clearing in pasture systems. Each of *A. harpophylla, E. melanophloia* and *E. populnea* communities showed different responses to clearing. Impact of clearing was more evident in *A. harpophylla* and *E. populnea* than in *E. melanophloia* based upon the cluster analysis. Further, the CVA across communities showed that *E. populnea* and *E. melanophloia* were different than *A. harpophylla* based primarily on species diversity, litter production and soil organic carbon (Fig 8.3 and Table 8.2). The *A. harpophylla* community, which has been mostly cleared in the past, was resilient to change (as demonstrated by integrated effect of 11 parameters in CVA; Fig 8.3 and Table 8.2) compared to *E. populnea* and *E. melanophloia* which are now the primary targets for clearing in central Queensland.

Clearing did increase the pasture production, but initial higher yields of pastures were not sustained in any of the cleared sites. There was a decline in pasture yield over time after the initial yield increase, in association with degradation of ecosystem functions (primarily litter production and hence the nutrient release to a system, the species diversity, soil NO_3-N and soil microbial biomass for C and N, and soil pH). The key parameters affected due to clearing were:

- Pasture biomass and species diversity

- Litter production

- Soil NO_3-N and soil pH

The CVA (Fig 8.2) and results presented in the previous Chapters showed that the soil nutrient availability decreased and soil pH increased at the oldest age of clearing. The oldest clearing did not sustain the same amount of pasture biomass as the recent or medium age of clearing. The erosion of top soil with clearing would have facilitated the depletion of nutrients as gullies were observed at the oldest cleared sites (no data were collected). The increase in soil compaction with grazing could also reduce the availability of soil moisture for plant growth. Soil pH increased significantly with clearing, especially with time post clearing in all cleared pastures for the three tree communities. The increased pH influenced the nutrient availability for plant growth as illustrated in Chapter 5. The increased Na concentration at the oldest cleared pastures for *E. melanophloia* is also a major concern raised in the present study. There was slow return of nutrients from litter produced at uncleared sites compared to cleared sites, however the potential content of C, N and P stored in litter was greater at the former than the latter. Litter production had no significant relationship with production of pasture biomass at unfenced plots, while the relationship was slightly significant for production of pasture biomass at fenced sites (r= 0.18 at $P<0.05$). The grazing pressure was the major confounding factor because litter was collected from unfenced sites.

This study provides the first evidence for the region that clearing is unlikely to impact on total soil CO_2 emissions once the pastures are established after clearing operations. In total soil respiration, root respiration was the major component and was 1.5 times greater at cleared compared to uncleared pastures. Root respiration is more sensitive to change in temperature than the total soil respiration (based on the literature and also observed in the current polyhouse experiment). This greater responsiveness of root respiration than soil respiration *per se* may contribute to an increase in total soil CO_2 emissions with rise in temperature, particularly for systems that have greater root biomass (e.g. recent and medium cleared treatments). However, the sensitivity of root respiration to change in temperature deserves detailed study in tropical climates.

The gains in pasture yield were associated with loss of ecosystem functions over time of clearing which may offset the initial benefits of pasture yield, since reparation of these functions may take even longer than the time taken to harvest the benefits. This suggests that a compromise is needed for production and maintenance of ecosystem functions for long-term sustainable production gains. The study was conducted on one property so the results should be carefully interpreted by landholders and policy makers to understand the impacts of clearing over time.

Results from hypotheses testing in previous Chapters:

Different tree communities showed a specific degree of response to clearing for different parameters studied, as presented in the previous Chapters. The following hypotheses were tested during the study (Table 8.3):

Table 8.3. Hypotheses for various parameters set for cleared pastures and their results.
If an intact woodland is cleared then, :

Hypothesis	**Result**
i. for pasture above-ground-biomass and composition (Chapter 4)	
• pasture above-ground-biomass will initially increase.	True for all three tree communities.
• over a longer term pasture above-ground-biomass will decrease after the initial increase.	True for all three tree communities.
• the extent and duration for increase or decrease in pasture above-ground-biomass following clearing will be different between different cleared tree communities.	True, the increase was over a short period in *E. melanophloia* compared to *E. populnea* and *A. harpophylla*.
• The composition of pasture species will change over time-since-clearing.	True for all three tree communities.
• pasture species diversity will decrease.	True for all three tree communities.
• pasture quality for cattle production will remain the same.	False, chemical composition varied as cleared pastures had lesser N but greater P compared to uncleared pastures. Pasture digestibility increased while non-grass component decreased with clearing.
ii. for soil properties (Chapter 5 and 6)	
• there will be an initial increase in available nutrients for plant growth.	True for all tree communities.
• a decline in nutrient availability will occur over a longer term.	True, but different responses of tree communities for different nutrients.
• soil organic carbon will increase with time-since-clearing.	False, it decreased with age-since-clearing especially at 30-60 cm depth in old and medium clearing for *E. populnea* and *A. harpophylla*, while it did not differ over time in *E. melanophloia*.
• Bulk density will increase.	True for all, except for recent clearing in *E. melanophloia*.
• soil pH, electrical conductivity (EC) and cation exchange capacity (CEC) will remain the same.	False, soil pH increased with age-since-clearing in all tree communities; EC increased at recent clearing in *E. populnea* and *A. harpophylla* but not in *E. melanophloia*; CEC increased in all tree communities.
• sub-soil properties will be negatively affected.	True, especially Na concentrations increased at old clearing for *E. melanophloia*, and pH increased at 30-60 cm depth in all the tree communities.
• soil microbial respiration, and soil microbial biomass for C (SMB-C) and for N (SMB-N) will decrease.	True, soil microbial respiration declined by 29 %, SMB-C declined by 34 % and SMB-N declines by 25 % with clearing (average values for three times-since clearing).
iii. for ground litter production, decomposition and nutrient return (Chapter 7)	
• litter production will decrease.	True, the decline was 40 % in *E. populnea*, 34 % in *E. melanophloia* and 58 % in *A. harpophylla* (averages across three ages of clearing).
• litter decomposition rates will increase.	True, 1.1 – 1.4 times faster with clearing.
• the amount of nutrient release from the amount of litter produced will remain the same.	The nutrient release increased with clearing, however decomposition process was incomplete with greater mass left to decompose at uncleared compared to cleared sites over one year period of decomposition.

The key findings of the study are:

- Pasture yield increased upon clearing but gains were not sustainable i.e. pasture yield increased from recent (5 yr) through the maximum at medium (11-13yr) to a decline at the oldest (33 yr) cleared pastures in *E. populnea* and *A. harpophylla*. For *E. melanophloia*, clearing led to an increase in pasture yield only at recent age of clearing. This was followed by a decline at medium and old age of clearing.

- For uncleared communities, an open woodland of *E. melanophloia* showed similar levels of pasture biomass as its medium or old cleared pastures.

- The monetary value of pasture biomass produced per year (assuming if all to be consumed) was calculated according to the amount of beef produced from the amount of pasture consumed over a year per cattle (182.5 kg weight gain/yr from 3139 kg/yr dry matter uptake; Minson and McDonald 1987, valued at the rate of AUD 1.50 /kg of livestock weight). The maximum monetary gains were obtained at medium age of clearing for *E. populnea* (AUD 331/kg/ha) and *A. harpophylla* (AUD 371/kg/ha), and at recent age of clearing (AUD 552/kg/ha) for *E. melanophloia* (Fig 8.4). However, the increased benefits were not sustained over time-since-clearing and showed a decline at old clearing in all tree communities. Indeed, for *E. melanophloia*, after 33 yrs of clearing, the monetary gains (AUD 246 /ha/yr) declined even below that at uncleared site (AUD 328 /ha/yr) (Fig 8.4). Hence, the benefits from increased pasture yield due to clearing in *E. melanophloia* were short-term only.

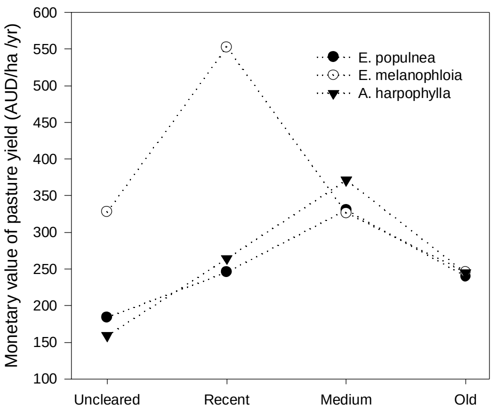


Fig 8.4.The monetary value of pasture yield produced at uncleared, recent, medium and old cleared sites for *E. populnea*, *E. melanophloia* and *A. harpophylla* communities.

- Clearing led to a decrease in species diversity that was associated with an increase in pasture biomass.
- Clearing increased nutrient concentrations in soil (i.e. organic carbon, total and available N and P) in recently cleared treatments, but those decreased at old clearing in all tree communities.
- Soil pH, electrical conductivity and concentration of Na were the main soil parameters affected by clearing. The increased pH at the oldest clearing, especially at 30-60 cm depth, influenced nutrient availability. The increased Na concentration at the oldest clearing of *E. melanophloia* indicated that clearing led to significant increases in sodicity.
- Soil microbial biomass decreased with clearing. This could reduce the mineralization of organic reserves to sustain the content of available nutrients.
- There was greater litter production with a slower rate of release of nutrients at uncleared compared to cleared sites.

- Clearing did not increase total soil CO_2 emissions once the cleared pastures were established.

Future perspectives

- A detailed study on soil biological properties (composition of microbial communities) at different age-since-clearing, and a longer term study on nutrient release from litter could provide more knowledge for the causes of pasture run-down in cleared pastures.

- Total value assessment based on all ecological parameters determined in the present study over time-since-clearing for cleared pastures and its comparison with uncleared pastures would help to explain the impacts of clearing more clearly for landholders and policy makers.

- Modelling of the present data over time-since-clearing would help to interpret the effects of clearing over longer periods of time. The estimation of these impacts would provide useful information to calculate the costs of damaged ecosystem functions.

- Estimation of total soil CO_2 emissions in various woodland and cleared (across a range from 10 to >50 years old) pasture communities across Queensland, and to determine the root respiration component of total soil respiration during different seasons. This would assist in evaluation of the impacts of post-clearing on global CO_2 emissions in tropical and subtropical regions of Queensland.

- Research on soil pH, Na concentration and electrical conductivity for a range of properties (under different management practices) in the central Queensland region could explain the change in ecosystem functions due to clearing, and to confirm the present findings on the effect of tree clearing on soil pH.

- A detailed study on pasture production at different levels of tree retention (20, 40, 50 or 60 percent) in major vegetation communities in the region could help to determine the most favourable level of tree retention for optimum productivity while sustaining the ecosystem functions.

- To assess suitable tree types according to the climates in central Queensland for introduction in cleared grazing properties to restore ecosystem functions while maintaining production gains.

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Appendix 1a (Chapter 4) - Grazing time

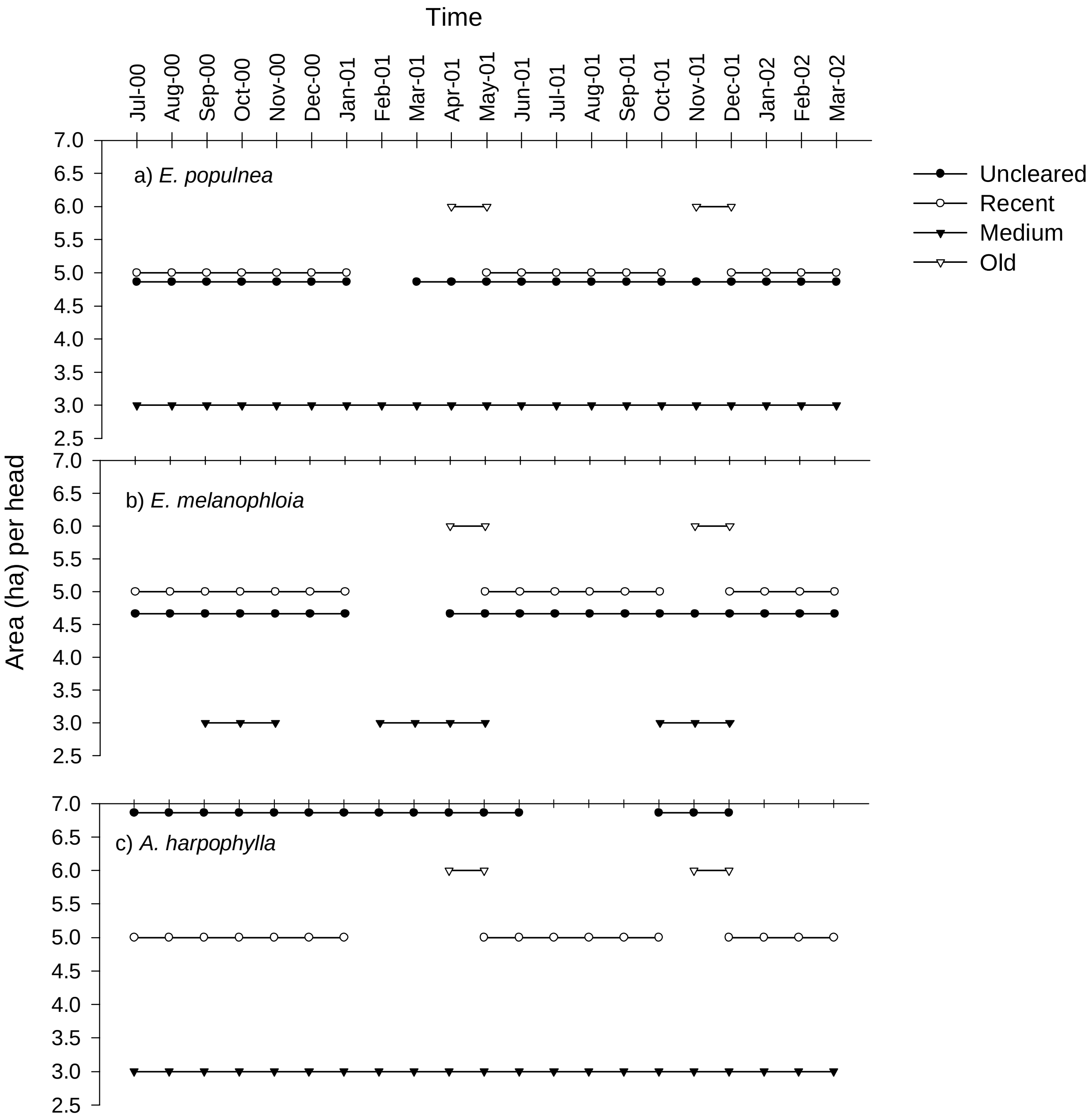

Fig 1. Grazing time at cleared and uncleared (mean values for 3 uncleared sites per tree community) sites for *E. populnea, E. melanophloia* and *A. harpophylla* communities.

Appendix 1b (Chapter 4). Pasture biomass in relation to tree basal area and tree density

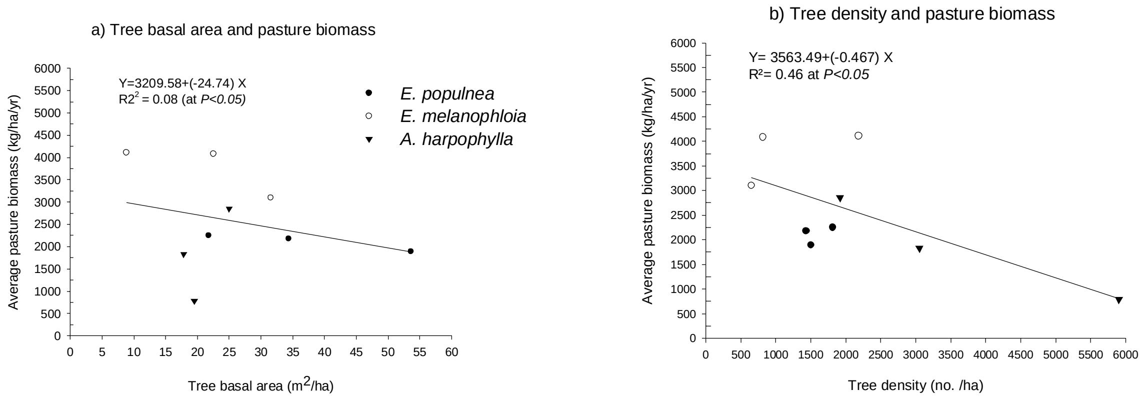

Fig 2. Pasture biomass at uncleared sites for *E. populnea*, *E. melanophloia* and *A. harpophylla* in relation to:

a) Tree basal area
b) Tree density

Appendix 2 (Chapter 5). Details of various methods used for soil analysis at soil laboratory Incitec Ltd., Queensland.

Soil parameter	**Method**
pH_w (1:5 water)	1 to 5, soil to water dilution, stirred, stand 1 hour and read while stirring, using combination electrode.
pH_{CaCl2} (1:5 $CaCl_2$)	1 to 5, soil 0.01 M Calcium chloride, stirred, read using combination electrode.
Organic carbon	Walkey and Black, using H_2SO_4 and $K_2Cr_2O_7$ in 1:100 dilution, measured colormetrically.
NO_3-N	1:5 soil to water, intermittent stirring over 1 hour, centrifuged nitrate measured colorimetrically in segmented flow analyser.
NH_3-N	1:5 soil to solution of 2 M KCl plus 0.05 M H_2SO_4 -1 hour shake centrifuged-measured on segmented flow analyser.
Total N	Using CHN analyser at Central Queensland university, Rockhamtpon.
P (Colwell)	1:100 soil to solution of 0.5 M $NaHCO_3$ end-over-end 16 hour shake, centrifuge measured colorimatrically in segmented flow analyser.
Total P	Using Kjeldahl method, samples were digested using H_2SO_4, and analysed with ICP.
K, Ca, Mg and Na	1:10 neutral, normal ammonium acetate, ½ hour vigorous shake, centrifuged, measured on ICP AES.
Electrical conductivity	1:5 soil to water, stirred, stand 1 hour and read by conductivity meter.
Cu, Zn, Fe and Mn	1:10 soil solution of DTPA, triethanolamine and $CaCl_2$ ½ hour vigorous shake, centrifuged, measured on ICP AES.