Deforestation for Pasture Development – Has It Been Worth It?

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The XIX International Grassland Congress took place in São Pedro, São Paulo, Brazil from February 11 through February 21, 2001.

Proceedings published by Fundacao de Estudos Agrarios Luiz de Queiroz

DEFORESTATION FOR PASTURE DEVELOPMENT – HAS IT BEEN WORTH IT?

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Abstract

Differing scenarios leading to deforestation for pasture development in savanna (woodland) and closed forest communities in the tropics – sub-tropics are compared and contrasted. Australian and Brazilian examples are highlighted. No simple answer is given to the question of whether deforestation for pasture development has been worth it, since both commercial and non-commercial values have equal validity and need to be taken into account. These issues are addressed in the context of land assigned by governments for agricultural purposes. It is concluded that technology and ecological understanding are now available to maintain sustainable production from converted forest systems. However emphasis should be on delivering this within the framework of existing deforested areas – rather than in expanding the area of forest conversion.

Keywords: Deforestation, tropical forest, savanna, pasture, cattle production, environment, economics, carbon sink

Introduction

Deforestation is an ongoing vexatious issue in both humid tropics and tropical/sub-tropical savannas throughout the world. Tropical forests constitute an ecological biome of global importance to carbon cycles, patterns of climate and biodiversity (Serrão *et al.* 1996). So the annual deforestation rate in the tropics, which increased from 0.6% in the 1980’s to 1.2% in the 1990’s (Kramer *et al.* 1997), remains a major international concern. However an alternative view (Osei 1996) is that the cutting down of forests constitutes an integral part of providing for human development needs, whether for cropping, grazing or commercial wood products.

Differing scenarios lead to deforestation for pasture development in savanna (woodland) and closed forest communities in the tropics – sub-tropics. Most savannas have an inherent grazing capacity in the absence of tree clearing, with tree removal or thinning simply promoting the growth of existing ‘native’ pasture. Meanwhile woody plants have proliferated in many savannas as a result of changed fire regimes which followed the introduction of domestic livestock (Scholes and Archer 1997). In these situations tree clearing is often employed to restore the carrying capacity and composition of the original plant community (e.g. Blainey 1982).

Conversion of rain forests and some closed woodlands to pasture brings about a more fundamental change in the structure and composition of the vegetation (Grainger 1999). Burning the felled forest is usually followed by the introduction of exotic pasture grasses capable of exploiting the enhanced nutrient levels released in the soil and from the overstorey ash – or of providing some continued agricultural production, once fertility is no longer adequate for cropping.

Ranching has historically served as an effective means of occupying frontier areas throughout the world (Serrão and Toledo 1990) and the Australian savannas and Brazilian Amazon are no exceptions. However a vocal environmental lobby has decried cattle production in tropical forests and many savanna areas subject to tree clearing, but this has become a moot point in the Amazon at least, given the number of people and cattle already there (Faminow 1997).

In the past the "success" of tree clearing to promote pasture development and livestock production was judged almost exclusively on the perceived economic responses. Although 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. Some of these considerations include habitat loss and fragmentation, nutrient decline, greenhouse/climate impacts and the loss of other ecosystem services provided by forests and woodlands.

These broad issues were comprehensively addressed at the XVIIth International Grassland Congress by Serrão *et al.* (1993) for the humid tropics and Burrows (1993) for savanna areas. This paper draws upon the findings of these previous reviews, together with more recent data and viewpoints, to evaluate the benefits and costs of such forest/woodland conversion for pastoral purposes. Brazilian and Australian examples are highlighted.

The role of cattle in tropical forest deforestation

There are many causes of deforestation throughout closed forests and woodlands in the tropics/sub-tropics. Serrão *et al.* (1996) consider the most important land use systems leading to recent deforestation in the Amazon have been logging (contributing c. 10%), shifting agriculture (30-35%) and extensive cattle ranching (>50%). The significant influence of pasture development for cattle raising on tropical forest deforestation is noted by Buschbacher (1986); Hecht (1993) and Aide *et al.* (1996).

In closed forests and woodlands some deforestation is obviously necessary for agricultural development, as was the case in many temperate countries during the early phases of economic growth (Cronon 1983; Walker 1994). However modern machinery and arboricides allow a much larger scale of forest conversion than was possible prior to the 1960's.

Faminow (1998) draws upon the prior studies of Hecht (1993) and Kaimowitz (1995) to detail factors thought to underlie the role of cattle in Amazon deforestation. These include:

- favourable markets for livestock products, especially amongst local urban dwellers
- government subsidies for livestock credit and road construction
- land tenure policies which promote deforestation to establish property rights
- slow technological development which favours extensive systems
- policies which depress timber prices
- improved personal security in remote frontiers
- special characteristics of cattle as a low labour, low maintenance, flexible commodity
- advantages cattle provide to small holder operators
- indirect financial benefits cattle provide through wealth maintenance, cashflow and risk reduction through diversification

Parayil and Tong (1998), *inter alia*, highlight the fact that land was a useful hedge against inflation. Thus inflationary pressures and high market values for land under pasture provided incentives to clear as much land as possible.

The role of cattle in savanna deforestation

The tropics contain about 50% of the earth's grazing lands (Squires and Vera 1992). Much of this tropical grazing land is found in South America and Australia and cattle production developed as a way to utilise the vast amounts of 'natural' grassland available. However historical accounts, photographic records and quantitative studies have demonstrated extensive invasion of grasslands by woody plants and increases in woody plant density in savannas over the past 200 years. (See comprehensive bibliographies in Archer 1994; 1995; Idso 1995 and at http://cnrit.tamu.edu/rlem/faculty/archer/bibliography.html).

This proliferation of woody plants is sometimes known as tree thickening, bush encroachment, shrub encroachment, woody weed increase, woody weed invasion or woody regrowth. Secondary forest development, following initial clearing of intact forests and woodlands, is a more obvious and direct management induced analogy of woody plant proliferation.

Traditional explanations offered to account for the historic displacement of grasses by woody plants have centred around changes in climate, livestock grazing and fire regimes. Idso (1995) claimed that the worldwide invasion of grassland by trees and shrubs that began c. 200 years ago has closely followed the upward trend in the air's CO_2 content. However Archer *et al.* (1995) examined this argument in detail and provide convincing evidence that the correlation between CO_2 and woody plant proliferation is spurious.

Fensham and Holman (1999) have recently suggested that the observed woody plant proliferation in Northern Australia is part of a normal tree death – regrowth cycle in this region, primarily driven by climate variation. But $\delta^{13}C$ analyses of soil organic matter in thickening woodlands indicate that these woodlands had a more open structure in the past (Burrows *et al.* 1998) – not a more wooded one as Fensham and Holman's fluctuating climate model implies. Boutton *et al.* (1998) also employed the $\delta^{13}C$ technique and reached similar conclusions about the nature of *Prosopis* invasion into the desert grasslands of the South-West USA.

Growing experimental evidence suggests all mixtures of trees and grass are unstable in savanna environments. In the absence of disturbances such as repeated fires, clearing by humans or feeding by large herbivores the tree cover increases at the expense of grass production, until it is limited by tree – tree competition (Scholes and Archer 1997). The rarity of the latter situation provides credence to the suggestion that both in Australia and elsewhere in the tropics/sub-tropics the savannas (woodlands) were maintained as a fire mediated sub-climax prior to the "recent" introduction of domestic livestock (Burrows *et al.* 1998). The prime agent changing the structure of these communities has been the conversion of land use from hunter-gathering or nomadism to the raising of sheep, cattle or goats – a practice which has often been associated with increased grazing pressure, reduced fuel loads and the containment of fire. In the absence of governmental regulations or incentives not to clear, the livestock raiser will eventually seek to control tree/shrub densities where woody plants are proliferating in savannas because, in most instances (Burrows *et al.* 1990; Scanlan 1992), this is associated with reduced pasture production and lowered livestock carrying capacities.

Pasture development pathways

In both savannas and tropical forests tree clearing usually results in increased nutrient and moisture availability for native or oversown replacement pastures. Improved light penetration would also be beneficial to replacement vegetation in tropical forests, but of less significance in savannas. There is, therefore, an initial flush in pasture production which is one of the major incentives for forest conversion.

It is appropriate for the livestock raiser to capture this post clearing pasture flush with raised animal carrying capacities and/or shorter turn-off-times. But in savannas the sustainability and value of clearing trees to promote native pasture production should be based on the pasture production potentially attainable in the intercanopy areas prior to tree removal, and not on the old canopy zones of trees killed (Burrows 1993). Analogously, claims that the introduction of exotic pastures following forest and closed woodland conversion represents non-sustainable land use (Fearnside 1993; 1997b; Hecht 1993; Fairfax & Fensham 2000) are partly based on the inevitable decline in pasture production as post clearing nutrient release returns to a new equilibrium level. This productivity decline may be accompanied by the invasion of less palatable weedy species and/or secondary forest (Scanlan and Anderson 1981; Fearnside and Guimãres 1996).

Agriculturalists have commonly exploited nutrient flushes released after clearing and burning of the felled forest/woodland by sowing nutrient demanding grasses into the ashbed such as the Panicoid genera (e.g. *Brachiaria, Cenchrus,* and *Panicum*). The foreseeable and inevitable 'decline' of these 'first cycle' pastures (Serrão *et al.* 1993) does not mean that sustainable pasture systems cannot be developed on these lands. An intensive and usually expensive response is to use fertiliser and better adapted 'second cycle' pastures to replace those suffering from nutrient run-down and the influx of weeds and pests (Mattos and Uhl 1994). A more long lasting alternative may be to introduce less nutrient demanding grasses into the pastures or to reconstruct the induced pasture ecosystem to more closely mimic the vegetation which it replaced (Ewel *et al.* 1991; Smith *et al.* 1996). The first goal can be achieved by utilising species (e.g. Andropogonoid cf. Panicoid grasses) more in tune with new equilibrium nutrient levels, once established post clearing. The second approach could lead to the incorporation of legumes and fodder trees into the system to maintain or restore ecosystem processes (nutrient and hydrologic cycles, carbon fixation) present in the original community.

[Insert Table 1]

Putting such theory into practice is a major challenge for pasture researchers. A successful example can be found in the beef production area of the Central Highlands, Queensland, Australia (Table 1). Here the original closed woodland community has been reconstructed to have greater agricultural productivity by replacing the dominant tree (*Acacia harpophylla*) with leaves unpalatable to cattle with a fodder tree (*Leucaena leucocephala*) which is highly palatable and nutritious. Likewise the low productivity understorey native grasses (*Chloris* spp., *Paspalidium* spp.) have been replaced by *Cenchrus ciliaris* planted within the inter rows of widely spaced (5-10 m) fodder trees. Leucaena and the tree it replaces (*A. harpophylla*) are both nitrogen fixers with similar rooting patterns. Soil carbon reserves in the *C. ciliaris* grass system 20-30 years after initial tree clearing have been shown to be little different to that under the original closed woodland community (Gifford 1999).

Much of the Australian leucaena based pastures are located on soils (Vertisols) of reasonable fertility. Brazilian tropical forests cleared and sown to pasture based systems for cattle production are usually located on infertile Oxisols and Ultisols. Legumes and fodder trees (e.g. *Calliandra, Erythrina, Gliricidia* and *Indigofera* spp. – Shelton 1998) tolerant of acid soils should be more widely evaluated for such systems.

Fearnside (1999) has questioned the practicality of using high phosphorus demanding pastures as a replacement for tropical forests in Brazilian Amazonia, because the region has to import most of its fertiliser requirements. However these needs can be met for fodder trees by specifically placing fertiliser within tree rows to minimise the total required per hectare. An alternative means of meeting the P requirements of stock is to use P supplements. This is a very effective and inexpensive option utilised in Australia where low nutrient demanding legumes, such as *Stylosanthes* spp. have been oversown into grass pastures on very P deficient soils (Partridge and Miller 1991). Provided erosion and deep leaching are minimised the overall removal of P in animal products from cattle based systems is relatively small

(Burrows 1993), but such losses still need to be taken into account and replenished if system productivity is to be maintained (Serrão and Toledo 1990).

Environmental concerns

Most conservation groups are affronted by the spectre of deforestation, whether it occurs in dry savannas or wet tropical forests. Nevertheless Vitousek *et al.* (1997) and Tilman (1999) note that all of earth's ecosystems are now impacted by humans to various degrees. If anything, such observations emphasise the need for appropriate conservation measures to be put in place to avoid unnecessary loss, rather than providing an excuse for ongoing deforestation.

Future biodiversity losses (i.e. extinctions) should not occur if tree clearing controls are implemented to protect significant proportions of the landscape. For example, in Queensland, Australia current legislation protects all endangered/vulnerable communities (with <10% of their original (pre-european) distribution remaining) and a yet to be implemented section of the Vegetation Management Act 1999 (Anon. 1999) would protect all communities 'of concern' (i.e. <30% of their original distribution remaining). Likewise it appears that Brazilian law has required that 50% of farm holdings remain in forest and this may now have increased to 75% (see Faminow 1997). Given such restrictions it should only be lack of government will to enforce current laws in these respective countries that would lead to complete 'loss' of forest and woodland organisms – rather than tree clearing *per se*.

Faminow (1998) notes that the key difference between most forest re-organising events (wind fall, death, heavy storms, wild fire, extensive logging) that create gaps and deforestation for cattle production is that the latter is usually an intentional, extensive and permanent change in land use. He further suggests that evidence supporting the belief that there is a risk of an imminent mega-extinction is almost non-existent. Yet there is no denying that land clearing can have many flow-on impacts that are not always immediately apparent to the instigator.

Schroeder and Winjum (1995) have noted that tropical forest deforestation accounts for the equivalent of c. 15-25% of greenhouse gas emissions that are attributable to fossil fuel use. Such emissions from felling and subsequent burning of forests cannot be denied (Fearnside 1997a) but recent data throw some doubt on below ground losses of carbon. The Revised 1996 IPCC Guidelines for Greenhouse Gas Inventories: Reference Manual (Chap. 5) (Houghton *et al.* 1996) suggests, in the absence of empirical data, a default loss of 30% of below ground C stocks can be assumed on conversion of forest to pasture. However recent reports (e.g. Koutika *et al.* 1997; Neill *et al.* 1997; Gifford 1999; Hughes *et al.* 2000; Post and Kwon 2000) indicate net below ground 'losses' could be close to zero wherever a healthy pasture is established on the previously forested land.

The proliferation of woody plants in woodlands/savannas grazed by domestic livestock is a worldwide phenomenon (see previous citations) and is a prime motivation for ongoing tree clearing in these grazing lands. In Queensland, Australia, the C sink attributable to this woody plant proliferation exceeds the C source returned to the atmosphere via tree clearing activities (Burrows *et al.* 1998), although only the latter is currently acknowledged in Australia's National Greenhouse Gas Inventory.

Deforestation and its contribution to greenhouse gases is thought to be a significant precursor to climate change (Fearnside 1995). The latter will manifest itself at global as well as regional scales. Expected impacts include temperature increases from global warming, increases in carbon dioxide concentrations, rainfall changes both from global warming and reduced evapotranspiration, increased cloudiness in some parts of the Amazon and extra-regional transport of dust and smoke (Fearnside 1999). Pastures are clearly much more fire prone systems than the forests they replace (Serrão *et al.* 1993). Many of these broad impacts would also be influenced by tree clearing in woodlands/savannas. In these areas the prospect of increased salinisation and erosion following deforestation are additional concerns (Webb 1984, Burrows 1993).

Economics

The accepted wisdom in the literature, with notable exceptions, is that cattle ranching is an unprofitable (and unsustainable – Fearnside 1997b) activity in tropical forests, which owes much more to speculative land profits and government subsidies, than any underlying economic rationale (Faminow 1998). Danckwerts and King (1984) had earlier made analogous observations for South African rangelands, where they pointed out that the investment value of land often exceeds its productive value. Likewise surveys of economic performance on Australian farms are carried out annually by the national government. Results commonly show that the capital appreciation of land used for beef production matches or exceeds the return generated by the farm business (e.g. ABARE 1997).

However Faminow (1998) questions why, if cattle are not profitable, the Amazon herd is so large (> 15M head) and why so many small and large scale farmers have invested in cattle? For example in Hecht's (1993) publication it is reported that there were then > 50 000 livestock operators in Amazonia at all scales of production. Indeed the pessimistic view of returns from cattle depicted by Browder (1988) and Hecht *et al.* (1988) is considered by Faminow (1998) to be at odds with much of the observed experience that has evolved since the mid 1980's, when the former studies were conducted. Over time, cattle ranching has grown and investors (large and small) have continued to make investments in cattle production, despite that fact that subsidies have either diminished, been discontinued altogether or were never received in the first place (Faminow 1998).

Many of the concerns raised over pasture development after deforestation of tropical forests have targeted the alleged unsustainability of the practice (e.g. Fearnside 1997b). However Serrão *et al.* (1993) have pointed out that improved pasture husbandry and germplasm have become available since the initial pioneering phase of pasture development in the Amazon. This led Uhl and his co-workers (Mattos and Uhl 1994; Almeida and Uhl 1995; Toniolo and Uhl 1995; Arima and Uhl 1997) to conclude that more intensive production systems can lead to improved financial returns and more sustainable use of the land base.

Even arch critics of deforestation (e.g. Hecht 1993) acknowledge that cattle production extends the economic life of a given cleared area when slash and burn agriculture sites are converted to pasture. Hecht concluded that, while the productivity of such pastures is amongst the lowest in the Amazon, they provide a marginal return on land that would otherwise be generating nothing for the small holder colonist. Indeed Serrão *et al.* (1993) and Faminow (1997) have each drawn upon the detailed studies of Mattos and Uhl (1994) to suggest that most stages of pasture development can be profitable when viewed from a landholder perspective.

Deforestation for pasture development remains a widespread activity in north-eastern Australian grazing lands with the area cleared (including regrowth) averaging around 300 000 ha/yr (Anon. 1997). Most of the pasture response is from the growth of native grasses following the removal of overstorey competition (see Burrows 1993). Significant areas are also sown with exotic species (mainly *Cenchrus ciliaris*).

[Insert Box 1]

Savanna deforestation for pasture development does not always aim to completely remove all overstorey trees. A variety of techniques, including chemical injection with arboricide and mechanical felling, are used to reduce tree competition. A recent study (Burrows *et al.* 1999) has used net present value analysis to evaluate potential economic returns from various tree clearing scenarios experimentally applied in eucalypt woodlands (Box 1). These indicate that tree clearing can be beneficial for beef production in these communities even when 20% of the tree cover is retained in intact blocks.

Simulation studies in a similar woodland suggested that reducing tree basal area from 6 to 3.5 m^2 per ha could increase cash flow by around $90 000 per year for a self-replacing cattle property of 20 000 ha (Stafford Smith *et al.* 1999). This analysis highlights the continuing need for landowners to be able to reduce tree competition with pasture in all woodlands where trees and shrubs are proliferating. Earlier research (Bartholomew and Wilson 1995) had suggested that an 80% tree retention rate would result in an annual profit around $40 000 lower than for 20% tree retention on an 18 000 ha eucalypt woodland property grazing beef cattle in Central Queensland.

Collectively these analyses indicate that deforestation to promote pasture and cattle production 'pays' in economic terms – at least from a landholder's perspective. The ubiquity of the practice from semi arid savanna to tropical rain forest and its enormous continuing extent (>1 M ha/yr in the Amazon (Fearnside 1999), 0.3 M ha/yr in Australian savannas (Anon. 1997)), can brook no other conclusion. But resource use is no longer judged on economic grounds alone.

Fearnside (1999) presents a simplified summary of the types of values in Amazonian forests (Table 2). He points out that many of the values of ecosystems are not marketed in to-day's human economy, and therefore receive little weight when business and political decisions are made. For example it is difficult to assign meaningful monetary values to biodiversity (Stirling 1993; Harris 1995) while no person would deny that there is an intrinsic, but unquantifiable existence value associated with Amazonian rain forest.

[Insert Table 2]

Non market values of environmental services are often assessed through contingent valuation techniques such as a willingness to pay for keeping the service or a willingness to accept its loss (Carson 1998). Choice Modelling (Morrison *et al.* 1996) is another approach that has been specifically applied to evaluate the effect of tree clearing on non market valuations for eucalypt woodland in Australia (Rolfe and Bennett, 2000). Fearnside (1999) emphasises that the monetary values generated by such techniques are not real values, while the people most interested in maintaining natural ecosystems often cannot afford to pay anything. Nevertheless these techniques serve a useful purpose in highlighting important ethical and moral questions about resource use which humanity must increasingly address (see Bormann and Kellert 1991).

Discussion

There is no doubt that deforestation for pasture development remains a topical and contentious issue throughout the world's tropical forest and savanna regions. The present review has only briefly canvassed the broad considerations raised by Serrão *et al.* (1993) and Burrows (1993) *inter alia*, and the in depth treatments provided by Faminow (1997; 1998) and Fearnside (1993; 1997a, b; 1999).

A critical point for this discussion is that much of the land at the centre of the deforestation debate has been assigned by governments, overtly or covertly (where pasture development leads to the granting of tenure), for agricultural purposes. Whether governments should have allocated land for such use is moot, especially for existing landholders who now depend on the land for their livelihood. Use of land for agriculture is a profound human necessity, particularly in developing countries with large and growing rural populations (Serrão *et al.* 1996). But this is no less in lightly populated regions such as Australia, where the need to occupy land is seen as essential to the nation's security.

A central theme of Faminow's (1998) book is that rain forest conversion to pasture and cattle raising is financially viable from the perspective of farm decision makers and that the environmental defence of the Amazon should start with an acknowledgment of this reality. A similar conclusion would be reached by any detailed consideration of tree clearing for pasture development in Australian savannas.

No responsible and thinking landholder would deliberately set out to destroy the land which is the foundation of his capital and livelihood. Yet there is no denying that serious problems have arisen in many areas following the removal of tree cover. In hindsight these problems are more a reflection on the lack of pre-clearing planning and post-clearing management then on the tree clearing itself. For example it is not difficult to plan to retain adequate wildlife corridors prior to clearing. Likewise areas that may contribute to exceptional erosion hazard or salting can be identified before clearing is undertaken and should not be cleared; overgrazing should be avoided in all circumstances etc. A common problem is that often a greater area is cleared than can be successfully managed to bring it into stable production post-clearing. The fear of government controls or resumptions has apparently been a factor in this 'over-clearing' in both Australia (Anon. 1999) and Brazil (Ledec 1992).

Bach (1999) has noted that sustainable management of tropical forest areas has been high on the international agenda for more than a decade. According to Fearnside (1997b; 1999) sustainability can best be assured by avoiding the substitution of pasture for the rain forest. He suggests that cattle pasture, the dominant system following forest conversion, is unlikely to prove sustainable over the long run. Rather than try to extend the life of cattle pastures by means of fertilisers and changes to grass species, it is better to start with tropical forest, which has proved itself sustainable for thousands of years of existence, and find ways to market the services the rain forest provides (Fearnside 1997b).

The argument that land use for pasture production is unsustainable ignores ongoing agricultural research. This increasingly demonstrates that better management, technology, germplasm and recognition of the need to mimic ecological processes of systems replaced can provide a sound basis for pastures and the cattle industry founded on them (see Mattos and Uhl (1994) for tropical forests; Shelton (1998) for leucaena based systems in savannas).

Yet it would be extremely naive to ignore the fact that much of the land cleared in both tropical forest and savanna regions in the past is presently in an unacceptable condition. Gleeson (1999) has suggested that up to 80% of the extensive land clearing in north-east Australia in recent years was of tree and shrub regrowth on land previously cleared. Fearnside (1997b) maintains that up to 45% of land cleared in the Amazon is reverting to secondary forest. Such realities led Serrão *et al.* (1993; 1996) to conclude that cattle ranching should be developed on already deforested lands and, in the medium and long term, the unsustainable extensive ranching model should be gradually transformed into more sustainable semi-intensive beef cattle systems, semi-intensive dual-purpose cattle ranching models and intensive agro-silvipastoral models.

Resource economists now generally concede that tropical forests are excessively exploited (e.g. Pearce 1990; Thiele and Wiebelt 1993). Responsible development of savannas has long included recommendations for retention of suitable areas of intact woodland (Burrows 1974; 1990) and this is currently enshrined in land administration, such as the

Queensland Vegetation Management Act 1999 (Anon. 1999). Nevertheless the greatest challenge remains to design equitable government policies to encourage the adoption of sustainable systems (Serrão *et al.* 1996). Central to this challenge is the need for all parties to accept that the only sustainable agriculture is profitable agriculture (Ainesworth 1989).

Thomson and Harris (2000) note that when native forests and woodlands have no economic value to landholders the incentive is to clear land for agricultural purposes. No value is placed on factors which give public good benefits (e.g. maintenance of biodiversity, stabilisation of the water cycle, carbon fixation) which lack property rights (Thiele and Wiebelt 1993) and so do not enter the individual's decision making process.

Governments can pass laws to prevent the clearing of land, but such laws have also to be enforced. It is increasingly recognised that incentive based schemes (Braithwaite 1996; MAFF 1996; Holden 1997; Latacz-Lohmann 1998) offer the best way forward to protect the interests of landholders and those of the wider community. Likewise the development of carbon trading rights for forest and woodland vegetation could conceivably be a way to maintain the economic viability of individual landholders owning such vegetation, while addressing broader environmental concerns.

In the final analysis there is no simple answer to the question of whether deforestation for pasture development has been worth it. The human population of the Brazilian Amazon has increased ten fold since the 1960's (Laurance *et al.* 2000) and it now supports a beef herd of > 15 M head. Analogous but smaller scale changes in human and cattle populations followed deforestation schemes in the brigalow (*Acacia harpophylla*) woodlands of north-east Australia (Lloyd 1984). These realities challenge the one-sided argument that the environmental costs of deforestation have not been shored up by human development gains (Faminow 1998). Yet there is considerable wisdom in the conclusions of Serrão *et al.* (1993) that future intensification of cattle ranching should be on already deforested lands – and that governments and the world community should now reduce the rates of deforestation in the interests of conserving our remaining forest resources (Serrão *et al.* 1996).

Acknowledgement

Meat and Livestock Australia provided financial support for the savanna clearing strategies trial reported in this review.

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Table 1 - Performance of steers in three feeding systems in inland Central Queensland, Australia (adapted from Esdale and Middelton 1997).

Feeding system (adult stocking rate range – ha $animal^{-1}$)	Steer liveweight gain kg day^{-1}	Steer liveweight gain 150 day total (kg)
Native pasture (4-10)	0.33	50
Buffel grass sown after clearing the brigalow woodlands (1.5-2.5)	0.86	129
Buffel grass oversown with leucaena in rows 10m apart (1.5-2.0)	1.26	189

Buffel (*Cenchrus ciliaris*); Leucaena (*Leucaena leucocephala*).

Table 2 - Types of value of Amazonian forests (after Fearnside 1999)

Type of value	*Examples*
Commercial value of presently-marketed commodities	Timber, beef
Commercial value of commodities not presently marketed	Pharmaceutical products
Environmental Services	Carbon storage, water cycling
Existence values	Non-utilitarian value of biodiversity and cultural diversity

Box 1

Woodland Clearing Strategies Trial – Economic Analysis Model
(Poplar box (*Eucalyptus populnea*) woodland – Central Queensland, Australia)

The purpose of this analysis is to compare **relative responses** between each treatment – not to prepare a statement of income. Therefore fixed costs are not considered. However interest on herd capital is charged at 10%.

Assumptions:

- Paddock size 1000 ha (Intact woodlands have grazing value but tree clearing usually enhances it)
- Stocking rates are determined by the GRASSMAN model (Scanlan and McKeon 1990) – tree basal area resulting from each treatment (actual field data) determines potential native pasture production (e.g. Scanlan and Burrows 1990)which is stocked so as to consume 30% of the pasture on offer
- Steers enter paddock at 180kg – leave at 450kg
- Average rainfall is assumed for each year of a modelled 15 year time frame
- Net Present Value (NPV) and Internal Rate of Return (IRR) are based on the 15 year time span and a 6% discount rate

Treatment	Response **relative** to Control	
	NPV	IRR
Control (Intact woodland – initial tree basal area 10 m^2/ha)		
Treat 100% trees with stem injected arboricide – picloram based	\$51.5 ha^{-1}	24%
Treats 100% trees with stem injected arboricide + stickrake regrowth in Year 8	\$37.0 ha^{-1}	19%
Treat 100% of area with soil absorbed arboricide – tebuthuiron (1.5kg a.i. ha^{-1})	\$50.0 ha^{-1}	16%
Treat 100% of area with soil absorbed arboricide – tebuthuiron (1.0kg a.i. ha^{-1})	\$79.5 ha^{-1}	28%
Pull 100% of area with tractor and chain	\$18.0 ha^{-1}	14%
Pull 100% of area with tractor and chain + burn regrowth	\$59.0 ha^{-1}	22%
Pull & burn 100% of area + stickrake regrowth in Year 8	\$47.0 ha^{-1}	19%
Retain 20% trees scattered over paddock, stem inject remainder	(\$21.0 ha^{-1})*	
Retain 20% trees in intact woodland strips – pull and burn remainder	\$47.0 ha^{-1}	22%
Retain 20% trees in intact woodland strips – treat remainder with tebuthuiron (1.5kg a.i. ha^{-1})	\$40.0 ha^{-1}	16%
Retain 20% trees in intact woodland strips – treat remainder with tebuthuiron (1.0kg a.i. ha^{-1})	\$63.6 ha^{-1}	28%

( )* = negative value
[Adapted from Burrows *et al.* 1999]