Landscape as Process:
An Exploration of Social and Physical Aspects of Ecology and Change in
the Alligator Rivers Region, Northern Australia

By

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Abstract

The Alligator Rivers Region is situated within the tropical savanna region of northwestern Australia. During European settlement the region remained on the periphery of colonial efforts and was not fully consolidated within state control until the establishment of Kakadu National Park in 1979. This thesis describes the ecological factors that maintain the forest-woodland mosaic in riparian communities in the South Alligator River catchment, and then presents two works that focus on significant events of colonization that had a particular influence on the physical and social landscape of the Alligator Rivers Region: (1) the introduction of the Asian water buffalo (*Bubalus bubalis*); (2) the depopulation and dispossession of Aboriginal people; and (3) changes in the use and experience of space as a result of colonization. These studies reveal a pattern of linkages that allows an understanding of the processes of landscape change not only as a physical phenomenon but as a social one as well.

Buffalo were initially introduced to northern Australia by Europeans as part of an effort to render a foreign landscape more familiar. Buffalo quickly expanded their range and became a feral problem for Europeans, but were incorporated into Aboriginal cosmology as a feature of the natural landscape. After World War II, rapid buffalo population growth and subsequent eradication each set in place different sequences of ecological effects that altered fire regimes as well as plant abundance and composition.
Among Aboriginal people, depopulation and the history of colonization dramatically shifted the perception and use of space and also impacted ecological processes. After colonization, riparian areas changed from interrelated and continuous movement corridors threading through the landscape to discrete sites connected by automobile travel. Among the major shifts in ecological processes, fire management changed from the activities of dispersed bands of humans to the creation of fire breaks along creek lines by helicopter. There remains today a strong disconnect between management policy and the ecological knowledge of Aboriginal people. I argue that these underlying differences are in fact differences in the perception of landscape and that joint management must take these fundamental differences into account.
Haiku

(if the abstract is too long)

Blackfella been here
Buffalo belong here now
Whitefella – know law
Acknowledgements

This dissertation began in Papua New Guinea, with a vague desire to understand different epistemologies and ways of interacting with the natural world. It’s hard to imagine that, with all of the twists and turns of this dissertation it could have turned out at all, let alone turned out so well. It began as a concept to study socio-natural systems in the Kimberley region of Western Australia and substantial funding from the University of California Pacific Rim foundation, as well as support from NSF dissertation Grant #211377. However, the difficulties of fieldwork in a remote area, particularly while trying to meet the needs of a spouse also working on her Ph.D., soon became apparent and I shifted my research focus to Kakadu National Park.

This is a decision I have in no way regretted. A project of this magnitude required substantial support and help from numerous people. I was welcomed into the fold of the now sadly defunct Key Centre for Tropical Wildlife Management at Charles Darwin University (then Northern Territory University) by the ever gracious and generous Peter Whitehead. I was swept in to David Bowman’s landscape change project, and would like to thank Prof. Bowman for advice early in the project and in-kind support of aerial photographs, as well as the joy of spending time with his wonderful family. His students, Daniel Banfai and Caroline Lehmann, provided friendship, advice, and a helpful sounding board. It has been a joy working with them. Donald Franklin was generous with his time and knowledge, and his enthusiasm for the natural history of the Top End was catching. Lynda Prior is modest to a fault, but I’m grateful for her esoteric knowledge of quantitative ecology – well, of ecology in general. Finally, Jeremy Freeman and Kelly Menadue provided extensive GIS and mapping services and were a bonzer couple with whom I spent many an enjoyable evening.

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Introduction: Landscape and change in context

This dissertation explores the physical, social and mental construction of land, landscape, and landscape change in post-settlement Australia. It also examines the role of history in shaping ecological and human communities. Beyond the importance of history for understanding ecological patterns and processes, thoughtful consideration of the relation between history and ecology reveals an important fact – change in human perception and action within the physical environment impacts on both the function and composition of ecological systems and our understanding of ecosystems and ecological change (Moran 1993, Balee 1994, Descola 1994). This is ‘landscape change’ where landscape is both materially and socially produced (Sauer 1963), and, in a human dominated world, social processes can be as important as physical processes in shaping ecological systems.

An understanding of the relationship between historical and environmental processes has been well developed by Balée (2006), who articulates the view that “human interactions with landscapes in a broad variety of historical and ecological contexts may be studied as a total (integrative) phenomenon.” The complex interrelation between
historical processes and ecological change can be understood as a pattern of linkages between history, ecological processes, and cultural perception and practice. These linkages manifest as landscape, a concept that captures the dialectic between mental, social and physical aspects of the lived world (Layton and Ucko 1999), and where changes in any one aspect will affect the others. Although numerous studies within the discipline of landscape ecology concern physical patterns and processes, the explanations that they generate are limited by their framing of the landscape concept. Full consideration of the patterns of landscape change requires an understanding of the processes of sociocultural change, but these can be difficult to discern.

The linkages between sociocultural and physical aspects of landscape are especially highlighted in situations of abrupt and profound cultural change. Colonization studies illuminate the linkage between physical, social and conceptual aspects through the juxtaposition of settler and indigenous communities (e.g. Vansina 1990, Balée 1994, McGregor 1995, Neumann 1998, Davis 2001, Braun 2002, Porro 2005). Colonization not only reflects the physical appropriation of the space of one culture by another, but also the appropriation of patterns of living and thinking. Indeed, not only does colonization place into conflict the symbolic and material cultures of indigenous and settler peoples, but ecological knowledge, particularly the interpretation of ecological change, is often itself a subject of conflict (McGregor 1995). The colonizer often controls the narratives of post-colonial ecological impact, while indigenous understandings and perceptions of ecological change receive less attention in both discourse and policy (Head 2000).
This dissertation considers the patterns and processes of landscape change in the context of the colonization of the Alligator Rivers Region of northern Australia. Our understanding of the relationship between people and their environment is well developed for Australia. This may be because of the antiquity and relative isolation of Aboriginal cultures in Australia (Head 2000), the peculiarity of Australia as a particularly fire prone continent and the ubiquity of anthropogenic fire as a management tool (Bowman 1998), or Aboriginal people themselves, who have been particularly forthright in emphasizing the link between landscape, culture and self (Morphy 1995).

The Alligator Rivers Region is proximal to southeast Asia and has long been a site of interaction between Australia and the rest of the world. Today it is among the least densely populated areas of Australia, but historically the resource-rich coastal floodplains sustained a pre-contact population that may have been among the highest of any hunting and gathering society in the world (Jones 1985). Not only are the processes of colonization, occupation and landscape change that occurred across the Top End exemplified by the recent history of the Alligator Rivers Region, but its ancient history also illustrates the substantial changes that have occurred since the Pleistocene.

Australia and New Guinea are in fact one continent connected by a broad shelf that has, in geologic terms, only recently flooded (Fig. 1.1a-b). The exposure of the Sahul shelf placed the present location of the Alligator Rivers Region several hundred kilometers inland, making it climatically drier when humans first colonized the Australian continent some 45,000 years before present (BP [O'Connell and Allen 2004]). The dry woodland vegetation of the time likely supported a much sparser population than the coastal floodplains that later developed (Allen 1994). A rapid Holocene rise in sea
Figure 1.1: (a) Changes in mean sea level from the Pleistocene to 3,000 BP (when sea level stabilized at the current level). The dashed line indicates 45,000 BP the earliest reliable date of human occupation in Australia (O’Connell and Allen 2004). Data from 85,000 to 13,000 BP are from the NOAA Millennial-Scale Glacial Sea Level Reconstruction (Arz et al. 2007). Data from 13,000 to 3,000 BP are from the NOAA Tahiti Glacial Relative Sea Level Reconstruction (Bard 2003). (b) Exposure of the Sahul shelf at 95m below mean sea level, the average level from 45,000 BP until the Holocene glacial retreat at 10,000 BP. The image is from Sahul Time (http://www.sahultime.monash.edu.au), and is reproduced with the permission of Matthew Coller, Monash University.
level starting in 10,000 BP (Fig. 1.1a) caused the South Alligator River valley to flood with salt water, and coastal mangroves appeared in the geological record around 8000 BP (Woodroffe et al. 1989). Over time, the river valley filled in with sediments from tidal deposition and the mangrove swamps gradually retreated (Woodroffe et al. 1985). It is likely that the resource rich freshwater floodplains that exist today along the South Alligator River are relatively young and did not establish until 1300 BP (Clark and Guppy 1988, Woodroffe et al. 1989). This time is coincident with a marked increase in the number of Aboriginal occupation sites on the floodplain, as well as an increase in the amount of occupation material within already established sites (Jones 1985, Meehan et al. 1989, Allen 1994).

When humans first colonized Australia, they probably induced changes to vegetation through the use of fire. This is often conceptualized as a continent wide transformation of vegetation (Kershaw 1986, Lewis 1994), perhaps contributing to the extinction of the marsupial megafauna (Koch and Barnosky 2006). However it is unlikely that humans shifted fire regimes to the degree that is often claimed. In particular the antiquity of the monsoonal weather pattern that drives fire regimes in the northern tropics and the ancient phylogeny of plant adaptations to fire there indicate that vegetation evolved to cope with frequent late dry season fires long before humans arrived (Bowman 2002), possibly coincident with a global increase in aridity in tropical latitudes (Cane and Molnar 2001). This is also evidenced by the persistence of fire sensitive vegetation communities in moist or sheltered regions where it is difficult for fire to penetrate. Head (1994b, 1996) suggests the duration and aridity of the dry season in northern Australia further intensified within the past 6,000 years (presently 90% of
annual rainfall falls between November and March and mean July and August rainfall is close to zero). Head (1996) postulates that the strong Aboriginal ethic of cleaning country through fire may have arisen relatively recently as a response to large scale, intense fires resulting from a prolonged dry season. The implication is that Aboriginal fire management has had a key role in maintaining vegetation in a state similar to what we see today.

The impact of a fire regime under current climate conditions but in the absence of humans can be seen today in depopulated areas of northern Australia where large and intense late season fires are much more common (Russell-Smith et al. 2003b). This is particularly evident in the Arnhem Land Plateau, where depopulation has led to a decline in obligate seeder communities (Price and Bowman 1994, Russell-Smith et al. 1998). If Head (1996) is correct, then not only did Aboriginal culture change in response to increased dry season aridity in the mid-Holocene, but fire sensitive communities in the Top End may now be dependent upon anthropogenic fire for persistence. This management tool is in fact a management relationship where anthropogenic fire is required to maintain ecological conditions quite different from those that would prevail in the absence of humans.

Compared with the dramatic climatic and cultural changes before European colonization, it might seem that the environmental and social changes after colonization were not particularly profound. However, several events indicate that European colonization was indeed uniquely transformative. The rich record of Aboriginal art in the Alligator Rivers Region suggests that the region has been more or less continually occupied by relatively homogeneous cultural groups for at least 23,000 years (Jones
Although there were technological and cultural changes during this period (some of which, particularly changes in art and tool production, are well recorded in the archaeological record while others are matters of speculation), none compared with the impact of European settlement on the physical and social landscape of Aboriginal people. First, Europeans introduced several large placental mammals to Australia, and, particularly, introduced grazing mega-herbivores at least 16,000 years after the extinction of the last marsupial mega-herbivores. Second, European colonization resulted in the depopulation of Aboriginal homelands, both from attrition and from Aboriginal settlement in localized communities. Third, social colonization dramatically shifted the perception and use of space among both Aboriginal and European societies, and hence their relationship with the natural world.

In this dissertation I explore these three aspects of European settlement, with particular attention placed on riparian communities. These communities are both socially and ecologically significant. They provide an important resource base and movement corridor for humans and continue to be central to Aboriginal interactions with their land. Additionally, they are an important dry season refuge for many animal communities (Woinarski and Braithwaite 1993, Woinarski et al. 2000). As Petty and Douglas describe in Chapter 2, the forest-woodland mosaic of riparian communities is susceptible to changes in fire management and thus is a useful indicator of management shifts. While wet season flooding is the primary determinant of woodland species composition, fire is important and frequent fires in woodland communities also reduce basal area and cover. Riparian forest composition doesn’t appear to be effected by fire, but the combination of fire and water availability during the dry season limits the outward extent of closed forest
from the water channel. Thus we would expect that any changes in the frequency or intensity of fires would have a demonstrable impact on riparian community structure and composition.

These riparian plant communities developed under Aboriginal land management and in the absence of large grazing populations for millennia until the Asian water buffalo (*Bubalus bubalis*) arrived on the mainland with the first European settlers in 1838 (Spillett 1972). As Petty et al. review in Chapter 3, the buffalo is the quintessential introduced mega-herbivore of the Top End. These early settlers unwittingly initiated a landscape scale experiment in savanna-grazer dynamics, one that neatly mirrored a twentieth century ‘natural experiment’ in Africa where grazer populations collapsed after a rinderpest epidemic (Sinclair and Norton-Griffiths 1979). The expansion and ecological impact of buffalo occurred within the context of settler attempts to colonize the Top End. Indeed, after the collapse of the hide industry following World War II, the nascent Northern Territory Agricultural and Animal Industry Branch attempted to use the ‘wild’ water buffalo as a lever to develop a domestic cattle industry in the northernmost floodplain regions of the Top End:

The branch policy is to foster this small industry and to try and place it on a sound permanent basis in the hope that the returns obtained by producers in an otherwise unproductive area might lead in time to more profitable pastoral development which would benefit the whole of the Top End (Anonymous 1961).

The policy ultimately failed, and buffalo became symbolic of the incomplete process of colonization in the Alligator Rivers Region (Robinson 2005). Arguably, the Alligator Rivers Region wasn’t brought under state control until the removal of buffalo in an eradication campaign. Thus, buffalo were subject both to the constraints of the physical
landscape and to the decisions of settler society. These decisions directly impacted on local buffalo populations which in turn had detectable effects on the physical landscape.

In addition to a post-buffalo era, the Alligator Rivers Region is now in a post-occupation era, where population densities are much lower than they were before contact, and those who live in the region, whether of European or Aboriginal descent, see the world in profoundly different terms. Chapter 4 explores the relationship between colonization and ecological perception and change. I draw on the dissimilar historical circumstances of three Aboriginal groups to explore the social and mental components of landscape and how these relate to the physical record of landscape change.

The historical record of settlement in the Alligator Rivers Region is poor, and we see the past through a dark glass indeed. Anecdotal observations of floodplain occupation sites indicate that post-contact material, particularly glass and metal, are usually absent (D. Lindner, pers. comm.). This would indicate a significant population shift in traditional occupation patterns before settlement began in earnest during the middle of the nineteenth century. Certainly there was widespread depopulation of the Alligator Rivers Region by the 1900s (Keen 1980).

The initial ecological impact of depopulation is difficult to determine as there are few records from the era. There was almost certainly an increase in large, hot fires in woodland and savanna communities, and possibly a decrease in fires on floodplains. The increased incidence of hot late season fires would likely have had a detrimental effect on riparian forest boundaries and riparian woodlands. As buffalo populations expanded grazing probably kept fuel loads and hence fire frequency and intensity down. Soon after the eradication of buffalo, Kakadu Park managers embarked on an intensive program of
prescribed burning that focused on burning beside riparian corridors to create fire breaks. Both of these factors have inscribed a series of changes in the landscape that is evident in the aerial photographic record. Other changes, including the abandonment of tended resource rich areas such as yam beds and the decreased hunting of game, would likely have had ecological impacts but are poorly studied.

In addition to these ecological changes, Aboriginal people experienced profound cultural changes that affected their relationship with the landscape. As I detail in Chapter 4, riparian corridors were conceptualized as continuous and interrelated spaces threading the landscape. Through their concentration in centralized communities and reliance on motor transport, Aboriginal movement through and knowledge about riparian places shifted to discrete ‘sites’ dotted across the landscape, and knowledge about places beyond individual experience became more diffuse and generalized.

However, perceptions of landscape and landscape change are not monolithic between Aboriginal groups. Those groups who lived in ‘cattle country’ in the south of Kakadu were disenfranchised from their land. For them road improvement in the 1970s and the establishment of Kakadu in 1979 meant the opening up of spaces that were previously restricted under European land regimes. Those who lived in the north of the Park were able to maintain a more continuous presence on their homelands by engaging with the buffalo industry. For them, the development of roads and the establishment of the Park resulted in the constriction of space to accommodate the demands of tourism and Park control. Not surprisingly, these groups have very different attitudes from European society about the ‘ferality’ of buffalo, and most concur that buffalo belong on country.
Indeed, some maintain that by keeping fuel loads down buffalo managed the land when people could not.

The results of this dissertation underscore the importance of understanding not only the ecological patterns and processes at work in a system (Chapter 2), but also the role of history and contingency in shaping landscape change (Chapters 3 and 4). Further, it illustrates the process of change not just in a natural system but a social one as well. European management paradigms, from the introduction of feral animals to ‘Europeanize’ the frontier to a mechanistic approach to the use of fire as a means of hazard reduction, have had visible and measurable effects on the landscape. Equally important was the shift in Aboriginal management, from dispersed communities traveling overland via riparian corridors to concentrated communities traveling by road. Differences in perception are also important, particularly within the context of Kakadu as a jointly managed national park. The concerns of Aboriginal people that today there are too many fires, that streamside vegetation is too thick, and that buffalo were land managers looking after a country depopulated of its original inhabitants, become clearer when one considers the context of the historical changes they have witnessed over their lifetimes. What remains to be seen is whether future management practices can respond no only to patterns of ecological change but also to the dramatic social changes that constitute part of the process of landscape change in the region.
Transverse and longitudinal variation in woody vegetation along the South Alligator River

Aaron M. Petty and Michael M. Douglas

ABSTRACT

Riparian vegetation responds to changes in ecological processes across both transverse (across channel) and longitudinal (parallel to channel, throughout the watershed) dimensions. Vegetation responses have been well described across transverse gradients, but comprehensive studies of longitudinal variation are rare and mostly restricted to moist temperate or arid and semi-arid subtropical regions. We sampled riparian vegetation along a transverse and longitudinal gradient within the South Alligator River catchment in Kakadu National Park, located in the wet/dry tropics of northern Australia. Riparian vegetation was highly heterogeneous across both transverse and longitudinal gradients, but formed distinct woodland, floodplain and closed forest communities. Woodland and floodplain communities varied according to both fire frequency and measures of stream size and topographic position (presumably responding to the degree of wet season inundation). Woodland recruitment and stand basal area were negatively correlated with fire frequency. Riparian closed forest was restricted to water
channels and presumably dependent upon dry season water table access, but otherwise there was little transverse variation in community composition. As pressure to develop Australia’s tropical region grows, land managers should consider the impact of increased groundwater use on riparian closed forest vegetation.

INTRODUCTION

Environmental processes within riparian communities change across both longitudinal (parallel to stream flow, throughout the watershed) and transverse (perpendicular to stream flow, across the stream channel) gradients (Bendix 1994b, Church 2002). Longitudinal variation in flow duration and rates of sediment erosion and deposition contribute to the creation of distinct structural typologies within river systems (Church 2002). In general, as rivers move longitudinally from the headwaters to lower elevations, they gain complexity in the transverse dimension (Naiman and Decamps 1997). The topographical variation of transverse profiles determines water table access during low flows and the period and depth of inundation during high flows (e.g. Hupp and Osterkamp 1996), and influences disturbance regimes, particularly fire and flooding (e.g. Busch and Smith 1995). Thus transverse and longitudinal river structures have a direct impact on plant recruitment and select for varying plant life history strategies.

It is well known that riparian plant community composition varies as species respond to longitudinal changes in geomorphological processes, channel constraint, and fluvial regimes. These patterns have been described worldwide, although most studies focus on the moist temperate regions of Europe and North America (e.g. Hupp and Osterkamp 1996, Cordes et al. 1997, Tabacchi et al. 1998, Naiman et al. 2005, p. 101), or arid, semi-arid and sub-tropical regions of Europe, Africa, Australia and southwestern
North America (e.g. Harris 1988, Hughes 1988, Bendix 1994a, Busch and Smith 1995, Hupp and Osterkamp 1996, Tabacchi et al. 1996, Van Coller et al. 1997, Petit et al. 2001, Friedman et al. 2006). Many of these studies occur at the extremes of a climate continuum. On one end are river systems in moist regions where water flow is relatively continuous with periodic floods of varying magnitude. In these systems susceptibility to flooding and frequency of inundation are the key determinants of vegetation composition (Hupp and Osterkamp 1996). At the other extreme are arid systems where rainfall is highly variable and stochastic and access to the water table as well as periodic recruitment events drive vegetation composition and structure (Hupp and Osterkamp 1996). Riparian communities within such biomes provide a clear contrast to surrounding vegetation (Busch and Smith 1995). The prevalence of fire as a structuring force on riparian vegetation distinguishes semi-arid and arid zones from moist temperate regions, and the arid and semi-arid zones of southwestern North America and Southern Africa have a particularly well developed literature with regard to the contrasting influences of fire and flood on riparian vegetation (e.g. Oba 1990, Bendix 1994b, Busch 1995, Busch and Smith 1995, Bendix 1998, Mosugelo et al. 2002, Friedman et al. 2006, Pettit and Naiman 2007b).

Tropical and sub-tropical savanna river systems sit midway between the extremes of mesic and arid river systems. Within these biomes, rainfall is more or less consistent, but highly seasonal, and riparian vegetation must contend with extreme hydrological variation (Dodds 1997). Moreover, the combination of sufficient annual moisture to promote substantial vegetative growth during the wet season and prolonged dry season drought make tropical savannas the most fire-prone biome on Earth (Dwyer et al. 2000).
Perhaps because of the remoteness of many tropical savanna rivers, longitudinal studies are rare and largely restricted to semi-arid savanna systems in Africa (Hughes 1988, Van Coller et al. 1997). Studies within mesic tropical savannas (i.e. savannas that receive more than 1200 mm mean annual rainfall) are predominantly limited to descriptions of transverse patterns at single sites (but see Douglas et al. 2003, Franklin and Bowman 2004), where surface and subsurface hydrology (Bowman and McDonough 1991) or isolation from fire (Biddulph and Kellman 1998, Kellman et al. 1998) segregate vegetation into mesic/xeric or fire intolerant/tolerant communities.

In this paper we investigate vegetation patterns across the South Alligator River catchment, a relatively undisturbed 9,000 km² watershed almost entirely contained within the boundaries of the World Heritage listed Kakadu National Park and situated within the mesic tropical savanna zone of Northern Australia. To our knowledge this is the first comprehensive longitudinal study of vegetation patterns in a mesic tropical savanna river system. Our aim is to describe dominant vegetation and recruitment patterns of 28 sites located in upland, valley and lowland reaches of this catchment, and correlate these patterns with biotic and environmental variables. We describe the variation in vegetation patterns across transverse (within site) and longitudinal (between site) gradients, as well as the relative importance of environmental correlates. We argue that extreme seasonality in rainfall is the principal environmental driver of the complex mosaic of riparian communities within the South Alligator catchment wherein individual plant species segregate by their ability to withstand varying degrees of drought and fire in the dry season and scouring and inundation in the wet.
Figure 2.1: Map of the South Alligator River catchment and study locations.

Study area

The riparian communities of Kakadu National Park thrive amongst annual disturbance cycles of flood and fire. Rainfall is highly seasonal – over 90% of the mean annual rainfall (1100-1500 mm) falls between November and March and water levels commonly exceed stream bank height (Moliere et al. 2002). Riparian vegetation exists within a matrix of *Eucalyptus miniata* and *E. tetrodonta* dominated savanna (Wilson et
al. 1991). The 1-2 m high grass layer cures in the dry season to carry fires that burn 50% of the landscape annually (Russell-Smith et al. 1997b).

The flora of the region lies along a moisture and topographic gradient that varies with dry season access to the water table and depth and length of wet season inundation (Langkamp et al. 1981, Bowman and Minchin 1987, Bowman and McDonough 1991, Finlayson 2005), although soil (Langkamp et al. 1981, Bowman and Dunlop 1986, Franklin et al. 2007) and fire (Russell-Smith et al. 2003b, Woinarski et al. 2004) also contribute significantly to the complex patchwork of ecological communities. Broadly speaking, the hydrological gradient extends from low lying annually submerged floodplains, then intermittently flooded woodlands, and finally more elevated woodlands that are not inundated.

The South Alligator River catchment crosses a south-north elevational gradient that crosses three landsystem types – the Arnhem Land Plateau, the Gimbat Valley, and the lowland plains (Figs. 1, 2a-d). These elements are analogous to the erosional, transitional and depositional river provinces described by Tabacchi et al. (1998) but with two important qualifiers: (1) The gradient across the Arnhem Land Plateau is slight compared to the headwaters of the river systems described by Tabacchi et al. (1998) and the rate of upland erosion is extremely low (Saynor and Erskine 2006); (2) Most deposition comes from downstream via tidal influx, which extends 105 km inland (Woodroffe et al. 1989) rather than from upstream sources.

The headwaters of the South Alligator River are located on the Arnhem Land Plateau, a mass of uplifted proterozoic sandstone lying 150-250 m above the surrounding lowlands (Fig. 2.1, 2.2a). Soils are generally veneers of sand seldom more than 150 mm
thick (Russell-Smith et al. 1995). Creeks are typically confined within very narrow channels with the sandstone block forming steep, sometimes vertical, banks. The porous sandstone also supports a large aquifer and numerous fissures produce springs that maintain dry season flow within the major channel of the South Alligator River and some major tributaries. However, most of the lower order streams in the catchment are seasonal and cease flowing during the dry season.

The South Alligator River descends from the Plateau into the Gimbat Valley, a large valley running roughly southeast-northwest (Fig. 2.2b). Stream structure within the Gimbat Valley is often braided and complex with alternating depositional and erosional reaches. Step-pools are more abundant in the upper reaches in the valley, and pool-riffle-bar structures (Church 2002) are common in the lower reaches. Much of the watercourse is constrained to alluvial channels by vegetated sandy ridges (Wende and Nanson 1998).

After the Gimbat Valley the river crosses into large undulating lowland plains comprised of highly eroded and laterised soils. Flow is contained by a deeply incised channel bounded with alluvial ridges that are regularly breached in the wet season (Fig. 2.2c). Some 120 km from the coast, the river joins two major tributaries, Jim Jim and Barramundie creeks, and the active river channel increasingly interconnects with a series of palaeochannels and cracking-clay floodplains that fill during the wet season and remain underwater for a substantial portion of the year (typically 4-7 months [Finlayson 2005]; Fig. 2.2d). Below its confluence with Jim Jim Creek, the active river channel drains into a vast floodplain. A tidal channel reforms some ten kilometers downstream but most wet season flow is carried as surface flow across floodplains that reach up to 20 km wide.
We define the riparian zone as the limit of annual flooding adjacent to a water bearing channel (Gregory et al. 1991). The vegetation adjacent to the main water bearing channel forms a zone that is distinctive by its high canopy cover and dense vegetation. We define this region as the “riparian closed forest”, or simply “riparian forest”, community. The adjacent vegetation is more open, but subjected to annual flooding. This we refer to as “riparian woodland”.

Figure 2.2: (a) A South Alligator River tributary within a ravine of the Arnhem Land Plateau. The well developed forest is dominated by *Melaleuca leucadendra* and *Xanthostemon eucalyptoides*. (b) A depositional stretch of the South Alligator River containing sand bars in the Gimbat Valley. Numerous *Melaleuca leucadendra* saplings are growing along the sandbar, with more developed riparian forest in the background. (c) A representative section of the South Alligator River in the lowland plains. Large *Melaleuca argentea* and *M. leucadendra* trees are abundant along the stream edge, intermixed with *Barringtonia acutangula*. (d) A perennial backwater lagoon (billabong) fringed with *Melaleuca viridiflora* that connects to the South Alligator River in the wet season.
METHODS

Riparian zone community structure

We surveyed 28 sites (Fig. 2.1) along either active channels of the South Alligator River and its tributaries or backwater depressions on floodplains that connect to the South Alligator River in the wet season and retain water throughout the dry season (perennial billabongs). Surveys were conducted from April to September in 2003 and 2004. Sites were selected on the basis of interviews with both Aboriginal residents and park staff to reflect a representative sample of communities throughout the Arnhem Land Plateau, Gimbat Valley and lowlands.

Within each site we established four to six transects of 200 m length, each running parallel to the main direction of the channel. We typically ran three transects on each side of a channel although if the opposite bank was inaccessible we established an additional three transects on a nearby riparian area within 2 km of the initial transects. Two transects were established within riparian forest that was greater than 10 m wide, one adjacent to the main water-bearing channel and the second at a distance marking the midpoint of the forest community. In forests narrower than 10 m (22 out of 59) only one transect was used, and placed within the midpoint of the forest community. At all sites an additional transect was placed within the mid-point of the riparian woodland community adjacent to the riparian forest (or 50 m from the forest boundary if the distance was immeasurable).

At the site of each set of transects we recorded the widths of the riparian forest and woodland communities, the distance to the opposite bank of the water bearing channel, the bank slope, the slope of the riparian forest and woodland communities
Variation in riparian vegetation

(measured orthogonally to the active channel), and the aspect of the active channel. If we were unable to calculate the riparian woodland distance in the field, the distance was estimated from 1:25,000 aerial photos taken in 2004. Woodland distances were positively skewed and were log transformed to approximate normality. Soil type was determined at each transect using the ribbon test of McDonald et al. (1998) and then classified into five groups: clay, clay-loam, loam, sand-loam, and sand.

We recorded the species, diameter at breast height (DBH), and distance of the four closest trees above 5 cm DBH along each transect at 20 m intervals (the point-center-quarter method [Mueller-Dombois and Ellenberg 1974]). For multi-stemmed trees, individual stems were recorded and basal area summed. Each tree was also scored for the presence or absence of charring on the trunk. At each 20 m interval we recorded the canopy cover using a densitometer and recorded the cover of seedlings (<1 m height), saplings (>1 m height and <5 cm DBH), grasses, sedges, forbs, bamboo, vines, shrubs, litter, bare soil, rock, coarse woody debris (>10 cm diameter), and pig damage within a 1 m quadrat. Species names were recorded for seedlings, saplings, vines and shrubs. Cover was recorded using a 5-point Braun-Blanquet scale (Kent and Coker 1992, p. 45). Finally, if any portion of the quadrat was burned, that quadrat was scored as burned. Strahler stream order (Strahler 1952) of the channel associated with each transect was calculated from 1:100,000 topographic map sheets covering the South Alligator watershed (Royal Australian Survey Corps, Australian Defense Force, 1988).

Fire frequency was determined from a fifteen year Landsat MSS (1989 – 1995) and Landsat TM (1996 – 2004) derived fire history provided by the Bushfires Council of the Northern Territory. The year 2002 was excluded due to cloud cover. All fire data
were resampled to 100 m, the scale of the lower resolution Landsat MSS derived fire scars, and converted to polygons. The start and end of transects were located using a handheld GPS (approximate error 5-10 m). Each transect was considered burned in a given year if over 50% of the transect lay within a burned polygon. The year 2004 was ignored for sites sampled in 2003.

Data analysis

Species groups and a-priori transect classifications. – A total of 121 woody species over 5 cm DBH were recorded. We combined species with highly similar functional attributes and/or characteristics difficult to distinguish in the field to create eight groups, or a total of 105 taxon groupings, as follows: Group “Acacia sp.” – Acacia difficilis Maiden, A. gonocarpa F.Muell., A. hemignostica F.Muell., A. holosericea A.Cunn. ex G.Don, A. lacertensis Pedley, A. lamprocarpa O.Schwartz, A. montfordiae Specht, A. plectocarpa A.Cunn. ex Benth., A. sericoflora Pedley, A. torulosa Benth., A. tropica (Maiden & Blakely) Tindale; group “Calytrix” – Calytrix arborescens (F.Muell.) Benth., C. brownii (Schauer) Craven; group “Corymbia latfoel” – Corymbia foelscheana (F.Muell.) K.D.Hill & L.A.S.Johnson, C. latifolia (F.Muell.) K.D.Hill & L.A.S.Johnson; group “Ficus” – Ficus brachypoda (Miq.) Miq., F. coronulata Miq., F. racemosa L., F. virens Aiton; group “Ficus sandpaper” – Ficus aculeata Miq., F. scobina Benth.; group “Gardenia” – Gardenia fucata R.Br. ex Benth., G. kakaduensis Puttock; group “Pavetta” - Pavetta brownii Bremek. Pavetta sp.

Each transect was attributed into one of eight groups based on (1) vegetation structure (e.g. riparian closed forest and woodland), (2) the landsystem association, or “region”, of the site (Plateau, Gimbat Valley, lowland). All transects within sites at
backwater depressions (billabongs) were classified as “floodplain” as they were often surrounded by homogeneous vegetation without a distinct closed forest fringe. To test for transverse variation in riparian closed forests, transects occurring within forest were also further classified by whether they were closest to the watercourse (“stream”) or in the middle of the riparian zone (“middle”). Where there was only one forest transect it was attributed as “stream”.

**Multivariate analyses.** – We used two techniques to explore the biogeographic differences between riparian community composition: (1) Ordination using non-metric multidimensional scaling (MDS) with the packages MASS 7.2-29 and Vegan 1.8-5 in R 2.4.0 (http://cran.r-project.org). Bray-Curtis dissimilarity was calculated between the fourth-root transformed dominance scores of all taxon groups within each transect. (2) The percentage of similarity between each transect class (stream, middle, savanna and floodplain) as well as the contribution of each taxon to the overall similarity between classes was quantified by the SIMPER routine in PRIMER (Version 6.1.2, Plymouth Marine Labs, Plymouth, UK [Clarke and Warwick 1994]). These differences were statistically tested using one-way analysis of similarities (ANOSIM) to non-parametrically test the average rank similarities of samples between classes.

To explore the correlation between dominant vegetation and other environmental variables we first divided the environmental data set into three groups:

(1) **Soil type:** a categorical environmental variable.

(2) **Width of the riparian woodland and forest zones; distance to the opposite bank; slope of the bank, riparian forest zone and riparian woodland zone; channel aspect; stream order; rock cover; fire frequency, percent of**
trees charred and percent of quadrats burned: continuous environmental variables that were normalized by subtracting from the mean and dividing by the standard deviation for each variable. This gives the each variable the same mean (0) and variance (1) so that comparisons between variables of different scales and origins were meaningful.

(3) Cover of saplings, seedlings, pig damage, grass, forbs, bamboo, vines, shrubs, litter, bare soil and coarse woody debris; and total basal area: continuous measurements of biological abundance that were standardized by dividing each variable by its maximum value. Standardization was used to place each variable within the same range (i.e. the proportion of the maximum observed value) because abundance values varied widely, e.g. grass, litter, bare soil and basal area had relatively high values while pig damage, coarse woody debris, bamboo and vines were sparse and had low values.

In each of the latter two data sets variables were selected that maximized the rank correlation between a similarity matrix of the variable set and the transformed dominant vegetation scores (the BEST procedure in PRIMER 6.1.2). Subset similarity matrices of taxa that contributed >10% to total dominance in any transect were also correlated with the overall vegetation matrix to determine the subset of taxa that most explained the variation between communities. Soil was fitted to the MDS ordination scores as a generalized additive model (GAM) using the R package “vegan.”

The BEST procedure is considered more robust than other methods that typically rely on linear regression (Clarke and Ainsworth 1993). However, it is difficult to visually interpret. We decided to use BEST to select the most important variables and then plot
Table 2.1: Percent similarity of mature woody species between and within groups (within-group similarities are indicated by bold). Brackets indicate the level of difference between groups using the ANOSIM non-parametric test statistic R in Primer 6.1.2 (0 indicates complete similarity, 1 indicates complete dissimilarity). The significance of R against the null hypothesis of no difference is indicated by an asterisk (P < .001).

<table>
<thead>
<tr>
<th></th>
<th>Floodplain</th>
<th>Woodland</th>
<th>Middle</th>
<th>Stream</th>
</tr>
</thead>
<tbody>
<tr>
<td>Floodplain</td>
<td>29.87</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Woodland</td>
<td>9.02 (R=0.546*)</td>
<td>19.69</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Middle</td>
<td>9.20 (R=0.606*)</td>
<td>12.23 (R=0.368*)</td>
<td>23.87</td>
<td></td>
</tr>
<tr>
<td>Stream</td>
<td>10.05 (R=0.628*)</td>
<td>9.05 (R=0.585*)</td>
<td>25.16 (R=0.037)</td>
<td>26.68</td>
</tr>
</tbody>
</table>

those variables as linearly correlated vectors. This is useful for interpretation, but can be misleading when it masks non-linear patterns. To avoid misinterpretation we separately plotted each variable as a fitted two-dimensional GAM surface. Where there are significant deviations from linearity we discuss them in the explanatory text.

RESULTS

Dominant vegetation structure

The SIMPER-derived measurement of similarity within transect groups was very low, indicating a high degree of diversity between sites (the most similar group, floodplains, had a within-group similarity of only 30%; Table 2.1). Moreover, despite what would seem to be very clear distinctions between forest, woodland and floodplain vegetation communities, there was intergradation of taxa between community types, and most transect groups shared about 10% of species with other groups. There was no distinguishable difference between the dominant taxa of stream and middle transects within riparian forest.
Table 2.2: Fire history and cover by transect type. “Fire frequency” is the mean percent of years where a fire intersected over 50% of the transect, as determined from Landsat derived fire scars from 1989 to 2004. Stream and middle transects were too close spatially to derive separate fire frequencies and were combined. “% burned” indicates the mean percentage of quadrats scored as burned during field surveys. “% charred” indicates the mean percentage of trees with indications of charring scored during field surveys. Standard errors are provided in parenthesis.

<table>
<thead>
<tr>
<th>Transect type</th>
<th>% Cover</th>
<th>% Fire Frequency</th>
<th>% burned</th>
<th>% charred</th>
</tr>
</thead>
<tbody>
<tr>
<td>Floodplain</td>
<td>52.6 (±7.0)</td>
<td>21.2 (±5.0)</td>
<td>11.2 (±6.0)</td>
<td>30.2 (±8.3)</td>
</tr>
<tr>
<td>Closed Forest - stream</td>
<td>77.9 (±3.6)</td>
<td>2.6 (±2.4)</td>
<td>25.6 (±6.4)</td>
<td></td>
</tr>
<tr>
<td>Closed Forest - middle</td>
<td>73.4 (±5.5)</td>
<td>28.6 (±2.9)</td>
<td>3.7 (±3.7)</td>
<td>30.9 (±9.1)</td>
</tr>
<tr>
<td>Woodland</td>
<td>38.9 (±4.5)</td>
<td>34.2 (±3.0)</td>
<td>44.5 (±8.0)</td>
<td>72.3 (±7.2)</td>
</tr>
</tbody>
</table>

Forest and woodland vegetation segregated by cover and the occurrence of fires (Table 2.2, Fig. 2.3a-b), as well as floristic composition (Table 2.3). Soil texture segregated with community type along both longitudinal and transverse axes: woodlands were typically found on sandy loams or sands, riparian communities on sand, and floodplains on clay and clay loams (Fig. 2.4a). Both higher basal area and ground cover (forbs, vines and saplings) were associated with riparian forests, suggesting greater productivity (Fig. 2.4b).

The transverse profile of streams differed between regions (Table 2.4). Bank slope was relatively similar across regions, but this was an inadequate indication of the transverse profile. Floodplain banks often had a very slight profile (on the order of 10 cm), whereas lowland banks along the main channel of the South Alligator River were frequently 5-10 m high (personal observation). A better, albeit indirect, indicator of vertical relief was the slope within the forest vegetation zone, which typically abutted the dry season watercourse and rarely extended more than 10 m beyond the channel bank. Average slope within forest vegetation was similar between the Plateau and Gimbat Valley, and then increased dramatically in the lowland plains (Table 2.4). By contrast,
Figure 2.3: Fire along the woodland – closed forest boundary. (a) An aerial view of Gerowie Creek, a small tributary of the South Alligator River. A fire through upland and riparian woodland in the foreground stopped short of the darker band of closed forest following the creek horizontally across the upper third of the photo. The adjoining riparian woodland in the background is unburned. (b) A fire along a different section of Gerowie Creek stopped after burning into the edge of the riparian forest.
the average slope of the adjacent woodland vegetation declined, and the width of the woodland vegetation zone increased, an overall indication of the increased extent of the seasonal inundation zone in the lowlands (Fig. 2.4a). Closed forests along the more developed banks of lowland plains did not differ in width from their upstream counterparts (Table 2.4), but were structurally different with greater basal area, less ground cover and bamboo stands (Fig. 2.4b).

In all sites, the species composition in riparian closed forests was largely independent of the transverse profile (Fig. 2.5a-c). Forests were generally dominated by *Melaleuca leucadendra*. *M. argentea* was a common dominant or co-dominant with *M. leucadendra* along lowland forests but rarely occurred in the Plateau or Gimbat Valley, and the competitive advantage of *M. leucadendra* vis a vis *M. argentea* in these regions is unclear. Rainforest species were common within most riparian forests, and associated rainforest species broadly conformed to the geographic provenance of the rainforest classification schema developed for northwestern Australia, suggesting vicariant or edaphic affiliations with regions (Russell-Smith 1991, Russell-Smith and Bowman 1992). Two riparian forest sites were floristically distinct from each other and all other sites and contained solely rainforest species with no *Melaleuca* spp. present. These highlight the enormous diversity within and between the relatively small rainforest pockets that line streams in monsoonal north Australia. Woodland species (*Lophostemon lactifluus*, *Grevillea pteridifolia*, *Erythrophleum chlorostachys*, and *Corymbia ptychocarpa*) were more commonly associated with Plateau forest sites than with sites in other regions.

The composition of open woodland varied greatly. Stream order and woodland slope were negatively correlated (Fig. 2.6a) such that high order lowland streams often
had low woodland profiles, and hence were particularly susceptible to flooding. Low-order streams on the Plateau were frequently abutted by *Eucalyptus tetrodonta* woodlands (Fig. 2.6c). *Syzygium suborbiculare*, *Erythrophleum chlorostachys* and other upland vegetation were more common within woodlands on high slopes. *Corymbia bella*, *C. ptychocarpa*, *C. polycarpa*, *C. grandifolia* and *Lophostemon lactifluus* occupied lower positions. At the lowest point of the topographic gradient, *Melaleuca viridiflora* commonly occupied poorly drained depressions within lowland woodlands.

Floodplain vegetation separated along two orthogonal axes corresponding to water availability: (1) a longitudinal gradient of wet season water volume (stream order) and (2) internal variation in the capacity of depressions to hold water through the dry season (channel width; Fig. 2.7a). Higher basal area, leaf litter and sapling cover were more associated with high stream order locations (Fig. 2.7b). *Melaleuca viridiflora* tended to be the dominant component of closed forests along high order backwater depressions while *Vitex glabrata* and *Corymbia bella* were associated with both higher fire frequencies and depressions associated with lower order streams (Fig. 2.7c). Two shrub species, *Cathormion umbellatum* and *Asteromyrtus symphyocarpa* (F.Muell.) Craven, segregated by channel width with the former more closely associated with smaller channels.
Table 2.3: Taxa contributions to similarity within each vegetation community. Grouped taxa are not italicized; groupings are explained in methods. Average basal area (m²/ha), average within-group similarity, the ratio of mean similarity to standard deviation, the contribution of each taxa to total within-group similarity, and the cumulative percent contribution are shown for each taxa. Deciduousness and the ability to resprout after fire are derived from the Tropical Savannas Fire Response Database (Gardener & Marrinan 2007).

<table>
<thead>
<tr>
<th>Riparian forest taxa</th>
<th>Basal Area</th>
<th>Av. Sim.</th>
<th>Sim/SD</th>
<th>Cum.%</th>
<th>Deciduous</th>
<th>Resprout</th>
</tr>
</thead>
<tbody>
<tr>
<td>Melaleuca leucadendra L. MYRTACEAE</td>
<td>0.37</td>
<td>5.63</td>
<td>0.83</td>
<td>19.11</td>
<td>-</td>
<td>+</td>
</tr>
<tr>
<td>Pandanus aquaticus F.Muell. PANDANACEAE</td>
<td>0.20</td>
<td>3.96</td>
<td>1.33</td>
<td>32.57</td>
<td>-</td>
<td>+</td>
</tr>
<tr>
<td>Melaleuca argentea W.Fitzg. MYRTACEAE</td>
<td>0.31</td>
<td>3.76</td>
<td>0.67</td>
<td>45.34</td>
<td>-</td>
<td>+</td>
</tr>
<tr>
<td>Syzygium armstrongii B.Hyland MYRTACEAE</td>
<td>0.21</td>
<td>2.43</td>
<td>0.65</td>
<td>53.61</td>
<td>-</td>
<td>+</td>
</tr>
<tr>
<td>Barringtonia acutangula (L.) Gaertn. LECYTHIDACEAE</td>
<td>0.16</td>
<td>1.46</td>
<td>0.45</td>
<td>58.57</td>
<td>+</td>
<td>+</td>
</tr>
<tr>
<td>Pandanus spiralis R.Br. PANDANACEAE</td>
<td>0.13</td>
<td>1.33</td>
<td>0.55</td>
<td>63.08</td>
<td>-</td>
<td>+</td>
</tr>
<tr>
<td>Acacia auriculiformis A.Cunn. ex Benth. MIMOSACEAE</td>
<td>0.13</td>
<td>1.13</td>
<td>0.45</td>
<td>66.9</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Lophostemon lactifluus (F.Muell.) Peter Wilson &amp; J.T.Waterh. MYRTACEAE</td>
<td>0.13</td>
<td>1.06</td>
<td>0.42</td>
<td>70.51</td>
<td>-</td>
<td>+</td>
</tr>
<tr>
<td>Acacia sp. MIMOSACEAE</td>
<td>0.11</td>
<td>1.01</td>
<td>0.46</td>
<td>73.95</td>
<td>(mostly)</td>
<td>(mostly)</td>
</tr>
<tr>
<td>Syzygium forte (F.Muell.) B.Hyland MYRTACEAE</td>
<td>0.15</td>
<td>0.91</td>
<td>0.31</td>
<td>77.03</td>
<td>-</td>
<td>+</td>
</tr>
<tr>
<td>Taxon</td>
<td>Family</td>
<td>Basal Area</td>
<td>Av. Sim.</td>
<td>Sim/SD</td>
<td>Cum.%</td>
<td>Deciduous</td>
</tr>
<tr>
<td>--------------------------------------------</td>
<td>--------------</td>
<td>------------</td>
<td>----------</td>
<td>--------</td>
<td>--------</td>
<td>-----------</td>
</tr>
<tr>
<td><em>Xanthostemon eucalyptoides</em> F. Muell.</td>
<td>Myrtaceae</td>
<td>0.13</td>
<td>0.87</td>
<td>0.33</td>
<td>80.00</td>
<td>-</td>
</tr>
<tr>
<td><em>Melaleuca viridiflora</em> Sol. Ex Gaertn.</td>
<td>Myrtaceae</td>
<td>0.11</td>
<td>0.87</td>
<td>0.37</td>
<td>82.94</td>
<td>-</td>
</tr>
<tr>
<td><em>Nauclea orientalis</em> (L.) L. Rubiaceae</td>
<td>Rubiaceae</td>
<td>0.11</td>
<td>0.68</td>
<td>0.31</td>
<td>85.24</td>
<td>+</td>
</tr>
<tr>
<td><em>Grevillea pteridifolia</em> Knight</td>
<td>Proteaceae</td>
<td>0.07</td>
<td>0.57</td>
<td>0.36</td>
<td>87.19</td>
<td>-</td>
</tr>
<tr>
<td><em>Carallia brachiata</em> (Lour.) Merr.</td>
<td>Rhizophoraceae</td>
<td>0.09</td>
<td>0.55</td>
<td>0.39</td>
<td>89.06</td>
<td>-</td>
</tr>
<tr>
<td><em>Ficus</em></td>
<td>Moraceae</td>
<td>0.10</td>
<td>0.45</td>
<td>0.22</td>
<td>90.58</td>
<td>-</td>
</tr>
</tbody>
</table>

Riparian Woodland Taxa

<table>
<thead>
<tr>
<th>Taxon</th>
<th>Family</th>
<th>Basal Area</th>
<th>Av. Sim.</th>
<th>Sim/SD</th>
<th>Cum.%</th>
<th>Deciduous</th>
<th>Resprout</th>
</tr>
</thead>
<tbody>
<tr>
<td><em>Erythrophleum chlorostachys</em> (F.Muell.) Baille.</td>
<td>Caesalpiniaceae</td>
<td>0.17</td>
<td>2.85</td>
<td>0.64</td>
<td>14.23</td>
<td>+</td>
<td>+</td>
</tr>
<tr>
<td><em>Corymbia bella</em> K.D.Hill &amp; L.A.S.Johnson</td>
<td>Myrtaceae</td>
<td>0.14</td>
<td>2.41</td>
<td>0.46</td>
<td>26.24</td>
<td>-</td>
<td>+</td>
</tr>
<tr>
<td><em>Syzygium suborbiculare</em> (Benth.) T.G.Hartley &amp; L.M.Perry</td>
<td>Myrtaceae</td>
<td>0.12</td>
<td>1.73</td>
<td>0.49</td>
<td>34.88</td>
<td>-</td>
<td>+</td>
</tr>
<tr>
<td><em>Lophostemon lactifluus</em></td>
<td>Myrtaceae</td>
<td>0.13</td>
<td>1.62</td>
<td>0.44</td>
<td>42.93</td>
<td>-</td>
<td>+</td>
</tr>
<tr>
<td><em>Corymbia polycarpa</em> (F.Muell.) K.D.Hill &amp; L.A.S.Johnson</td>
<td>Myrtaceae</td>
<td>0.11</td>
<td>1.09</td>
<td>0.34</td>
<td>48.34</td>
<td>-</td>
<td>+</td>
</tr>
<tr>
<td><em>Melaleuca viridiflora</em></td>
<td>Myrtaceae</td>
<td>0.10</td>
<td>1.00</td>
<td>0.30</td>
<td>53.32</td>
<td>-</td>
<td>+</td>
</tr>
<tr>
<td><em>Corymbia ptychocarpa</em> (F.Muell.) K.D.Hill &amp; L.A.S.Johnson</td>
<td>Myrtaceae</td>
<td>0.09</td>
<td>0.99</td>
<td>0.34</td>
<td>58.24</td>
<td>-</td>
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</tr>
<tr>
<td>Species</td>
<td>Family</td>
<td>Dominant</td>
<td>Abundant</td>
<td>Subordinate</td>
<td>Dominant</td>
<td>Abundant</td>
<td>Subordinate</td>
</tr>
<tr>
<td>-------------------------------</td>
<td>--------------</td>
<td>----------</td>
<td>----------</td>
<td>-------------</td>
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<td>-------------</td>
</tr>
<tr>
<td><em>Pandanus spiralis</em></td>
<td>Pandanaceae</td>
<td>0.08</td>
<td>0.97</td>
<td>0.44</td>
<td>63.06</td>
<td>-</td>
<td>+</td>
</tr>
<tr>
<td><em>Eucalyptus tetrodonta</em> F.Muell.</td>
<td>Myrtaceae</td>
<td>0.10</td>
<td>0.76</td>
<td>0.28</td>
<td>66.85</td>
<td>-</td>
<td>+</td>
</tr>
<tr>
<td><em>Eucalyptus patellaris</em> F.Muell.</td>
<td>Myrtaceae</td>
<td>0.08</td>
<td>0.61</td>
<td>0.21</td>
<td>69.91</td>
<td>-</td>
<td>+</td>
</tr>
<tr>
<td>Ficus sandpaper</td>
<td>Moraceae</td>
<td>0.05</td>
<td>0.50</td>
<td>0.31</td>
<td>72.42</td>
<td>-</td>
<td>+</td>
</tr>
<tr>
<td>Acacia sp.</td>
<td>Mimosaceae</td>
<td>0.05</td>
<td>0.50</td>
<td>0.31</td>
<td>74.91</td>
<td>(mostly)</td>
<td>(mostly)</td>
</tr>
<tr>
<td><em>Buchanania obovata</em> Engl.</td>
<td>Anacardiaceae</td>
<td>0.06</td>
<td>0.49</td>
<td>0.28</td>
<td>77.32</td>
<td>-</td>
<td>+</td>
</tr>
<tr>
<td><em>Corymbia grandifolia</em> (R.Br. ex Benth.) K.D.Hill &amp; L.A.S.Johnson</td>
<td>Myrtaceae</td>
<td>0.06</td>
<td>0.46</td>
<td>0.27</td>
<td>79.6</td>
<td>-</td>
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<td><em>Terminalia platyphylla</em> F.Muell.</td>
<td>Combretaceae</td>
<td>0.05</td>
<td>0.45</td>
<td>0.22</td>
<td>81.86</td>
<td>+</td>
<td>+</td>
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<td>Corymbia latifolia</td>
<td>Myrtaceae</td>
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<td>0.37</td>
<td>0.22</td>
<td>83.70</td>
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<td><em>Barringtonia acutangula</em></td>
<td>Lecythidaceae</td>
<td>0.05</td>
<td>0.28</td>
<td>0.20</td>
<td>85.10</td>
<td>+</td>
<td>+</td>
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<td>0.05</td>
<td>0.25</td>
<td>0.16</td>
<td>86.34</td>
<td>-</td>
<td>+</td>
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<tr>
<td><em>Vitex glabrata</em> R.Br.</td>
<td>Verbenaceae</td>
<td>0.04</td>
<td>0.25</td>
<td>0.18</td>
<td>87.57</td>
<td>+</td>
<td>+</td>
</tr>
<tr>
<td><em>Terminalia grandiflora</em> F.Muell.</td>
<td>Combretaceae</td>
<td>0.04</td>
<td>0.24</td>
<td>0.20</td>
<td>88.77</td>
<td>+</td>
<td>+</td>
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Variation in riparian vegetation
<table>
<thead>
<tr>
<th>Floodplain Taxa</th>
<th>Basal Area</th>
<th>Av. Sim.</th>
<th>Sim/SD</th>
<th>Cum. %</th>
<th>Deciduous</th>
<th>Resprout</th>
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</thead>
<tbody>
<tr>
<td><em>Eucalyptus miniata</em> F.Muell.</td>
<td>0.05</td>
<td>0.24</td>
<td>0.17</td>
<td>89.96</td>
<td>-</td>
<td>+</td>
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<tr>
<td><em>Planchonia careya</em> (F.Muell.) Kunth</td>
<td>0.03</td>
<td>0.21</td>
<td>0.20</td>
<td>91.03</td>
<td>-</td>
<td>+</td>
</tr>
<tr>
<td><em>Melaleuca viridiflora</em> MYRTACEAE</td>
<td>0.35</td>
<td>14.22</td>
<td>1.20</td>
<td>49.30</td>
<td>-</td>
<td>+</td>
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<tr>
<td><em>Barringtonia acutangula</em> LECYTHIDACEAE</td>
<td>0.20</td>
<td>5.56</td>
<td>0.65</td>
<td>68.57</td>
<td>+</td>
<td>+</td>
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<tr>
<td><em>Cathormion umbellatum</em> (Vahl.) Kosterm. MIMOSACEAE</td>
<td>0.19</td>
<td>4.51</td>
<td>0.50</td>
<td>84.22</td>
<td>+</td>
<td>+</td>
</tr>
<tr>
<td><em>Melaleuca cajuputi</em> Powell MYRTACEAE</td>
<td>0.09</td>
<td>0.99</td>
<td>0.27</td>
<td>87.65</td>
<td>-</td>
<td>+</td>
</tr>
<tr>
<td><em>Corymbia bella</em> MYRTACEAE</td>
<td>0.06</td>
<td>0.63</td>
<td>0.19</td>
<td>89.83</td>
<td>-</td>
<td>+</td>
</tr>
<tr>
<td><em>Acacia auriculiformis</em> MIMOSACEAE</td>
<td>0.06</td>
<td>0.54</td>
<td>0.26</td>
<td>91.70</td>
<td>-</td>
<td>-</td>
</tr>
</tbody>
</table>
Figure 2.4: NMDS ordination of all transects (N=144, stress = 24.5). Data points are displayed according to a-priori classification of transects. (a) Best-fit of environmental vectors and GAM fit of soil type. Four variables: stream order, the log distance of the riparian woodland zone (logWood), percentage of trees charred (Char) and percentage of quadrats burned (PetBurn), provide a Pearson’s correlation of $\rho = 0.418$. (b) Best-fit of biological abundance vectors. Six measures of cover: seedlings, saplings, forbs, vines, bamboo, pig damage (Pigs) and total basal area (BA) provide a Pearson’s correlation of $\rho = 0.308$. (c) Vector fit of eleven species that provide a Pearson’s correlation of $\rho = 0.900$. *Lophostemon lactifluus* and *Pandanus spiralis* do not follow the gradient indicated by their vectors but their greatest abundances are centered on the tip of their respective vectors, and generally decline away from that point.
Variation in riparian vegetation
Figure 2.5: NMDS ordination of riparian forest transects (N = 39, stress = 17.2). Basal area scores of stream and middle transects were combined within each site. Data points are displayed according to biogeographic region. (a) Best-fit of environmental vectors and GAM fit of soil type. One variable, stream order, provides a Pearson’s correlation of $\rho = 0.454$. (b) Best-fit of biological abundance vectors. Eight measures of cover: saplings, grass, shrubs, forbs, sedges, bamboo, bare soil and total basal area (BA) provide a Pearson’s correlation of $\rho = 0.447$. Grass and forbs have non-linear gradients. Grass cover increases toward the upper center and lower left of the ordination. Forb cover increases toward the upper right and lower left of the ordination. (c) Vector fit of eleven species that provide a Pearson’s correlation of $\rho = 0.750$. The gradient of *Melaleuca leucadendra* does not follow the gradient indicated by its vector, but is most abundant towards the center left of the ordination and declines away from that position.
Variation in riparian vegetation
Figure 2.6: NMDS ordination of woodland transects (N = 38, stress = 21.9). Data points are displayed according to biogeographic region. (a) Best-linear and GAM fit of environmental vectors and soil type. Six variables: stream order, width of the riparian forest zone (Rip), fire frequency (FF), percentage of quadrats burned (PetBurn), percentage of trees charred (Char) and slope within the riparian woodland zone (WoodSlope) provide a Pearson’s correlation of $\rho = 0.409$. The gradient of riparian width is non-linear, with higher values trending towards the upper right and lower center of the ordination. Char is also non-linear, and the GAM contour line representing 80% of trees charred is shown in place of a vector to avoid misinterpretation. (b) Best-fit biological abundance vectors. Five measures of cover: bare soil, bamboo, seedlings, forbs and total basal area (BA) provide a Pearson’s correlation of $\rho = 0.190$. The vector for bamboo is not drawn as it was present – and dominant – in only one site. (c) Vector fit of eleven species that provide a Pearson’s correlation of $\rho = 0.907$. The gradient of *Erythrophleum chlorostachys* is non-linear with the greatest abundance tending toward the center of the vector.
Variation in riparian vegetation
**Figure 2.7:** NMDS ordination of floodplain transects (N = 22, stress = 17.20). (a) Best-fit of environmental vectors. Points are classified by soil type. Five variables: stream order, width of the riparian forest zone (Rip), fire frequency (FF), percentage of quadrats burned (PctBurn) and distance to the opposite bank (OppBank) provide a Pearson’s correlation of $\rho = 0.718$. (b) Best-fit of biological abundance vectors. Three measures of cover: saplings, litter, and total basal area (BA) provide a Pearson’s correlation of $\rho = 0.477$. (c) Vector fit of eight species that provide a Pearson’s correlation of $\rho = 0.955$. Two species, *Barringtonia acutangula* and *Melaleuca cajuputi*, have non-linear gradients. The greatest abundance of *B. acutangula* centers around its vector. *M. cajuputi* is dominant in transects in the center of the ordination.
Variation in riparian vegetation

(b)

NMDS 2

-2 0 2 4 6 8

NMDS 1

NMDS 2

-2 0 2 4 6 8

NMDS 1

(c)

NMDS 2

-2 0 2 4 6 8

NMDS 1

Cathormion umbellatum

Melaleuca dealbata

Melaleuca cajuputi

Barringtonia acutangula

Melaleuca viridiflora

Asteromyrtus symphyocarpa

Vitex glabrata

Corymbia bella
**Fire**

Fire occurrence was strongly associated with woodland communities and within woodland communities formed an orthogonal axis to slope (Fig. 2.6a). Judging from the occurrence of burned quadrats and charred trees, fire occurrence is relatively rare within riparian forests, although the Landsat record indicates more frequent fires (Table 2.2). However, we suspect this result is an artifact of the spatial resolution of our fire frequency data (100 m) as this is larger than the width of most riparian forest zones. In within-community ordinations of woodlands and floodplains, both fire frequency and observed fire occurrence were closely correlated. The charred tree percentage in woodlands was generally very high (>60%) and was not correlated with other fire indicators. Rather it split bimodally into one group positively correlated with frequent fires and another negatively correlated (Fig. 2.6a). Among woodlands, communities in the Plateau were the least burned, possibly reflecting topographic protection. In woodland sites overall, *Eucalyptus patellaris*, *Corymbia bella* and *Terminalia platyphylla* were associated with high fire frequencies, while *Corymbia ptychocarpa* and *Lophostemon lactifluus* were associated with lower fire frequencies. Fire frequency was negatively correlated with basal area and seedling cover in woodlands (Fig. 2.6b).

**Recruitment**

In forest communities there were declines in seedling cover with distance from the active channel, but tree size class distribution tended to skew towards larger classes away from the channel (Fig. 2.8). Contrasting the recruitment pattern for riparian forests at the main water bearing channel edge and forest middle shows greater germination at the channel’s edge with decreased recruitment into larger size classes. This is likely the
Figure 2.8: Comparison of recruitment and stand structure between stream and middle transects within riparian forest. Mean ground cover of seedlings and saplings is presented along with the percentage of mature (>5 cm dbh) trees within each of five size classes (N [stream] = 1,856, N [middle] = 1,080). Difference between groups was tested using Student’s T-test (seedlings, p = 0.04; saplings, p = 0.16). The probability of similarity between size class distributions was tested and rejected using the Komolgorov-Smirnov test (p = 0.007).

result of decreased stability at the channel’s edge providing more open strata for germination and inhibiting growth into larger size classes (Busch and Smith 1995), but may also be the result of less frequent fires close to the active channel, as suggested by Table 2.2.

Stand structure in woodland communities, particularly the suppression of juvenile trees (Fig. 2.9), was skewed in a manner consistent with the recruitment bottlenecks typical of savanna ecosystems (Higgins et al. 2000, Sankaran et al. 2004). By contrast, floodplain communities showed both suppressed recruitment of seedlings and saplings
Figure 2.9: Comparison of recruitment and stand structure between vegetation communities. Mean ground cover of seedlings and saplings is presented along with the percentage of mature (>5 cm dbh) trees within each of five size classes (N [forest] = 3,016, N [floodplain] = 1,480, N [woodland] = 1,572). Analysis of variance (ANOVA) showed the difference between vegetation communities was significant for seedlings (p = 0.01) and highly significant for saplings (p < 0.001). The likelihood of similarity between pair-wise size class distributions was tested and rejected using the Komolgorov-Smirnov test (p < 0.001 for all comparisons).

Relative to forest sites and a skew towards smaller adult trees relative to woodland sites. This suggests that both the establishment and the persistence of woody vegetation is difficult on floodplains, a finding mirrored in Melaleuca forests more generally (Franklin et al. 2007).
Figure 2.10: Schematic representation of dominant vegetation and transverse (x-axis) and longitudinal (y-axis) position of all transects from NMDS ordination (Fig. 2.3). The dashed line encloses all NMDS points. Rainforest groups corresponds to Russell-Smith (1991).
DISCUSSION

The catchment of the South Alligator River encompasses a large variety of riparian vegetation communities that segregate by both longitudinal and transverse position (Fig. 2.10). The transverse variation between riparian communities of the South Alligator River is abrupt, with *Melaleuca*-dominated closed forest quickly giving way to seasonally inundated open woodland which extends to the limit of the annual flood zone. Floristic variation within closed forest occurs along a longitudinal gradient and transverse processes appear less important within the riparian forest itself. Woodland vegetation varies greatly and intermixes with floodplain and forest vegetation depending upon both the transverse profile as well as longitudinal variation in wet season stream flow, but generally follows a similar pattern to other hydrological gradients in northern Australia (Bowman and Dunlop 1986, Bowman and Minchin 1987, Bowman and McDonough 1991).

The sharp transverse boundary between closed forest and woodland likely forms due to interactions between declining water-table access from the active channel outward and the impact of fires carried from woodland towards the water channel. Despite some evidence for higher fire frequency closer to riparian forest margins (Table 2.2), we found no evidence of a “protective fringe” of fire tolerant vegetation as reported for riparian vegetation in tropical savanna streams in Africa (Melanson 1993, p. 48) and Central America (Kellman et al. 1998). Moreover, gallery forests in Central America comprise upland mesic forest species that are intolerant of fire (Kellman et al. 1998). By contrast, the dominant closed forest vegetation along the South Alligator River is distinctive, albeit with moist forest elements, and nearly all riparian vegetation readily resprouts in
response to fire and other disturbances (Table 2.3). Unfortunately, the spatial resolution of historical data is too low to determine the impact of longer term fire frequency on closed forest vegetation. Longitudinal variation in forest composition may be partially explained by variation in topographic protection from fire, and there is some circumstantial evidence that frequent fires are destructive to closed forest vegetation. For example, the rainforest tree *Xanthostemon eucalyptoides* was found nearly exclusively in fire protected ravines within Plateau sites, where riparian forests were also generally wider (Table 2.4). The woodland species *Terminalia platyphylla* was associated with forest vegetation in Gimbat Valley sites where woodland fire frequency was high. Overall, however, there is little evidence here to suggest that fires have a major effect on the structure or composition of closed forests.

The nearly universal dominance of *Melaleuca* along streams in this study contrasts a study by Douglas et al. (2003) of small intermittent streams along lower reaches of the South Alligator River. There streamside vegetation primarily comprised woodland species (e.g. *Erythrophleum chlorostachys*, *Corymbia polycarpa*, *Lophostemon lactifluus*, *Melaleuca nervosa*, *M. viridiflora*, and *Terminalia platyphylla*), and cover was lower than that reported for riparian forests here. Our study did not include smaller intermittent streams, and taken together both studies suggest that a minimum level of stream size, or perhaps more critically dry season water availability, is needed to form *Melaleuca* closed forest. Both *M. leucadendra* and *M. argentea* are relatively fire tolerant (Franklin et al. 2007) and may establish the closed canopy under which other, less fire tolerant vegetation can develop; both species are absent from the riparian vegetation described by Douglas et al. (2003). Once riparian forest establishes, higher
humidity and lower grass cover inhibit incursion of fires into the forest zone (Pettit and Naiman 2007a).

Our findings on basal area and recruitment in woodlands and floodplains are consistent with Douglas et al. (2003; Figs. 2.6a-b, 2.7a-b) where vegetation was strongly impacted by fire intensity. In that study, both the abundance and diversity of riparian vegetation declined with increasing fire intensity and the relative abundance of woodland to forest taxa (*sensu* this study) increased. However, in this study, while fire varied independently of other factors within woodlands, in floodplains it was negatively correlated with stream order, likely because of the increased persistence of water in the dry season on high order floodplain sites. Whether decreased fire, increased water retention, or both impacted on floodplain stand structure and basal area remains unclear.

Like other high rainfall areas of both temperate and tropical zones (Salo et al. 1986, Hupp and Osterkamp 1996), the period of wet season inundation is a key determinant of vegetation composition along the South Alligator River. This is particularly the case for woodland vegetation where longitudinal changes in species composition occur as stream order interacts with channel profile to favor more flood tolerant species in lower reaches. However, because of a prolonged dry season where rainfall is essentially nil, the South Alligator River system also shares many of the characteristics of arid and semi-arid systems elsewhere (Hughes 1988, Oba 1990, Bendix 1994a, Busch 1995, Hupp and Osterkamp 1996, Van Coller et al. 1997). In particular, transverse variation in dry season access to ground or surface water and exposure to fire are strongly correlated with vegetation patterns. Indeed, the results presented here and elsewhere (Douglas et al. 2003, Lamontagne et al. 2005) indicate that dry season access
Variation in riparian vegetation

To water is the most important factor in the establishment of closed forest vegetation. The boundary between closed forest and adjacent woodland vegetation is likely determined by dry season limitations to groundwater access and regular fires that limit recruitment of forest species within the woodland zone.

There is increasing pressure from the more densely populated and drier southern regions of Australia to develop the water resources of the north. The belief of abundant water in the tropics is belied by the reality of a system that oscillates between mesic and arid conditions annually. It is likely that the structure and function of riparian communities, particularly riparian forests, will be critically impacted if the extraction of ground or surface water leads to a reduction in dry season water availability.
Savanna responses to a rapid increase and decline of feral buffalo in the Alligator Rivers Region

Aaron M. Petty, Patricia A. Werner, Caroline E. R. Lehmann, Jan E. Riley, Daniel S. Banfai, and Louis P. Elliott


ABSTRACT

Savannas are the major biome of tropical regions, spanning 30% of the Earth’s land surface. Tree-grass ratios of savannas are inherently unstable and can be shifted easily by changes in fire, grazing, or climate. We synthesize the history and ecological impacts of the rapid expansion and eradication of an exotic large herbivore, the Asian water buffalo, on the mesic savannas of Kakadu National Park (KNP), a World Heritage Park located within the Alligator Rivers Region (ARR) of monsoonal north Australia. The study inverts the experience of the Serengeti savannas where grazing herds rapidly declined due to a rinderpest epidemic and then recovered upon disease control.
Buffalo entered the ARR by the 1880’s, but densities were low until the late 1950s when populations rapidly grew to carrying capacity within a decade. In the 1980s, numbers declined precipitously due to an eradication program. We show evidence that the rapid population expansion and sudden removal of this exotic herbivore created two ecological cascades by altering ground cover abundance and composition which in turn affected competitive regimes and fuel loads, with possible further, long-term effects due to changes in fire regimes. Overall, ecological impacts varied across a north-south gradient in KNP that corresponded to the interacting factors of precipitation, landform, and vegetation type but was also contingent upon the history of buffalo harvest. Floodplains showed the greatest degree of impact during the period of rapid buffalo expansion, but after buffalo removal, largely reverted to their prior state. Conversely, the woodlands experienced less visible impact during the first cascade. However, in areas of low buffalo harvest and severe impact there was little recruitment of juvenile trees into the canopy due to the indirect effects of grazing and high frequency of prescribed fires once buffalo were removed. Rainforests were clearly heavily impacted during the first cascade, but the long term consequences of buffalo increase and removal remain unclear. Due to hysteresis effects, the simple removal of an exotic herbivore was not sufficient to return savanna systems to their previous state.

INTRODUCTION

Savannas cover almost a third of the world’s land surface: over 50% the area of Africa, 50% of tropical Australia, 45% of South America, and about 10% of India and South-East Asia (Cole 1986, Werner et al. 1991). They are the major biome of tropical
and subtropical regions, occurring wherever rainfall is highly seasonal. Vegetation is sparsely wooded with a continuous C4 grassy understory (Huntley and Walker 1982, Tothill and Mott 1985a, Sarmiento 1984, Frost et al. 1986, Walker 1987, Solbrig et al. 1996), although they also incorporate other elements such as scattered heavily forested patches, riverine communities, and floodplains (Scholes and Archer 1997). The differences among savannas are often described in terms of tree : grass ratios. Grassy savannas predominate in east Africa (e.g. the Serengeti) and central and northern South America (e.g. the Gran Chaco and Venezuelan llanos), while in south eastern Africa (miombo), southeastern Brazil (cerrados), the coastal plains of southeastern USA and Central America (pine savannas), and northern Australia (eucalypt savanna) wooded savannas predominate.

The determinants of savanna tree : grass ratios have been much studied (Belsky 1990), most recently by Scholes and Archer (1997), Higgins et al. (2000), Jeltsch et al. (2000), House et al. (2003), van Langevelde et al. (2003), Sankaran et al. (2004, 2005), and Gardner (2006). These studies are consistent in demonstrating that what seems like a node of stability in time is mainly a product of disturbance (e.g. fire, grazing, drought). It is the disturbance-dependent nature of savannas that makes them particularly susceptible to anthropogenic change, whether from direct management or the introduction of novel species. Indeed, savannas on every continent have experienced some degree of change in tree : grass ratios over recent decades or centuries, due especially to changes in management affecting grazing and/or fire (Sinclair and Norton-Griffiths 1979, San José and Fariñas 1983, 1991, McNaughton 1985, 1993, Menaut et al. 1985, 1990, Archer 1990, Stott 1990, Werner 1991, Scholes and Walker 1993, Glitzenstein et al. 1995,
introduced buffalo 53


The majority of these studies on savanna processes have focused on the role of grazing or fire in shaping savanna structure, but only a few have considered both simultaneously (e.g., Norton-Griffiths 1979, San José and Ferinas 1983, 1991, McNaughton 1993). The most applicable contrast to our study is that of Norton-Griffiths (1979) who richly detailed the cascading effects of an 1890-1895 rinderpest epidemic that eliminated 95% of the ruminant grazers of the Serengeti in East Africa. The number of ruminants remained well below historical levels for some fifty years until disease control measures gradually restored the population. The sudden removal of grazers from the savanna had a profound impact on plant, animal and human populations as well as fire regimes, and the effects were still evident some four decades after rinderpest was brought under control.

We report here on circumstances that invert the Serengeti experience. In the mid 1800s English settlers introduced Asian water buffalo (*Bubalus bubalis* Lydekker), a novel large herbivore, into the tropical savannas of central far north Australia, a continent whose biota have evolved, at least since the Pleistocene, in the absence of large grazers (Mulvaney and Kamminga 1999). As early as 1885, Captain L. Carrington warned that, left unchecked, buffalo would become a “serious evil” (Carrington 1885). Indeed, buffalo expanded rapidly across several major river systems of the north within a century. Numbers grew rapidly to carrying capacity from 1960 to 1980, and their ecological impact was recorded (Stocker 1972, Letts et al. 1979, Braithwaite et al. 1984, Taylor and

Only now has there been enough research at sufficient spatial and temporal scales for us to appreciate the extent of the cascading effects and long-term nature of the ecological changes caused by the rapid increase and subsequent decrease of buffalo. In this paper, we aim to (1) establish the historical pattern of buffalo abundance in a region of significant biological and cultural importance, using published, archival and oral sources and (2) investigate the secondary impacts of buffalo on structure and dynamics of three major vegetation types in the savannas of the region: floodplains, monsoon forests, and woodlands.

Like the rinderpest epidemic of the Serengeti, this situation provides “a rare opportunity to understand an ecosystem” (Sinclair and Norton-Griffiths 1979). The savannas of north central Australia are broadly similar in climate and structure to the mesic savannas of Africa and South America, with over 1000mm of rainfall and an extended dry season (Sarmiento 1984, Menaut et al. 1985) but are generally oligotrophic and support a relatively low biomass of vertebrates (Mott et al. 1985, Braithwaite 1990). Unlike the savannas of Africa, but like the savannas of South America, Australia does not have native flora adapted to large ungulates. This had profound consequences for perennial and annual grass responses to grazing pressure in Australia.

This work also yields insights into the nature of niche competition and demographic processes in shaping savanna ecosystems in general, and informs current debate regarding the interaction between fire and grazing on shaping Australian savanna
Introduced buffalo

Plate 3.1: A buffalo cow and calf in the buffalo farm (central sector, Kakadu National Park). The exotic weed *Hyptis suaveolens* forms a clump to the right of the buffalo and also infests the woodland edge in the background. Photo: A. Petty.

composition and structure (Fensham and Holman 1999, Sharp and Whittaker 2003, Archibald et al. 2005, Mills and Fey 2005). It also provides an example of the cascades of effects that can flow through ecosystems when a major restructuring agent (large grazer) is added or subtracted from the system. Finally, it is a case study in the application of historical ecology to contemporary management questions, demonstrating that both environmental and historical factors must be considered when attempting to make sense of observed landscape scale changes.
Methods and Frameworks

Location and Biome

Buffalo ranged across most of north-central Australia – colloquially referred to as the “Top End” (Fig. 3.1). The densest populations of buffalo and the longest management history were in the Alligator Rivers Region (ARR) (Finlayson et al. 1997), a region of approximately 28,000 km² that encompasses three large tidal river systems: the East Alligator, South Alligator, and West Alligator Rivers. Climate, hydrology, geology, and biotic elements of the ARR are described in detail in Finlayson and von Oertzen (1996). The present Kakadu National Park (KNP), a World Heritage area, is bounded by the ARR. Because KNP was established over three stages, from 1974 to 1987 (Press et al. 1995) we often refer to the ARR in historical terms as required, but with the understanding that most of this same area ultimately became KNP.

The ARR was the location of both the earliest buffalo hunting leases and the most thorough attempts at population control and eradication. The park is considered to be particularly pristine ecologically, with no known extirpations and relatively few exotic fauna (Braithwaite and Werner 1987, Press et al. 1995, Woinarski et al. 2001), although there are six non-native mammal species (Ridpath 1991). Exotic plants are mainly confined to disturbed ground and riparian communities (Cowie and Werner 1993). The ecological focus of this chapter is KNP, with references to the larger region as required by historical narrative.

In this chapter, we use the term “savanna” in the sense of Scholes and Archer’s (1997) term “savanna landscape,” a biome with many elements including open, sparsely-wooded vegetation, closed-canopy monsoon forest, and open, herbaceous floodplains.
Figure 3.1 The top map shows northern Australia and the maximal extent of the buffalo invasion in Australia (adapted from Letts et al. 1979). The bottom map shows the core buffalo region, including the Alligator Rivers Region and Kakadu National Park, the historical locations of abattoirs, and the location of the earliest settlements in the Top End, Port Essington and Palmerston (present day Darwin).

To avoid confusion we use the simple word “woodland” for the dominant sparsely wooded vegetation community commonly called “savanna” or “savanna woodland.”

**Sectors of Kakadu National Park**

In our work, we discovered that reconciling historical records and ecological studies of impacts made sense only when we divided KNP into three sectors: northern,
central, and southern. These roughly correspond to Stage 2 of KNP (declared a national park in 1984), Stage 1 (1979), and Stage 3 (1987), respectively (Levitus 1995; Fig. 3.2a). These sectors reflect the different social histories of KNP as well as the different dates of buffalo encroachment as they moved southward. They differ in relative cover of the main vegetation types (more floodplains in the north and woodland in the south; Fig. 3.2b), and soils (low lying, poorly drained, clay soils are common in the northern sector; uplifted Proterozoic sandstone with shallow, sandy soils dominate the southern sector [East 1996]). A rainfall gradient spans the sectors, decreasing from an average of 1380 mm/yr in the north to 1150 mm/yr in the south (Fig. 3.2c).

The northern sector is dominated by the floodplains and tidal channels of all three of the Alligator Rivers. By far the largest in extent and volume is the South Alligator River. In the wet season the South Alligator floodplains span some 5-10 km on either side of the central channel and in the dry season, a twice-daily tidal bore advances some 90 km inland, into the central sector. The major floodplains in the northern sector are interspersed with lateritic upland ridges of savanna woodland dominated by *Eucalyptus miniata* A. Cunn. ex Schauer and *Eucalyptus tetrodonta* F. Muell.

The central sector extends across a region of freshwater floodplains, seasonally inundated savanna, and drier upland savanna lying 60-80 km inland from the coast. It includes a special wetland complex of some 200 km² where the South Alligator River discharges freshwater into a vast network of floodplains and perennial and seasonal billabongs that in turn eventually drain into a tidal channel. The central sector also includes Jim Jim Creek, a major tributary of the South Alligator River characterized by braided streams, perennial billabongs, and an admixture of paperbark (*Melaleuca* spp.)
Figure 3.2: Maps of Kakadu National Park: (a) details the biogeographic and historical sectors we describe in the text, as well as the location of abattoirs within the Alligator Rivers Region, and the location of the former Woolwonga Reserve (1936-1979) and Kapalga Research Station (1974-1996); (b) illustrates the major vegetation communities we describe in the text (unshaded areas are generally other wooded savanna); (c) is a map of rainfall isohyets, in mm/yr. Vegetation data are from Schodde et al. (1987) and Wilson et al. (1991). Rainfall data are derived from gridded monthly climate data at 0.25° of latitude-longitude from the National Climate Centre of the Bureau of Meteorology Research Centre, Melbourne, Australia.
swamps, seasonally inundated open woodland savanna, and upland *Eucalyptus miniata*/*E. tetrodonta* woodland savanna.

The southern sector incorporates a large area of upland plateau, part of the Arnhem Land Plateau, to the east and south, and large outwash plains and lowland savanna woodlands to the west. Both the plateau and the lowland savanna woodlands are interspersed with numerous perennial and seasonal creeks. The southern sector was formerly two large pastoral properties, and was the last sector to be incorporated into KNP in 1987 (Press et al. 1995), well after eradication efforts were underway in the northern and central sectors.

*Ecological Cascades*

The ecological impacts of buffalo differed across sectors depending on the local buffalo population, vegetation community and edaphic factors. Our understanding of these complex ecological processes and impacts is greatly enhanced by recognition of the ecological chain of impacts initiated by the rapid increase and decline of buffalo. We have constructed two ecological cascades that operated during two different times: (1) The period of increase in buffalo densities, from 1960 to 1980; and (2) the period of rapid decline in buffalo densities, 1980 - 1994.

During each of these cascades, the rapid increase or decline of buffalo densities directly altered ground cover abundance and composition. Buffalo often stripped ground vegetation bare as they approached carrying capacity in the first cascade. Conversely, ground vegetation, particularly annual grasses, were released by buffalo decline. In general, the direct effect of buffalo on ground level vegetation and soils indirectly altered competitive relationships among trees, grass and forbs, as well as produced large changes
in fuel loads and fire regimes which, in turn, further altered species composition and overall structure of the savanna. The second cascade, with the reversal in grazing pressure, also included significant hysteresis effects within the vegetation (Werner 2005, Werner et al. 2006) and released resources to other feral animals which then increased in numbers (Corbett 1995).

THE HISTORY OF BUFFALO IN NORTHERN AUSTRALIA

Introduction, Dispersal, and Increasing Population Density

Buffalo were first introduced to mainland Australia from 1825 to 1829 in several small shipments totaling no more than 100 individuals to Port Essington and Raffles Bay on the Cobourg Peninsula (McKnight 1976, Ridpath 1991). As the original settlements were abandoned, these buffalo were left behind and their descendants expanded across the peninsula. In 1844 the explorer Ludwig Leichhardt reported buffalo grazing on the Murganella Plains, immediately south of the Cobourg Peninsula and approximately 80 km from the Alligator Rivers Region (Leichhardt 1884). Once they passed through the hilly and relatively inhospitable neck of Cobourg Peninsula, buffalo rapidly expanded across the coastal riverine plains. In an 1862 expedition along the Mary River coastal plain John McDouall Stuart reported no buffalo sightings, but by 1885 they had expanded at least as far as the Adelaide River to the west and the Liverpool River to the east (Lindsay 1884 as reported in Letts et al. 1979, Ridpath 1991). In the Alligator Rivers Region, buffalo were abundant around the East Alligator River by 1885, and were present but scarce around the South Alligator River (Fig. 3.1). Paddy Cahill, a famous early settler in the ARR, hunted buffalo around Kapalga, near the mouth of the South Alligator
Introduced buffalo

River by 1899, and likely 1895. He indicated that buffalo were common, if not abundant, near the South Alligator at this time (Mulvaney 2004).

The expanding feral buffalo population became an economic resource for the Top End with the rise in demand for buffalo hides for use as belting in industrial machinery. Hide hunting began in the 1880s, but did not gain momentum until the turn of the century, both from increased demand overseas and, locally, the introduction of a more efficient means of horseback shooting (McKnight 1976). In 1900, 1,645 hides were exported from the Northern Territory (Anon 1901), and exports climbed rapidly thereafter. Hide exports for the Northern Territory generally remained above 5,000 per annum from the early 1900s and remained high for over half a century. So important were buffalo to the economy that in 1920 the Northern Territory’s acting administrator recommended protecting buffalo as they were “fast becoming extinct on the mainland” (Anon 1921). However buffalo hide exports continued to climb unabated after 1920 with no discernable impact on populations. McKnight (1976) suggested that, contrary to administrative concerns, market demand, and not population fluctuations, drove buffalo exports.

The hide industry collapsed in 1956 soon after synthetic materials replaced buffalo leather for industrial belting. In the space of a year hide exports in the Northern Territory dropped from 5,663 to 100 hides (McKnight 1976). After a brief hiatus, exports again increased, and harvests in the 1960s and 1970s well exceeded those of the hide era (Fig. 3.3). This was due both to local demand for buffalo for meat and pet food, and an initiative of the Northern Territory Agricultural and Animal Industry Branch to
redevelop buffalo as a viable economic resource for the Top End (Anon 1961). Reported buffalo take in the Northern Territory rose from under 300 in 1959 to 3,077 in 1962 to over 13,000 by 1967 (Anon 1959, 1962, 1967, Fig. 3.3) although these estimates are likely conservative, given the largely unregulated trade in pet meat in the 1960s (Ridpath 1991). In 1961 the Northern Territory Animal Industry Branch reported that “the value of production from buffaloes was greater than since 1955, and the buffaloes killed were more fully utilised than ever before” (Anon 1961). In 1972 buffalo removal for commercial purposes had peaked at approximately 30,000 head.
Despite increasing rates of buffalo harvest, the aerial photographic record, anecdotal accounts, and government reports indicate that buffalo densities were increasing rapidly from 1960 to 1980 (Fig. 3.3), with peak densities reached throughout the region. Under ideal conditions of high dry season rainfall, abundant surface water, and no hunting pressure, buffalo populations have the capacity to increase exponentially to the carrying capacity of the environment within approximately two decades (Freeland and Boulton 1990, Skeat 1990, Skeat et al. 1996). In good rainfall years, harvest rates must approach 10% to have a chance of reducing buffalo populations within a reasonable time (Freeland and Boulton 1990, Boulton and Freeland 1991). Although no accurate regional counts of buffalo were made until 1981 (Ridpath 1991), several lines of evidence indicate that buffalo had reached carrying capacity before then. In general, population numbers increased, with some declines principally due to natural mortality from reduced landscape carrying capacity during years of low or late wet season rainfall. Tulloch (1968) estimated that over 20,000 buffalo died of starvation in ‘bog holes’ due to late rains in 1965. D. Lindner (personal communication) recalled a similar event killing large numbers of buffalo in 1972. Nevertheless, by 1981 the Northern Territory buffalo population had clearly rebounded, and an aerial survey by Graham et al. (1982) estimated the Northern Territory buffalo population at 280,000, slightly higher than Letts’ (1964) estimate of 150-200,000 buffalo within the lowland sub-coastal plains immediately after the collapse of the hide industry.

Despite their exploitation for economic gain, buffalo had expanded in less than 150 years from an isolated population of fewer than 100 individuals to become the most abundant large grazing and browsing mammal across the coastal plains of the Top End
Introduced buffalo (Bayliss and Yeomans 1989), and a significant environmental concern (Letts et al. 1979). It took the concerted effort of a large scale government campaign to reduce buffalo numbers to the point that they were no longer a visible environmental force in the Alligator Rivers Region.

Eradication Campaign and Rapid Decline

Buffalo eradication began as a means of enabling the certification of Top End beef for export by controlling tuberculosis and brucellosis infection rates among domestic cattle (Letts et al. 1979). A Board of Inquiry was established in 1970 and determined that feral animals, particularly buffalo, were a major source of concern for the cross infection of domestic bovine herds. The Brucellosis and Tuberculosis Eradication Campaign (BTEC) began with the mustering of live animals and then ground shooting of the densest feral buffalo herds on the northernmost floodplains in 1980. By 1987 buffalo density in KNP had been reduced to fewer than 1 buffalo/km$^2$ (Skeat et al. 1996), and eradication efforts turned towards the last remaining populations of buffalo, located in the isolated rainforest pockets and ravines of the rugged Arnhem Land plateau. In 1989, KNP turned to helicopter eradication to remove any remnant populations. By the completion of the campaign in 1992 the buffalo population had been reduced to less than 0.01 buffalo/km$^2$ (Robinson and Whitehead 2003), although parts of neighboring Arnhem Land retained substantial populations of buffalo (Bayliss, unpublished data).

BTEC was a remarkable success in eradicating feral buffalo from KNP and the Top End at large, but remains highly controversial to this day, particularly among both tour operators and Aboriginal communities (Lucas and Russell-Smith 1993, Robinson and Whitehead 2003, Robinson 2005), who have seen the campaign as a threat to their
livelihoods. With the exception of a few domestic herds, most remnant buffalo populations have been left largely unchecked since BTEC, and they are again expanding in KNP, particularly within the southern sector as buffalo immigrate from Arnhem Land.

THE HISTORICAL ECOLOGY OF BUFFALO IN KAKADU NATIONAL PARK

Introduction and Dispersal within KNP

As elsewhere in the Top End, the closest and best habitats for buffalo in the Alligator Rivers Region were along the northern coastal plains, and this was also the first point of contact between hide hunters and buffalo. Explorers’ records indicate buffalo herds were in the northern sector of KNP by the late nineteenth century. By the early twentieth century, this sector supported a thriving buffalo hide industry. Traces of buffalo tracks and wallows are abundant in 1950 aerial photos of northern reaches of the Mary River, a river system comparable to the South Alligator River, but some 100 km to the west (Petty et al. 2005, Stocker 1972). Movement of breeding herds southward probably did not occur until buffalo had expanded to relatively high population densities in the north. Not only were southern areas less suitable, but individual buffaloes, particularly cows and calves, typically have restricted ranges and outward dispersal of breeding herds can be relatively slow (Tulloch 1978). Indeed, our visual inspection of the aerial photographs of the floodplains of the central sector of KNP shows little evidence of buffalo tracks and wallows in 1950 although they are abundant there, and in the southern sector, by 1964.

The anomalies in population dynamics of buffalo herds across the three sectors are correlated to both habitat quality and human activity. Although the area of habitat most suited to buffalo is most extensive in the north and decreasing southerly through
Introduced buffalo

KNP, a history of abattoirs in the northern sector and a conservation reserve in the central sector acted to reduce local population numbers. By contrast, dispersal into the southern sector occurred quite late, but buffalo numbers were left relatively unchecked.

Northern Sector

The vegetation and topography of the north coastal sector are ideal for buffalo as the woodlands on the lateritic ridges of floodplain margins provide camping and resting habitat for groups of cows and juveniles, as well as access to fodder during the end of the dry season and during peak floods, when food resources are scarce on the floodplains (Tulloch 1969, 1970, 1978).

Before 1960, buffalo hides were intensively hunted on both sides of the South Alligator River, although the western side was arguably better known as the site of Kapalga, the hunting lease of the famed buffalo hunter Tom Cole (Cole 1988). After the collapse of the hide industry, the western sector saw little buffalo management apart from informal hunting for pet meat and domestic consumption. By contrast, on the eastern section, three abattoirs were developed in the 1960s, at Mudginberri, Cannon Hill and Munmarlary (Fig. 3.2a). At their peak, these abattoirs processed over 50% of the Northern Territory’s buffalo meat (Anon 1967). The last of the abattoirs, at Mudginberri, was closed in 1987.

By the early 1980s the north-west had the highest population densities of buffalo in the whole Alligator Rivers region – as high as 34 buffalo/km² on the South Alligator River floodplains at Kapalga (Ridpath et al. 1983, Ridpath 1991) (Fig. 3.4). In aerial counts taken in 1985, Bayliss and Yeomans (1989) reported population densities that
were 5-10 buffalo/km$^2$ for the west, and 0.1-5 buffalo/km$^2$ in the east, although both sides had localized regions with much higher population densities.

Within the Alligator Rivers region, the tuberculosis infection rate among buffalo was highest in the northern sector, and this sector received early and sustained attention from the BTEC campaign (Robinson and Whitehead 2003). This caused a precipitous drop in buffalo numbers (Fig. 3.4). In 1983, the population averaged 4.3 buffalo/km$^2$ across both east and west portions of the northern sector, but by 1988 the population density had dropped to fewer than 1 buffalo/km$^2$ (Skeat 1990, Robinson and Whitehead 2003).

**Central Sector**

Overall, buffalo never attained the levels seen in the northern sector (Fig. 3.4). However in the freshwater floodplains of the central sector buffalo were as numerous, and their impact as extensive, as in the larger northern sector.

The growing impact of buffalo on the floodplains of the central sector was the subject of increasing concern for the land managers and Aboriginal residents of the Woolwonga Aboriginal Reserve (Fig. 3.2a), established in 1936 as the first protected area in the ARR and in 1964 designated a conservation reserve (Lucas and Russell-Smith 1993). Yorky Bill Alderson, a famed buffalo hunter who was raised in the ARR, also attested to the impact of buffalo in the Jim Jim Creek area:

The Wet is all right for them [buffalo], but in the Long Dry every year, from May to October and sometimes November, they must reckon they’re in the wrong country, too. They’re a damn pest, a menace. When I was a boy the Jim Jim was much better country than it is now; cattle won’t graze with buffs, and they’ve destroyed hundreds of miles of it (from Lockwood 1979, pp. 49-50).
In 1974, reserve managers instituted a program of buffalo removal both in the reserve and in neighboring areas (Letts et al. 1979). Dave Lindner, who was employed on the reserve at the time of the program, estimated that 90% of the buffalo were eradicated from floodplains, and that this action created a marked return of grasses that had previously been grazed out from these areas (D. Lindner pers. comm., Petty et al. 2005). The decline in numbers of buffalo during this culling action is reflected in the aerial photographic record of the area. By 1984, tidal channels damaged by buffalo showed signs of self-repairing, a date too early to be a result of the later BTEC reductions (Fig. 3.5a). In contrast, on the western side of the South Alligator River, vegetation seemed to have suffered heavier damage from buffalo than the east side (Fig. 3.5b). As in the northern sector, the differences in environmental impact on the two sides of the South Alligator River are consistent with the history of the sector, as there were no extensive buffalo hunting activities, either for culling or trade, on the western side.

Also like the northern sector, virtually all wild stocks of buffalo were eradicated by BTEC. However, in 1993 a local Aboriginal corporation set up a domesticated herd of some 350 buffalo as a source of buffalo meat for local Aboriginal communities. Today, this buffalo farm manages a herd that varies between 600 and 1000 head within fenced 12,000 hectares. It provides valuable information on the impact of buffalo on vegetation as well as best management practices in controlled herds in these types of landscapes (Riley 2005).
**Figure 3.4:** Estimated buffalo population numbers in the ARR. Data points marked with a double asterisk (**) are population estimates obtained directly from published sources. A single asterisk (*) indicates data inferred from population estimates of the ARR in published sources or from oral history. Unmarked data are inferred estimates from remarks and reports on buffalo in historical written records and oral history. Data are derived from Carrington 1885, Letts 1964, Stocker 1972, Christian and Aldrick 1977, Letts et al. 1979, Cole 1988, Skeat 1990, Ridpath 1991, Robinson and Whitehead 2003, Mulvaney 2004, T. Hill (*personal communication*), and A. Fisher (*personal communication*).

**Southern Sector**

The numerous perennial and seasonal creeks of both the plateau and lowlands of this sector provide suitable, if not ideal, habitat for buffalo. Data on the sector are sparse, but anecdotal accounts indicate that buffalo did attain high population densities locally, particularly in the lowland woodland savannas to the west of the escarpment (T. Hill, A. Fisher, *personal communication*). The ruggedness and isolation of the plateau provided buffalo a refuge from hunters and they likely achieved carrying capacity rather quickly once they had dispersed into the area (Fig. 3.4).

Historically, cattle were the focus of economic activity in the sector, and the closest abattoir, Mary River, located just outside the present day boundary of KNP, ran a small operation that never approached the processing rates achieved at the abattoirs to the
north (Anon 1967). Alan Fisher worked as a musterer for both pastoral properties prior to their incorporation into KNP, and was retained to de-stock the properties when they were acquired by the Park. He witnessed buffalo becoming increasingly common throughout the 1980s: “Back in the old days [late 1970], we’d go out in the dry season to bring in over three-year bulls for Mary River. It took a while for us to find some. A few years later it wasn’t uncommon to find a few small groups. Then, later, we’d see hundreds of them” (A. Fisher, personal communication).

When the BTEC campaign turned its attention to the southern end of the Park in 1987, unexpectedly high numbers of buffalo were found. Their numbers made buffalo too valuable of an economic resource and the strategy turned from helicopter shooting on site to helicopter mustering for shipping to an abattoir (T. Hill personal communication). In the space of three years, Alan Fisher (personal communication) estimates that over 45,000 buffalo were removed from this sector. Other reports indicate the population density dropped from over 10 buffalo/km$^2$ to less than 1 buffalo/km$^2$ from 1986 to 1991 (Skeat 1990, Robinson and Whitehead 2003).

The plateau region of western Arnhem Land, which abuts the eastern side of the southern sector, never received much attention during the BTEC campaign as brucellosis and tuberculosis infection rates were very low, and there was little motivation to shoot buffalo on Aboriginal freehold land. A. Fisher (personal communication), found it difficult to meet the BTEC destocking quota because of continual migration of buffalo into KNP from Arnhem Land. Today this region remains a major source of buffalo migration into KNP, particularly in the southeastern corner, which has the highest densities of buffalo in KNP today (R. Muller, P. Bayliss personal communication).
ECOLOGICAL IMPACTS OF BUFFALO

Most of the research on the ecological impact of buffalo has focused on wetland environments, with little attention to the impact on habitats such as woodland savanna and rainforests. Further, the substantial anecdotal and photographic evidence of the dramatic effects of buffalo on the landscape were almost entirely confined to the floodplains, seasonally inundated fringing lands, and other riparian areas where buffalo impact was most visible and human-buffalo coexistence was closest. Here, we consider the impact of buffalo expansion and decline in three major habitat types – floodplains, monsoon rainforests, and savanna woodlands.

Given that the three sectors of KNP are made up of different relative areas of these three major habitat types, one would expect that the overall impact per sector would be quite different. Further, given the general north to south gradient in buffalo numbers (Fig. 3.4), one might assume a greater ecological impact in the northern sectors of KNP. However, the differences in both the resilience of the various habitats and the carrying capacity across the north-south gradient must be taken into consideration, with any impact of numbers scaled relative to the capacity of the land to produce green biomass.

We posit that lower rainfall and lower plant productivity in the southern sector of the Park, as well as long-sustained buffalo populations at or near carrying capacity, historically produced as great an ecological impact in the southern sector as in the northern sector. The evidence of impact in the north is well recorded, whereas the record of environmental impacts in the south is limited to aerial photography, which shows no change in woody cover. However, aerial photography says nothing of the condition of understory vegetation, which surely changed with buffalo grazing. The situation is also
Figure 3.5: Buffalo impact on floodplain vegetation and channels. All data were derived from 1950, 1964, 1975 and 1983/1984 orthorectified aerial photomosaics of the South Alligator River floodplains in the central sector of KNP. (a) Linear channel change rate was calculated as the difference in total channel length between time periods for all channels which were absent in the earlier photo (formed) and for all channels which were absent in the more recent photo (disappeared). Data are from Petty et al. (2005). (b) Woody vegetation cover was calculated from two different studies of geostationary points spaced across the South Alligator Floodplain in the Central Sector. The arrows illustrate the initiation points of the two ecological cascades. Data are from Riley (2005) and J. Freeman (unpublished data). (c) Rate of channel formation compared with percent of geostationary points where aerial photos showed visual signs of buffalo impact (tracks and wallows). Data are from Petty et al. (2005) and J. Freeman (unpublished data).
Introduced buffalo

confounded by the impact of other feral animals, including cattle and horses, which, in the southern sector, reached numbers as high as buffalo in some places (A. Fisher personal communication). Nevertheless, A. Fisher (personal communication) recalls that most visible environmental damage on the plateau was restricted to riparian corridors and other areas of dense canopy cover that were the preferred habitat of buffalo. Skeat et al. (1996) and East (1990) reported extremely high sediment loads in the South Alligator catchment along the Arnhem Land plateau which they attributed to buffalo activity. Similarly, buffalo were implicated in increased fire damage within the monsoon rainforests that line the gullies of the plateau of the southern sector (Russell-Smith 1984).

Floodplains

Several broad surveys of floodplains were produced during the period of peak buffalo impact, especially concentrating on lowland seasonally flooded black clay plains, and perennial billabongs, where buffalo numbers were generally very high and soils typically highly erodible (Fogarty 1982, Graetz 1989, East 1990). The South Alligator River floodplains and the Cooinda/Jim Jim flood basin, formed over silty and sandy alluvium, are particularly susceptible to erosion and degradation, whereas the East Alligator and West Alligator River floodplains, located over sand and clay-sand soils, were much less severely degraded (Graetz 1989, East 1990). It is from heavily-grazed wetlands such as these that we have the most information about the ecological impacts of buffalo (Stocker 1972, Letts et al. 1979, Fogarty 1982, O'Neill and Matthews 1983, Friend and Taylor 1984, Taylor and Friend 1984, Graetz 1989, Corbett and Hertog 1996), including more recent reviews of broad scale buffalo impact on floodplains produced after the BTEC era (Skeat et al. 1996, Finlayson et al. 1997). Lucas and Russell-Smith

A 1919 description of the floodplains at Cannon Hill near the East Alligator River describes “wonderful grassy plains” covered with “beautiful green, water couch grass” (Warburton 1944), whereas in the dry season of 1979, the same flood basin was black clay heavily pug marked from buffalo activities and almost completely bare of vegetation (Skeat et al. 1996). Paddy Cahill describes crossing the tidal head of the South Alligator River on horseback in 1901 at a location that is today scoured by a deep tidal channel (Mulvaney, 2004). Grasses such as *Hymenachne acutigluma* (Steud.) Gilliland and *Phragmites vallatoria* (Pluk ex L.) Veldkamp were apparently common across the area prior to the explosion in buffalo numbers between 1960 and 1980. Lewis (1922) describes stands of “reeds” (likely *Phragmites*) from which he constructed a raft while trying to cross the East Alligator in 1887. In 1946, Christian and Stewart (1953) described grasses as very abundant on the coastal plains, including *Hymenachne acutigluma* at Cannon Hill, but by 1972–1973, Story (1976) found that grasses were very scarce in the same estuarine alluvium, noting the great change over the previous 26 years. More recent recollections of Aboriginal residents in KNP recall widespread stands of *Phragmites* from which they made spears (O'Neill and Matthews 1983). By 1996, *Hymenachne* was completely absent from Cannon Hill, and *Phragmites* was virtually absent from both Cannon Hill and the South Alligator flood plain (Skeat et al. 1996).
First cascade: buffalo population eruption. – Most documented observations of the direct buffalo impact on the floodplain surface come from scientific reports during the 1960s through 1980s, and are well summarized by Skeat et al. (1996). These impacts, including wallowing, track and pad formation, and compression of soils, resulted in the siltation of streams, increased turbidity and eutrophication of water bodies, reduced vegetation and pedestal, rill and gully erosion (Skeat 1990). Buffalo also impacted hydrology directly by creating small channels and depressions which were further deepened by wet season runoff, and increasing sheet erosion from decreased water-retention capacity in compacted soils (Noble and Tongway 1986, Skeat et al. 1996).

However, the largest landscape-scale buffalo impact in the Alligator Rivers Region was mediated through grazing and its impact on standing plant biomass, both green and litter. At the high population densities seen in the South Alligator River floodplains, buffalo removed most of the surface vegetation, starting with freshwater perennial grasses, including Hymenachne and Phragmites, then turning to the less palatable but more abundant Paspalum distichum L., a salt-tolerant grass which stabilizes tidal levees and is key in preventing the formation of saltwater bearing channels (Petty et al. 2005).

Channel formation rates were correlated with signs of buffalo impact on the South Alligator floodplain (Fig. 3.5c). The denuded floodplain surface was scoured by freshwater draining off the floodplains at the end of the wet season. This created channels that connected freshwater floodplains directly to tide-bearing channels. These channels directly and substantially impacted floodplain ecology by permitting the influx of tidal water to regions of salt-intolerant vegetation, and permitting early drainage of
Figure 3.6: These photographs highlight differences in floodplain woody vegetation cover in buffalo impacted and non-impacted regions, separated by fence lines. Top: A 2004 aerial photograph with a highlighted fenceline diagonal across the image. The fence separates the buffalo farm in central KNP (right) from KNP proper, which at present has very low buffalo numbers. Bottom: A ground photograph taken at Kapalga in 1984 contrasting a buffalo excluded area (right) from an area with buffalo (Photo by D. Tulloch).

Introduced buffalo impounded freshwater, thus leading to early drying of floodplains in the dry season (Finlayson et al. 1997), and increased subsoil salinity (Stocker 1972). High sediment
Introduced buffalo loads carried by incoming tides also filled in permanent freshwater lakes as has occurred on at least two occasions on the South Alligator River floodplain (Petty et al 2005). The aerial photographic record of the floodplains of the East Alligator and South Alligator Rivers indicates severe and widespread vegetation loss, buffalo tracks, and a marked increase in channel formation after 1950 (Heerdegen and Hill 1999, Petty et al. 2005).

From 1950 to 1975 there was a general decline in woody vegetation on the central floodplains: a notable exception being the eastern side of the South Alligator River where there was more active buffalo management (Fig 3.5b). Data on the direct impact of buffalo on woody vegetation of the floodplains has not been reported, although there are numerous photographic records (e.g. Letts et al. 1979; Fig 3.6), as well as reports of buffalo eating the seedlings and saplings of the shrub Cathormion umbellatum (Vahl) Kosterm (Lucas and Russell-Smith 1993). Similarly, contrasting aerial photographs inside and outside of the current buffalo farm (central sector) show a marked decline in woody vegetation inside the boundary of the farm where buffalo graze (Riley 2005; Fig. 3.6a). In every case, photographic records and other studies show a negative relationship between woody vegetation and buffalo grazing.

The capacity of buffalo to browse and otherwise inhibit the growth of woody vegetation on floodplains contrasts with widely reported cattle mediated woody thickening, including recent findings in the Victoria River District (VRD), approximately 400 km southeast of the Alligator Rivers Region (Sharp and Whittaker 2003). On both the VRD and KNP, grazing significantly reduced fuel loads. On the VRD, however, woody cover in seasonally-inundated pastures increased with cattle grazing due primarily to the growth of Eucalyptus microtheca F. Muell. and Exoecaria parvifolia Muell. Arg.
On the South Alligator River, where those species are absent but *Melaleuca* spp., as well as several shrubs (*Cathormion umbellatum*, *Barringtonia acutangula* (L.) Gaertn., and *Strychnos lucida* R. Br.) are prominent, woody thickening did not occur. Instead, in the South Alligator floodplains all vegetation decreased with buffalo grazing.

As elsewhere in KNP, the ecological impact of buffalo interacted strongly with fire. Prior to European settlement, these floodplains were a rich resource for Aboriginal people who would have burnt the floodplains extensively (Jones 1980, Haynes 1985, Meehan 1991, Lucas and Russell-Smith 1993). The arrival of buffalo in the region coincided with European settlement and Aboriginal depopulation and hence a decrease in both human sources of ignition and fuel loads. During the peak buffalo period “burning in many floodplain situations was no longer practicable” (Lucas and Russell-Smith 1993). However, once buffalo were removed, traditional fire regimes were not resumed on the depopulated floodplain.

There is little from the literature to inform us on the impact of buffalo on native floodplain fauna. Reports of long term Aboriginal and European residents of the area indicate that Magpie Geese (*Anseranas semipalmata* Latham) numbers declined in the South Alligator floodplains during the buffalo era (Frith and Davies 1961, Tulloch and McKean 1983, Lucas and Russell-Smith 1993), due in part to widespread habitat destruction from buffalo grazing and wallowing (Corbett and Hertog 1996) and saltwater intrusion onto freshwater floodplains (Petty et al. 2005). The dusky rat (*Rattus colletti* Thomas) and other small vertebrates appear to be negatively impacted by buffalo, mainly through reduced vegetation cover at high buffalo densities (Friend et al. 1988).
Table 3.1: Summary of impacts of the two buffalo-mediated cascades on floodplain, rain forest and woodland vegetation types.

<table>
<thead>
<tr>
<th></th>
<th>Cascade 1: 1960 - 1985</th>
<th>Cascade 2: 1985 - present</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Floodplain</strong></td>
<td>Increased grazing pressure on grasses. Trampling and compaction of soils and physical damage to woody vegetation.</td>
<td>Increased fuel loads with release from grazing pressure, but decreased ignition due to human depopulation.</td>
</tr>
<tr>
<td>Driver</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Principal effects</td>
<td>Decreased woody and grass cover. Rapid increase in saltwater bearing channels. Loss of some freshwater vegetation communities, siltation of some ponds.</td>
<td>Increase in woody cover. Grass cover largely restored although some species (e.g. Phragmites karka) have not. Channels restabilized.</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><strong>Rainforest</strong></td>
<td>Trampling and compaction of soils. Grazing pressure on grasses at rainforest boundary.</td>
<td>Increased fuel loads are managed by protective burning at rainforest margins.</td>
</tr>
<tr>
<td>Driver</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Principal effects</td>
<td>Understory cover reduced in interior. Increased invasion by native and exotic herbs at rainforest boundary and interior. Reduction of grass fuels at forest boundary may have caused expansion of forest edge due to reduced fire intensity.</td>
<td>Apart from exotic weed legacy, buffalo removal is unlikely to have had a significant impact on continued rainforest expansion.</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><strong>Woodland</strong></td>
<td>Increased grazing pressure on grasses and, to a lesser degree, woody juveniles.</td>
<td>Release of ground vegetation, particularly annuals initially, with perennials coming later. High fire frequency promotes annual sorghum.</td>
</tr>
<tr>
<td>Driver</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Principal effects</td>
<td>Decreased total ground vegetation and a shift to annuals. Reduced intraspecific competition and lower fuel loads result in increased growth and lower mortality in woody species.</td>
<td>Increased competition with understory reduces growth and survival of woody plant. More frequent and intense fires lead to increased mortality in juvenile woody species.</td>
</tr>
</tbody>
</table>

*Note:* Cascade 1 corresponds to rapid buffalo population increase, while cascade 2 follows the elimination of buffalo through a deliberate eradication campaign.
Second cascade: buffalo eradication. – The hydrology of the floodplains appears to have largely recovered from the era of peak buffalo impact. Following buffalo removal in the region of the Woolwonga reserve in the 1970s, infilling by tidal sediment caused a marked contraction in channels within the South Alligator floodplain (Fig. 3.5a). Tidal sediments were stabilized by the re-establishment of freshwater and saltwater grasses across the floodplain, with the exception of Phragmites, which has yet to re-establish (Petty et al. 2005).

The total biomass of both herbaceous and woody vegetation on the floodplains increased with the removal of buffalo grazing (Petty et al. 2005; Fig. 3.5b). Relative abundances of herbaceous species also shifted very rapidly (Minchin and Dunlop 1989). For example, on the South Alligator floodplain in the northern sector, the grass *Hymenachne acutigluma* increased from 1984-1988 which in turn increasingly displaced the deep water spike-rush *Eleocharis dulcis* (Burm.f.) Trin. ex Hensch (Corbett and Hertog 1996). This result is comparable to the marked increase in total biomass and the relative increase in *Paspalum distichum* over grazing tolerant perennial herbs when Asian buffalo were removed from Keoladeo National Park in India (Vandervalk et al. 1993).

However, further changes in relative species abundances, especially among woody plants, were most likely confounded by the fire regime that followed buffalo removal (Table 3.1). Although fuel loads increased rapidly, historical fire regimes were not restored due to the fact that the original ignition source (independent bands of humans) was not replaced. Because of the greater availability of fuel, fires did become more extensive after buffalo were removed (Russell-Smith et al. 1997b). Gill et al. (2000) reported a gradual increase in area burnt on floodplains from 5 million ha in 1980
Introduced buffalo to approximately 9 million ha in 1992. Nevertheless, fires were not frequent enough to prevent re-establishment of floodplain shrubs (principally *Barringtonia acutangula*, *Cathormion umbellatum*, and *Strychnos lucida*). These shrubs increased sharply in aerial cover and now exceed that found before the buffalo population eruption (Fig. 3.5b). We postulate that the subsequent change in fire regime, from traditional aboriginal fires to park-management fires, has also played a role in any expansion of woody vegetation on the KNP floodplains once buffalo were removed.

Upon removal of buffalo, the nests of Magpie Geese shifted toward deep water areas at the edge of the floodplains (Corbett and Hertog 1996) and the numbers of the native dusky rat increased (Skeat et al. 1996). Feral pig populations increased at least three-fold across the region, due to release from competition with buffalo (Corbett 1995). In general, the removal of buffalo on the Kapalga floodplains initiated a cascade of events (Table 3.1) with detectible changes in vegetation within one year, in small herbivores within two years, and in small predators within three years (J. A. Taylor, G. Friend, L. Corbett personal communication).

**Monsoon rainforests**

Monsoon rainforests are evergreen, closed-canopy forests; non-Eucalypt, fire-sensitive species are the dominant woody species (Bowman 2000). In the Alligator Rivers Region they occur as an archipelago of hundreds of small fragmented patches and as such are particularly vulnerable to disturbance (Russell-Smith and Dunlop 1987, Bowman 2000). Buffalo frequently rest and camp in these densely wooded, closed-canopy areas, often as preferred habitat (Tulloch 1978, Taylor and Friend 1984), and so their density in monsoon rainforest can be much higher than that of the surrounding
Introduced buffalo landscape (Ridpath 1991). Ecological impacts ranged from severe to relatively undisturbed, varying from patch to patch depending upon the landscape setting, but always correlated to buffalo densities.

Although research on the mechanisms of buffalo impacts on monsoon rainforests is sparse, within the patches of monsoon rainforest themselves most of the impact appears to be due to the direct effects of buffalo activity: soil and root compaction, browsing, wallowing and physical destruction of vegetation (Braithwaite et al. 1984, Russell-Smith and Dunlop 1987, Russell-Smith and Bowman 1992; Table 3.1). Russell-Smith (1984) showed that in sandstone terrain, only those rainforest sites associated with springs or streamline habitats were seriously affected by buffalo, while in the lowlands, impact was widespread. The general pattern of impact in the sandstone springs was destruction of margin vegetation by wallowing, and erosion leading to tree death. In lowland habitat types Russell-Smith (1984) found that the most obvious impact was the elimination of the lower strata of woody plants as well as the death of canopy trees. This is consistent with the impacts recorded by Braithwaite et al. (1984), who sampled 30 monsoon rainforest patches in KNP and compared sites with varying intensity of use by buffalo. They concluded that buffalo had a fundamental impact on vegetation structure and composition, including reduced lower canopy cover, and postulated that soil compaction by buffalo caused the death of large trees by hindering groundwater recharge. The density of vegetation under 3 m height was negatively related to buffalo density in the dry season although weedy annual species, particularly Hyptis suaveolens (L.) Poit. and Cassia spp., increased with buffalo presence (Braithwaite et al. 1984, Cowie and Werner 1993).
Secondary effects of the type found in floodplains (above) or savanna woodlands (below) have not been examined other than those related to fire. Changes in fuel loads at the periphery of rainforests, due to buffalo grazing and wallowing, in turn affect fire regimes. Any changes in peripheral fire regime can affect the boundaries of the monsoon rainforest patches and/or penetrate into the interior of the forest; there are examples of both increases and decreases of patch sizes.

Although there has been some localized contraction due to buffalo impact and introduced weeds, overall there is clear evidence from aerial photography that most patches of monsoon rainforest have expanded in Kakadu National Park (Banfai and Bowman 2006). Reduced fire frequency surrounding monsoon rainforest patches has been implicated as a major cause of monsoon forest expansion elsewhere (Hopkins 1992, Swaine et al. 1992, King et al. 1997, Bowman et al. 2001b). Although buffalo may have played a role in this recent expansion, it is unlikely that they were the primary and/or only driver as there was little evidence of a change in the rate of expansion during the peak period of buffalo populations in the 1970s, or following the dramatic reduction in buffalo numbers in the 1980s (Banfai and Bowman 2006).

The apparent contradictory nature of buffalo impact on monsoon rainforests can be partially resolved by separating their impact on forest interiors (mediated by behaviors such as grazing and trampling) from forest boundaries (mediated by their indirect impact on fire regimes). Buffalo appear to contribute substantially to the structural decline of forest interiors through grazing, trampling, and acting as weed vectors. On forest boundaries their impact is more mixed. Buffalo promote the expansion of monsoon rainforest boundaries by reducing fuel loads outside of monsoon rainforests and hence the
severity of fires which might impact on rainforest boundaries. At the same time, buffalo also promote rainforest contraction by allowing grass establishment and weed invasion in gaps (as more canopy trees die with buffalo present) and creating buffalo thoroughfares through the vegetation, both of which permit greater fire penetration into the interior of rainforest patches. Such “fire tongues” along buffalo tracks have been observed at several sites in KNP (Russell-Smith 1984).

The long-term legacy of the buffalo-era seems to be increased weed populations along rainforest margins. Otherwise, there is no known evidence of a “second cascade” phenomenon within monsoon rainforests.

_Savanna Woodlands and Open Forest_

Eucalypt-dominated woodland and open forest cover over 60% of the Alligator Rivers Region (ARR) (Wilson et al. 1996, Fig. 3.2b). Unfortunately, little has been published on the impact of introduced buffalo on the drier upland woodlands, despite clear evidence that buffalo used these areas extensively and floodplains and woodlands are intricately tied with regard to buffalo life histories (Tulloch 1969, 1970, 1978, Tulloch and Cellier 1986, Werner 2005).

Buffalo are broad-spectrum grazers whose grazing patterns are strongly determined by grass phenology, not necessarily relative abundance or even nutrition (Tulloch and Cellier 1986, Tulloch 1970). During the wet season, when the flood plains are inundated, buffalo eat underwater vegetation in the shallower areas, retreating to the upland savanna woodlands for nurseries and overnight camps and grazing of newly germinated annual grasses in the woodlands. At the end of the wet season, as the floodplains dry out, the newly exposed sedges and grasses there become a major food
source for buffalo. In the latter part of the dry season when floodplains are no longer lush, buffalo again rely heavily on the savanna woodlands, although this time most of the annuals are senescent, and buffalo graze perennial grasses.

Given these daily and seasonal movements, population densities of buffalo in woodlands would have exceeded those on the floodplain at certain times of day and seasons. In the early 1980s, at peak population numbers, the mean density of buffalo in the woodlands of the northern sector was estimated at 7 animals/km$^2$ in Munmarlary (Bowman et al. 1988) and a substantial 15 animals/km$^2$ at the Kapalga Research Station (Ridpath et al. 1983, Ridpath 1991).

Most of what is known about the impact of buffalo on upland wooded savannas comes from two bodies of research, which we integrate in this section:


2. Broad field surveys and analysis of aerial photographs of sites covering all sectors of KNP (Lehmann 2007). The aerial photographic analysis covered the period from 1964 to 1984 (when buffalo numbers rapidly increased) and from 1984 to
2004 (when buffalo numbers rapidly decreased). Patterns of demography and recruitment were also investigated by ground surveys of vegetation on these same sites in 2003-2004 (Lehmann 2007).

First cascade: buffalo population eruption. – The only direct evidence of vegetation changes in woodlands during the period of rapid increase in buffalo densities is the photographic record, where Lehmann (2007) compared woody cover in 1964 and 1984. This shows a general trend of increased woody cover, with some differences among sectors of KNP (Fig. 3.7). Increases in tree cover can be the result of increased recruitment of new trees and/or the enlargement of established trees. Insights into the degree that each of these and other changes in the understory may be important, as well as the mechanisms of these changes, have been gleaned from experimental work in buffalo-absent and buffalo-present plots during the second cascade.

In 1982, the frequency size distributions of tree populations in Kapalga showed a bias toward middle and large trees: in six study plots spread over a topographic gradient, there were no trees < 5 cm diameter breast height (dbh) and fewer juvenile trees (trees < 1.4 m height) and trees < 20 cm dbh than expected in a stable population (Werner 1986). From size-specific growth rates, it was estimated that there had been almost no recruitment of juveniles into the sapling stage for approximately 20 years prior to 1982 – a time period that coincides with the buffalo eruption (Werner and Murphy 1987). Further, the results of a nine-year, longitudinal study of individually-marked trees showed that both the growth and survival of established trees were significantly enhanced in the presence of buffalo (but no fire), but that recruitment into the canopy was negligible (Werner 2005).
Figure 3.7 Trajectories of change in tree cover by sector. For comparison, the northern sector is divided into west of the South Alligator River (North West) and east of the South Alligator River (North East). Error bars are ±1 SE. The data are from Lehmann (2007).

Experimental studies have shown that the direct effect of buffalo on woodlands was to reduce herbaceous vegetation (mainly grasses) which in turn enhanced the growth and survival of established trees of all sizes (Table 3.1). Competition for water among grasses, juvenile plants and canopy trees is very strong in the top 50 cm of the soil during the early dry season (Chen et al. 2004). The dense herds of buffalo in woodlands at Kapalga removed an estimated 2-6 tonne/Ha of herbaceous biomass per year through grazing and trampling (Fig. 3.8). In a six-year experimental study (1983-1988) of marked juvenile (<1.4 m height) trees at Kapalga juveniles grew significantly faster and had significantly higher survival rates in the presence of buffalo (and no fire), despite the
Figure 3.8: Mean total ground-level biomass in paired plots of woodland, with buffalo present (solid bars) and buffalo removed (gray bars) on the ridge top, slope and flat, at the end of the growing season three years after buffalo were removed, and without any intervening fires. Error bars on means are 2 SE. The figure is from Werner (2005).

fact that 10–12% of them showed signs of mammalian herbivory, compared to only 0–1% where buffalo had been removed (Werner et al. 2006). Similarly, in a concurrent eight-year study (1982-1989) of 750 marked established trees on the same site, all trees had higher growth rates and lower mortality rates in the presence of buffalo (no fire), and only the late season fires affected them adversely (Werner 2005). Indeed, the growth and survival of established adult and juvenile trees have been enhanced by the removal of herbaceous vegetation by a variety of means, including buffalo grazing, experimental hand clipping, early dry season fires (Werner 2005, Werner et al. 2006), and even poor rainfall years (Prior et al. 2006).
Figure 3.9: Mortality of juvenile trees in savanna woodland in buffalo-present vs. buffalo-absent plots, averaged across all time intervals over the six years of the experiment, in plots that received no fire vs. plots that received high-intensity, late season fires. The figure is from Werner et al. (2006).

In spite of the advantages in growth and survival of established trees in the presence of buffalo, however, there was a recruitment bottleneck where juvenile eucalypts did not move into the sapling sizes (Werner et al. 2006). The most likely explanation is that overstory competition was sufficient to prevent juvenile Eucalypts from moving into sapling stages (Fensham and Bowman 1992). Juvenile Eucalypts are particularly susceptible to water stress (Prior et al. 1997). Such was not the case for species of mid-storey non-eucalypts, however, wherein populations doubled to
quadrupled from 1982 to 1990 where buffalo were present, but declined where buffalo had been removed (Werner 2005). Fire was also particularly damaging to juvenile trees where buffalo had been removed, due to increased fuel loads (Fig. 3.9).

Using aerial photographs of wooded savanna in the Kapalga section, in the western side of the northern sector, Lehmann (2007) found that total tree cover increased from 1964-1984, but then decreased thereafter (Fig. 3.7). This is consistent with the demographic shift where the lowered recruitment of juveniles to adults resulted in the long-term decrease in the number of adults in the overstory. By contrast, on the eastern side of the South Alligator River (edaphically identical to the western side, but with historically lower buffalo numbers), and in the central sector of KNP, tree cover increased over the entire period from 1964 to 2004 (Fig. 3.7).

In the southern sector, there has been no change in the average woody cover over the past 40 years (Fig. 3.7), although there has been dynamic change at local scales. This is true despite clearly having buffalo densities nearly as great as the central sector (Fig. 3.4) and evidence of buffalo impact on soil erosion and understory vegetation (East 1990, Skeat et al. 1996). Similarly, Sharp and Bowman (2004) found little change in upland savanna woodlands in cattle grazed areas of the Victoria River District (VRD). Both the VRD and the southern sector of KNP are xeric relative to the northern sector of KNP (Fig. 3.2c). Rainfall in the southern sector lies close to the annual water use requirements for Eucalyptus woodland (Hutley et al. 2000), which, given the high interannual variation in rainfall in the Top End, makes the southern KNP savanna woodlands particularly vulnerable to severe stress during dry years. Higgins et al. (2000) have suggested that more xeric systems have less capacity for large variance in the fire regime, and
Introduced buffalo correspondingly, for grazing impacts on fire regime. To the extent that Higgins et al.'s (2000) suggestion holds across small gradients (e.g. a difference of 250 mm/yr, north to south, in KNP), it may be that in the southern sector of KNP, rainfall plays a much larger role relative to fire and/or grazing in defining tree/grass coexistence, as has been found in other xeric savanna systems (Sankaran et al. 2005),

Buffalo shifted the relative proportion of annual and perennial herbaceous plants in favor of the former (Minchin and Dunlop 1989, unpublished data discussed in Skeat et al. 1996). This had significant consequences once buffalo were removed from this system due mainly to hysteresis effects of grasses, and hence fire regime (below). Further, Hodda (1992), in a survey of termites in Kapalga, found that the decrease in total herbaceous vegetation and the relative shift to annual grasses negatively affected “harvester” termites which preferentially feed on perennial grasses. Conversely, the species richness and abundance of termites that feed on the wood of live trees or forage in wood litter (and on buffalo dung) were positively correlated with buffalo presence (Braithwaite et al. 1988, Hodda 1992).

Second cascade: buffalo eradication. – The second ecological cascade was driven by an ecological “rebound” of the herbaceous vegetation (Werner 2005, Werner et al. 2006): a temporary but very large increase in primary production common during secondary succession (Horn 1974). In Kapalga biomass more than doubled within three years (Fig. 3.8). In locations where both buffalo and fire were removed from the system, biomass increased several fold to a steady state in three to six years (Cook 2003). Most of the large increase in biomass was due to an immediate increase in annual grasses. Only later did perennial grasses become a significant part of the understory, through
competitive advantage, but only if an area was left unburned and ungrazed for three to eight years or more. Perennial grass cover increased the longer a site was left unburned, but never rose above an average of 26% during the study period (P. A. Werner and J. S. Cusack, *unpublished data*). In general, annual grasses dominate stands that are continuously burned or grazed.

By far the most abundant annual grass in woodlands is the native annual sorghum (*Sorghum intrans* F. Mueller ex Benth). The removal of buffalo is widely perceived as the driver of an overall increase in native annual sorghum across KNP within the last decade (Miles 2003). To a great extent this is true, as buffalo selectively grazed perennials to the advantage of annuals such as sorghum, and certainly fostered its dominance in the region. But this view must be blended with other observations on the role of fire since buffalo removal, as sorghum is fire promoting and fire dependent (Cook et al. 1996) and the abundance of sorghum is strongly associated with fire frequency (Elliott 2005, Bowman et al. 2007). The sorghum dominated understory is now heavily entangled with increasing fire frequencies in KNP, where the mean fire frequency in woodland has increased from 55% to 70% in the past decade (Bowman et al. 2007, Petty and Bowman 2007).

These shifts in biomass and species composition of herbaceous vegetation had set the stage. What happened next to woodland structure depended mainly on fire treatments once buffalo were removed. In KNP, the woodlands were burned by both park managers and Aboriginal residents, resulting in fire return intervals of one to two years (Price et al. 2005). Starting around 1984, fires tended to occur earlier in the dry season (Russell-Smith et al. 1997b), perhaps in part because annual grasses, particularly sorghum, cure
Introduced buffalo

earlier than most perennial grasses thus permitting earlier fires. Woodland areas of Kapalga (northern sector) where buffalo had been removed for some 9 years, and fire excluded for 3 years after removal, had an abundance of eucalypt juveniles and saplings (Prior et al. 2006). This is in contrast to areas of Kapalga that had been burned annually from 1990-2003, following buffalo removal. Here the woodland remained dominated by an understory of annual sorghum with few sapling trees, and the eucalypt canopy cover had declined by more than half (Werner 2005; Werner unpublished data).

Without fires, marked trees in experimental plots in Kapalga had lower growth and survival rates when buffalo were removed, compared to trees where buffalo still grazed (Werner 2005, Werner et al. 2006). After late dry-season fires, however, growth rates of established eucalypts in plots where buffalo had been removed did not return to unburned levels over the next two years, whereas they did where buffalo grazed (Werner 2005). Juvenile mortality was 22% in burned plots and 6% in unburned plots where buffalo had been removed, but there was no statistical difference with respect to fire where buffalo continued to graze (Werner et al. 2006; Fig. 3.9). The differences in responses to fire with buffalo removal are most probably linked to the differences in understory vegetation, and hence fuel loads.

In a 2005 survey of woodland stand structure throughout KNP, Lehmann (2007) found that sapling recruitment into the overstory was limited by fire as well as basal area. However, beyond speculating on the effect of increasing fire frequency and intensity due to increasing annual grasses during the second cascade as well as increases in basal area during the first cascade, the degree to which buffalo removal has played a role in limiting sapling recruitment across Kakadu is not known.
Figure 3.10: Conceptual models illustrating resistance and resilience within savanna ecosystems in KNP. Ecosystem resistance refers to the strength of feedback loops that prevent change in the system; the precise mechanism varies with the state of the system, and is explained in more detail below. (a) This panel illustrates high resistance but low resilience in the mesic savanna woodlands and rainforests of the northern sector. The savanna woodlands have been quite resistant to change, but as illustrated by the divergence of the west and east portions, the systems are less likely to return to their previous stage once disturbed. The west portion, in particular, seems to have entered a state where the understory is dominated by annual sorghum, and recruitment of new trees continues to be inhibited by high fire frequencies and fuel loads (a grass/fire cycle). Conversely, in the eastern portion, where fires occurred less frequently from at least 1975 to 1990, the canopy continues to increase, and there is a more even mix of annual and perennial species in the understory. The monsoon rain forest is a non-disturbance-dependent climax community; in areas with deep soils, a very long fire-free interval could drive woodland communities to a monsoon rain forest state, while in other areas, overall cover and relative abundance of some rain forest species would increase (Woinarski et al. 2004). (b) This panel illustrates the low resistance but high resilience of the freshwater floodplains of the tidal region of the central sector. Floodplains have changed rapidly under impact from buffalo grazing (low resistance) towards a highly resistant state where mud deposition is no longer able to counteract incoming tidal force and the floodplains are dominated by extensive saltwater marshes and mangrove channels. Such a state did exist under higher sea level conditions 7,000 to 5,000 yr BP (Woodroffe et al. 1985). Following buffalo removal the floodplains are quickly returning to historical equilibrium (high resilience).
a) Mesic woodland (North sector):

- Fire frequency
  - annual
  - 1984
  - 2004 (west)
  - 2004 (east)
  - < 1 / 20 years

- Ecosystem Resistance
  - Woodland with grass/fire cycle
  - Woodland
  - Monsoon rainforest

- Overstory cover
  - 50%
  - 100%

b) Central sector floodplains

- Ecosystem Resistance
  - Freshwater floodplain mixed sedge/shrublands

- Saltwater channels
  - Fewer
  - More
DISCUSSION

Resistance and resilience of KNP vegetation types

The stability of ecosystems within a given ecological state has two aspects: resistance, the capacity of a system to resist change, and resilience, the capacity of a system to recover to a prior state after change (Westoby et al. 1989, Suding et al. 2004). Evolutionary history can have a profound impact on stability as evidenced by savanna grasses in Africa which seem to be resilient to grazing (McNaughton 1985). In contrast, perennial savanna grasses in northern Australia appear to have low resilience to sustained grazing pressure by introduced ungulates, as evidenced by the deliberate introduction of African perennial grasses into Australian pastures (Tothill and Mott 1985b, Winter 1991). Additionally, as we illustrate here, native perennial grasses are reduced in favor of highly-flammable native annual grasses where large grazers are present. This low resilience of Australian perennial grasses to grazing by large herbivores is most likely due to evolutionary history (Ridpath et al. 1993).

Resistance and resilience differ with vegetation type in KNP. Floodplain systems exhibit low resistance to change—rapid changes in grass species composition, reduction in grass cover, and expansion of channels under high buffalo population densities (Knighton et al. 1992, Finlayson et al. 1997, Petty et al. 2005), but high resilience in returning to the previous state—rapid increase in grass cover, recovery of most floral assemblages, and reduction of channels after buffalo removal (Fig. 3.10b, Finlayson et al. 1997, Petty et al. 2005).

In the monsoon rainforests, overstory cover in the interior of rainforest patches is
quite resistant to change, perhaps due in part to the long lifespans of most dominant rainforest tree species, as well as some protection from disturbance (Fig. 3.10a). Within the monsoon rainforest stands understory dynamics are unknown, although it is clear that buffalo caused extensive damage (Braithwaite et al. 1984, Lucas and Russell-Smith 1993), but recovery has not been assessed. On monsoon rainforest boundaries, park managers have aimed to reduce fire frequency to counter the effects of increased fuel loads with the removal of buffalo, and may have played a major role in increasing the total rainforest cover in the park (Banfai and Bowman 2006).

Woodlands are highly resistant to change, at least initially. However, high resistance is accompanied by low resilience, due in part to changes in the understory that prevent the recruitment of young canopy trees, both due to hysteresis effects from buffalo removal and from subsequent fire management. Indeed, the changes documented over the past 40 years indicate a shift in some areas to an alternate “grass-fire” state in woodlands where the understory is dominated by annual sorghum, increased fire frequencies, and, potentially, a long term decrease in woody cover (Fig. 3.10a).

The magnitude of change is different depending on historical buffalo population levels. Most of the evidence for buffalo-caused inhibition of overstory recruitment, post-buffalo release of annual sorghum, and juvenile tree/sorghum interactions comes from experiments performed on the western side of the South Alligator River, the region of KNP that historically had the highest buffalo populations. As if to underscore the role of buffalo in any state-shift, we do not see a decline in overstory cover on the eastern side of the South Alligator where buffalo population levels were historically lower. By contrast, in the southern sector of KNP, where buffalo numbers were also historically lower, there
was little change overall. However, this must be tempered by consideration of likely lower carrying capacity in the less productive South, as well as the possibility that lower rainfall makes Southern savannas even more resistant (Sankaran et al. 2005; Fig. 3.10b). Unfortunately, little is known of recruitment patterns or understory vegetation dynamics in the southern sector, and it is difficult to make definitive conclusions at this time.

*Long term consequences and management*

KNP is internationally renowned as simultaneously a pristine landscape and a landscape culturally constructed by Aboriginal people who continue to live in the park. However, continued management of the region requires the recognition of European settler efforts to “domesticate” this area of the frontier; efforts that were both facilitated and hampered by buffalo (Robinson 2005). Successful future management decisions for KNP and the Top End as a whole require an understanding of the characteristics of the present ecosystem in relation to the pre-buffalo “reference ecosystem” (*sensu* Egan and Howell 2001), as well as the consideration of means to manage the long term effects of buffalo removal.

Today, feral buffalo are once again dispersing into KNP and continue to grow in numbers in neighboring Arnhem Land. They have reached particularly high population densities in the vicinity of the Blythe River and the Arafura Swamp in central Arnhem Land, a wetland complex as extensive as the Alligator Rivers region. The long term prospects for controlling buffalo across a wide landscape are currently unknown and it is very possible that increasing feral buffalo populations will become a significant factor in KNP and beyond, and perhaps in KNP initiate a “third ecological cascade.”
Bond and Keeley (2005) have drawn parallels between fire and herbivores as alternative consumers of vegetation. Although grazing doesn’t necessarily replicate fire in terms of behavior or pattern, we show multiple examples in this paper where both are equally effective at consuming herbaceous biomass. Historically, most studies of mesic savannas have focused on grazing effects alone (with a vast literature on African savannas, Bond and Keeley 2005). Alternatively, empirical studies of fire and/or edaphic factors rarely address how grazing may have played a role in the vegetation dynamics of mesic savannas (e.g. Menaut et al. 1990, Lonsdale and Braithwaite 1991, Swaine et al. 1992, Bowman and Panton 1993, Glitzenstein et al. 1995, Williams et al. 1998, 1999, 2002, Moreira 2000, Andersen et al. 2003, Gardner 2006), although Russell-Smith et al (2003) mention that buffalo may have played a role in interpreting a 24-year study of the effects of fire on savanna woodland in KNP. The general lack of consideration for grazing in fire studies is perhaps understandable because fire effects are immediate and dramatic, grazers may no longer be present, and the impact of hysteresis effects can be easily overlooked or underestimated without a historical perspective. Notable exceptions are studies of Norton-Griffiths (1979), McNaughton (1993) and Mills and Fey (2005) on interactions between fire and native large grazers in Africa, and the studies of Silva et al. (2001) and San José and Fariñas (1983, 1991) on changes in a cattle- and fire-protected savanna in Venezuela. In general, our study shows consistency with the African studies (above), but comparisons with the Venezuelan work are inconclusive.

The results of this KNP study are consistent with models of fire-grazer interactions in savannas developed from an east African perspective (Higgins et al. 2000, van Langevelde et al. 2003, Archibald et al. 2005) as well as field studies in drier
Introduced buffalo savannas of northern Australia (Fensham and Skull 1999, Sharp and Whittaker 2003, Sharp and Bowman 2004). Further, both time lags and rainfall gradients in KNP show a similar pattern of change to that reported along a rainfall gradient in the Serengeti (Norton-Griffiths 1979). There Norton-Griffiths (1979) found a greater decline in woody vegetation from 1958 to 1972 in the wetter north, which he primarily attributed to a time lag from repressed recruitment due to intense fires during the rinderpest epidemic. As in KNP, grazer-induced changes in fire regime had less of an impact on woody cover in the drier southern regions of the Serengeti.

Still, however, the long-term cascading impact of grazing differs between Australia and Africa. The subsequent release in Australia of a fire-promoting native annual has created the possibility of a long term decline in the tall woody canopy (Fig. 3.10a). Globally, the savannas of KNP (and the ARR) are most analogous to the moist savannas of South America, particularly the cerrados of Brazil (Furley 1999), as neither support populations of large native grazers. It would be of great interest, and immediate importance, to investigate the impact of introduced grazers on cerrados dynamics in South America.

Additionally, comparisons would be fruitful with the Amazonian floodplains of Brazil, which currently have the highest rate of buffalo population growth in the world (Sheikh et al. 2006). The response of vegetation on the floodplains of Keoladeo National Park in India, where buffalo were removed from their native habitat, would also be informative. There, the floodplains contained a rich mixture of grazing-tolerant plant species when buffalo were present. The relative abundance of these species declined after buffalo were removed (Middleton et al. 1992).
Successful management of savanna ecosystems requires a thorough understanding of the historical range of variability in the system, the factors that drive that variability, and an understanding of potential alternate states. This is critical, not only for understanding the range of potential management targets, but also for understanding how best to manage savannas within desired thresholds (van Wilgen et al. 2004) as the importance of savannas to human economies increases in the next century.
Changing perceptions and perceptions of change: conceptions of landscape in Kakadu National Park

Aaron M. Petty

ABSTRACT

As pressure grows to incorporate the management of protected spaces with recognition of indigenous tenure (“joint management”), the understanding of landscape, as the intersection between space and the cultural understanding of space, is emerging as an important field of study. In this paper I analyze cultural constructions of landscape among three groups of Aboriginal landowners resident within Kakadu National Park, a jointly managed national park located in the tropical savanna region of far northern Australia. I focus on rivers and the forest surrounding rivers (collectively known as riparian corridors) as they are of immense importance to Aboriginal people in Kakadu. Historically, they provided pathways for people through the landscape, and were a critical resource base that provided both food and water throughout the lengthy dry season. Since European settlement, the role of riparian corridors in Aboriginal livelihoods has changed in response to changing patterns of movement and interaction with country. Among other changes, riparian areas have changed from interrelated and continuous
spaces threading through the landscape to discrete “good sites” connected by automobile travel. I discuss the impact that this change has had on ecological patterns in the region, as well as the conceptions of ecological change by each Aboriginal group. I compare and contrast case studies of each group and argue that underlying differences in the perception of ecological change, both between groups and between the groups and the dominant society, reflect a fundamental diversity in the perception of landscape. Any attempt to satisfactorily resolve questions of ecological change and ecosystem management within a joint management context must take into account these varying perceptions.

INTRODUCTION

Kakadu National Park is a jointly managed park located in the tropical savanna region of Australia’s Northern Territory (the “Top End”; Fig. 4.1) whose history exemplifies the post-colonial processes of symbolic and physical change in the region. Although there has been more or less continuous enthusiasm for developing Australia’s tropical north into an economic powerhouse, the process of colonisation has failed to completely restructure the Top End in European terms. This is highlighted in the ecological discourse of the region, where there is as great or greater concern about the ecological impacts of changes in the Aboriginal management of remote areas than from the direct impacts of colonization and industrialization (e.g. Bowman and Panton 1993, Russell-Smith et al. 1998, Franklin 1999, Whitehead 2000, Woinarski et al. 2001, Whitehead et al. 2005).

Prior to the establishment of Kakadu National Park in the late 1970s, the region was situated at the north-eastern edge of European settlement and was largely resistant to
state control. As elsewhere, disease and dispossession lead to significant Aboriginal depopulation in the region that became Kakadu. This restructured the social relations of Aboriginal people and opened the way for European settlement. However cattle and mining, the enterprises that facilitated colonialist expansion elsewhere in the Top End, were marginal in the south and non-existent in coastal plains of northern Kakadu. There a unique industry, the processing of Asian water buffalo (*Bubalus bubalis*) hides, supported Aboriginal lifestyles within a quasi-European economic framework (Keen 1984).

These differences in the pattern of European settlement lead to differences in the individual social histories of Aboriginal people that were contingent upon their geographic position. This has lead in turn to different individual perceptions of change, and also different perceptions of the ongoing process of colonisation and state control. In particular, the development of improved roads and the establishment of the national park (concurrent with the granting of land rights) lead to a feeling of dispossession for those people located in the north who had previously maintained significant autonomy within the buffalo economy of the region. For those who associated with territories in the southern part of Kakadu road development and Park establishment resulted in access, for some the first time in their adult lives, to areas of great cultural significance from which they had previously been dispossessed.

In this paper I use cultural perceptions of landscape, particularly landscape as understood within historical ecology (Balée 2006) as a lens through which to view these different understandings of ecological and social change. My intent here is to show how landscape is historically constructed as both an ecological and a social phenomenon. I
am attempting, in part, to reconcile different interpretations of landscape change, but I also wish to develop an understanding of landscape as both interpreted and contested. This is critical if the project of joint management is to be successful. At present, European interpretations and values are predominant in management discourse in Kakadu, and the lack of recognition on the part of European managers that alternate landscapes coexist in Kakadu and other jointly managed parks in Australia creates space for considerable conflict (e.g. DeKoninck 2005).

Although I discuss environmental and social change broadly, my focus is on the floodplains, rivers and other riparian environments of Kakadu. As I show, riparian areas constitute some of the most significant lived spaces in Aboriginal life, and, insofar as Aboriginal knowledge and understanding are embedded within lived experience (Merlan 2005), are integral to Aboriginal constructions of landscape. As in other areas of northern Australia, riparian corridors were historically the ‘core’ area of both social and economic interaction with the environment (Pickering 1994). As the locus of interaction with country, riparian areas are places of memory and history where Aboriginality and tradition are reinforced (Merlan 1994). As one Aboriginal woman told me, “We tell children their story and history when we come camping by rivers.”

THE MEANING OF LANDSCAPE

‘Landscape’ had its roots in sixteenth century art and was originally considered static – a fixed and idealized background upon which played the human drama. In 1926 Sauer (reprinted 1963) articulated the notion of a ‘cultural landscape.’ This reflected a growing awareness among geographers that culture both physically shapes and mentally interprets space to produce landscape, and expanded the concept of landscape from the
realm of paintings and maps to a dialectic between the culturally constructed ideal of landscape and the everyday experience of and interaction with landscape (Hirsch and O'Hanlon 1995).

This understanding of landscape was particularly well developed in Aboriginal studies (e.g. Stanner 1979, Rose 1992, Morphy 1995). Indeed, Morphy (1995, p. 206) suggests the agency of Aboriginal people themselves in making ‘landscape a key concept in the study of their society’. The Dreaming, widely interpreted as a parallel space coexisting with our own world (Stanner 1979, Rose 1992, Morphy 1995), can be understood as the encoding onto the landscape of idealized human relations to one another and the natural world, a point articulated by Rose (1994), “Dreamings demarcated a spatially identified moral universe.”

Landscape is continually reshaped and reinterpreted in terms of contemporary social reality (Morphy 1993). For Aboriginal people few realities could have been as confronting and challenging as that of European colonization and conquest. For example, in the vicinity of the Roper River, on the southern boundary of the Arnhem Land Aboriginal Reserve:

Aborigines reconstructed their own regional linkages and continually interacted across the boundaries set by Europeans. The colonial process ended up structuring the Aboriginal population to fit into the landscape as it had been transformed by European settlement and gave the people no option but to articulate with the life of the cattle stations. One of the consequences of this was that the sacred landscape became focused on the cattle stations and their immediate environs (Morphy 1993, p. 237).

A similar process occurred in the cattle station regions of southern Kakadu where specific knowledge of place became concentrated along the locations of stations, towns, settlements and mining camps. General knowledge of country travelled through corresponded with seasonal migration during periods of unemployment (Merlan 1992),
while knowledge of places outside of normal experience, particularly places to be avoided by all but initiated males, become generalized to broad regions that were rarely if ever visited (Palmer 2004).

By studying the process of material change within landscapes, the field of archaeology added a historical dimension to landscape, as well as articulating the importance of the physical environment in shaping culture and the capacity of culture to transform the physical environment; landscape as both ‘created and creating’ (Gosden and Head 1994). As with cultural geography, Aboriginal engagement with archaeologists helped to shift the focus of Australian archaeology from material culture to how Aboriginal people view landscape (Moser 1995). In particular, the Aboriginal use of fire and fire as a landscape management tool has been well developed (e.g. Jones 1969, 1980, Kershaw 1986, Head 1994b, Lewis 1994, Head 1996, Russell-Smith et al. 1997a), but Australian archaeology has also contributed to our understanding of the relationship between ‘social’ landscapes and settlement and subsistence patterns (Pickering 1994 and references therein).

In this chapter I explicitly define landscape as the space that is interpreted and shaped by human action (Hirsch and O’Hanlon 1995) and thus changes over time in response to human action and the underlying human interpretation. This process has been described as ‘inscription’ (Balée 2006) and as ‘sedimentation’ (Moore 1998), and is particularly useful as it encompasses the physical construction of space, human perception of space and (particularly) human agency within space. Layton and Ucko (1999) describe three dialectical aspects of landscape: (1) the physical environment, (2) the social organisation of the environment and (3) perceptions of the environment.
Understanding landscape change therefore includes an understanding of the symbolic processes underlying the formation of landscape as well as the material processes traditionally associated with ecological change.
The history of the region that became Kakadu National Park illustrates how differences in the physical environment, particularly the difference between the northern floodplains and drier southern portion of Kakadu, shaped the European vision to transform the region into economically productive land. The different economic activities in the north and south lead to different social relations between Aboriginal residents and European settlers that reshaped both European economic activity and the Aboriginal perception of and interaction with landscape in different ways.

METHODS

I focus on two historical eras in Kakadu National Park, using different methodologies for each era: (1) published and archival documents that reflect on the Aboriginal experience in the Alligator Rivers Region after European contact and before World War II; (2) the life stories of three Aboriginal groups who are recognized as traditional owners of the southern and central portions of Kakadu and who narrated their own post-World War II experiences.

The history and geography of Kakadu intersect to create two distinct regions of Kakadu (Fig. 4.1). The northern and western portions Kakadu, some two-thirds of the Park, comprise lowland plains with very little vertical relief and numerous meandering creeks and rivers. Large floodplains – up to 20 km across – form along major rivers beginning 100 km from the coast. This region was ideal habitat for Asian water buffalo, which rapidly expanded to carrying capacity in the region (Chapter 3). The southern and eastern portions of Kakadu are dominated by an uplifted sandstone massif, the Arnhem Land Plateau. The canyons and ravines of the Plateau make access to the area difficult and skeletal soils make the region largely unsuitable for cattle. However the geology of
the region makes it particularly rich in minerals. The southern region of Kakadu is drier than the north and the lowland plains to the south and west of the Plateau were better suited for cattle than the north.

Between January 2003 and September 2004, I conducted 36 semi-structured interviews with 15 Aboriginal residents of Kakadu National Park. These participants self-identified into three groups based on their affiliation with a particular language or clan name. The first group comprises several families who claim their identity as a language group, the Jawoyn, that encompasses the area from southern Kakadu to Katherine. The particular Jawoyn I interviewed have recognized traditional ties to Kakadu. The second and third groups comprise two families who separately identify as Murrumburr and Ngomburr and who have maintained ties to the South Alligator Floodplains in central Kakadu (Fig. 4.1).

I conducted two different types of interviews, ecological and biographical. Interviews were open ended, but structured around separate techniques of mapping and separate lists of core questions. Although there are many other Aboriginal families and identified groups residing in Kakadu, the individuals I interviewed for this research are comprehensive samples of both those Jawoyn who have claim to Kakadu and of the Ngomburr and Murrumburr families.

Ecological interviews were held “in-country” (at stream locations that were selected by participants as being particularly desirable to visit). We would begin by constructing a sketched map of the area, highlighting resources and areas that had changed. Often this would lead to a discussion of plant and animal abundance and management of the area (couched in terms of ‘looking after country’). I would ask
participants for their opinions on the nature of change, and what they thought the trend was (i.e. what would the place look like in 10 years’ time?). I would also ask specific questions about resource management and use of the area, including what plants and animals would be gathered from this area, how they were gathered, and whether any other steps were taken to manage them. Finally, we discussed Park management of the area. These visits were often done in conjunction with a vegetation survey (Chapter 2) that would take about two days to complete. This afforded plenty of time for discussion outside of the formalized context of the interview.

Biographical interviews were conducted within participants’ households or at Kakadu district ranger stations where some participants were employed. They centred on individual life history and past and present travels across the landscape, including the location of visitation sites and sites of significance, and the locations of food and other resources. I used 1:100,000 topographic maps to facilitate the conversations about history and to contextualize peoples’ biographies in space (Rose 1987).

CHANGING PERCEPTIONS: SOCIAL AND ECOLOGICAL HISTORY

Gardens and Sacred Groves: pre-contact interaction with riparian areas

Balée (2006) contrasts the “garden:” “the underlying premise of a landscape, for there humans habitually interact with other forms,” with the “sacred grove”, a space where human disturbance is excluded. Sacred groves are important ecologically as they provide refugia for disturbance sensitive species. Balée’s (2006) formulation perhaps reflects an Amazonian perspective, however it is a useful lens through which to view riparian areas in Kakadu, as they take on characteristics of both. They are clearly foci for
human interactions with the environment and yet are simultaneously important refugia from drought and fire for many plant and animal species.

The riparian forests of Kakadu are situated within a matrix of tropical eucalypt savanna. The vegetation composition of the savanna is largely determined by the degree of wet season inundation, and extensive floodplains form in regions of persistent inundation (Bowman and Minchin 1987). These wetlands have nurtured one of the longest continuous hunting and gathering cultures in the world (Jones and Negerevich 1985), and, with the evolution of vast freshwater floodplains some 1200 years ago (Clark and Guppy 1988), supported one of the highest population densities for a hunting and gathering culture (Jones 1980, 1985). Before purchased foods became widely available, fish, shellfish and water lilies provided the staple wet season food for people in eastern Arnhem Land, and presumably elsewhere (Thomson 1949).

Perhaps reflecting on landscape as garden, Jones (1969) dubbed Aboriginal fire management in northern Australia ‘fire-stick farming’. Over 90% of the annual rainfall of the region falls during the Austral summer (November to April) and the region is highly fire prone. Human initiated fires burn some 50% of the landscape annually. Fire is a strong determinant of vegetation structure and composition in northern Australia (Bradstock et al. 2002), and helps shape the boundary between fire sensitive forest and fire insensitive woodland communities along riparian corridors (Chapter 2; Fig. 4.2). Beginning perhaps as soon as first arrival some 40,000 years ago (Lewis 1994, Bowman 1998, 2002), or in the late Holocene when rain seasonality intensified (Head 1994b, Head 1996), Aboriginal people shifted the fire regime. Without human intervention, extensive lightning ignited fires will occur under very hot and dry conditions as convection storms
Figure 4.2: Photo contrasting the dense riparian forest adjoining the South Alligator River with more open riparian woodland. The woodland is burned in this image. Photo by A. Petty.

build up at the end of the dry season and start of the wet season (October to December). By igniting smaller fires occurring throughout the dry season (April to December), humans can create a patchwork mosaic that limits the spread and intensity of late season fires. Fire is used by Aboriginal people today to renew grasses and attract wildlife (Haynes 1985, Ybarbuk et al. 2001, Murphy and Bowman 2007), to prevent the incursion of fires into sensitive areas, particularly those with yams (Russell-Smith et al. 1997a), to clear living areas and travel corridors of grass (Haynes 1985), and to “clean” long unoccupied country and mark presence within the landscape (Lewis 1994, Head 1996).

Although the open woodlands adjoining riparian forests were frequently burned, the active fire management of riparian forests appears to be uncommon. Aboriginal
groups in the Kimberley region of Australia describe burning ‘right up to the edge’ of small spring-fed closed forest patches, but fire would not get inside the forest because it was too wet (Head 1994b). Among the respondents in this study, there was a nearly universal opinion that riparian forest was not burned; the sentiment expressed was typically passive, i.e. “there’s no need to burn the forest,” or, “we just leave the forest alone.” Most reported that when they burn by riparian forest they always pay attention to the wind and burn so that fire moves away from the forest and into adjoining savanna. Riparian forest tends to be wet and difficult to burn, and small forest patches will degrade under repeated fires (Douglas et al. 2003) so it should not be surprising that fire would neither be attempted nor desired in riparian areas. Nonetheless, I have known Aboriginal people to burn small patches by a stream, particularly the dried leaves on pandanus palms (*Pandanus aquaticus*), to ease access to the waterfront, to clear out snakes, to clear out camping and hunting areas. One Jawoyn woman emphasized fire as a means of clearing access to riparian areas, “All across Kakadu, women burn their grass by fishing areas.” Moreover, many expressed the sentiment that vegetation has become too dense and that this has made access to the water difficult. It is not improbable that before depopulation, the combined impact of such small scale management, multiplied by thousands of individuals regularly walking through and camping in riparian corridors may have been enough to cause a net decrease riparian canopy cover.

Foot travel along riparian corridors would not have occurred within the riparian forest itself, which would be nearly impassable even if vegetation was historically less dense. Rather, people travelled through the more open riparian woodland adjacent to the forest. These woodlands are akin to upland savannas but consist of species tolerant of
occasional wet season flooding (Chapter 2). Like upland savannas they are actively burned – sometimes to my chagrin when a particularly overgrown area would be burned before I could complete the vegetation survey! The principal reason given for burning woodlands was to clear the vegetation, although some respondents did suggest that burning the woodlands would protect the forest, again suggesting the sacred grove character of riparian forest.

The Gimbat Valley and the Plateau region in the southern part of the Park is collectively known as ‘sickness country’ by the Jawoyn and other Aboriginal communities within Kakadu and Arnhem Land. It contains sacred sites of paramount importance that are believed to have the power to unleash devastating destructive power if they are disturbed. Within this region, only initiated elders could traditionally burn upland areas near sacred areas. These fires would burn from the top down to lowland areas. People would not burn near lowland creeks as the fires might travel up and damage sacred areas.

There was little evidence of other forms of management of riparian forests and woodlands. Tending yam beds to maintain and enhance productivity is mentioned in ethnobiological accounts of northern Australia (Russell-Smith et al. 1997a). However, these are often found in rainforest patches or upland along well-drained scree slopes of the Plateau adjacent to riparian areas. Only one, less desirable, species of yam, cheeky yam (*Amorphophallus galbra*), is commonly found in riparian bottomlands. Women would break off the tops of these or replant the tendrils after harvesting, but generally less care was taken with them than with the much sought after long yam (*Dioscorea transversa*), which is found in rainforest and dense woodland. It was not directly related
to me that the substantial number of plant species utilized within riparian communities were manipulated or managed. However, given the ubiquity of such practices in other foraging societies (e.g. Vansina 1990, Anderson and Moratto 1996, Rival 1998), I would not be surprised if these practices once existed but have escaped the notice of researchers. As with small scale fires, intensive management, including the tending of existing patches of plants and the planting of “anthropophytes” (Pinkley 1973 referenced in Balée 1994) within riparian corridors, when applied by thousands of people over generations may have had a profound impact on forest structure.

*European contact and conquest*

The people of Australia’s north coast have had sustained contact with overseas cultures since at least the 1500s, when Macassans from the island of Sulawesi likely began regular trading voyages (Berndt and Berndt 1954). However, these voyages, predicated on relationships between relative equals, did little to prepare Aboriginal people for the European invasion. European settlement of the Top End was virtually non-existent until the mid- to late- 1800s when sporadic economic opportunities from mining brought the first wave of settlers.

Gold deposits were first found in the vicinity of Pine Creek (about 50 km south-east of Kakadu) in the 1870s, about the time of the construction of a north-south railway between Adelaide River (90 km south of Darwin) and Pine Creek as well as an overland telegraph line linking Darwin with the more populated regions of Australia (Fig. 4.1). Opium brought by Chinese immigrants, as well as tobacco and alcohol attracted Aboriginal people to mining camps, and some sought economic opportunities in menial labor or in the many brothels set up for the overwhelmingly male settler population.
These early gold rush days had devastating consequences for the Aboriginal population. Poor health and hygiene from drug and alcohol abuse and concentration within towns facilitated the spread of introduced diseases through the population. Violence and disunity between groups, and infertility from sexually transmitted diseases, made the situation worse (Keen 1980, Levitus 1982). Keen (1980) estimates that the Aboriginal population in the Alligator Rivers Region declined to about 3% of their number at the time of contact. Today, all of the gunmogurrgurr, or patrilineal descent groups, of two language groups, the Ngombur and Ngumbarkala, within the South Alligator floodplains are extinct (Keen 1980), as are most of the Jawoyn mowurrwurr (their analogue for gunmogurrgurr), particularly those at the northern end of their territory (Merlan 1992).

The ecological impact of depopulation and concentration within townships has been well demonstrated, particularly for the dryland vegetation of the Arnhem Land Plateau. Indeed, it’s arguably better understood in this region than for most other parts of the world that have experienced the devastating impacts of European conquest. The decline of the conifer Callitris intratropica (Price and Bowman 1994, Bowman et al. 2001a), threats to fire sensitive heath species (Russell-Smith et al. 2002), and expansion of the rainforest species Allosyncarpia ternata (Prior et al. 2007) have all been linked to changing fire regimes as a result of depopulation. Depopulated areas are in general more likely to have large and intense late dry season fires (Russell-Smith et al. 2003b, Bowman et al. 2004). It is highly likely that there were similar impacts on riparian vegetation, at least initially after depopulation, but this impact was later confounded by the effects of the increasing population of introduced buffalo and cattle.
Pastoralist expansion in the Top End began soon after the first gold rush and was facilitated by the construction of the railway and telegraph line. Jawoyn territory in the southern part of the Park was situated in the northern periphery of “cattle country.” As elsewhere in the Top End, pastoralism was never economically successful in Kakadu, but was a means of territorial control and conquest (Morphy 1993). Many Jawoyn were employed on cattle stations, often for access to store goods and to avoid being killed, a common fate for ‘wild’ Aborigines. Merlan (1992) suggests that prior to World War II clan-level relationships were more sustained in the Plateau region of Kakadu than in other Jawoyn areas because the region was not yet under European control and was a refuge both for people living outside of the European sphere and a destination for seasonal migrants let go from cattle station work.

The first marginally successful cattle station located within what is now Kakadu was Goodparla. It was established in the 1930s in the lowlands adjacent to the Plateau, in the south-western portion of Kakadu (Fig. 4.1). Its owner, George Cooke, had little to do with Aboriginal people and generally regarded them as a threat to his livelihood (Levitus 1982, pp. 39-40). Gimbat was established adjacent to Goodparla in 1937, and these two properties later formed Stage III of Kakadu National Park, annexed in 1987. Successive owners in both Gimbat and Goodparla had more relaxed attitudes to Aboriginal people, and employed a modest number of mostly Jawoyn people, as well as a few Myali from the vicinity of the East Alligator River, up until the late 1960s when mandatory award wages decreased the incentive for hiring Aboriginal people.

The Ngomburr and Murrumburr country in the northern section of Kakadu never had even a marginally productive cattle station (Levitus 1995). Rather, it was the hub of
a uniquely Top End industry: the hunting of feral water buffalo for their hides. Buffalo were first introduced to the Australian mainland at Cobourg Peninsula, some 100 km to the north of the Alligator Rivers Region (Fig. 4.1). By 1885 they were abundant in the vicinity of the East Alligator River and by 1895, the first known buffalo hunter, Paddy Cahill, was hunting buffalo at Kapalga, on the west side of the South Alligator River (Mulvaney 2004). The buffalo hide industry quickly gained pace after Aboriginal workers adopted horseback riding and were trained in Paddy Cahill’s technique of horseback shooting (Levitus 1982). Hide exports for the Northern Territory generally remained above 5,000 per annum from the early 1900s and remained high for over half a century. So important was the buffalo to the economy that in 1920 the Northern Territory’s acting administrator recommended protecting buffalo as they were “fast becoming extinct on the mainland” (Anonymous 1921).

Unlike cattle, buffalo did not become an agent of expropriation, but rather to Europeans became symbolic of the failed “domestication” of the frontier (Robinson 2005). The buffalo hide industry took place at the margins of state control, and the nomadic and seasonal qualities of the industry permitted much more Aboriginal autonomy. It allowed Aboriginal people to preserve their relationships to country and to each other to a greater degree than was possible to the south (Levitus 1995). The synergy between Aboriginal and European lifestyles was such that buffalo hunting attracted a “westward migration” of people from western and central Arnhem Land to the Alligator Rivers Region to participate in the economic opportunities provided. Disease and alcohol, although still prevalent, were reduced in buffalo camps relative to the crowded “town camps” that on the outskirts of Darwin, Katherine and Pine Creek.
Grazers such as cattle and buffalo preferred creeks and floodplains for many of the same reasons humans did. They provided cool shade, water and green forage through the dry season. Through browsing and trampling they likely had a profound impact on vegetation. Some of the impacts on riparian forest would have been analogous to the impacts on rainforest detailed in Chapter 3, including the destruction of small trees and seedlings, the introduction of exotic plants, and the establishment of ruderal plants in spaces opened up from trampling. At sufficiently high population densities buffalo grazing would have reduced fuel loads, in some ways mitigating the impact of depopulation on fire management.

Post-war

World War II was a watershed event in European settlement that consolidated state control in the Top End and marked the beginning of more intrusive efforts to colonize and control Aboriginal life in the region. Economic and technological changes, including the collapse of the hide industry, the development of crocodile skin, timber and tourism industries, and road construction and improvement, restructured social relations in the region. Before World War II there were two dirt roads coming into the Alligator Rivers Region, one from Pine Creek to Oenpelli and the other entering the Alligator Rivers Region from the northwest and joining the Pine Creek/Oenpelli road. Uranium mining, a new industry from the aftermath of World War II, facilitated the construction of new roads and the improvement of existing ones, and a network of bush tracks developed to accommodate safari-based tourism. These developments increased access to Kakadu for both Aboriginal and European people. Several major uranium deposits were found in the northern edge of the Arnhem Land Plateau, and the resulting environmental inquiry
(Fox et al. 1977) suggested a jointly managed national park with excisions around the uranium deposits as a means of reconciling the competing interests of mining, tourism, and the recognition of Aboriginal land rights. The latter had entered national consciousness with the passage of the Aboriginal Land Rights (Northern Territory) Act in 1976. Stage I of Kakadu National Park, which encompassed most of the land associated with the Ngomburr and Murrumburr, was recognized in 1979 as Aboriginal land to be leased as a park. Recognition and annexation of the principally Jawoyn lands of Stage III didn’t begin until 1987, and was finalized in 1992 after a lengthy land claim process that was somewhat contentious because of the unresolved status of several mining leases within the Gimbat Valley. These changes occurred within the living memory of the Jawoyn and Murrumburr/Ngomburr I interviewed, and their life histories illustrate the scale of change to Aboriginal life after World War II.

Life histories

Australian government policy during the war was to place “loose” (unemployed) Aboriginal people in camps (Levitus 1982, p. 64) and many Jawoyn were removed to Bamyili (now Barunga) settlement, an Army camp some 50 km east of Katherine. Bamyili soon developed into a major Aboriginal community and reference point for Jawoyn (Merlan 1998). All of the male custodians of Stage III went to school there, and few had any sustained connection with Kakadu after childhood. Much of their adult lives were spent in Bamyili working on a nearby peanut farm and cattle stations, or doing various jobs for the Australian Native Affairs Commission.

The family of Bessie Coleman and Beryl Smith maintained residency within Stage III after World War II, and Bessie was born on Goodparla station in 1955. Beryl
was born in Pine Creek in 1956. Their parents worked doing domestic chores and cattle handling at the two pastoral stations. Both Bessie and Beryl were removed from their parents and placed in the Rita Dixon boarding school in Darwin, part of general Aboriginal policy at the time. Bessie later returned at the age of 13 to work at the Mary River tin mine. She worked through her young adult life as a buffalo catcher throughout Stage III and the Mary River and South Alligator floodplains to the north. As she grew older she settled in Pine Creek and had infrequent contact with the Park until 1997 when she took up employment as a Park Ranger and lived in Gimbat. She has since maintained sporadic employment within the Park, either as a ranger or with tourist operators. Beryl worked in Darwin and Pine Creek as an adult, and first returned to Stage III as a Park Ranger in 1998. She still lives and works in the Park as a ranger. Of the males I interviewed who had traditional inheritance ties to parts of Stage III, only one, Roy Anderson, had any contact with the region before the 1980s, when the stations were being claimed as Aboriginal land. Roy accompanied his parents on trips across the Plateau in the 1950s, returned in the 1960s to build stockyards in Gimbat, and did not return again until the land claim period. The other male interviewees had parents, now deceased, who travelled and worked within the present day boundary of Kakadu, but they themselves didn’t visit until they were adults, during the annexation period.

As in other pastoral regions of northern Australia, the Jawoyn altered their interactions with country to conform to a landscape that had been reconstructed by European society. Access to country, and hence knowledge of country, crystallized around the realities of pastoral life. Movement within southern Kakadu centred on the pastoral stations of Gimbat and Goodparla (Fig. 4.3). When released from work, people
Figure 4.3: Jawoyn travels across the landscape reconstructed from biographical maps. Disused paths and campsites were not used by the respondents, but were recalled as being places of travel and camping. Historical sites and overland paths were used by respondents when as children or young adults they lived within the present boundary of Kakadu, but are now no longer (or very rarely) used or visited. Today travel to contemporary sites takes place by vehicle.
followed streams and horse and buffalo tracks to places near the stations, but removed from European oversight. The Plateau, which was a refuge for Jawoyn before World War II, receded from experience (Merlan 1992, Palmer 2004).

The language that Bessie, Beryl and Roy use to recall historical, and indeed contemporary, travel around country is structured by cattle station life and labor. Travel to favored sites for fishing or camping was referred to as occurring on ‘weekends and holidays.’ These sites are also often referred to as ‘refuges’ or ‘quiet places’ from the intrusion of white people into Aboriginal life. Travel emanated from the cattle stations at Gimbat and ‘old’ and ‘new’ Goodparla to other points (Fig. 4.3). The South Alligator River/Kambolgie Creek region, including the Gimbat Valley, contains some of the most significant Jawoyn sacred sites, some of which were situated along a cattle station and road corridor that were regularly visited during the cattle station era. Travel through other regions of the Plateau was virtually abandoned after World War II. This increased the importance of the South Alligator and other rivers in navigating the Plateau, as rivers were known to be a safe corridor for women and uninitiated males in territory that is riddled with sites of great power and significance (a point that was reinforced by Bessie on multiple occasions). As a result of disassociation with large areas of country, names and knowledge about specific places outside of safe travel corridors became generalized (Palmer 2004).

By contrast, although the Murrumburr and Ngomburr were transformed by settlement, they and the country they occupy partly resisted European attempts at transformation into productive European space. However, this is qualified because the Murrumburr and Ngomburr engagement with the cash economy almost certainly changed
livelihood patterns. It is difficult to reconstruct pre-settlement movement across the landscape, but it is likely that Murrumburr and Ngomburr travel contracted after the war to focus on economic activity on the floodplains. Accounts of other Alligator Rivers peoples (Chaloupka 1981, Levitus 1982) indicate that regular travel pre-contact was more extensive, and people would often travel from the headwaters of the Katherine River to the Cobourg Peninsula in the space of a single year. This travel was not so much to engage in the food quest as to renew social relations with people. It is likely that Aboriginal depopulation and population shifts westward removed the motivation for these journeys.

Among the Murrumburr and Ngomburr, the Alderson family, who identify as Murrumburr, maintained the strongest continuous relationship to Kakadu. Yorky Bill Alderson was born in 1897 to a European man and an Aboriginal woman. He later moved from Western Australia to become a buffalo shooter in the Top End and was adopted into the Murrumburr clan. He incorporated an “entrepreneurial” way of life grounded in European notions of productivity and labor and this probably saved him from being removed to the Aboriginal camps during World War II. He was permitted to stay in the Alligator Rivers Region while others were forcibly removed (Levitus 1982, p. 64). In the 1950s, Yorky Bill Alderson worked at Munmarlary, on the eastern side of the South Alligator River (Fig. 4.4) and travelled around the northern floodplains of the South Alligator River hunting buffalo. His first child was born in 1948 at Ngurrkdu (Spring Peak), an area that would later become the “base camp” for the Alderson family. His second child, Jessie, was born at Nurdulgulgul, a floodplain on the western side of the South Alligator River. His third, Violet, was born at Munmarlary, a buffalo camp on
**Figure 4.4**: Murrumburr travels across the landscape as reconstructed from biographical maps. Historically visited locations are locations that were visited by respondents until approximately the late 1960s and early 1970s. Today, virtually all travel to currently visited sites is done by road and very few overland paths are used (except from Ngurrkdu to Yirrirri and Gulindji).
the northern floodplains of the South Alligator River. In the late 1950s there were no more employment opportunities in the northern floodplains and the family moved to Anlarrh (Nourlangie Camp). There Yorky Billy worked long enough to buy horses and begin collecting and selling dingo and crocodile skins independently. The family settled at Ngurrkdu and would make regular trips to sell skins at Pine Creek in the wet season and Anlarrh in the dry. Throughout the 1960s they travelled by horseback extensively throughout the South Alligator floodplains and the catchments of Jim Jim and Nourlangie Creeks, two major tributaries of the South Alligator River. In the dry season the family would move around the floodplains, and often camp at Gindjala to hunt the abundant geese, ducks and turtles. In the wet season, when travel was restricted, the family would stay at Ngurrkdu and make day trips or short camping trips to nearby sites such as Yirrirri and Gulindji. Their youngest daughter, Judy, was born at Spring Peak in 1968 and related to me that “[my parents] were getting old so were much less mobile than when Violet was a child.” This also would have been around the last time the family visited the floodplains on the western side of the South Alligator River. She remembers visiting Anlarrh by motor vehicle, but otherwise by the early 1970s their movements were largely restricted to the environs of Ngurrkdu.

The Cahill family, who identify as Ngomburr, had a quite different engagement with the economic and social changes after World War II. Walter Patrick Cahill (who was given his name after the first European buffalo hide shooter in the region, Paddy Cahill) was born at Alligator Billabong, and lived and travelled with his family around the Western side of the South Alligator River. As an adult, in the 1950s and 1960s, Walter Patrick took up buffalo shooting and moved between Wildman River, Pine Creek,
Oenpelli, Gimbat, Mudginberry and Munmarlary. His daughter, Eileen Cahill, married a
buffalo shooter and lived and worked between Goodparla and Jim Jim Creek. She later
remarried to a European meat supplier in Darwin, Johnny Bell, and the family travelled
as far as Broome (on the west coast of Australia) before eventually settling near Darwin
where Johnny Bell set up a small sawmill. In the late 1970s Eileen heard about the land
claim for Stage I of Kakadu and returned in 1985 with her three children to Kakadu.
After their return, the family moved between Darwin and a sequence of old buffalo and
safari camps in Kakadu, many of which are now Aboriginal outstations: Nourlangie
Camp, Cannon Hill, Oenpelli, Mudginberry, Jabiru and Patonga.

Post-war changes

It is tempting to caricature the Jawoyn as hapless victims of European conquest,
while the Murrumburr maintained their Aboriginal lifeways in a buffalo-mediated
cultural zoo and the Ngomburr pursued new economic opportunities. However, the
changes of the post-World War II era, in particular the development of roads, increased
vehicle access, and the restructuring of daily economic activity around cattle stations,
buffalo camps and safari camps reshaped the spaces of Murrumburr, Ngomburr and
Jawoyn life, albeit in different ways. Merlan (1998), in her study of post-colonial change
in Katherine, details the process of condensation and fragmentation of the landscape for
Jawoyn living through the transformational period of World War II. While older
generation Jawoyn would speak of movement between places by referring to intermediate
places, the younger (post-War) generation would speak into terms of “visits (mostly by
vehicle) to places conceived of as good spots in the landscape, rather than in terms of
their relationship to each other” (Merlan 1998, p. 107). The Aboriginal people of Kakadu
also spoke about country in terms of discrete locations without reference to intermediate location. However as we discussed travels through country in detail some, particularly Bessie, Violet and Judy, recalled an earlier era of walking across country and, in particular, through riparian areas to reach ‘good spots.’ For Violet and Judy, the experience of country as something “travelled through” rather than “travelled to” was a much more significant part of their lived experience, and their recollection of the pathways and places between sites was much more detailed than others who spent less of their life in country.

As for most Aboriginal peoples in northern Australia, pastoralism provided the only means for Jawoyn to assert connection to country and provide for their communities (Head 1994a). Pressure to assimilate and centralize Aboriginal communities removed many of the traditional owners of southern Kakadu from their homelands. Mining, until recently heavily concentrated in the southern regions of Kakadu, was an exclusive enterprise that had little need for Aboriginal knowledge or labor. The establishment of the Park provided for many Jawoyn the first opportunity to visit country that was of great significance but had been removed from day to day experience.

By contrast, the successful enterprises in the north of the Park – hide shooting and safari tourism – all required Aboriginal participation and knowledge. Moreover, the flexibility afforded by these enterprises allowed more autonomy by Aboriginal communities. That’s not to say that there weren’t significant cultural adaptations – both Murumburr and Ngomburr participated heavily in the cash economy, intermarried with Europeans, and adopted many aspects of settler society. Indeed the Cahills left the Alligator Rivers Region and didn’t return until Johnny Bell was deceased and the Park
was established. For the Aldersons, however, the Park meant a loss of autonomy and a new era of negotiation with European culture, one that was reflected on by Violet as a necessary compromise with increasing pressure to develop the region, “We had to have the Park, after we got that [Ranger uranium] mine.”

Today Jawoyn and Murrumburr (but not Ngomburr) are represented with other indigenous groups in Kakadu by a Board of Management that is intended to be the main instrument of joint management of Kakadu. Also sitting on the board are representatives of the tourist industry, a conservation scientist and two senior executives from the office of the Director of National Parks, a federal office (Wellings 1995). The Board is charged with directing Park policy which is implemented by a professional staff of predominantly European rangers. It operates under a stakeholder approach that is itself problematic for representing Aboriginal values and authority (Carter and Hill 2007, deKoninck 2007). In short, Aboriginal people, although given nominal land rights and significant autonomy within the Park, must contend with a bureaucratic and distinctly European social structure. Moreover, Park policy must be set within the wider Australian political context, where Kakadu is viewed more as a “National Park” and hence the domain of every Australian than as Aboriginal land. For example in 2004 an attempt to restrict access to Twin Falls, a popular tourist destination located in an area regarded by Aboriginal people as both sacred and dangerous, was undermined in the interest of appeasing to package tour operators and mollifying the perception that Kakadu was kowtowing to Aboriginal “special interests” (deKoninck 2007).

The difficulty of exerting Aboriginal control and power within the new institutional structures was also exemplified by the Brucellosis and Tuberculosis
Eradication Campaign (BTEC) in the 1980s. This was a campaign to eliminate the feral buffalo populations in Kakadu and elsewhere that were believed to be a reservoir for bovine tuberculosis, a threat to the cattle industry. By 1992, bolstered by improved infrastructure and with the support of Park staff, the campaign had virtually eliminated all of the buffalo population in Kakadu. Widely hailed as a success in European circles, the campaign was seen as a disaster by many Aboriginal people who had grown up with buffalo: “‘Those BTEC mob did not consider that buffalo has a history. … Some of us work with them [in shooting camps] on country. Plenty of meat on buffalo – feed everyone. He [buffalo] has an aboriginal name ... as long as not too many – buffalo can stay’” (Jacob Nayinggal as quoted in Robinson and Whitehead 2003 [brackets in original]). The campaign was carried out to meet the economic needs of the larger society, with little regard for, or communication with, the Aboriginal population (Robinson and Whitehead 2003). In the end, Aboriginal concerns were met by allocating a buffalo farm in the center of the Park to supply buffalo meat to Aboriginal residents, a move that seems to indicate that Park administrators at the time understood the Aboriginal perception of buffalo in terms of an economic resource rather than as a creature with a ‘history’ of belonging on country.

Ecological change along river courses embodies the social changes arising from depopulation and the eventual restructuring of daily life around outstations, roads and discrete visiting sites. However, these changes are poorly understood, perhaps because each “regime shift” in management has been replaced by another, muddying the historical waters. Buffalo reduced fuel loads, decreasing the likelihood and intensity of late season fires. The removal of buffalo occurred concurrently with the establishment of
the Park and the woodlands adjacent to riparian forests are today frequently burned by helicopter to create fire breaks (Price et al. 2007). These fires occur exclusively in the early dry season, and are generally of low intensity. Aboriginal started fires are today rare along creek lines – although abundant along roadsides, again reflecting the changing use of space.

These changing patterns of land use are reflected by patterns of change evident from the aerial photographic record. L. Nielsen and I. Dixon (*unpublished data*) analyzed a series of aerial photographs taken in 1950, 1964, 1984 and 2004 of the riparian sites detailed in Chapter 2. They marked the width of the riparian forest boundary by the stream margin at each time period, 1950-2004, and woodland cover at each time period, 1964-2004 (Fig. 4.5a-b). Overall, forest cover increased, particularly in the floodplain and lowland reaches of the South Alligator River (Fig. 4.5a). The Gimbat Valley didn’t change, most likely because of very high fire frequencies and frequent disturbance within an unstable channel. Woodland cover increased from 1964 to 1984, but then declined from 1984 to 2004 (Fig. 4.2b).

The expansion of riparian forest may be a signature of this long history of low intensity fires within the riparian zone. The increase in the width of closed forest vegetation may be explained by both decreased fire intensity at the forest boundary during the buffalo era and protection from fire by burning along creek margins after buffalo were removed. This could also explain why woodland vegetation expanded in the first era and then declined in the second. When buffalo were present they decreased fuel loads and fire frequencies in the woodland margins. Buffalo were removed
Figure 4.5: Comparison of vegetation change in riparian forest and woodland communities along the South Alligator River catchment. Changes were normalized within sites, and sites were grouped by region along a gradient that roughly follows from downstream to upstream (clockwise from upper left: floodplain, lowland, South Alligator Valley [SAV], and Plateau). Full details on site locations and regions are provided in Chapter 2. (a) Relative change in riparian forest width (I. Dixon, unpublished data). (b) Relative change in woodland cover (L. Nielsen, unpublished data).
Conceptions of landscape concurrently with the establishment of the Park, and fuel loads not only increased but riparian woodlands were specifically targeted by Park staff to create fire breaks.

PERCEPTIONS OF CHANGE

I have speculated some of the ecological changes that may have arisen in the immediate aftermath of European settlement, particularly from the initial holocaust that unravelled the social structure underpinning Aboriginal land management, and have described some changes in riparian forest structure during the post-war period. During my visits to stream sites with Aboriginal people I asked for their impressions of changes that have occurred during their lifetimes (summarized in Table 4.1). Our discussions ranged from specific ecological changes along watercourses to general issues of Park management and the new social and ecological terrain of the Park. Although there is a broad concurrence in views of ecological change between the groups, there are also significant differences, particularly in the causes of ecological change that I suspect reflect their different post-contact experiences.

When adjusted for sample size, Murrumburr, Ngomburr and Jawoyn were consistent in their assessment of ecological change across sites, although Jawoyn reported change less often than Murrumburr or Ngomburr (Table 4.1). The most commonly noted facet of change, and a point of concern, was change to flow regime and/or bank erosion. People were also particularly responsive to change in the abundance of animals, reflecting their importance as a food resource but also a sense that particular animals belonged in certain places – they “should be” or “should not be” there. For Jawoyn the loss of animals such as goannas (Varanus spp.), blue tongued skinks (Tiliqua spp.), and snakes was particularly acute as the cane toad (Bufo marinus) had recently established in
large numbers on their country. The cane toad is a highly toxic exotic pest that has a devastating impact on native predators (Phillips and Shine 2005). Apart from the impact of cane toads on native fauna, Murrumburr and Jawoyn differed in their perceptions of the causes of change (Table 4.1). Erosion and changes in fire regimes were perceived as major factors of change by the Jawoyn, while for Murrumburr, feral animals (typically pigs and buffalo) and tourism were the primary cause of change.

The Jawoyn were particularly concerned about changing fire regimes, particularly at some locations which they thought were burned too much. There was also grave concern about the use of helicopters in fire management:

Parks mob is now burning the grass, not the Traditional Owners. Because in the Park, they’ve got the technology. The technology, it overrides our way of doing things. Of course this is true, it’s there you can see it. All of our information has to come through technology now; at the press of a button (SW, a deceased Jawoyn man).

This statement is telling as it highlights a general perception by Jawoyn in particular but also Murrumburr and Ngomburr that the bureaucratic and technological approach of the Park is not an appropriate way of engaging with country, and that it undermines Aboriginal identity and authority. This became particularly clear to me from a statement by an Aboriginal Park ranger: “Burning from helicopter isn’t the Aboriginal way. It’s like getting a chook [chicken] from the supermarket rather than hunting for meat.”

Although too frequent fires were blamed for declining vegetation at some sites, many observed that there had been a general increase in woody vegetation (“thickening up”) across their country, making access to some places difficult. This was particularly reported by Murrumburr and Ngomburr on floodplain country, where they attributed vegetation thickening to the removal of buffalo. The general trend toward vegetation
Table 4.1: Responses to site-specific questions of ecological change. I interviewed 5 Jawoyn and 9 Murrumburr/Ngomburr traditional owners at 15 and 21 riparian sites, respectively, on their impressions of ecological change and the cause of that change. The number of responses indicating change in each of six categories is indicated in bold. Where a cause was given the cause and the number of times it was given is indicated in plain text below each category.

<table>
<thead>
<tr>
<th>Category</th>
<th>Jawoyn</th>
<th>Murrumburr / Ngomburr</th>
</tr>
</thead>
<tbody>
<tr>
<td>Erosion / Changes in Hydrology</td>
<td>7</td>
<td>11</td>
</tr>
<tr>
<td>Tourism</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Weather</td>
<td>1</td>
<td>1</td>
</tr>
<tr>
<td>Increased vegetation</td>
<td>4</td>
<td>7</td>
</tr>
<tr>
<td>Fire</td>
<td>1</td>
<td></td>
</tr>
<tr>
<td>Feral Animals</td>
<td></td>
<td>1</td>
</tr>
<tr>
<td>Decreased vegetation</td>
<td>3</td>
<td>6</td>
</tr>
<tr>
<td>Fire</td>
<td>2</td>
<td></td>
</tr>
<tr>
<td>Feral Animals</td>
<td></td>
<td>3</td>
</tr>
<tr>
<td>Floods / Hydrology</td>
<td>1</td>
<td>1</td>
</tr>
<tr>
<td>Increased animal abundance</td>
<td></td>
<td>3</td>
</tr>
<tr>
<td>(no cause given)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Decreased animal abundance</td>
<td>3</td>
<td>5</td>
</tr>
<tr>
<td>- Fire</td>
<td>1</td>
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<tr>
<td>- Feral Animals</td>
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<tr>
<td>- Floods / Hydrology</td>
<td></td>
<td>1</td>
</tr>
<tr>
<td>No change</td>
<td>2</td>
<td>6</td>
</tr>
</tbody>
</table>

thickening is belied somewhat by Table 4.1, where it appears that respondents reported as much decreased as increased vegetation at specific sites. However, where people referred to decreased vegetation they spoke about specific species that had disappeared or decreased in abundance. Increased vegetation almost always referred to a general increase in vegetation density. Vegetation thickening was also often mentioned in the course of discussions not related to specific sites.

Pig diggings are highly visible on the landscape and all Jawoyn, Murrumburr and Ngomburr I interviewed thought pigs were undesirable, particularly because they dig up yams. By contrast buffalo have a special place, both in memory and in the conception of landscape of many Aboriginal residents of Kakadu (Bowman and Robinson 2002,
The people most positive about buffalo were Judy Alderson, Violet Lawson and Bessie Coleman, all of whom had spent their childhood and young adulthood in Kakadu when buffalo were at their highest population (Fig. 3.4). All three were involved in the buffalo industry in one form or another.

Many Jawoyn were ambivalent about the ecological impact of buffalo. Roy Anderson’s response, where he referred to a small creek located near the Arnhem Land Plateau, was typical: “Buffalo wouldn’t damage country like this much. Mostly they’d rub their horns. Pigs do most of the damage. Buffalo tear up billabongs.” Jawoyn perceived a differential impact between buffalo living in the Plateau (and other upland streams) and the lowland floodplains. In lowland areas they were perceived as very destructive to billabongs and other waterways where they would muddy waters, destroy water lilies and other macrophytes and erode banks through wallowing, trampling and overgrazing. It is interesting to note that, prior to BTEC, buffalo were at their lowest population density in the Plateau region (Chapter 3).

Most Murrumburr, who generally had a more intimate association with buffalo than either Jawoyn or Ngomburr, believe that the removal of buffalo has caused several undesirable ecological changes, including vegetation thickening and more, hotter fires. Judy Alderson explained, “Buffalo keep the place even. Some areas are too thick, others are too clear – trees are getting burnt.” Other Murrumburr expressed the sentiment that by keeping fuel loads down and the country clear, buffalo did what the depopulated people of the northern floodplains couldn’t. Violet Lawson specifically stated that buffalo were “land managers.” Generally, the perception among both Murrumburr and
Ngomburr was that the removal of buffalo had allowed too much vegetation to grow on the floodplains.

This position might seem to imply a generational shift in attitudes toward buffalo. Yorky Billy Alderson is quoted as saying, during the height of the buffalo era when buffalo cleared the floodplains of vegetation and gullies and rills pocked the landscape:

The Wet is all right for them [buffalo], but in the Long Dry every year, from May to October and sometimes November, they must reckon they’re in the wrong country, too. They’re a damn pest, a menace. When I was a boy the Jim Jim was much better country than it is now; cattle won’t graze with buffs, and they’ve destroyed hundreds of miles of it (from Lockwood 1979, pp. 49-50).

However, I think their perspective is similar to that described by Rose (1994) for the Yarralin of Daly River: “They do not privilege objectivity, nor do they privilege the specific. The question of what ‘really’ happened is not a question of fact but a question of meaning.” For Judy and Violet, and indeed most Aboriginal people of Kakadu, buffalo do not effect ecological change because they have been incorporated into the socialized landscape, i.e. they are not ‘feral’. This was succinctly stated by Bessie Coleman, “To us, buffalo are like people.”

Jawoyn and Murrumburr/Ngomburr also differed in their perception of tourism, although there were some commonalities. Both expressed concern for tourists’ safety, as the death or harm of a guest in one’s own country is to be avoided at all costs. Also, both expressed frustration with tourists not acting properly on country, particularly with regard to sacred areas. However, tourists were generally seen as positive by the Jawoyn, who welcomed the economic opportunities provided by tour companies and the Park as well as the opportunity for Jawoyn to visit country as tour guides. There was also a sense of
pride in country and a desire for people to see their country and have a positive experience there.

Such sentiments weren’t entirely lacking amongst Murrumburr or Ngomburr, but there were more negative comments about tourism from both Murrumburr and Ngomburr than from Jawoyn. There was a sense that tourism and the infrastructure of tourism has impacted negatively both on their quality of life and on the physical environment. Tourism precluded access to several sites of great significance to Murrumburr and Ngomburr because they didn’t feel comfortable visiting when tourists were there. These include the Yellow Waters complex (Ngurrungurrudjba) as well as the Maguk plunge pool, a place mentioned by many as “one of our favourite places,” and a place that was only visited in the wet season when closed to public access.

Tourism was also believed by many Murrumburr to be responsible for downstream hydrological changes in Barramundi and Jim Jim Creeks. Erosion and sand infilling on Barramundi Creek are linked to the construction of a bridge over the Pine Creek highway crossing. On Jim Jim Creek, progradation of a billabong at Kingfisher was directly linked to a crossing further upstream along a very popular four wheel drive track that leads to Twin Falls. The construction of this road, and the increase in tourist numbers, was linked by Judy Alderson to Alan Stewart, who ran the safari operation at Patonga: “Alan Stewart came through here and brought more people to this country – that’s how Jim Jim Road was started.”

The perception of tourists as exotic to the landscape while buffalo belong may also explain an incident at a symposium at Kakadu National Park. I was to give a presentation on historical changes in the configuration of saltwater channels on the South
Alligator River floodplains – an intricate mixture of freshwater and saltwater systems that lies in the center of Kakadu. My talk would, in my mind, present convincing evidence that buffalo had a direct impact on the freshwater systems that were valued by tourists, Park staff, and the Aboriginal residents of Kakadu. Violet Lawson came up to me, visibly upset, and said, “Buffalo didn’t cause any changes in Kakadu. It was tourists who did that.” At the time the statement struck me as clearly normatively false, if not absurd. However, it is clear that Violet was trying to articulate to me a conception of the landscape where the biggest changes do not involve buffalo but rather the imposition of European settlement. For many Aldersons the transformation of space brought about by Park establishment and roads has radically reconfigured the landscape within living memory. This transformation has not generally been received positively, and Judy Alderson expressed the wish of many Murrumburr to see the conditions of their childhood return, “I hope time goes back, I really want to see them times again, olden days time.”

DISCUSSION

The process of colonization is as much perceptual as physical. As the colonizing society attempts to recreate the landscape in its own terms, it not only occupies and controls physical space, but also reconstructs of the conception of space and social organization in relation to space. In northern Australia, the creation of pastoral stations was predicated on the conceptual transformation of “empty” land to “productive” land on the part of European settlers (Morphy 1993). The introduction of Asian water buffalo, pigs (*Sus scrofa*) and other novel mammals to coastal settlements along the Top End was likewise an attempt at Europeanizing what to early settlers was a foreign landscape
Conceptions of landscape (Spillett 1972). However, in this regard the colonizer was not successful. Buffalo, who expanded within 100 years across most of the Top End, were quickly incorporated into the social configuration of the Aboriginal landscape, but, having failed in their civilizing mission, became a feral ‘problem’ to Europeans (Robinson 2005).

Keen (1984) states that what has been constructed under the land claim process as patrilineal obligations to demarcated territories (an obligation to place) is as much or more a social obligation to renew ties between groups who associate with different sites of significance spread across the landscape (also an obligation to place, but place as socially constructed). Seen in this light, the holocaust of first contact with Europeans would have radically restructured land/social relationships as many patrilineal groups went extinct, and others moved westward and concentrated around European settlements. The motive for visiting remote areas was reduced or eliminated. This had ecological consequences as well as fewer people concentrated into fewer places on the landscape fire regimes shifted such that areas distant from settlement had fewer fires occurring primarily in the late dry season while places close to settlement had more fires in the early dry season.

The different histories of the northern and southern regions of the Park have their root in physical differences in the landscape. Essentially, the floodplains of northern Kakadu were remote from settlements, difficult to access, and were favoured habitat for buffalo. This led to social differences in the relationship between Aboriginals and Europeans as Aboriginal people were an indispensable part of the labor force shooting wild herds of water buffalo for their hides. The rocky country to the south was less suitable for buffalo, but was closer to the railway and telegraph lines and more suitable
for cattle. It contained gold, uranium and tin deposits that fuelled various small scale mining rushes through the late nineteenth and much of the twentieth centuries. This set the stage for post-War displacement of the Jawoyn while the Murrumburr, and to a lesser degree Ngomburr, retained a strong experiential attachment to country.

The gradual condensation of the Aboriginal population around distinct communities likewise restructured Aboriginal spatial relationships, and motor vehicle travel simultaneously contracted the space of Aboriginal action while condensing the loci of action to distinct nodes on the landscape. The space that was “travelled through” shifted from riparian corridors to roads, while the condensation of space to distinctive “sites” created a stronger sense of place as “travelled to”; as a destination rather than as a place connected and related to other places (Merlan 1998). This shift also had ecological consequences, as fire frequencies increased along roads (Price et al. 2005, Petty and Bowman 2007) while Aboriginal use of fire and other forms of management in riparian areas declined.

The improvement and expansion of the road network, and later the concurrent recognition of Aboriginal land rights and establishment of Kakadu as a jointly managed national park signified very different changes to members of each group. To the Jawoyn, motor vehicles and the recognition of land rights signified by Park establishment provided renewed access to places that were highly significant within cultural memory, but with few personal associations. For the Murrumburr/Ngumbur these changes resulted in greater mobility which did result in a return to country for some, but also a loss of autonomy as they negotiated their lives within the framework of a national park.
The different histories and different conceptualisations of landscape between actors in jointly managed parks are seldom recognized. Many suggest the use of a “two tool-box” approach to successfully implement joint management projects (Altman and Allen 1992, Aslin and Bennett 2005). The presumption is that by combining two different empirical methodologies, one indigenous and the other scientific (more accurately, “positivist”), the outcome will be better natural resource management while engaging with and recognizing the needs of indigenous peoples. One difficulty with this approach is that it ignores the social constructions of landscape; the ‘tools’ that each culture brings to the table are only relevant within a particular conception of landscape. A positivist approach, such as I and others have adopted elsewhere in describing landscape change in Kakadu (Banfai and Bowman 2006, Chapter 3), looks at the landscape as a physical entity and uses documents such as aerial photographs, the distribution of vegetation size classes, and other measurable impacts on the physical landscape. One can ascribe mechanistic events to observed or inferred changes: vegetative thickening as a result of changes in the timing and frequency of burning, the increase and decline of saltwater bearing channels on floodplains as a result of the population explosion (itself a metaphor) and removal of buffalo, and so on.

An alternative conception, one that I believe is adopted by the Aboriginal people of Kakadu, and one that Park managers must come to terms with, is that landscape is produced through social action and interaction. Place is both a temporary locus of lived experience and an enduring reference to the identity of the self in relation to the group. The reconstruction of lived spaces and movement through the landscape that has resulted from colonisation and post-war development led to “the pulling away of … ideal
concepts of place or person linkage from the everyday experience of places” (Merlan 1998, p. 77). Landscape, the intersection between place and the conception of place, has both expanded to include places beyond the everyday experience of pre-contact Aboriginal people (Pine Creek, Darwin, Katherine) and changed within Kakadu from a web of places connected by riparian corridors to a series of distinct nodes connected by roads. Knowledge of and action on country accretes around discrete sites, while the roads themselves accrete meaning that would not have existed before. Thus, although the substance of cultural interaction with the landscape is similar, the construction and use of space within the landscape has changed.

Many of the conflicts arising from different perceptions of the landscape are realized in the contentious issue of the management of exotic mammals (DeKoninck 2005, Robinson et al. 2005). Exotic mammals are a cause of great concern among Park staff and the Park is charged, under the plan of management, to “[minimize] the adverse effects of domestic and feral animals on the natural and cultural values of the Park” (Anon. 2007, p.67). For Aboriginal people, the concept of ‘ferality’ itself is problematic. That is perhaps not surprising given the tendency of Aboriginal people to render distant historical events as synchronous and actualised in physical space (Morphy 1993, Rose 1994). Thus, animals which are present, are thriving on the landscape, and to which one has a strong affiliation and attachment, are not feral; they have a history and belong in country.

Europeans, however, refuse to operate within the Aboriginal system that defines landscape in terms of the social relationships between people and country, and defines social identity, behaviour and morality in terms of the relationship between people and
country (Rose 1992). The technological approach of European land management undermines the appropriate forms of interaction with country. Tourists behave recklessly and endanger themselves by not understanding the “rules” that govern social interaction with country.

By understanding these crucial conceptual differences, and by understanding landscape as physical, perceptual and social (Layton and Ucko 1999) we can begin to understand landscape change within Murrumburr, Ngomburr and Jawoyn contexts. There can be areas of agreement, for example aerial photographs may show a record of physical change that can be interpreted as changing fire regimes and concurrently a change in the relation of people to country (Fig. 4.5). Likewise there can be disagreement, feral buffalo are on the one hand effective land managers and on the other powerful agents of ecological change (Chapter 3). These differing views of landscape change are not contradictory if they are comparing different landscapes. For Aboriginal people, the greatest change to their landscape has been the submergence of their cultural space with that of a dominant society. This has resulted in a contraction of space and condensation of place to discrete sites, the appropriation of Aboriginal interaction with country by new technologies (some, such as the automobile, Aboriginal people have readily adopted, others, such as the helicopter, they are more sceptical of) and the physical appropriation of space.

In recognition of the implications of the Aboriginal Land Rights Act (Northern Territory) 1976, and the reality that Aboriginal people can contest the space now controlled by a settler society, the Northern Territory is currently engaged in a process of changing to a joint management framework for all of its parks. This process reflects a
wider global recognition of indigenous rights, concurrent with increasing pressure to preserve global “biodiversity hotspots” in the face of increasing environmental pressure (Myers et al. 2000). In the case of the founding of Kakadu National Park, the project of joint management for some conferred nominal land rights to people who had been denied recognition of tenure, but for others was an extension of colonization into a landscape that had previously resisted it.
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