

Forest mammals of northern Queensland: is their conservation status improving?

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ABSTRACT

Changes in conservation issues relating to forest mammals in northern Queensland over the past 12 years are assessed for the major forest types. Considerable advances have been made in our understanding of the biology of northern Queensland mammals and in their conservation at a local scale. The habitat of rainforest mammals is mostly protected by World Heritage listing and other legislative measures, the ecology and distribution of species in tall open forest and woodlands is much better known, but mammals in mangroves are mostly overlooked. However, broad scale major threats are now apparent. Global warming may catastrophically reduce the habitat of upland rainforest and tall open forest species, and climate change may also be an issue in lowland and drier habitats; a rise in sea level accompanying global warming may eliminate extensive areas of mangroves; and more intensive use of tropical woodland may lead to large scale declines of many native species in this habitat unless mitigating measures are taken.

Key words: forest mammals, conservation, impacts, northern Queensland.

Introduction

What has changed in northern Queensland in the 12 years since I last wrote about the conservation of forest fauna (Winter 1991a) for Lunney's (1991) compilation of papers on the subject of the conservation of Australia's forest fauna? What gains have occurred in the conservation of forest mammals and what losses? Are we seeing an overall improvement or is it one of decline?

I treat northern Queensland as that part of the state north of 20°S. Most of the area is covered with eucalypt forest and woodland types with extensive areas of paperbark *Melaleuca* and wattle *Acacia* woodlands, divided into western and eastern sections by the open grasslands extending south of the Gulf of Carpentaria (Fig. 1). I concentrate on the eastern section, which is the one I know best. Two major occurrences of rainforest exist along the east coast between Townsville and Cooktown (Wet Tropics bioregion) and in the McIlwraith/Iron Ranges area of the Cape York Peninsula bioregion (Fig. 1). Smaller patches of rainforest occur at the tip of Cape York Peninsula, the Lockerbie Scrub, and as small patches of riparian, interfluvial and dunefield vine forest right across northern Queensland. Mangroves are too limited in extent to map at a small scale other than as part of the littoral complex, which includes salt pans. However, substantial areas occur in the Hinchinbrook Channel and at the mouths of major rivers along the coastline of northern Queensland.

During my 31 years as a resident practising wildlife ecologist in far north Queensland I have seen huge changes in the presence and emphasis of both management and research in the region. Emphasis has shifted from the management of National Parks and wildlife as beauty spots and game to be hunted to management of biodiversity in its own right and of sustainable ecosystems. In the research area the number of resident researchers has grown exponentially within the Townsville and Cairns campuses of James Cook University, within CSIRO at Atherton and within various

Queensland Government departments. The rainforests of northern Queensland now have an international standing and through the Rainforest Cooperative Research Centre attracts national and international researchers. The savanna woodlands, the long-overlooked biome of the north, are now the focus of wildlife research through the Savanna Cooperative Research Centre. In addition there is a huge tourist industry in the north concentrated on the Great Barrier Reef and the rainforest, but with the savannas beginning to attract more visitors.

It is exciting and stimulating times to be living in the north because of the greater understanding of its unique fauna and flora. It is also a sobering time because of the greater realisation of the major issues we are facing if we are to conserve the natural ecosystems of the region. In this chapter I will not attempt to cover all faunal groups, let alone all mammals. Instead, I will use the species I know best, the possums and some ground dwelling mammals, to examine issues affecting forest mammals.

Rainforest

Wet Tropics bioregion

Conservation of rainforest had a major gain with the placing on the World Heritage List in 1988 of much of the Wet Tropics bioregion. It brought to an end the very controversial rainforest selective logging in extensive areas of State Forest. One side of the debate equated selective logging to mining of the rainforest with major implication on the conservation of its fauna. The other side argued that logging was sustainable over the long-term and equivalent to natural ecological processes in the forest with little effect on the fauna (Winter 1991b).

Rainforest now has a very high global profile, such that throughout northern Queensland the habitat is largely protected from clearing except on freehold land. Even

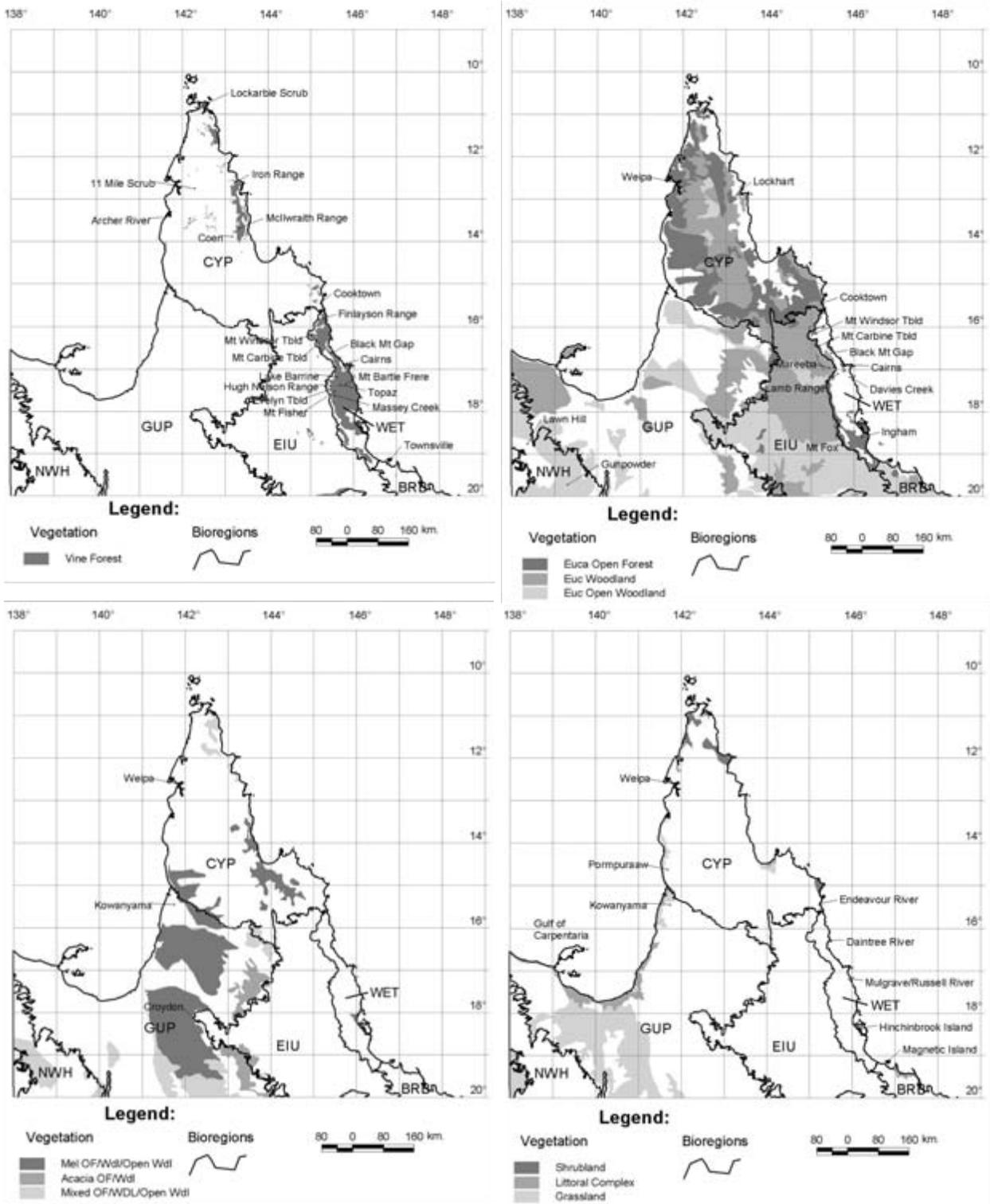


Figure 1. Major vegetation types of Queensland north of 20° S latitude and bioregions and places mentioned in the text: CYP – Cape York Peninsula, WET – Wet Tropics, EIU – Einasleigh Uplands, GUP – Gulf Plains, NWH – Northwest Highlands, BRB – Brigalow Belt. Bioregions after Sattler and Williams (1999); vegetation after Carnahan (1997).

on freehold land, declared endangered types, such as Type 5b (Tracey 1982) commonly known as *Mabi forest*, receive protection (EPBC ACT 1999). In addition, huge emphasis is placed on replanting schemes involving both state government departments (e.g. Department of Primary Industries, Environmental Protection Agency) and community groups (e.g. Trees for the Atherton and Evelyn

Tablelands, Tree Kangaroo and Mammal Group). Creek lines tend to be the target of such schemes on farmland with particular emphasis on maintaining connectivity through the landscape for arboreal species (Laurance and Laurance 1999), particularly icon species such as Lumholtz’s Tree-kangaroo *Dendrolagus lumholtzi* (Schmidt *et al.* 2000, Tucker 2000, Kanowski and Tucker 2002, Kanowski *et al.* in press).

On the negative side, clearing of rainforest on freehold land still persists which gradually reduces the remaining areas. The contention that *any rainforest on freehold land will eventually be cleared* is still the norm, except for the one protected type, i.e. Type 5b. Encouragingly, private individuals and trusts such as the Bush Heritage Trust, Australian Wildlife Conservancy and Australian Rainforest Foundation, are buying rainforest properties and placing conservation agreements on the title in a move to preserve rainforest on freehold land. In addition, many private landholders are placing conservation agreements over areas of rainforest on their properties. This practice is paramount in maintaining islands of relatively high rainforest biodiversity values within the mosaic of land developed for agriculture and rural residential blocks. Rainforest replantings by contrast are expensive, confined to a limited suite of rainforest species, usually only to trees, and take many years, probably decades, to approach the level of biodiversity of natural fragments (Catterall *et al.* 2004). Nevertheless, replantings are important for enhancing connectivity of the rainforest habitat mosaic in the altered landscape and are seen as an integral part of restoring connectivity of habitat for Lumholtz's Tree-kangaroo on the Atherton Tablelands (Kanowski *et al.* 2001a, Kanowski *et al.* in press). Doherty's Corridor between the Lake Barrine isolate of rainforest and continuous rainforest to the south-east is one of the best documented of such plantings with evidence that certain elements of rainforest fauna have begun to move between the original rainforest areas at each end after five years (Tucker 2000, 2001, 2002).

Most non-volant mammal species living in the rainforests of northern Queensland are not classified as threatened. The exception is the Spotted-tailed Quoll *Dasyurus maculatus*, classified as endangered in both state (QNCA 1992) and commonwealth (EPBC ACT 1999) legislation. Burnett and Marsh (2004) identified three ecological constraints that make the quoll particularly vulnerable. One constraint is its short life span (three years) and rigid breeding system, which places a requirement upon populations for annually low levels of extrinsic mortality and predictably high breeding success each year. Another is a low population number, estimated to be 550 individuals, divided into three core areas. Third is the opportunistic carnivory of quolls which exposes them to two new mortality factors, being killed by humans when the quolls raid chicken runs and almost certain death when ingesting the introduced cane toad *Bufo marinus*. Their wide ranging behaviour, at least 6100 m for males and 2450 m for females, means that a poultry yard has the potential to attract quolls from a wide area. With these constraints, the prognosis for the extinction of the quoll is high.

Looming as a major threat to the rainforest fauna of north-eastern Australia is climate change, particularly global warming which has the potential to drive the climate at a speed and direction possibly not experienced in the past. It is of particular concern to the Wet Tropics bioregion because of the high proportion of endemic mammals confined to the uplands (Winter 1984, 1997a, Williams *et al.* 1996). Of the ten mammals endemic to the region (Rusty Antechinus *Antechinus adustus*,

Atherton Antechinus *A. godmani*, Musky Rat-kangaroo *Hypsiprimnodon moschatus*, Bennett's Tree-Kangaroo *Dendrolagus bennettianus*, Lumholtz's Tree-kangaroo *D. lumholtzi*, Lemuroid Ringtail Possum *Hemibelideus lemuroides*, Green Ringtail Possum *Pseudochirops archeri*, Daintree River Ringtail Possum *Pseudochirulus cinereus*, Herbert River Ringtail Possum *P. herbertensis*, Masked White-tailed Rat *Uromys hadrourus*), only three extend their range down onto the lowlands (Musky Rat-kangaroo, Bennett's Tree-Kangaroo and Lumholtz's Tree-kangaroo), with the remaining seven confined to elevations above 300 m. With increased global warming, upland species will be restricted to higher and higher elevations, resulting in progressive fragmentation of populations with increased probability of local extinctions from stochastic effects and finally the disappearance of habitat at suitably cool temperatures (Hilbert *et al.* 2004, Williams *et al.* 2003, Krockenberger *et al.* 2003). Williams *et al.* (2003) estimate that with an increase of 1° C in annual mean temperature, one species of the 65 vertebrate species modelled will lose its core habitat and with an increase of 3.5° C all 65 species will undergo dramatic declines in distribution, with 30 of these species completely losing their core environment.

Any consideration of the effects of global warming on rainforest upland species in the Wet Tropics bioregions needs to be seen within the dynamic context of past climates. Since the height of the last glacial period, about 18000 years before the present, two significant contractions of rainforest have been postulated. The first, as a result of a cooler drier climate at the apex of the ice age, was a contraction of rainforest into two distinct blocks separated by sclerophyll forests of the Black Mountain Gap (Nix and Switzer 1991). This had a profound effect on the evolution of rainforest faunal species within the region (see Moritz *et al.* 1997 for overview) and is the most likely vicariant event leading to the divergence of the possum genus *Pseudochirulus* to two distinct species, the Herbert River Ringtail Possum and the Daintree River Ringtail Possum (Winter 1997a). However, it was a cooling event of some -3.5° C (Nix and Switzer 1991) and despite the overall contraction of rainforest, suitable habitat for cool-adapted species increased by expanding onto the coastal lowlands.

Of greater relevance to present global warming is the 2° C warmer and wetter climate postulated to have occurred 5000-3600 years before the present (Nix and Switzer 1991). Although rainforest expanded beyond today's limits, it was a warmer rainforest, and rainforest habitat suitable for cool-adapted species, such as the present day upland endemics, retreated to higher elevations than the present (Hilbert *et al.* 2004). It was a significant vicariant event for cool-adapted species and provides a best-fit model for the distribution of upland species, particularly for the absences of species from rainforest in the Finlayson Range at the northern extremity of the bioregion (Winter 1997a). It is this warm vicariant event that may have triggered genetic drift towards an unusually high proportion of a white colour morph in the Lemuroid Ringtail Possum on the Mount Carbine Tableland. It would have done this by isolating and reducing the population north of the Black Mountain Gap (Winter 1984, 1997a,

Trenerry and Werren 1993). These two vicariant events, one cool the other warm, are the bottlenecks through which present day rainforest species diversity of the Wet Tropics has passed and to which the relatively depauperate fauna of the region is attributed (Crome 1990).

Is there any evidence of upland species responding to global warming? Anecdotal reports of possum numbers declining in some areas may be a precursor of climate change, but the issue is complex as thickening of the forest canopy along the edges and roads, which are the most common sites for spotlighting, may be reducing visibility of the possums. Musky Rat-kangaroos have disappeared from Massey Creek at 1200 m in elevation, but still persist at lower altitudes (Winter pers. obs.) which seems a reverse situation to that expected, but could this reflect drier times or changes in climate in general and if so, is it merely a blip in the population which will return, or is it a precursor of a long-term trend. Golden bowerbirds *Prionodura newtoniana* appear to have abandoned their bowers in the Topaz area at the lower limit of their altitudinal distribution on the eastern escarpment of the Wet Tropics bioregion (S. Breeden pers. comm.) which is consistent with the prediction by Hilbert *et al.* (2004) that the bird's habitat in this area will be the first to disappear with climatic warming. Validation of these anecdotal observations requires properly designed studies to assess whether upland rainforest fauna is responding to long-term climatic changes.

So, why the concern that the current global warming will lead to species extinctions when a warming of two degrees may merely be returning the situation to that which occurred 5000-3600 years BP. A number of reasons are proposed as to why the present warming event differs from previous ones and why it is to the detriment of the upland cool-adapted fauna. The speed at which warming is occurring is greater than in the past (Haughton *et al.* 2001) thus allowing species less time to make adjustments to climate change either physiologically or ecologically. Clearing of rainforest by humans in the past 100 years has imposed an additional constraint on the ability of species to move in response to climate change (Winter *et al.* 1987). However, this only applies to upland areas of the Atherton and Evelyn Tablelands where large scale clearing has taken place above 700 m elevation (Winter *et al.* 1987). It does not apply to the other upland massifs, including the Mount Carbine Tableland or the Bartle Frere Range, both of which are major centres of biodiversity within the Wet Tropics region (Winter 1984, Williams 1997). The levels of carbon dioxide in the atmosphere have increased exponentially over the past 100 years (Haughton *et al.* 2001) with implications for the leaf-eating endemic rainforest ringtail possums of the Wet Tropics region because of reduced nutritional value and increased toughness of leaves (Kanowski 2001). In addition, increased water stress for upland species is expected from a rise in the average basal altitude of the orographic cloud layer (Pounds *et al.* 1999, Still *et al.* 1999, Williams *et al.* 2003), which in the Wet Tropics region of Queensland provides a significant contribution, as high as 30%, to the water entering the forest habitat (McJannet and Reddell 2001, 2002).

However, we must not forget that the global warming scenarios are just computer models, albeit based on best available evidence. Evidence for temperature increases over time is strong (Haughton *et al.* 1996), but the evidence used for predicting changes to rainfall patterns is much weaker (Walsh and Ryan 2000). Mean annual temperature is used as a surrogate for climate change in general and is therefore relatively coarse and emphasise the altitudinal upward creep of warmer temperatures. To date the models are at a regional scale and have not allowed for small variations influenced by within-regional topography (Hilbert *et al.* 2001) where moist gorges and the mountain side effect may buffer, though not prevent, the general temperature rise.

A number of solutions to the extinction problem of species appear to be gaining credence. One, gaining media coverage (see Australian Broadcasting Corporation's *The World Today* 31 July 2003 on the web), is to move species south to cooler climes. This totally ignores any ecological repercussions on the existing biota of more southerly rainforests with the introduction of a new species. A modification of the relocation scenario that has greater merit, is the implanting of an upland tropical forest, not just a few elements of its fauna, in more southerly latitudes (R. Russell pers. comm.). This would be a staged process beginning with tree planting in an area presently devoid of, but suitable for rainforest, and culminating with the transfer of elements requiring more mature forest, such as the Lemuroid Ringtail Possum. The extent of the implant would need to be in the order of 1750 ha, which is the area of the smallest naturally occurring upland rainforest isolate (Herberton Range) within the Wet Tropics bioregion that supports a full complement of rainforest possums (Winter 1984). The lead time to attaining at least the beginnings of a mature phase rainforest is 50-60 years (Hopkins 1978, Winter *et al.* 1991), which is faster than the time scale of global warming, and would require to be initiated within the next decade.

Another high profile solution proposed is the establishment of captive colonies of species to be saved, which often ignores the well known phenomenon of progressive genetic degradation of captive species (Woodworth *et al.* 2002), thus probably negating this as a solution because of the long time span of global warming before a return to temperatures compatible for the species. Preservation of a species' genes in gene banks with the intention of reconstituting the species once favourable conditions return is another preservation scenario which obtains high media coverage. One could argue that it at least preserves genetic material and even if conditions favourable for the original species do not return, new species could be created.

The relocation and high-tech solutions are desperate measures designed to save only a minute fraction of the upland rainforest biodiversity. Even an attempted full forest transplant will not preserve the full suite of species from microbes to higher vertebrates. Thus to pursue these measures is to accept a major loss of biodiversity. They are appealing in being high profile and provide media opportunities for politicians to give the impression of the government actively combating the loss of species diversity. They divert attention, however, from the much harder decisions regarding combating global warming through changes to human activities and life style.

The most ecologically sound solution is to take measures that reduce, even reverse, the rate of global warming and to buffer the effects for as long as possible by management of existing forests. Within the Wet Tropics region, a high priority management option would be to identify potential refugial areas and to reduce impacts on these areas or to enhance them by appropriate management options. Webb and Tracey (1981) identified deep gorges and mountain-tops as rainforest refugial areas, areas in which rainforest has continued to flourish even in adverse conditions. Hopkins *et al.* (1993), on the basis of charcoal found within rainforest, have suggested that the refugial areas may have been smaller than generally expected and possibly limited to valleys and their lower slopes. Nevertheless, the presence of upland specialist rainforest species, such as the endemic rainforest ringtail possums, indicate that rainforest has had a sufficiently long continuous presence for these species to survive (Winter 1988). Current climatic and vegetation modelling is sufficient to identify the upland cool refugial areas. With refinement, it may also be able to determine whether deep valleys and steep mountain slopes contribute to refugia.

A suite of cool-adapted endemic species, presently restricted to higher elevations, characterises the terrestrial vertebrate biodiversity of the Wet Tropics (Winter 1988, Williams 1997). This assemblage can be attributed to the bottleneck effect at the height of the most recent glacial period during which the warmer forests and their associated fauna were obliterated from the region (Nix and Switzer 1991, Winter 1997a, Hilbert *et al.* 2004). It is this upland, cool-adapted suite of species that is at greatest risk from global warming through the declining area of habitat as it becomes progressively more restricted to mountain tops. Managing for the continuing survival of this suite of species for as long as possible within the region comes down to managing cool refugial areas. Both mountain tops and gorges can act as cool refugial areas. Mountain tops are refugial because of their cooler temperatures at higher elevations. Gorges act as refugia from higher temperatures and drier conditions because of the cooling effects of shorter sunlight hours, cool air flowing into them from higher elevations, the protection they provide from drying winds and increased humidity from water courses.

Within the Wet Tropics most of the high elevation refugial areas are relatively undisturbed by clearing of rainforest. Human impacts are restricted to roads, television towers and walking tracks and careful management of these can ensure that impact is minimal. An important upland refugial area, that has suffered as the result of substantial clearing of rainforest for farming purposes above 900 m in elevation, is the Evelyn Tableland and the slopes of the adjacent Hugh Nelson Range and Mt Fisher (Winter *et al.* 1987). It lies within the Atherton Upland subregion, which has the highest species richness of endemic vertebrate fauna for the Wet Tropics region (Williams 1997) and represents a high proportion of the subregion above 900 m in elevation. Furthermore, much of this upland area is on the higher fertility basaltic soils, which are known to support higher densities of arboreal marsupials such as the rainforest possums and Lumholtz's tree-kangaroo (Kanowski *et al.* 2001b).

It is within the Evelyn Tableland area that the greatest improvements can be made to the ecological sustainability of the area for cool-adapted species. A high priority can be placed on converting all cleared land above 900 m in elevation back to rainforest. If this were to take effect, the Mt Fisher/Hugh Nelson Range/Evelyn Tableland would provide a substantial refugial area for cool-adapted species. The manner in which this was done would depend largely, but not entirely, on buy-back schemes particularly if there is a shift in private industry from the present dairy industry to others such as eco-tourism, timber production and carbon sequestration. There are programs, such as the Community Rainforest Reafforestation Program, Wet Tropics Region, Queensland (Commonwealth Department of the Environment and Heritage 'Farm Forestry for Green and Gold' program), which encourage the planting of trees as an integral part of property management. However, I am not aware of any schemes in which trees are the prime industry on a property.

While cool refugial areas, such as the Evelyn Tableland, may completely disappear with a sufficiently high rise in global temperatures, their management as refugial buffers to the global warming effect, would thus prolong the survival time of cool-adapted species. It is possible that this local action could prolong the life of the cool refugial area to a time when global warming might be reversed by global conservation actions.

In addition to the management of refugial areas, a high priority is to understand how the fauna reacts to its environment and how it might respond to the postulated changes to global climate. Studies on temperature strategies of Green Ringtail Possums (Krockenberger 2002), biodiversity models (Williams and Pearson 1997), rainforest vertebrates as seed dispersers (Dennis *et al.* 2002), autecology of the Musky Rat-kangaroo (Dennis and Marsh 1997, Dennis 2002a), responses to fragmentation (Laurance and Yensen 1991, Laurance 1990, 1994, Laurance and Laurance 1996), ecological correlates of arboreal mammals (Kanowski *et al.* 2001) and biogeography (Winter 1997a) all contribute to our understanding of mammalian biology and ecology of rainforest species. However, there is a need for properly constituted studies focusing on the responses of species to climatic changes.

Cape York Peninsula bioregion

On Cape York Peninsula, clearing of rainforest is not a major issue as most of it is within protected areas or is restricted from clearing by state legislation. Global warming will have an immediate impact on a small area of rainforest above about 500 m in elevation on the McIlwraith Range. No mammals are restricted to this upland area, but it contains three upland endemic frogs, the Cape York Nurseryfrog *Cophixalus peninsulae*, Northern Nurseryfrog *Cophixalus crepitans*, Long Snouted Treefrog *Litoria longirostris*, an upland endemic leaf-tail gecko *Orraya occultus*, and an upland population isolate of Lewin's Honeyeater *Meliphaga lewinii*. Global warming will have a lesser impact on the rainforest mammals of Cape York Peninsula, particularly those known to occur further north on the lowlands of New Guinea, such as the Common Spotted Cuscus *Spiloglossus maculatus* and Common

Southern Cuscus *Phalanger intercastellanus*, because they are presumably preadapted to warmer temperatures.

Of particular interest is a series of interfluvial vine forest patches, up to 400 ha in size though usually 20-100 ha, spread across Cape York Peninsula north of the Archer river and usually on lateritic surfaces between watersheds (see O'Neill *et al.* 1988 for map). They are remnants of a once more-extensive vine forest, the most obvious sign of their contraction being disused megapode mounds in adjoining woodland (Stocker 1971). They contain a suite of rainforest vertebrate species, most of which have good powers of dispersal across intervening woodland, with the possible exception of the Common Spotted Cuscus and the Red-legged Pademelon *Thylogale stigmatica* which are known from a few patches only (J. Winter unpublished data). These patches provide an opportunity to examine the ability of the contained fauna to survive in such a fragmented system and whether the fauna is a remnant one from a once more widely-dispersed habitat or one with powers of dispersal between patches. The pademelon has been recorded from four patches over the past 30 years, two of them at high densities. In at least one patch, the Eleven Mile Scrub north of Moreton, the pademelon has declined from abundant in 1973 and 1975 to apparently absent in 1993 and one sighting in 2003 (J. Winter unpublished data). Is this a case of local stochastic extinction and recolonisation or contraction of the population to extremely low density difficult to sample? Will it recover to the high densities recorded in the 1970s? A meta-population study of the pademelon in this naturally fragmented landscape, together with genetic markers, should give us the answer.

Mangroves

Mangroves line the Queensland coast with substantial concentrations in large estuaries or bays north of Rockhampton, but our knowledge of their mammalian fauna has progressed little over the past twelve years. The most systematic surveys of mangroves have targeted the False Water-rat, now more positively called the Water Mouse *Xeromys myoides* in south-eastern Queensland (Van Dyck 1996) and the Central Queensland coast (D. Ball pers. comm.). These surveys have established the mouse's distribution at a number of localities along the Queensland coast south of the Proserpine district, just outside the area considered in this chapter. The Australian distribution of this species is shown as two regional populations, one in south-eastern Queensland, the other along the coast of the Northern Territory (Woinarski *et al.* 2000). However, it seems inconceivable that the mouse does not also occur round the coastline of Cape York Peninsula and Van Dyck (2000), based on habitat similarity, considering they are probably in the Cairns area. A current project, run by the Far North Queensland branch of the Wildlife Preservation Society of Queensland, has found unconfirmed signs of the animal's presence along the Wet Tropics coast (S. Clague pers. comm.). Unfortunately, clearing for sugar cane farming has destroyed large areas identified as potentially original habitat for the mouse within the Wet Tropics region (S. Clague pers. comm.).

Previously (Winter 1991a), I reported on a population of Common Ringtail Possum *Pseudocheirus peregrinus* and Common Brushtail Possum *Trichosurus vulpecula* living in mangroves from the stretch of coastline between Shoalwater Bay and Proserpine and suggested that these two species did not occur in mangroves further north. This may still hold true for the Common Ringtail Possum as I know of no observations of this species in mangroves north of Mackay. There are, however, records of Common Brushtail Possums from mangroves along the much of the northern coast. There is a population found living, or at least foraging, in mangroves on Magnetic Island, off Townsville (W. Foley pers. comm.). In the course of spotlighting for crocodiles along the northern rivers, Common Brushtail Possums have been sighted in mangroves on the east coast of Cape York Peninsula at Lakefield and on the western side of the Peninsula at Weipa, Pormpuraaw and Kowanyama, but only as very occasional sightings (M. Read pers. comm.). The low numbers likely to be encountered in the more northerly estuaries is emphasised by the lack of records for this species in a preliminary fauna survey of three northern mangrove estuaries, Russell/Mulgrave, Daintree and Endeavour Rivers (Trenerry 1991).

S. A. Dalla Pozza and W. J. Foley (pers. comm.) examined the diet of the Common Brushtail Possum in mangroves on Magnetic Island. They found clear evidence that animals living in mangrove forests took in significantly more water than conspecifics in sclerophyll forests and that the water:sodium ratio within the mangrove possums was not significantly different from that found in the woodland possums, despite the higher sodium content of mangrove leaves. They also obtained a higher trapping success ratio within the mangroves than in neighbouring eucalypt woodland, suggesting that mangroves are a preferred habitat, possibly because of the higher water availability within the mangrove foliage.

The most obvious threat to mangrove communities is the loss, often misleadingly called reclamation, of the littoral zone for cane farming, aquaculture and marinas. In northern Queensland, such clearing is currently confined to the Wet Tropics region. Apart from some high profile, very localised, projects, virtually nothing is known about the impacts of this disturbance and destruction of habitat on the mangrove fauna, which highlights our poor knowledge of this suite of fauna in northern Queensland.

Another, more widespread threat to the mangroves is the rising sea level associated with global warming. This has the potential to obliterate mangrove communities, particularly where they are prevented from moving inland, to keep pace with the rising sea level, when blocked by agricultural and residential developments or by steep topography (Hughes 2003).

Tall open forest and moist woodlands

Tall open forest (Tracey 1982 Type 14) and moist woodland (Tracey 1982 Types 16a-g, 16p) are primarily adjacent to rainforest in the Wet Tropics bioregions with minor occurrences associated with the rainforests of the McIlwraith/Iron Ranges on Cape York Peninsula (McIlwraith Range 1:100 000 vegetation map). Three

of the four threatened non-volant mammals in northern Queensland, listed as threatened by both the state (QNCA 1992) and commonwealth (EPBC ACT 1999), are confined to this habitat. They are the endangered Mahogany Glider *Petaurus gracilis*, the endangered Northern Bettong *Bettongia tropica* and the vulnerable Yellow-bellied Glider *Petaurus australis*. All three have major issues affecting their conservation.

The issue for the Yellow-bellied Glider is the loss of habitat through natural processes. The glider is confined to a narrow band of tall open forest, also called wet sclerophyll forest, which in northern Queensland is mainly restricted to elevations above about 800 m along the western fringes of the Wet Tropics rainforest. Nowhere is the forest band greater than 5 km wide, usually about 2 km wide and can be as little as one or two trees wide (Harrington and Sanderson 1994). The continuity of the glider's habitat is interrupted by gaps of varying width, but with major discontinuities dividing the glider population into three sub-populations (Winter 1997b). The attenuated nature of the glider's habitat and the natural fragmentation of the population are the prime reasons for its listing as vulnerable (Maxwell *et al.* 1996) in the State and National legislation. The glider's habitat within the Wet Tropics represents an isolated occurrence of a type of forest that is widespread in south-eastern Australia. It is isolated from the next nearest occurrence to the south, on the Clarke Ranges in the Mackay hinterland, by a gap of 400 km and is the habitat for plants and animals that have a relatively wide distribution in south-eastern Australia and a small isolated population in the Wet Tropics region. The Yellow-bellied Glider is the mammal with a distribution most closely matching this pattern of distribution (Russell 1995, Winter 1997b, Queensland Electricity Commission 1991). The Wet Tropics population of the Yellow-bellied Glider is dependent on two tree species which in northern Queensland are characteristic of, and confined to, wet sclerophyll forest. The Rose Gum *Eucalyptus grandis* provides 90% of dens for the glider (Queensland Electricity Commission 1991), possibly because its hollows are sufficiently large to accommodate a family group of the gliders. The Red Stringybark *Eucalyptus macta* (previously known as *E. resinifera*) is the only tree in northern Queensland tapped for sap by the glider (Russell 1995, Winter 1997b, Queensland Electricity Commission 1991).

The greatest threat to Yellow-bellied Glider habitat is its eventual displacement by rainforest (Harrington and Sanderson 1994). They estimated that, during the 50 years from 1943-5 to 1991-2, 70% of the tall open forest dominated by Rose Gum and 50% of the tall open forest dominated by mixed species (Rose Gum, Red Stringybark and others) has been captured by rainforest. While emergent eucalypts continue to be a feature of the invading rainforest for many decades to come, the closed rainforest canopy prevents regeneration of the light-loving wet sclerophyll forest species. Although the gliders may continue to use emergent Rose Gum trees within the rainforest as dens, and may feed off the blossom of some rainforest trees (Queensland Electricity Commission 1991, Quin *et al.* 1996), their preferred habitat of wet sclerophyll is eliminated.

Capture of wet sclerophyll forest by rainforest will in time reduce and fragment the glider's habitat, thus making the glider more vulnerable to local extinctions, particularly as there is no indication that wet sclerophyll forest is expanding its distribution away from rainforest in response to changing fire regimes. With the percentages documented by Harrington and Sanderson (1994) of habitat captured, a reassessment of the glider's conservation status is likely to degrade it from Vulnerable to Endangered (Winter 1997b).

Fire maintains the sclerophyll and grassy nature of wet sclerophyll forest and determines its dynamic boundary with rainforest (Harrington and Sanderson 1994, Hopkins *et al.* 1993, Unwin 1989). Changes to the fire pattern since the arrival of Europeans is thought to have enabled the rainforest to invade the wet sclerophyll (Harrington and Sanderson 1994). Reduced fuel loads as a result of grazing and burning earlier in the dry season could lead to lower intensity fires, particularly in the wettest parts of the system which is the most susceptible to rainforest invasion (Harrington and Sanderson 1994). In addition, a road system which often runs lengthways along the narrow bands of wet sclerophyll forest provide effective fire breaks to fires originating in drier woodland to the west, thus hastening rainforest invasion from the east.

Fire, apart from halting the invasion of wet sclerophyll forest by rainforest, also results in the destruction of large Rose Gums. Successive fires can result in the formation of fire scars at the base of larger trees, leads to a chimney effect, and to the eventual collapse of the tree. The loss of large Rose Gums has a potentially high impact on the Yellow-bellied Glider population because of the importance of these trees as preferred den sites. The loss of large Red Stringybarks may not be as threatening to the gliders, because they are capable of sap tapping smaller trees less affected by fire scars. The dynamics between fire, rainforest and sclerophyll den trees is a complex issue that is not fully understood, and is one that needs to be addressed to conserve the gliders in the Wet Tropics region.

It is generally recognised that proactive management is required to maintain tall open forest and that the crucial management practice is to halt the invasion of rainforest primarily through the use of fire (Harrington and Sanderson 1994, Williams 2000, Queensland Environmental Protection Agency fire plans). At least half the tall open forest is within the World Heritage area under the management of the Queensland Environment Protection Agency, but inertia over burning prevails for two reasons. There is a strongly held perception by the general public that burning of rainforest, even along its edges, is undesirable because of the high profile afforded to rainforest protection. Consequently, when fire does occur, the agencies are castigated for either initiating it or not attempting to confine it once started. The Environmental Protection Agency is also nervous to use fire in tall open forest close to habitation because of the risk to property. Furthermore, fire entering the tall open forest from grassy woodlands to the west is much reduced, because of changed grazing practices associated with increased stocking rates,

which reduces the fuel load on the ground. However, in more remote areas where there is little danger of damage to buildings, the Queensland Environment Protection Agency has initiated a proactive fire management plan with associated monitoring of changes.

Another issue that has the potential to affect the conservation of the Yellow-bellied Glider is its taxonomic status, because conservation funding can be dependent on an animal's degree of taxonomic separation. Currently, this is not an issue because at all three levels of listing, international in the Red Data Book (Baillie and Groombridge 1996), nationally by the Australian Commonwealth (EPBC ACT 1999) and locally by the Queensland (QNCA 1992), the Wet Tropics glider population is recognised as a distinct conservation entity. However, a clarification of the glider's taxonomic status could strengthen the case for treating the Wet Tropics population as a separate entity if it was shown to be genetically distinct from populations further south (Moritz 1994). The nearest population is on the Clarke Ranges in the McKay region, a gap of 400 km. Russell (1995) identified the Wet tropics population as a distinct subspecies, *P. a. reginae*, but did so erroneously because the type locality of *P. a. reginae* is Gin Gin in south-eastern Queensland and the subspecies is considered to extend as far south as Coffs Harbour, New South Wales (McKay 1988). The Wet Tropics population may indeed be a sub-species because of its isolation, but the possibility exists that the three wet Tropics subpopulations may be equally separated genetically from each other as they are from the Clarke Ranges population (Winter 1997a). My contention is that the three sub-populations within the Wet Tropics may have become isolated from each other by the same vicariant event which lead to the isolation of the Wet Tropics population from that further south through loss of their wet sclerophyll habitat post the last glacial maxima.

Finally, global warming will reduce the yellow-bellied glider's core habitat, in a manner similar to that of the upland rainforest species (Williams *et al.* 2003), because the glider is a cool-adapted species confined to altitudes above about 800 m.

The overriding conservation issue for the endangered Mahogany Glider, like the Yellow-bellied Glider, is loss of habitat, but from direct human induced loss rather than natural loss. Habitat preference of the glider is eucalypt woodland with an open canopy, poorly developed understorey and the dominance of Myrtaceae species, including Clarkson's Bloodwood *Corymbia clarksoniana* and the Poplar Gum *Eucalyptus platyphylla* (Jackson 2000). The glider's geographical range is confined to elevations below 200 m between Tully and 30 km south of Ingham, a north-south distance of 108 km and east-west distance of 2-12 km, with a maximum available area of approximately 87 200 ha of potential habitat (Jackson and Claridge 1999). Most of the glider's original habitat was on the coastal plain and it has been severely fragmented by clearing for agriculture, pasture and plantation forestry, which has reduced it to approximately

20% of its original extent (Van Dyck 1993, Queensland Parks and Wildlife Service 2001). Only about 20% of the remaining habitat is in protected areas and much of the rest is at risk from further clearing (Queensland Parks and Wildlife Service 2001). A population viability analysis identified that a minimum area of 8000 ha containing 800 individuals is required for the long-term viability of the population (Jackson 1999). A Recovery Plan aims to manage a minimum of 3-4 populations for their long-term viability, but recognises that the survival of the species also requires habitat management outside protected areas to prevent the fragmentation of the glider population into isolates vulnerable to local extinction (Queensland Parks and Wildlife Service 2001). Because 80% of the glider's remaining habitat is on freehold or leasehold land, obtaining cooperation from landholders is imperative for the glider's long-term survival.

The conservation issues affecting the endangered Northern Bettong are more obscure than those affecting the two threatened gliders. Like the Yellow-bellied Glider, the bettong is distributed along the western margin of the rainforest within the Wet Tropics region, and its population is naturally subdivided into four known sub-populations (Winter and Johnson 1995, Laurance 1997, Winter 1997c, McIlwee and Freeman 1998, Dennis 2002b) with an unconfirmed fifth sub-population in the Ravenshoe district (Winter 1997c). The bettong's habitat includes tall open forest but, unlike the glider, it extends further down the moisture gradient into moist woodland often characterised by the Scented Gum *Eucalyptus citriodora* above about 700 m in elevation (Laurance 1997, Winter 1997c). There is no clear distinction between woodland inhabited by the Northern Bettong and that from which it is apparently absent (Laurance 1997, Winter 1997c). The only record of the bettong outside of the Wet Tropics region is of the original specimen collected in 1884 at or near Coomoolaroo, inland from Rockhampton (Collett 1887, Wakefield 1967). In recent times, the only bettong recorded in this area during fauna surveys has been the Rufous Bettong *Aepyprymnus rufescens* (T. Pulsford pers. comm.). However, no targeted surveys have looked for *B. tropica* between the original record and its present known distribution, although bioclimatic modelling has revealed potentially suitable climatic conditions exist in the intervening area (Winter 1997c).

A further complication is the significant difference in population densities between the population isolates. Highest densities occur on the Lamb Range and have done so consistently from the commencement of trapping for the species in 1985 to the present time, with trapping success ranging between 7 and 15% at Davies Creek (Winter 1997c, J. Winter and P. Latch unpublished data). In apparently similar habitat on the Mt Windsor Tableland and Mt Carbine Tableland, population densities are significantly lower with trapping success at 1% or less (Winter 1997c). On the Mt Windsor Tableland the bettong has not been recorded since 1989 despite two concentrated trapping efforts in 1991 (Winter 1992) and 1994 (Laurance 1997), but it

is too soon to conclude that the bettong has become locally extinct on Windsor. It is hoped that camera-traps, now used by Queensland Parks and Wildlife Service to monitor low density bettong populations (Dennis 2002b, P. Latch and S. Burnett pers. comm.), will confirm its continued existence on the Mt Windsor Tableland. Currently there is no explanation for these differences in population densities at the separate localities.

The sporocarps of hypogeous fungi (truffles) are the primary food of the Northern Bettong (Johnson and McIlwee 1997), comprising approximately 60% of the diet during all seasons (Vernes *et al.* 2001). The remainder of the diet consists mostly of subterranean vegetation such as the rhizomes of Cockatoo Grass *Alleteropsis semialata*, lillies (*Hypoxis spp*) and ground orchids (Johnson and McIlwee 1997, Vernes *et al.* 2001). It is the availability of truffles which may be the most critical habitat feature limiting the bettong's distribution (Dennis 2002b). Johnson and McIlwee (1997) suggested that a population density decline, while individual well-being did not decline, towards the limits of the bettong distribution, was indicative of a decline in its food resource. However, truffles are extremely difficult to detect which poses significant problems for measuring their seasonality and diversity (Dennis 2002b). The use of sniffer dogs to detect truffles, already used on commercial truffle farms in Tasmania, has been suggested as a technique to assess their availability in the wild (Dennis 2002b). Pilot trials within Northern Bettong habitat have proved successful, but the expense of maintaining or hiring trained sniffer dogs is proving an obstacle (P. Latch pers. comm.). Other mammalian truffle eaters, sympatric with the bettong, and therefore potential competitors, are the Northern Brown Bandicoot *Isoodon macrourus* and the White-tailed Rat *Uromys caudimaculatus* and some fine-scale habitat partitioning is exhibited by the three species (Vernes 2003). The Northern Bettong overlaps marginally with the Rufous Bettong, another truffle eater (McIlwee 1994, Johnson and McIlwee 1997), but the zone of overlap is narrow (Carpenter *et al.* 1993). This allopatric distribution of the two bettongs is considered to be a function of their ecological differences rather than one of competitive exclusion (Winter 1997c, Johnson and McIlwee 1997, Laurance 1997).

The role of fire is considered crucial in the management of grassy woodland in northern Queensland, both as a fuel reduction measure and to maintain a productive grassy ground cover for cattle grazing. Kangaroo Grass *Themeda australis* is the dominant grass in areas occupied by the Northern Bettong and it senesces if unburnt for a period of years (Morgan and Lunt 1999). In the tropics, the period at which this is reached is approximately four years (Mott and Andrew 1985). Several studies have indicated that the availability of some species of hypogeous fungi are influenced by fire (Christensen 1980, Taylor 1991, Johnson 1995). Vernes *et al.* (2001) have shown that on the Lamb Range the consumption of these fungi does not differ significantly between burnt and unburnt sites, but there is a significant increase in the quantity of one genus, *Mesophellia*, consumed at the

expense of other genera in burnt areas. This is consistent with increased time spent by the bettongs foraging in recently burnt areas (Vernes and Haydon 2001). There is some evidence that *Mesophellia* fruiting bodies are stimulated by fire (Christensen 1980, Taylor 1991, Johnson 1995). Vernes *et al.* (2001) suggest that the Northern Bettong is adapted to the common occurrence of fire within its habitat because of its ability to utilise this genus of fungus at a time when other resources could be scarce. What is not certain is whether the bettong has an obligate or facultative dependence on post-fire flushes of *Mesophellia* fruiting for long-term maintenance of its population. Vernes *et al.* (2001) propose a precautionary approach to fire management of bettong habitat by continuing a burning regime of fires every 3-4 years, which is in accordance with the regime adopted by the local grazing industry.

To obscure the issue of Northern Bettong conservation further, is its phylogeography which is not consistent with known present-day geographical or habitat discontinuities within the Wet Tropics region. Based on mitochondrial DNA the species can be divided into a northern and southern clade with an overlap zone on the Lamb Range to form a continuous population (Pope *et al.* 2000). The two divergent clades of the bettong are consistent with many other species within the Wet Tropics many of which have distinct genetic population north and south of the Black Mountain Gap (Moritz *et al.* 1997, Schneider *et al.* 1998) as a result of past wet to moist habitat contractions during and prior to the Pleistocene (Nix and Switzer 1991). However, there are two main differences identified by Pope *et al.* (2000). The first is that the degree of divergence between the north and south clades is more recent than for the rainforest taxa which these authors suggest is indicative of greater historical connectivity of wet sclerophyll habitat during the Pleistocene. Second, the position of the break between the clades is south of the usual Black Mountain break identified for rainforest species and in what is now continuous suitable habitat. Pope *et al.* (2000) propose that *B. tropica* colonised the Lamb Range quite recently, within the past 5000 years, in response to recolonisation of the range by suitable wet sclerophyll habitat as a result of climate changes since the height of the most recent glacial maxima.

Dry sclerophyll

Since I last wrote on this topic (Winter 1991a) the situation in the dry sclerophyll woodlands of northern Queensland is both better and worse. Better, because more is known about the mammals of the woodland as a result of several studies concentrating on this habitat. Worse, because there appears to have been a degradation of the habitat with a decline of some mammal species, which is little understood. Large scale clearing of habitat is not presently the issue, as the main clearing front has not reached northern Queensland (Fig. 2). Rather it is a subtle change in the habitat, which still retains its tree cover.

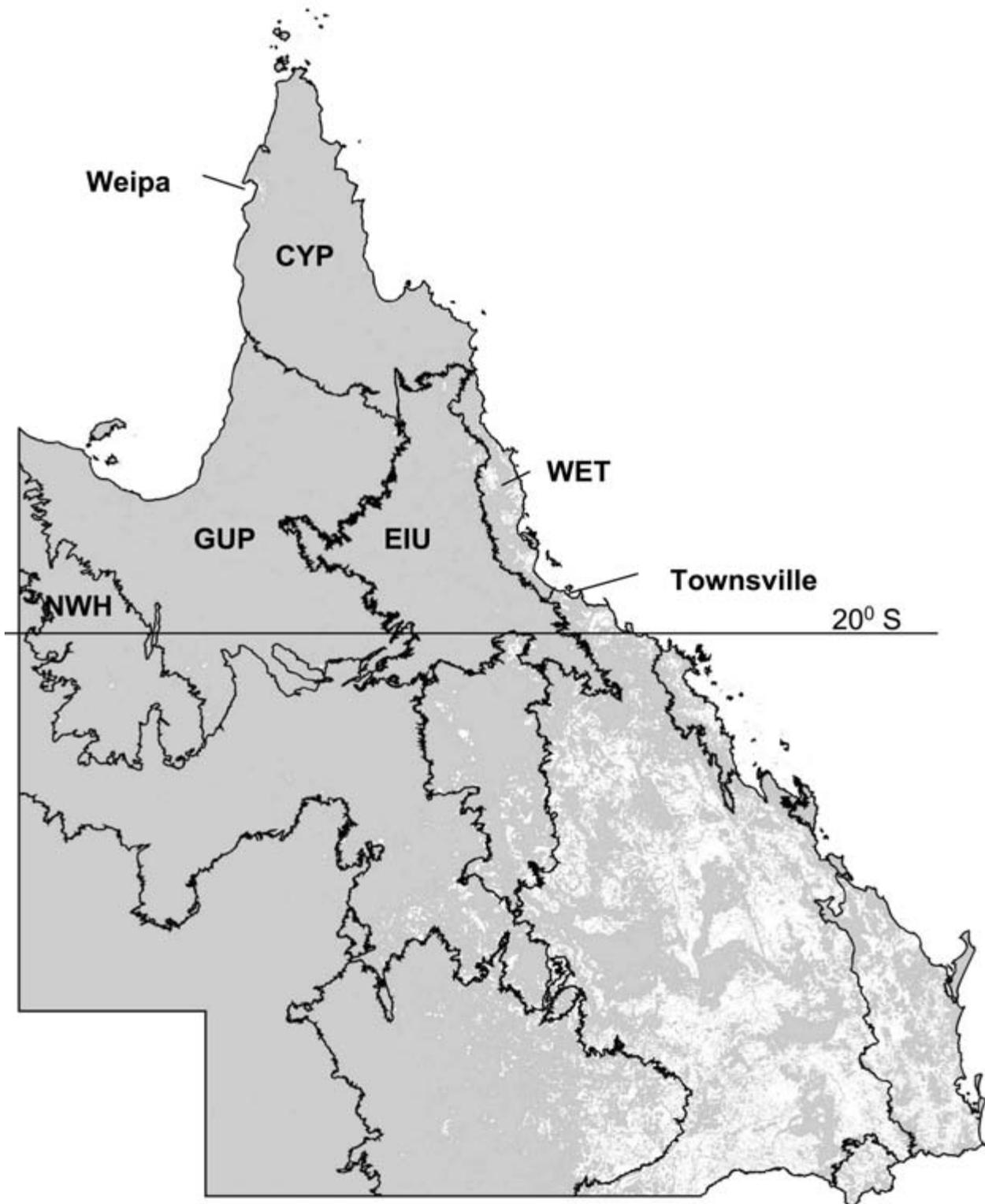


Figure 2. Remnant Vegetation Cover of Queensland and bioregions. Cleared areas not shaded, remnant (uncleared) areas shaded. Vegetation cover supplied by Queensland Herbarium, Environmental Protection Agency, Brisbane. Version 4.0 - 1995, 1997, 1999 and 2001.

Fauna surveys have increasingly concentrated on the very extensive sclerophyll woodlands of the region, rather than treating them as fringe habitats to rainforest, which was the main focus for many years. This resulted in substantial known range extensions of two mammals. During the first stage of the Cape York Peninsula Land Use Strategy (CYPLUS), fauna surveys specifically targeted the extensive sclerophyll woodlands in a start to plug many

of the huge gaps left from previous surveys (Winter and Lethbridge 1995, D.L. Storch pers. comm.). Prior to the 1990s, the Squirrel Glider *Petaurus norfolcensis* was known to occur only as far north as Mareeba and little further westward than the Great Dividing Range (Suckling 1995). With the surveys in the 1990s we found the glider in eucalypt woodland much further to the north and west (Fig. 3). It is confirmed as far north as Munkan-Kanju

Petaurus norfolcensis

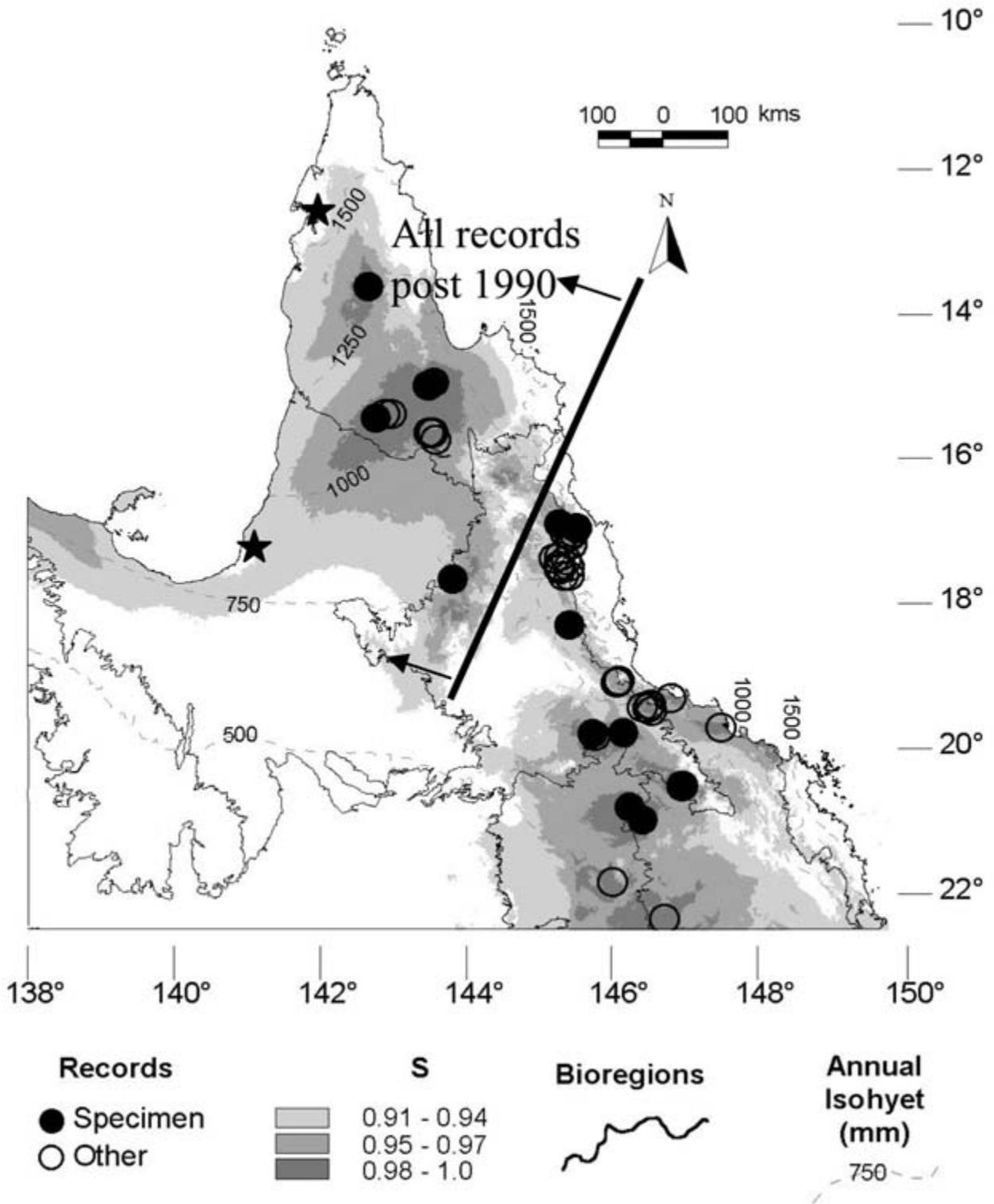


Figure 3. Known range extension of the squirrel glider *Petaurus norfolcensis* since 1990. Star = unconfirmed record (Modified from Winter *et al.* in press).

National Park about 70 km north-west of Coen on Cape York Peninsula, an extension of range of some 450 km. To the west, the glider is now known to occur on Bullaringa National Park in the Mt Surprise area and at Oriners

about 120 km east of Kowinyama. Unconfirmed records suggest an even more expansive range for the glider, to Weipa in the north and to about 50 km north-east of Karumba (Winter *et al.* in press).

An even more surprising extension of range was the discovery of the northern subspecies of the Southern Brown Bandicoot *Isoodon obesulus peninsulae*, on the western margin of the Wet Tropics bioregion a few kilometres east of Mareeba in the early 1990s (Pope *et al.* 2001), a southern extension of range of 350 km. Previously it was known to occur on Cape York Peninsula and southern Australia (Braithwaite 1995). There are two implications of this discovery. First, it emphasises how little survey work was undertaken in sclerophyll woodland, even close to the rainforest, prior to the early 1990s. Secondly, it raises the possibility that zoologists, including myself, were capturing the species but were so confident that only one member of the genus, the Northern Brown Bandicoot *I. macrourus*, was present that we failed to appreciate the not very great differences between the two species. A genetic analysis of the genus by Pope *et al.* (2001) suggests that the northern population of the Southern Brown Bandicoot is more closely related to the northern Golden Bandicoot *I. auratus*, than it is to the southern form of the Southern Brown Bandicoot, which reduces the apparent isolation of the northern Queensland form to about 700 km between it and the Golden Bandicoot's historical distribution, rather than the 2000 km separation from the southern form of the Southern Brown Bandicoot. Furthermore, (Pope *et al.* 2001) suggest *I. obesulus* and *I. auratus* be synonymised, which results in the species having one of the widest ranges of all Australian mammals.

A third example of a much improved understanding of the distribution of a woodland mammal is that of the Brush-tailed Phascogale *Phascogale tapoatafa*. Donald Thompson was the first to record the species in northern Queensland, at Lockhart River, Cape York Peninsula, in 1929 (Dixon and Huxley 1985). This was followed by a complete blank in observations until the late 1970s when sightings began to be recorded from Lockhart south to the Mareeba area. However, it was not until 1993 that its presence south of Lockhart was confirmed from specimens collected near Cooktown (Covacevich *et al.* 1994). Accumulation of records continues, not because of any suggestion of a population increase, but because of increased reporting as a result of a greater awareness of the species in northern Queensland by both scientists and the general public.

An example from north-western Queensland is the range of extension of the Rock Ringtail Possum *Petropseudes dahli* across the Northern Territory border into Queensland along the sandstone escarpments as far south-east as Lawn Hill and possibly further to the Gunpowder area (Winter *et al.* in press).

Despite, or because, of our increased emphasis on tropical woodlands there is a growing awareness that elements of the fauna in this habitat are in decline. Mammal populations are documented as declining in the Northern Territory (Braithwaite and Muller 1997, Woinarski *et al.* 2001) and for the Northern Quoll *Dasyurus hallucatus* right across its range (Braithwaite and Griffiths 1994). The Common Brushtail Possum is one of the species exhibiting a decline over much of its

Australian range in semi-arid and sclerophyll woodland (Kerle *et al.* 1992, Kerle in press, Woinarski *et al.* 2001) and there is evidence of severe population declines, possibly local extinctions in northern Queensland dry sclerophyll woodlands (Winter and Allison 1980, Winter, J. unpublished data). Initially, I thought the brushtail possum was in general decline throughout Cape York Peninsula. However, in a current study, while I have documented declines at some localities, there are nodes of relatively high populations at other localities. The general pattern appears to be one of widespread population at extremely low densities which makes them difficult to sample, with localised nodes of relatively high densities which may peak over the span of a number of years. There is no obvious explanation for the presence of high density populations and they are not necessarily located in permanent refugial areas as proposed for arid Central Australia by Morton (1990) because the high densities may be ephemeral, suggesting that they are responding to a combination of factors leading to population expansions. Braithwaite and Muller (1997) positively correlated mammal densities in the Northern Territory to ground water levels and Kerle (1985, 1998) related higher possum population densities in Kakadu to longer times since the last fire and the success of the wet season. Kerle (1998) is optimistic that the Common Brushtail Possum's ability to produce 1.7 young per year per female during favourable conditions in northern Australia will allow populations to recover from unfavourable times. I have yet to determine the effects of fire and/or rainfall on population dynamics of the Cape York Peninsula brushtail possums, but it would seem they have the ability to recover under favourable conditions.

The apparent plight of the common brushtail possum in the widespread dry sclerophyll woodlands of Australia emphasises the importance of understanding the population biology and socioecology of a species. Studies along these lines are underway and include a population study, by Chris Johnson and students from James Cook University, on the population parameters in a series of high and low density populations of Common Brushtail Possums and Rufous Bettong in the Mt Fox area (Isaacs and Johnson 2003, Johnson pers. comm.) and of possum densities in sclerophyll forest in northern Queensland (McIlwee 2002). A Savannah CRC project, part of a broader project across northern Australia, undertaken by Queensland Environment Protection Agency, is assessing changes to mammal populations by revisiting areas sampled 20 to 28 years previously (J.W. Winter, P. Latch and A.B. Freeman, unpublished data). Other Savannah CRC projects are assessing the impacts of grazing on terrestrial vertebrate fauna (Fisher and Kutt 2002), population densities of the Antilopine Wallaroo *Macropus antilopinus* in northern Queensland (E. Ritchie and C. Johnson, pers. comm.) and the Black-footed Tree Rat *Mesembriomys gouldi* (F. Ford pers. comm.).

Clearing of sclerophyll woodland for agricultural and pastoral purposes is a major conservation issue in Queensland with an average annual clearing rate over

the two year period 1999 to 2001 of 557,000 ha per year (Department of Natural Resources and Mines 2003). Luckily, most of northern Queensland has not been targeted except in localised areas, mainly concentrated south-west of Cairns and in the immediate vicinity of Cooktown and Weipa (Fig. 2). Nevertheless, broad-scale clearing may increasingly become an issue of concern in northern Queensland. The bioregion most affected is the Einasleigh Uplands with 2.03% cleared (Fox *et al.* 2001). Although the total area cleared is small, it has been concentrated on certain woodland types, notably those on Cainozoic alluvial plains (Map unit C in Fox *et al.* 2001), which are the more fertile soils and would be expected to have higher wildlife values. In addition, open-cut mining for bauxite in the Weipa area has the potential to eliminate tall eucalypt woodland on aluminous laterite over very large areas on the western side of Cape York Peninsula.

More subtle, because it does not seek to remove the tree cover, yet potentially having a significant impact on much of the tropical woodland environment of northern Queensland, is the increase in cattle stocking rates. This has occurred over the previous few decades through three factors (Tropical Savannas CRC 2003, web site <http://savanna.ntu.edu.au/information>). The introduction of new breeds of cattle, *Bos indicus*, more suited to tropical environments; supplementary feeding of nutrients to make up for low levels of nutrients in pastures, which has allowed cattle to eat more of the standing vegetation, particularly in times of drought; and the introduction of new pasture grasses. One such grass, the perennial exotic Buffel Grass *Cenchrus ciliaris*, has the capacity to exclude native ground cover species (Franks 2002), which may have detrimental ramifications on the ecological processes in tropical grassy woodlands. The effects of pastoralism on vertebrates in tropical woodlands are not well known, but the few studies confirm that the impacts may be substantial with a decline in biodiversity in reptiles, birds and mammals (Thurgate 1997, Woinarski and Ash 2002, James 2003).

The decline in mammalian populations within tropical sclerophyll woodlands is likely to be a complex interaction between climate, moisture availability, fire and land management associated with cattle grazing.

Conclusions

There is little doubt that the volume of research on forest fauna has increased substantially in northern Queensland, and we have a much better understanding of the conservation issues involved. However, as in the past, rainforest still receives most attention. Previously the rainforests of northern Queensland were targeted because of their affinities with New Guinea and were seen as exotic within the Australian context. They remain the focus of attention because of their high profile and the interest stimulated by the rapid destruction of the habitat globally. Direct human impact on the rainforest as a result of timber extraction and clearing is no longer a major conservation issue in northern Queensland. Direct impacts still occur

as a result of increased construction of road, power lines and other facilities required by the increasing human population, but they are much more localised. The biggest conservation issue facing the rainforest in northern Queensland is the indirect impact of global warming and the threat it poses on an upland fauna in a mountainous tropical region. It involves a major shift in dealing with conservation threats at a local level to more remote measures at a national and global scale. Despite this, local involvement is still important to understand the ecological processes within the forest and to undertake restoration measures, such as the reclamation to forest of upland areas, to buffer the effects of climate change.

Proactive management remains the main issue in the cool tall open forests on the western margin of the Wet Tropics, which is the habitat of the vulnerable population isolate of the Yellow-bellied Glider. Burning strategies to prevent capture of this forest type by rainforest without the destruction of large den trees or human residential property are still in the development stage. The lowland moist woodlands inhabited by Mahogany Gliders pose a different problem as the survival of this glider depends on success in convincing land holders to accept habitat retention for the species as a high priority in property management. In marked contrast to the two gliders, better understanding the basic ecology of the endangered Northern Bettong remains the issue for conservation management of this species.

Mangroves continue to be the neglected forest type in northern Queensland. Boot-sucking mud, sandflies, mosquitoes and possible encounters with crocodiles are potent inhibitors to working in these forests. The Water Mouse is waiting to be discovered in far northern Queensland and mangroves may prove to be important refugial areas for the Common Brushtail Possum if the possum can make use of the high water content of the mangrove leaves.

A greater focus on the extensive savanna woodlands of the north over the past decade has positive and negative implications for conservation of the fauna living there. We are extending known distributions of species, often dramatically, and appreciating the fine scale complexity of the woodlands. However, that increased knowledge is exposing our poor understanding of population dynamics of woodland fauna, particularly the reasons for declines in numbers and distribution of some species which is a major worry. A major conservation challenge for the next decade is to understand the often subtle interactions between natural forces, such as climate, with land management involving clearing, stocking rates and fire.

My conclusion is that over the past decade there have been gains in the conservation of forest fauna at the local level, most obviously in rainforest. However, these are overshadowed by broader scale threats: global climate change to all forest types, not just upland rainforest; and more intensive use of tropical woodland which, if not managed appropriately, may lead to large scale declines of many native species.

Acknowledgements

I particularly want to thank all those, too numerous to thank individually, who have shared their knowledge, data, commitment and love of the environment with me over the thirty years I have lived and worked

in northern Queensland. Their willingness to do so makes it a pleasure to be involved in this field of endeavour. Hans Dillewaard prepared the vegetation map for me.

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