Skull, Stephen David (1998) ***The ecology of tropical lowland plant communities with particular reference to habitat fragmentation and Melaleuca viridiflora Sol. ex Gaertn. dominated woodlands.*** **PhD thesis, James Cook University**

CHAPTER ONE - GENERAL INTRODUCTION

In our hands now lies not only our own future, but also that of all other living creatures with whom we share the earth (Attenborough 1979).

1. GENERAL INTRODUCTION

1.1 Introduction and background

Ongoing clearing and fragmentation of the tropical lowlands in north-eastern Queensland continue to pose serious threats to the biological diversity of the region (DEST 1995a). Although it is acknowledged that the ecosystems most at risk from these processes are those that have already been substantially cleared (Given 1994), no accurate figures with respect to either clearing or fragmentation are available for the vegetation types of this region. This lack of information remains apparent today despite:

- a long history of clearing in an area widely recognised for its unique biological values and national ecological importance (Webb 1966; Stanton & Godwin 1989; QDEH 1995a);
- the well known ecological effects of fragmentation on remnant plant communities including weed invasion, localised species extinction, and alterations to disturbance regimes (Saunders *et al.* 1991);
- continuing pressure on remnant plant communities from agricultural expansion (particularly sugar cane), urbanisation, and a range of other ecological impacts typical of many coastal regions around Australia; and
- a commitment from all levels of government to establish a comprehensive, adequate and representative national conservation reserve system based upon "careful survey of all Australia's major landscapes" (ACG 1995).

Current assessments of the relative significance of plant communities within the lowlands of the WTBR rely upon scattered mapping and field survey data (QDEH 1995a). More detailed baseline data is critical if management agencies are to formulate informed conservation objectives, and ensure that future development of the tropical lowland environment proceeds in an ecologically sustainable fashion. The lack of adequate information for the management of lowland vegetation extends to arguably the most important natural disturbance agent in these terrestrial ecosystems, fire. This remains a major concern of conservation management agencies in the region (Mr P. Stanton, pers. comm.).

Gill (1996) and Keith (1996) have recently documented the potentially significant impacts that fire can have on plant biodiversity, with fire, or its absence over long periods, responsible for the localised extinction of native plant species in Australia. The use of prescribed fires by government agencies to achieve specific conservation and management objectives has increased dramatically in north-eastern Queensland over the past decade. Many national parks in the region, however, have only had fire management plans drafted relatively recently (e.g. QNPWS 1991; QDEH 1995b; QDEH 1996a).

These often controversial fire management plans have primarily been introduced as a means of addressing the massive, rapid habitat changes that have been observed (e.g. Stanton 1992), which are chiefly responsible for a reduction in habitat diversity. In addition, this increased use arises from a raised level of recognition and acceptance that many native plant species require fire to complete critical stages of their life cycle. It is, however, widely acknowledged that the effects of fire on many of the plant communities in northern Australia are poorly understood (e.g. Gill *et al.* 1996). Similarly, few tropical plant species are listed on a national register that documents the responses of vascular plant species to fire (Gill & Bradstock 1992). Furthermore, both the past and present fire regimes (fire intensity, season and frequency; Gill 1981) affecting lowland plant communities in north-eastern Queensland remain largely undocumented.

Exotic species invasion is another major threat to biodiversity in Australia (Hobbs & Humphries 1995). Tropical lowland plant communities are in no way exempt from this threat, with many exotic species in the region already listed in one of the following categories:

- major environmental weeds of northern Australia (with species classified as either being capable of destroying, or affecting massive impacts on, terrestrial ecosystems);
- significant environmental weeds; or
- potentially invasive introduced plants (Humphries *et al.* 1991).

Adair (1995) documented examples of exotic species contributing to localised and regional plant extinctions, whilst Fox (1995) discussed the many ways in which exotic species can alter ecosystem functioning. As with many disciplines of ecological research, the invasion of tropical plant communities by exotic species (and their subsequent effects on ecosystem processes) has been poorly studied compared with temperate systems. Species which have been studied in detail are threatening agricultural productivity (e.g. Rubber vine) or extremely high profile conservation reserves (e.g. *Mimosa pigra* L. in Kakadu National Park).

The overall objective of this thesis was, therefore, to document some of these major threats to biodiversity in the tropical lowland environment of north-eastern Queensland. This has been achieved by an initial assessment of habitat loss and fragmentation for a selected section of ecologically significant lowland habitat, followed by more detailed investigations which have been restricted to a single plant community.

1.2 Scope of thesis

This thesis necessarily had a management focus. During the investigation of several management issues, a range of ecological concepts and processes were considered. Wherever possible, every attempt was made to include the appropriate ecological theory literature (including sampling methodologies, statistical analyses and issues of spatial and temporal scale) relating to these concepts and processes in each section of the thesis. This inclusion was, however, rationalised so that lengthy reviews of the many aspects and development of ecological theory (from their origins to the current "schools of thought") could be avoided.

1.3 Project objectives and structure of the thesis

This thesis had the following more specific objectives:

- to provide data on recent (past 50 years) patterns of clearing and fragmentation for certain vegetation types within the lowland habitat mosaic of north-eastern Queensland;
- to assess the structure of a vulnerable plant community (primarily located within the same lowland habitat mosaic) at a range of sites between Townsville and Cooktown;

and

to record the effects of soil moisture, soil type, prescribed fire management and exotic pine invasion on the same plant community.

Chapter 2 initiates the research component of the thesis with an assessment of habitat loss and fragmentation. These processes are documented in detail for lowland plant communities in the Cardwell region between 1942 and 1992. *Melaleuca viridiflora* Sol. ex Gaertn. dominated woodlands, one of the plant communities most affected by the clearing process, then become the focus of the remaining thesis chapters.

The *M. viridiflora* woodland community was selected for further study for two other reasons. Firstly, its conservation status in the region is currently considered vulnerable (QDEH 1995a), and is soon to be upgraded to endangered (Mr G. Morgan, pers. comm.). This suggests its current level of representation within the existing conservation reserve system is inadequate. Secondly, the community is relatively simple in structural terms as few other species are common in either the canopy or the midstorey. The latter makes it comparatively easy to document and assess changes in community structure associated with ecological disturbance.

The second broad objective of the thesis is addressed in Chapter 3, which outlines the structure and composition of *M. viridiflora* communities at 24 sites between Townsville and Cooktown. A combination of multi-variate statistical analyses is utilised to produce groups of sites, which are then tested and discussed in terms of soil types, fire histories and predicted climate. The effects of soil type and surface soil moisture on woodland structure are assessed in greater detail in Chapter 4.

Chapter 5 examines some of the ecological responses of *M. viridiflora* woodlands to single and repeated prescribed fires. The implications of the research findings are discussed and recommendations for future fire management initiatives are proposed. The invasion of plantation pine trees into *M. viridiflora* woodlands is investigated in Chapter 6. This ecological problem is a relatively recent addition to a substantial list of management challenges already associated with lowland remnants of this plant community. The spatial pattern of invasion is assessed, and the growth rates and germination responses of the native and pine trees compared. The use of fire as a potential control measure for this invasion process is also documented. Finally, in Chapter 7, the results of the entire thesis are discussed in a broader context and the possibilities for future research are outlined.

CHAPTER 2 - AN ASSESSMENT OF HABITAT FRAGMENTATION:

THE NORTH-EASTERN QUEENSLAND TROPICAL LOWLANDS - A CASE STUDY

Only after the last tree has been cut down, only after the last river has been poisoned,
only after the last fish has been caught, only then will you find
that money cannot be eaten (Cree Indian Prophecy).

2.1 INTRODUCTION

2.1.1 Habitat fragmentation and its ecological effects

Habitat fragmentation can be defined as a combination of habitat loss and the subsequent apportionment of the remaining habitat into smaller patches with increased levels of isolation (Noss & Csuti 1994). It remains the most serious threat to biological diversity and the ongoing process of species extinction (Wilcox & Murphy 1985; Harris & Silva-Lopez 1992; Purdie 1995), with the conservation of regional biota in some areas now depending almost entirely on the management of habitat fragments (Saunders *et al.* 1991).

The extent of habitat loss in Australia over the last 50 years is equivalent to all the clearing that occurred in the previous 150 years (AUSLIG 1990). Australia clears more remnant vegetation per year than Malaysia or Papua New Guinea, but less than Brazil, Indonesia, Mexico and Thailand (DEST 1995a). Despite inconsistencies in some Australian clearing figures because of changing definitions (Young 1996), many estimates of habitat loss are now available. In the Western Australian wheatbelt, for example, where habitat fragmentation has been the subject of a large CSIRO research program, some 93% of the original vegetation has been cleared, with even higher figures in some regions (Saunders *et al.* 1991).

In Queensland, clearing over a ten year period (1983-1993) has averaged 300,000 ha yr^{-1}, more than twice that of NSW and more than 12 times that of the state with the next closest figure (Western Australia) (DEST 1995a). All lowland (< 60 m altitude) plant communities in south-eastern Queensland have been acutely affected by clearing, with losses averaging 80% (Catterall & Kingston 1993). For the entire south-eastern Queensland area, the annual clearing rate for the past 160 years has been nearly 7,000 ha yr^{-1}, although in recent times this figure has risen dramatically to over 40,000 ha yr^{-1} (Catterall & Kingston 1993; Smith *et al.* 1994). One of the few figures that exists for northern Queensland is that of the rainforests of the Atherton Tablelands, which have been reduced to less than 20% of their original extent (Laurance 1987). House and Moritz (1991) added that clearing of rainforests had particularly affected lowland forest types and the upland communities of the Atherton and Evelyn Tablelands. The only exception to this broad-scale clearing of closed forests, they noted, was the large tracts of comparatively undisturbed forest in mountainous areas. Other figures relevant to the study area examined by this thesis are outlined further in Section 2.1.2.

In a comprehensive review of the consequences of fragmentation on terrestrial ecosystems, Saunders *et al.* (1991) report that this process causes both physical and biogeographic changes in landscapes. Catterall and Kingston (1993) also provide a list of the main ecological processes affected by habitat loss and fragmentation, and discuss these effects on various habitat complexes and riparian zones. Ecosystems considered most at risk from fragmentation include those that have already been reduced in terms of their occurrence or size, and those that remain similarly threatened (Given 1994). Noss and Csuti (1994) defined the various spatial and temporal scales at which habitat fragmentation can operate. These scales provide important background information for a discussion of the ecological effects of habitat fragmentation, and include:

- a biogeographic scale (tens to hundreds of kilometres) - this type of fragmentation may take place over a long time scale (hundreds of years) as regions are separated from others by intensive agriculture and/or urban development;
- an intermediate scale (tens of kilometres) - usually the scale at which the effects of this phenomenon are studied, and may operate on a ten year temporal scale; and
- a fine scale (< 10 kilometres) - the level at which the internal dynamics of fragments are most commonly studied over several years.

In addition, it is important to consider that these effects can operate at the level of individual species, populations or communities, and are influenced by the shape, size and position of remnants in the landscape (Saunders *et al.* 1991). Furthermore, the interaction of these effects drives specific biotic responses.

One of the most immediate effects of habitat loss is the creation of new edges, the primary effect of which is an increase in the area/perimeter ratio of remnant fragments (House & Moritz 1991). Changes in plant species composition at the edge of forest remnants have been documented by several authors (Ranney *et al.* 1981; Lovejoy *et al.* 1986). Ranney *et al.* (1981) noted a permanent increase in plant basal area and a transient increase in tree density. Increased rates of plant growth and animal predation have also been recorded (Andren & Angelstam 1988; Noss & Csuti 1994), as have changes in edge permeability to fauna (Stamps *et al.* 1987). Noss and Csuti (1994) concluded that the intensity of edge effects was related to the structural diversity of the adjacent habitats.

The creation of new edges also results in marked changes in the micro-climate of remnant habitat patches. Saunders *et al.* (1991) listed three main alterations to micro-climates on the edges of remnant habitats which can affect at least the outer 50 m of a fragment (Young & Mitchell 1994). The first of these effects is an increase in solar radiation fluxes, which has been recorded by Palik and Murphy (1990) in sugar maple/beech forests, and Hobbs (1993) in an agricultural landscape. This effect is listed as a serious concern for remnant habitats by the Australian Conservation Foundation (ACF 1995). Through increased day-time and decreased night-time temperatures (Saunders *et al.* 1991), and subsequent changes in relative humidities (Noss & Csuti 1994), this change can have flow-on effects including alterations to seed germination conditions (Hopkins 1990), changes in plant growth rates (Lovejoy *et al.* 1986) and increased reflectivity from bare soils and pastures in agricultural landscapes (Monteith & Unsworth 1990).

The second of the micro-climate effects is a change in the wind profile of a fragment, which has been shown to increase tree mortality, wind erosion and windthrows (Lovejoy *et al.* 1984; Laurance 1987; Nulsen 1993; Noss & Csuti 1994). The third effect is changes to water fluxes which can include altered rates of evapo-transpiration and soil moisture levels (Kapos 1989). On a large scale hydrological patterns are also affected following habitat loss and fragmentation. Studies in Western Australia have shown that catchment clearing leads to increased run-off rates (McFarlane *et al.* 1993) and a less buffered hydrological cycle (Peck

& Williams 1987). Other hydrological effects that have been recorded in Australia are increased levels of waterlogging (McFarlane & Wheaton 1990), which can be detrimental to many terrestrial plant communities, and salinisation following a rise in the watertable (George 1990). Salinisation continues to be a major land management problem in Australia. As a result of micro-climate changes, other major ecosystem processes such as nutrient cycling and decomposition can also be affected (Saunders *et al.* 1991), although few data are currently available. Extensive habitat clearing has also been shown to lead to changes in rainfall patterns (Williams 1991).

Fragmentation also results in the formation of new barriers to dispersal. The effect of barriers on the movement of some faunal groups (Mader 1984; Burnett 1992) and the dispersal patterns of seed (Hopkins 1990) is well documented. Genetic effects can result, including reproductive isolation (Myers 1994), reduced heterozygosity (Crome 1988), reduced population viability (Noss & Csuti 1994; Possingham 1995), and alterations to speciation processes, the most likely of which is an increase in the rate of extinction (Myers 1994). Obligate outcrossers can be particularly affected as a result of changing pollinator densities and/or movements (Crome 1988; Prof. R. Whelan, pers. comm.), and barriers can also result in changes to species composition (favouring exotic species), habitat structure and successional development (Johnson *et al.* 1981). In fact, the invasion of exotic species into remnant habitats is one of the most serious effects of fragmentation (Noss & Csuti 1994; ACF 1995; Purdie 1995). Somewhat disturbingly, it has also been noted that some species invasions may not be triggered for some time following fragmentation, hence some ecologists fear that for Australia the worst may be yet to come (Fox 1995). The issue of exotic species invasion into remnants is considered further in Chapter 6.

Changes in species richness, including local extinctions that can continue long after isolation, are some of the most studied effects of habitat fragmentation (Lovejoy *et al.* 1984; Recher & Lim 1990). This extinction process will continue even if the area of fragment being investigated remains constant (Levenson 1981). Island biogeographic theory (MacArthur & Wilson 1963, 1967) has formed the basis for much debate on island sizes and their associated species richness, as has the ensuing discussions centred on optimal reserve configuration. The pros and cons of this debate have been well documented within the literature (see for example Simberloff & Abele 1982; Wilcox & Murphy 1985) and will not be re-iterated here. Instead, the results of a range of scientific investigations are presented to illustrate the types and extent of losses that can occur.

Species relaxation in isolated remnants (reduced species "carrying capacities" of remnants) is one of the ecological effects predicted by island biogeographic theory (Saunders 1989). Drayton and Primack (1995) reported that 37% percent of the plant species in an isolated conservation reserve were lost over a 100 year period, and the proportion of native species declined from 83% to 74%. In Ecuador, 90 endemic species were lost on a single mountain ridge subject to broad-scale clearing for agriculture (Dodson & Gentry 1991). The degree of fragment isolation has also been negatively correlated with floristic richness in eastern deciduous forests of the United States (Johnson *et al.* 1981). Long-term vegetation effects, such as alterations to both the physical and genetic structure of populations, remain relatively unknown (Noss & Csuti 1994).

Throughout the literature, however, the effects on animal populations are much more widely documented. From a long-term study of birds on Barro Colorado Island in Panama, Karr (1994) illustrated that the process of species extinction within a fragment was not random, and that individual species survival rates were one of the most critical demographic attributes associated with extinction. Species that cannot adapt to fragmented landscapes are bound for eventual extinction, and those most at risk include naturally rare species, species with large home ranges, species with poor dispersal abilities and species with highly variable population sizes (Noss & Csuti 1994). The decline of small mammal populations and birds are well documented (e.g. Burbidge & McKenzie 1989; Saunders & Ingram 1995), with amphibians and reptiles expected to exhibit similar patterns of decline over time (Recher & Lim 1990). Indeed, clearance and fragmentation of habitat pose the highest threat to the survival of Australia's bird populations (Garnett 1992) and are the major reasons for the current status of 64% of Australia's threatened reptile species (ACF 1995). Interestingly, little data are available relating to natural rates of species extinction in undisturbed landscapes for comparative purposes.

2.1.2 Australian lowlands and those of the north-eastern Queensland tropics

Catterall and Kingston (1993) provided a recent example of the correlation between altitude and vegetation clearance. The authors provide a list of papers that all record a disproportionate loss of native vegetation from lowland areas in Australia. This is usually a result of clearing for either urban areas (on flat, low lying sections of the landscape) or agriculture (which clears the fertile soils of lowland areas).

Within Queensland, ongoing clearing in the lowlands of the WTBR, as defined by ANCA (1995), although less extensive than other regions of the State, is focussed on remnant vegetation and therefore no less significant for biodiversity protection (DEST 1995a).

A recent Commonwealth Government project utilised satellite imagery from 1990 and 1992 to map the type, severity and extent of landcover disturbance across the Australian continent at an approximate scale of 1: 1,000,000 (DEST 1995b). With an average pixel size of one hectare, this study classified the vegetation types into landcover classes, based on soil data, overstorey structural (estimates of projective foliage cover) and floristic attributes. Most of the vegetation types examined in this thesis could be considered to fall into four of the landcover types recorded, and the percentage that has been cleared for each of these is presented in Table 2.1.

Few figures exist as to the precise extent to which Australian tropical lowlands have been cleared. In his discussion of the ecology of the recently re-discovered Mahogany glider, van Dyck (1993) estimated that over 80% of the lowland vegetation complex had already been cleared, and continuing landuse (sugar cane expansion programs, aquaculture, forestry and urbanisation) in the region further threatened the few intact remnant communities. No accurate figures specific to each of the actual vegetation types were provided in this paper, nor a description of the methods utilised to obtain the 80% figure. Lavarack (1994) concluded that in terms of lowland plant communities, the re-discovery of the Mahogany glider had been a mixed blessing, as landholders of all tenures accelerated clearing prior to moratoriums (of clearing) being introduced. Braby (1992) estimated that 60-80% of the tropical lowland habitat mosaic had been cleared in the area studied here, but again provided

Table 2.1 **Vegetation clearance (%) figures determined by three previous studies (DEST 1995b-Entire Australian continent, Bianco 1994-Mulgrave Shire of north Queensland and QDPI 1993-the Tully-Murray catchment).**

DEST (1995b)		Bianco (1994)		QDPI (1993)	
Landcover type and code	%	Vegetation type after Tracey (1982)	%	Vegetation type	%
Tall, medium & low closed eucalypt forest (eTML3)	37.1	Complex mesophyll vine forest (1c)	100	Closed forest	19
Medium open eucalypt forest (eM2)	65.3	Mesophyll vine forest (2a)	25	Open eucalypt forest	14
Medium open non-eucalypt forest (xM2)	56.7	Mesophyll vine forest with palms (3a)	48.9		
Low open non-eucalypt forest (xL2)	9.7	Complex notophyll vine forest (6)	66.5		
		Notophyll vine forest (7a)	0		
		Vine forest with acacia (12a)	100		
		Coastal beach ridges and swales (17)	82.3		
		Mangroves (22a)	44.4		

no account of how these figures had been reached nor any details for specific vegetation types. In other areas of the WTBR, Braby (1992) estimated that these figures were even higher (90%) for intensively developed landscapes such as the Tully River delta.

Hamilton and Cocks (1994) also stated that significant losses of native vegetation had occurred in the Cairns-Townsville region, although no figures were quoted. In analysing habitat fragmentation in rainforests, Crome (1988) indicated that the tropical lowlands were the most reduced and fragmented of the wet forest ecosystems. This is also reflected in a recent assessment by the Queensland Department of Environment (QDE), of the conservation status of Queensland's bioregional ecosystems, with most lowland habitats for this region considered either endangered or vulnerable (QDEH 1995a). These habitats also often contain species considered rare and threatened (Thomas & McDonald 1989; Ingram & Raven 1991).

Within the WTBR, a study of the remnant vegetation in the Mulgrave Shire examined habitat loss (including lowlands) and provided accurate figures for specific vegetation types based on mapping of vegetation patterns derived from aerial photographs over a 25 year period (1965-1990) (Bianco 1994). This study was conducted in a more northern section of the WTBR and therefore included several vegetation types not found in the area examined by this thesis. A total of 1466 ha was lost in a 25 year period from the shire at an average of almost 59 ha yr^{-1}. The Queensland Department of Natural Resources (DNR) assessed the condition of all river catchments in Queensland, and calculated the reductions in some vegetation types (QDPI 1993). Within the Tully-Murray catchment (which covers the study area of this investigation), reductions in the total areas of two vegetation types (closed forest and eucalypt open forest) are estimated. The results from both these studies are presented in Table 2.1.

The overall conservation status of remnant terrestrial and wetland habitats within the Tully-Murray catchment of the WTBR was also assessed by Tait (1994). This report identified conservation management issues relevant to proposed expansions of the sugar cane industry as a result of the Sugar Industry Infrastructure Package (SIIP). Less than 20% of land systems with high agricultural suitability remain under natural vegetation. The gazettal of Edmund Kennedy National Park has ensured protection for some habitats which are restricted to the coastal province (mangroves, mixed dune forests, bulkuru swamps, littoral vine forest, swampy paperbark forest and marine couch grassland). Only 25% of the park, however, contains vegetation types that continue to be threatened and further diminished by agricultural development. These include eucalypt open woodlands, palm swamps, broad-leaved paperbark woodlands (addressed in detail in the remaining chapters of this thesis), paperbark/beach forests and acacia open forests. On the mainland, lowland habitats are also protected to some extent within both Lumholtz and Hinchinbrook Island Channel National Parks. Offshore, some habitats are protected within Hinchinbrook Island National Park (HINP), although these habitats are not considered typical of those on the adjacent mainland (Mr P. Stanton, pers. comm.).

The QDE is currently conducting the Coastal Lowland Vegetation Mapping Project, which will result in a Coastal Lowland Conservation Plan for lowland plant communities between Townsville and Tully (QDEH 1993, 1994, 1996b). The aim of this GIS-based project is to identify and recommend important remnant patches of lowland habitat for addition to the existing conservation reserve system. To date no data from this project have been published in the wider literature.

2.1.3 Indices of landscape pattern

The area and perimeters of remnant vegetation fragments have long been utilised to assess the shape of habitat "islands" or remnant fragments in terrestrial landscapes. Patton (1975) developed a diversity index now commonly referred to as the shape index (SI). This index describes the deviation of a fragment from circularity (Laurance & Yensen 1991) and is determined using the formula:

$$SI = P/2(\Pi A)^{0.5},$$

where A is the area of a fragment in square metres and P is the perimeter of a fragment in metres. A perfectly circular fragment will have a SI value of one and all other shapes have higher values (Laurance & Yensen 1991). This index has been used to:

- assess the shapes of rainforest fragments (Laurance 1989, 1991), rain clouds (Lovejoy 1982) and a variety of terrestrial landscapes (Ripple *et al.* 1991; Bianco 1994);
- compare different patterns of habitat reduction (Zipperer 1993); and
- formulate designs of nature reserve boundaries (Buechner 1987).

The shape index is still relevant to reserve design in predominantly undisturbed landscapes. A high SI indicates that most of the fragment will be susceptible to edge effects (and will therefore be difficult to manage). Wilcove *et al.* (1986) showed that as a result of edge effects associated with habitat fragmentation in temperate ecosystems, some habitat patches below a critical size and shape will have no central core representative of the original habitat.

Conversely, a SI value close to one indicates that a fragment may have a relatively large core area (depending on its size), which is potentially more suitable for conservation. This index has been shown to be a more robust measure of patch shape compared with simple perimeter-area ratios (Ripple *et al.* 1991).

Remnant habitat perimeter and area data have also been used in the development of algorithms for identifying critical remnant habitats for conservation purposes (e.g. Fensham submitted). This assessment of remnant vegetation patches in the Darling Downs region of south-eastern Queensland established the most efficient method of protecting 1% of the original area of all the mapped vegetation types within additional conservation reserves.

The dispersion of specific vegetation types across a landscape can be calculated using the formula derived by Clark and Evans (1954):

$$R = 2\rho^{0.5}r,$$

where R is dispersion, r the mean nearest neighbour distance and ρ the mean patch density (number of patches per unit area). The dispersion index indicates whether fragments are distributed at random ($R = 1$) or in an aggregated fashion across the landscape ($R > 1$) (Ripple *et al.* 1991).

The fractal geometry of fragments (particularly fragment perimeters) has also been utilised to assess spatial landscape patterns. The calculation of fractal dimensions (D) is based on work by Mandlebrot (1983), and can be used to indicate trends in landscape complexity (Odum & Turner 1990; Noss & Csuti 1994; van Hees 1994), dispersion (O'Neill *et al.* 1988), diversity (Kienast 1993) and the dominance of different vegetation types (Hulshoff 1995). Fractals are calculated using regressions of perimeter and area. One fractal dimension commonly used (e.g. O'Neill *et al.* 1988) was considered for use in this study, but a pilot study produced results not statistically significantly different from the SI described above.

Perimeters and areas of fragments in a landscape can also be used to calculate a fragmentation index (FI). This index is considered less robust than others such as the SI, but has been used in comparative studies of landscape pattern in conjunction with other indices (e.g. Ripple *et. al.* 1991; Bianco 1994). The index can be calculated using the formula:

FI = P/A (symbols as for SI formula above).

An increase in the FI indicates that the vegetation type concerned has become more fragmented over a particular time period (Bianco 1994).

Recently, a relatively simple yet objective method was utilised for assessing habitat fragmentation in both undisturbed and cleared landscapes (DEST 1995b). This assessment sorts patches according to their size, and then plots cumulative area against patch size rank (from largest to smallest) for each landscape. This produces two curves with shapes that will be markedly different in a highly disturbed landscape, or conversely, curves with similar shapes when two undisturbed landscapes are compared (DEST 1995b).

2.1.4 Aims of this investigation

An assessment of habitat reduction and fragmentation was undertaken within the Tully-Murray catchment of the WTBR near Cardwell, north-eastern Queensland. This assessment aimed to:

(i) quantify the extent of clearing of each lowland mapping unit between 1942 and 1992;

(ii) compare these findings with the few previous studies that exist; and

(ii) assess changes during this period for each mapping unit in terms of the total area remaining, the number of remnant patches present, their shape, perimeter length, area, dispersion and degree of fragmentation.

2.2 METHODS

2.2.1 Study area

Lowland vegetation (< 100 m altitude) was mapped between Dallachy Creek (north of Cardwell) and Sunday Creek (south of Cardwell). The study area (indicated on the final maps produced, Section 2.3.1) falls within the WTBR of Queensland and is located in the southern section of the Tully-Murray Catchment Area (QDPI 1993). This catchment has been subjected to extensive clearing, particularly of open eucalypt forest and closed forest communities (QDPI 1993). Conservation reserves (state forests and national parks) cover 64% of the catchment (QDPI 1993), although these are predominantly located to the north of the study area within the Wet Tropics World Heritage Area (WHWTA).

The study area includes several of the study sites investigated in Chapters 3-6, and some of the most important remnants of tropical lowland forest and woodland communities in the WTBR. Many of these communities are habitat for the recently re-discovered Mahogany glider (van Dyck 1993), which is now protected by the Nature Conservation (Mahogany glider) Plan (QDEH 1995c), a sub-ordinate piece of legislation under the Nature Conservation Act (1992). The rediscovery of this species lead to a moratorium on further clearing in the region, both on private property through the issue of Interim Conservation Orders and through a moratorium placed on the Queensland State Forest Service (QSFS) with respect to further clearing of remnant vegetation for plantation pine.

2.2.2 Vegetation mapping

The entire study area was mapped to the 100 m contour from 1: 25,000 aerial photographs. Two sets of photographs were analysed: a black and white set taken in August 1942, and a colour set taken in September 1992. A total of 70 photographs was analysed. Topographic detail from three 1: 50,000 topographic maps (Mt Graham, Cardwell and Kirrima) was transferred to four A3 sheets and used as reference data for the transfer of aerial photograph information. The sheets were then enlarged to enable the transfer of data from the photographs using a Zoom-Transfer Scope. This technique has been successfully tested by

Power and Jackes (1991). Stereo pairs of photographs were then analysed and likely vegetation boundaries drawn onto the prepared sheets.

The prepared sheets were then utilised in the field verification component (foot and vehicle traverse) of the mapping process. The field work enabled the mapping units to be classified structurally according to Walker and Hopkins (1990). Subsequently, the aerial photographs were re-examined and the vegetation boundaries re-appraised. This structural data allowed for an extrapolation of unsurveyed areas to produce final vegetation maps for the entire study area. This extrapolation involved the recognition of similar vegetation patterns, based on the density and colour of the canopy layer, the shape of certain tree species, vegetation height and the location of a given area. It should be noted that slight differences in the type (monochromatic compared with colour) scale and quality of the photographs may have produced small errors for some vegetation pattern boundaries. In some cases, time and access constraints have not allowed verification of all boundaries during the field work component of this investigation.

2.2.3 Digitising

The resultant vegetation maps were digitised using MicroMine V6.6, a graphical package used primarily for the presentation of geological and geographical data, and a Kurta IS/Three A0 digitising table. Individual polygons on the map were exported separately as .dxf files, and imported into MapInfo Professional Version 4.0.

Once imported into MapInfo, polygons were converted to regions and coloured according to mapping units. Polygons were saved as individual files, and then appended into a single combined file. Any overlapping boundaries between polygons were erased. A map browser was added which contained a numerical identification (ID) for each polygon, a mapping unit ID, the area in hectares for each polygon, the perimeter (km) of each polygon, and the Australian Metric Grid (AMG) co-ordinates for the centroid of each polygon.

At the completion of the mapping process, the MapInfo browser table was exported as a delimited ASCII file for statistical analysis. It should be noted that because the source material for the map was uncontrolled aerial photos, a certain degree of inaccuracy is inherent in the final product. This is because of parallax error and the scale of the original map, making location of accurate control points difficult. At its worst, this inaccuracy is approximately 0.2 km. As the total length of the map is approximately 46 km, this represents an error of about 0.4%.

2.2.4 Data analysis

The following variables were calculated for each mapping unit, time and specific area studied (Table 2.2); total area (ha), total number of patches, mean patch area (ha), mean perimeter length (km), and landscape indices relating to patch shape, fragmentation and dispersion. Where applicable, t-tests were conducted to assess whether observed changes were significant between time periods.

Table 2.2 Extent of the three areas examined and the associated level of landscape disturbance.

Area	Polygon co-ordinates included	Landscape disturbance*
Entire area	All polygons	Varied
1	All those < 18°22'S, 146°03'E (UTM E399636, N7968978)	High
2	All those > 18°21.5'S, 146°05'E (UTM E403157, N7969303)	Low

Based on an initial assessment of the final vegetation maps (Figures 2.1 and 2.2, Section 2.3.1). Below 18°22'S, landscape disturbance was considered to decrease sufficiently to warrant investigation as a separate area.

An unbalanced 3-way analysis of variance (ANOVA) was utilised to assess differences in variables across all areas, mapping time and mapping units. As Area 1 only contained a subset of the total number of plant communities (particularly in 1992), higher interactions from these analyses were suppressed in the computations. This suppression did not, however, prevent the analysis from providing a simultaneous assessment of the variances of the three variables examined. A one-way ANOVA was used to investigate the relationship between each variable and mapping unit for both mapping times in each of the three areas examined. To identify which mapping units had significantly different means, a Tukey's honestly significant difference (HSD) post-hoc multiple comparison test (SPSS 1993) was used for the one-way ANOVA. T-tests were then utilised to assess for significant differences between time periods for individual mapping units within each area.

For specific mapping units of interest (especially *Melaleuca* open woodlands, Chapters 3-6), chi-squared heterogeneity tests were used to compare expected (1942) and observed (1992) patch area size class distributions. Seven equally-sized classes were analysed, the overall range of which was dependent on the patch sizes of each mapping unit. Expected frequencies were pooled until all categories except one had values greater than five. Cochran (1963) has shown this method to be permissible. The distribution of the total number of patches for each mapping unit was also analysed in this fashion. All statistical analyses were performed using SPSS 6.0 for Windows (SPSS 1993).

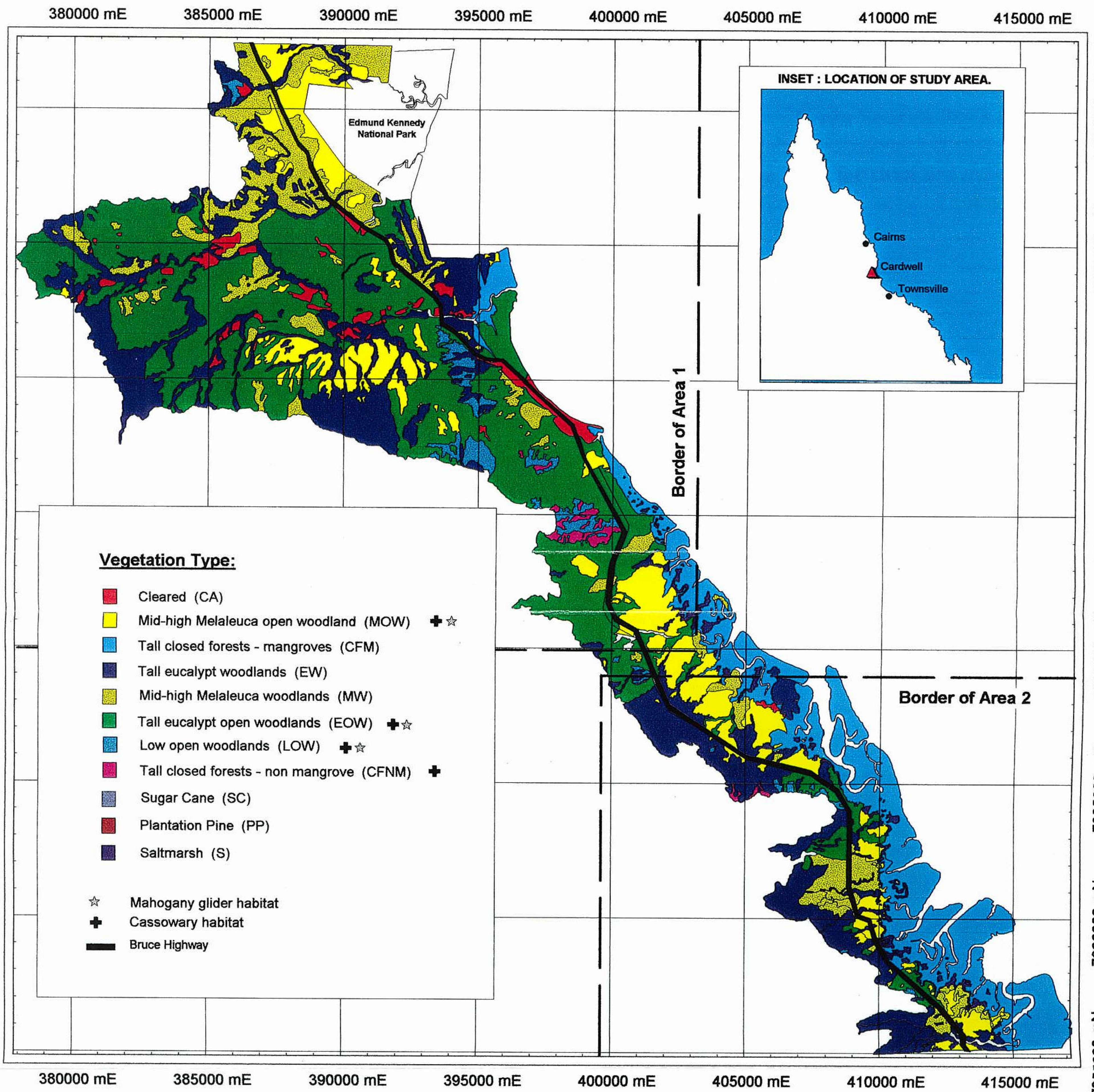
2.3 RESULTS

2.3.1 General

The final vegetation maps produced are presented as Figures 2.1 (1942) and 2.2 (1992). So that vegetation patterns are more readily identifiable, only main roads have been included on the final version of these maps. Data associated with the polygons of these maps form the basis of the subsequent analyses reported in this section. A total of eleven mapping units (hereafter vegetation types) were recognised by the mapping process (Table 2.3), eight of which were remnant plant communities.

The total area of remnant vegetation was reduced by nearly 7,000 ha (29%) across the entire study area (Table 2.4). This clearing occurred predominantly in the north-western section of the study area (Figure 2.2), and equates to an annual clearing rate of native vegetation of approximately 140 ha yr^{-1} (0.6%). A large decrease in the number of remnant patches of vegetation was also recorded, with the highest losses occurring for small (<20 ha) patches (Figure 2.3). The size class results were also significantly different between time periods ($p<0.05$, $\chi^2=42.66$, df=5). Average patch area decreased by over 12 ha, whilst mean

Figure 2.1 Vegetation map of the entire study area from aerial photographs taken in 1942.



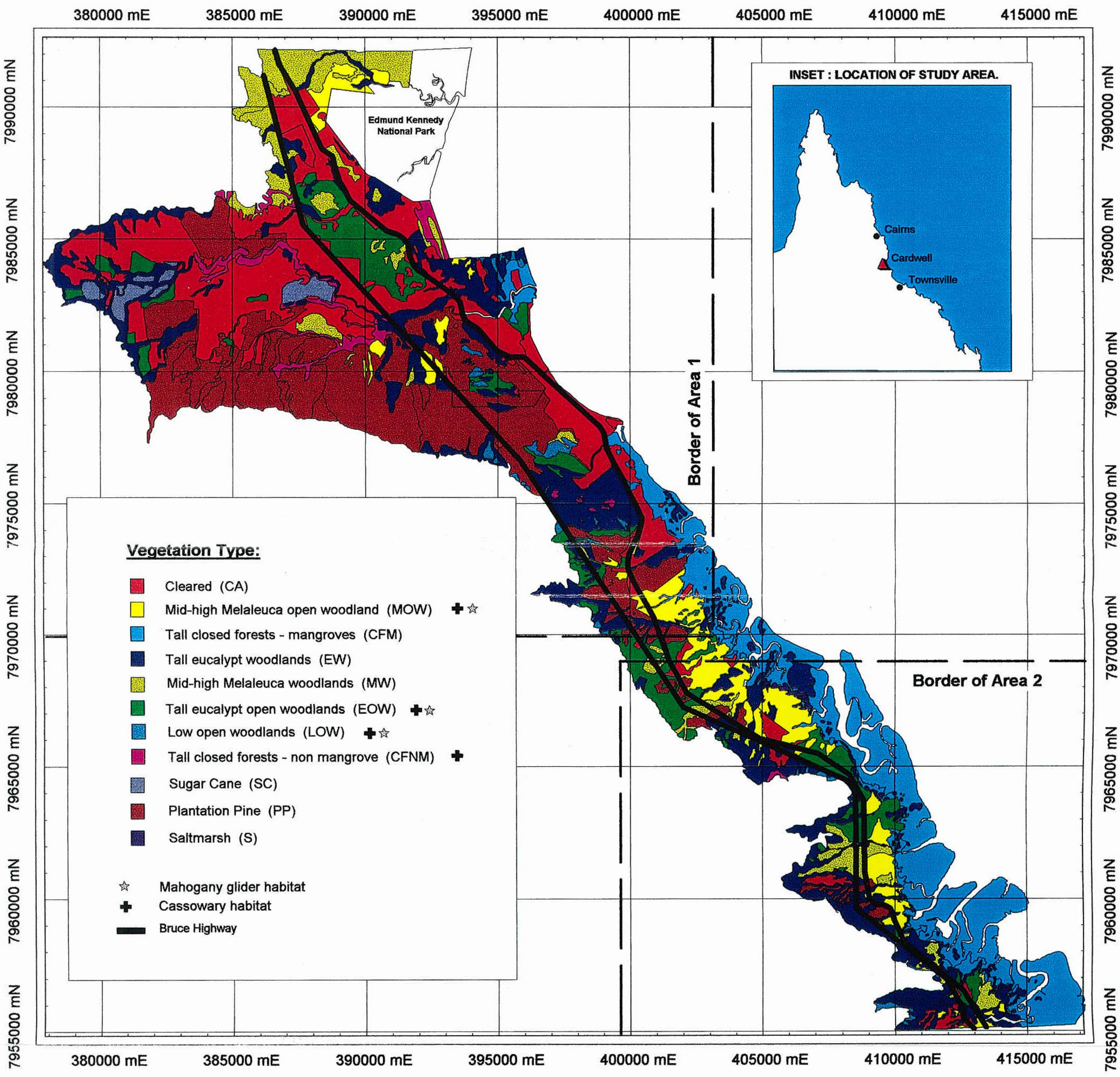
Terra Search Pty Ltd

Vegetation Map for the Cardwell Region Taken From 1942 Aerial Photographs

0 10 20 30 40 50 Km

Originator:	S.Skull	Date:	Oct 96	Drawn:	J.Brealey	Date:	April 98
Scale:	As Shown	Plan No:	Veg42.pgl				

Figure 2.2 Vegetation map of the entire study area from aerial photographs taken in 1992.



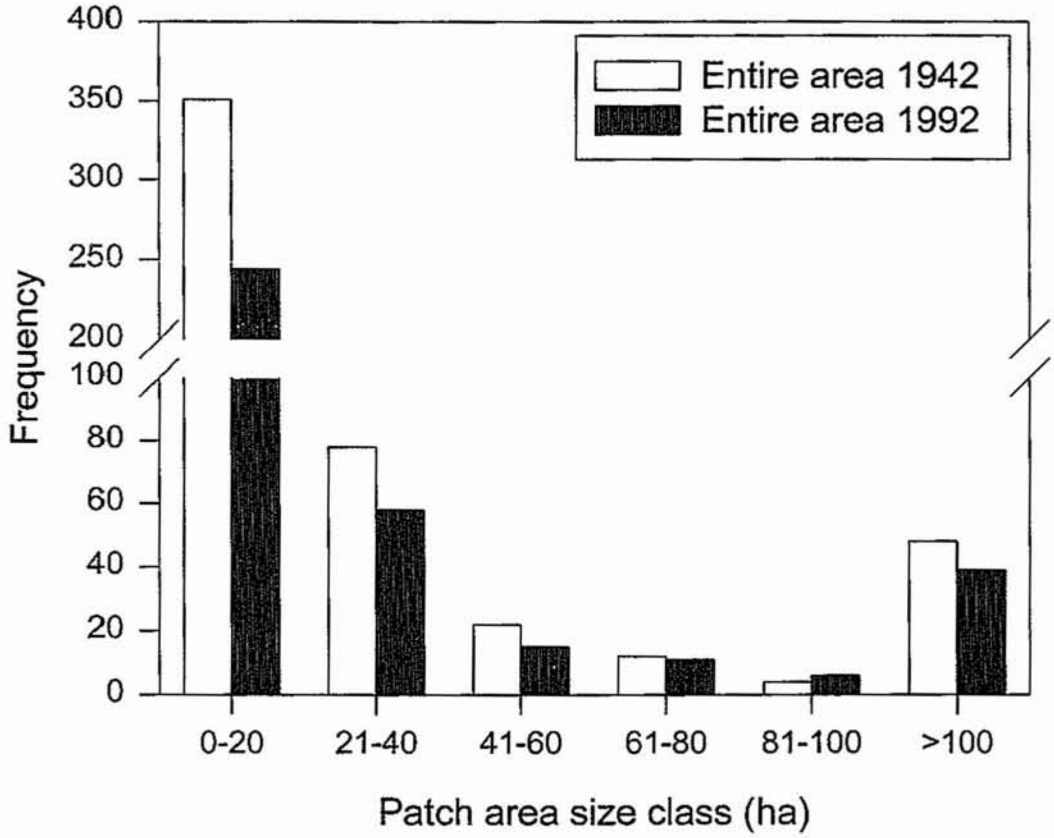
Terra Search Pty Ltd

Vegetation Map for the Cardwell Region Taken From 1992 Aerial Photographs

0 10 20 30 40 50 Km

Originator:	S.Skull	Date:	Oct 96	Drawn:	J.Brealey	Date:	April 98
Scale:	As Shown	Plan No:	Veg92.pgl				

Figure 2.3 Patch size class (ha) distribution results for all remnant vegetation types.


Entire area 1942
Entire area 1992
Frequency
400
350
300
250
200
100
80
60
40
20
0
0-20
21-40
41-60
61-80
81-100
>100
Patch area size class (ha)

perimeter length, shape index and dispersion index all recorded slight increases (Table 2.4). The fragmentation index decreased slightly over the 50 year period. Of all the variables tested, only the shape index changed significantly (Table 2.4).

Table 2.3 Mapping units (vegetation types) recognised by this investigation.

Vegetation type	Description (after Walker and Hopkins 1990)	Text abbreviation
1	Cleared areas (predominantly urban)	CA
2	Mid-high *Melaleuca* open woodlands	MOW
3	Tall closed forests (mangroves)	CFM
4	Tall eucalypt woodlands	EW
5	Mid-high *Melaleuca* woodlands	MW
6	Tall eucalypt open woodlands	EOW
7	Low open woodlands	LOW
8	Tall closed forests (non-mangroves)	CFNM
9	Sugar cane	SC
10	Plantation pine	PP
11	Saltmarsh	S

2.3.2 Total areas of vegetation types

For abbreviations used in the following text see Table 2.3. Over the entire study area the three disturbed "vegetation types" all exhibited a marked increase in total area over the 50 year period (Figure 2.4a). The single largest increase was recorded for plantation pine (6058 ha). Other large increases included almost 800% for cleared areas (CA) and over 300 ha of sugar cane (SC). The largest reductions of remnant vegetation types were recorded for tall eucalypt open woodlands (EOW) (78%) and low open woodlands (LOW) (56%). *Melaleuca* woodlands (MW) and *Melaleuca* open woodlands (MOW) were reduced by 30% and 53%

Figure 2.4 **Total area (ha) of each vegetation type in 1942 and 1992.**

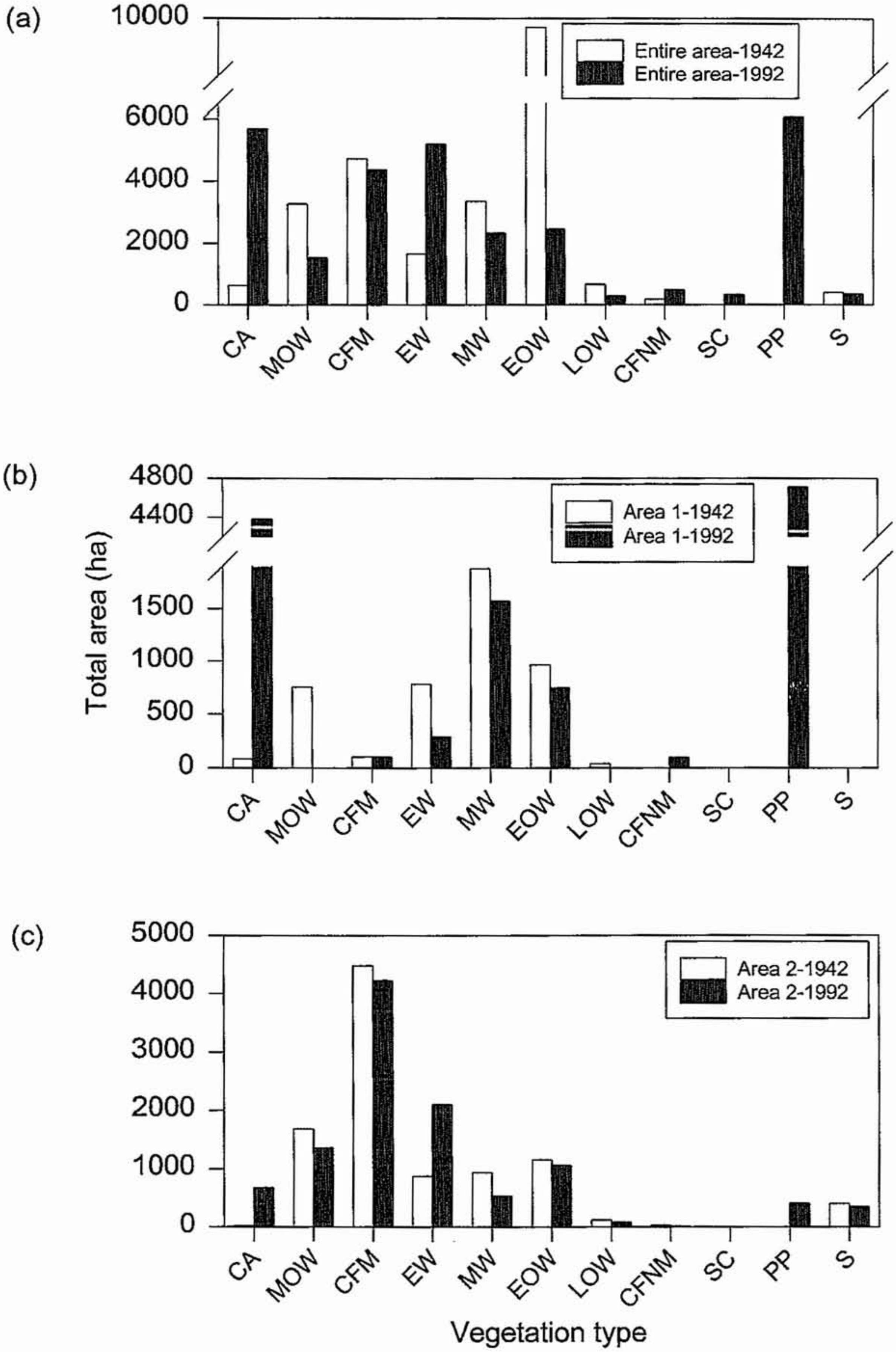
(a) **Entire area**

(b) **Area 1**

(c) **Area 2**

Legend (from Table 2.3):

CA	Cleared areas
MOW	*Melaleuca* mid-high open woodlands
CFM	Tall closed forests (mangroves)
EW	Tall eucalypt woodlands
MW	Mid-high *Melaleuca* woodlands
EOW	Tall eucalypt open woodlands
LOW	Low open woodlands
CFNM	Tall closed forests (non-mangroves)
SC	Sugarcane
PP	Plantation pine
S	Saltmarsh


(a)
10000
6000
4000
2000
0
Entire area-1942
Entire area-1992
CA
MOW
CFM
EW
MW
EOW
LOW
CFNM
SC
PP
S
(b)
4800
4400
1500
1000
500
0
Area 1-1942
Area 1-1992
Total area (ha)
CA
MOW
CFM
EW
MW
EOW
LOW
CFNM
SC
PP
S
(c)
5000
4000
3000
2000
1000
0
Area 2-1942
Area 2-1992
CA
MOW
CFM
EW
MW
EOW
LOW
CFNM
SC
PP
S
Vegetation type

respectively. Relatively small decreases were recorded for mangroves (CFM) and saltmarshes (S) (14% and 7.5 % respectively). Interestingly, both eucalypt woodlands (EW) and non-mangrove closed forests (CFNM) exhibited an overall increase over the 50 year period.

Table 2.4 **Results of variables describing remnant vegetation for the entire study area.** Where appropriate, standard errors are given in parentheses. Significant t-test results ($p \leq 0.05$) are highlighted in bold.

Variable	1942	1992	T-test p value
Total area (ha)	23,916	17,024	n/a
Total no. of patches	515	373	n/a
Mean patch area (ha)	58.30 (8.4)	45.65 (5.33)	0.25
Mean perimeter length (km)	4.64 (0.47)	4.72 (0.43)	0.91
Mean shape index	1.88 (0.04)	2.01 (0.06)	**0.05**
Mean fragmentation index	2.53 (0.08)	2.39 (0.02)	0.38
Mean dispersion index	1.19 (0.36)	1.22 (0.52)	0.37

In specific areas the trends were either similar, amplified or diminished when compared with the entire study area (Figures 2.4b and 2.4c). For example, MW were reduced by approximately the same relative amount in each of the three areas, whereas MOW were completely lost from Area 1, and less EOW were lost from Areas 1 and 2 compared with the entire study area.

2.3.3 Total number of patches of vegetation types

The total number of all vegetation patches (including disturbed types) decreased from 566 to 536 over the 50 year time period. In the majority of cases, the total number of patches of each vegetation type for each area reflected the changes in total area (Figures 2.5a-c). Some exceptions to this are noteworthy, including an increase in the number of patches for CFM and EOW, despite an overall reduction in the total area present. Over the entire study

Figure 2.5 **Total number of patches of each vegetation type in 1942 and 1992.**

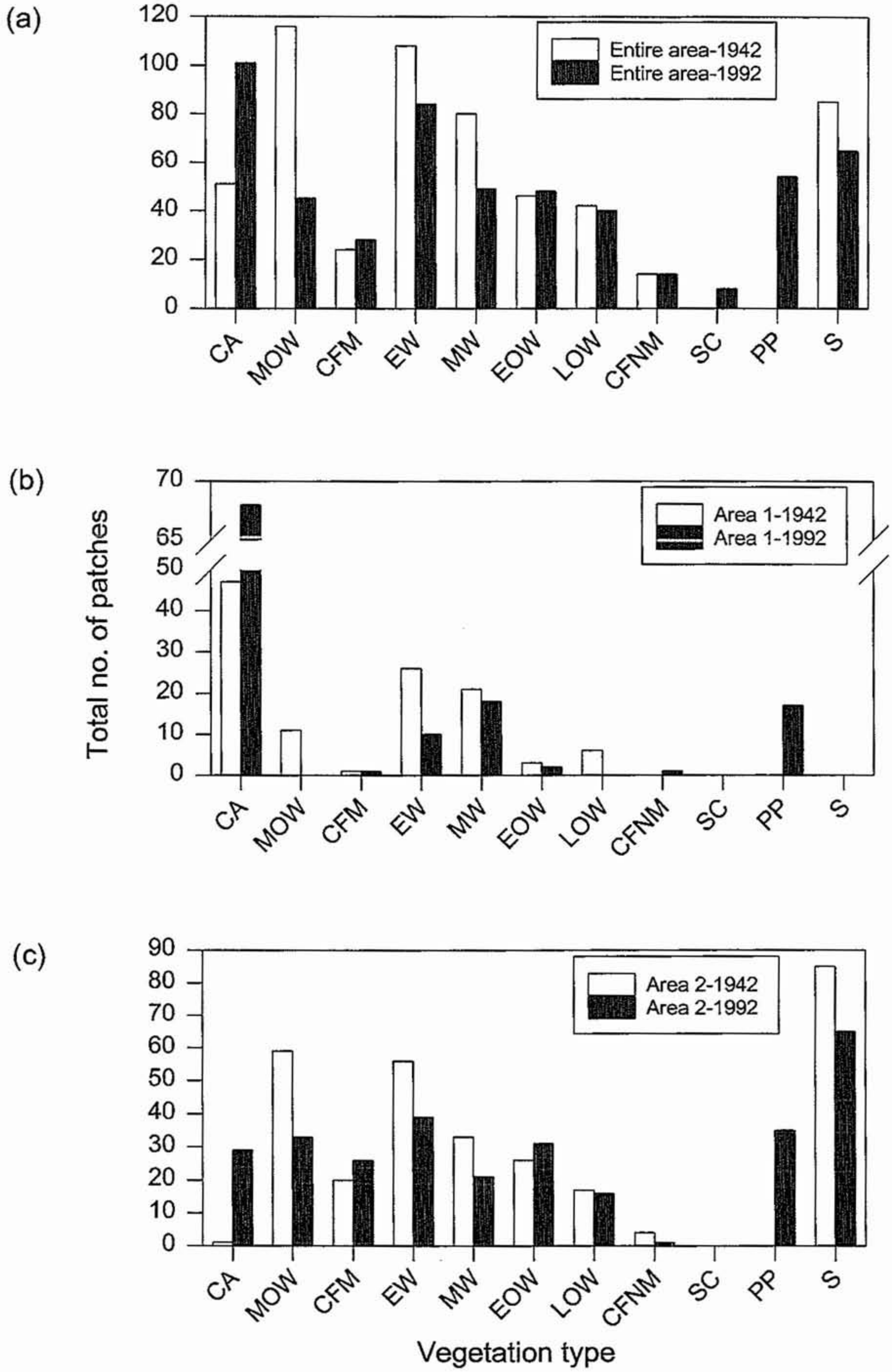
(a) **Entire area**

(b) **Area 1**

(c) **Area 2**

Legend (from Table 2.3):

CA	Cleared areas
MOW	*Melaleuca* mid-high open woodlands
CFM	Tall closed forests (mangroves)
EW	Tall eucalypt woodlands
MW	Mid-high *Melaleuca* woodlands
EOW	Tall eucalypt open woodlands
LOW	Low open woodlands
CFNM	Tall closed forests (non-mangroves)
SC	Sugar cane
PP	Plantation pine
S	Saltmarsh


(a)
120
100
80
60
40
20
0
Entire area-1942
Entire area-1992
CA
MOW
CFM
EW
MW
EOW
LOW
CFNM
SC
PP
S
(b)
70
65
50
40
30
20
10
0
Total no. of patches
Area 1-1942
Area 1-1992
CA
MOW
CFM
EW
MW
EOW
LOW
CFNM
SC
PP
S
(c)
90
80
70
60
50
40
30
20
10
0
Area 2-1942
Area 2-1992
CA
MOW
CFM
EW
MW
EOW
LOW
CFNM
SC
PP
S
Vegetation type

area, MOW lost the highest relative proportion of patches (61%), with MW rating second (39%). The other most affected vegetation types were S (25%) and EW (22%). As for total area, these figures varied between the areas examined, the most notable of which is the complete loss of MOW from Area 1, a highly disturbed landscape.

2.3.4 Patch size class distributions of selected vegetation types

The majority of MOW patches fell within either the 0-5 ha or >30 ha size class (Figure 2.6a). Losses in these categories were 74% and 48% respectively. As indicated above (see Section 2.3.2), MOW were completely cleared in Area 1, so all size classes in this region were lost (Figure 2.6b). Other classes with lower frequencies lost even higher percentages across the entire area, e.g. 83% of the 16-20 ha size class, with this class totally removed from Area 2 (Figure 2.6c).

Trends in the patch size class distributions of three other vegetation types (EW, MW and EOW) are presented in Figures 2.7a-c. Tall eucalypt woodlands (EW) were reduced in a similar pattern to MOW over the entire area, with the major losses occurring in both the smallest and largest size classes. Unlike the MOW, the third size class recorded an increase in frequency. Highest losses for MW occurred in the 41-60 ha category, with large proportional increases and decreases in the 0-20 and 21-40 ha size classes respectively. Tall eucalypt open woodlands (EOW) also exhibited a unique pattern of change, with the smallest category (0-25 ha) recording a 142% increase. The largest losses were recorded in the 51-75 ha (57%) and >150 ha categories (70%). As for EW, a slight increase was also recorded in the second largest patch size class.

Results of the chi-squared analyses of the size class frequency distribution and number of patches data are presented in Table 2.5. All observed (1992) distributions were significantly different from the expected (1942) distribution. Chi-squared values were particularly high for size class distributions over the entire area for MOW, EW and MW. No result is provided for MOW in Area 1 as all expected frequencies were less than 5, and the vegetation type was absent in 1992. For the number of patches, chi-squared values were exceptionally high for Area 1 and the entire study area.

Figure 2.6 Patch size class distributions for mid-high *Melaleuca* open woodlands (MOW) in 1942 and 1992.

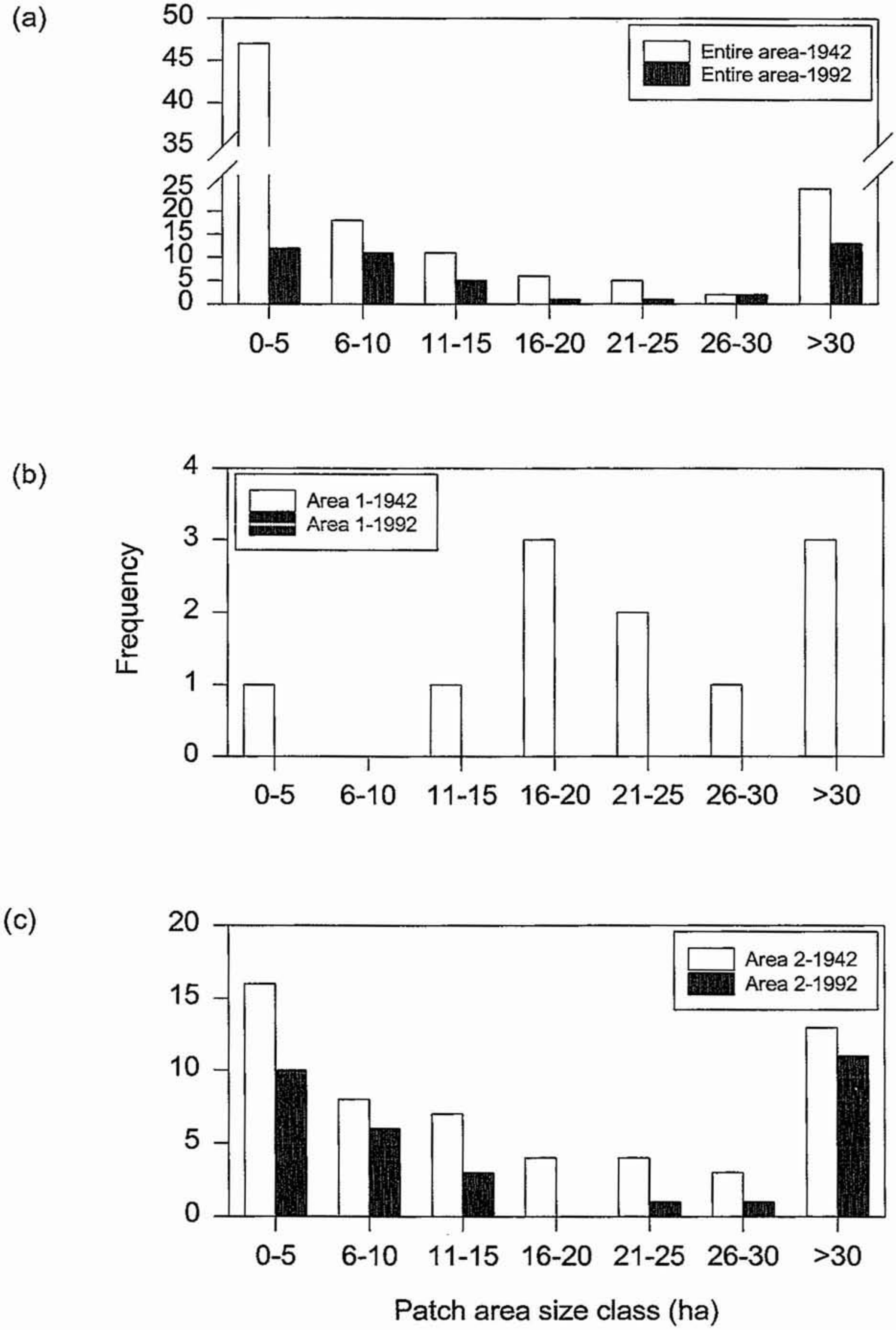
(a) Entire area

(b) Area 1

(c) Area 2

Legend (from Table 2.3):

CA	Cleared areas
MOW	*Melaleuca* mid-high open woodlands
CFM	Tall closed forests (mangroves)
EW	Tall eucalypt woodlands
MW	Mid-high *Melaleuca* woodlands
EOW	Tall eucalypt open woodlands
LOW	Low open woodlands
CFNM	Tall closed forests (non-mangroves)
SC	Sugar cane
PP	Plantation pine
S	Saltmarsh



(a)
50
45
40
35
25
20
15
10
5
0
Entire area-1942
Entire area-1992
0-5
6-10
11-15
16-20
21-25
26-30
>30
(b)
4
3
2
1
0
Area 1-1942
Area 1-1992
Frequency
0-5
6-10
11-15
16-20
21-25
26-30
>30
(c)
20
15
10
5
0
Area 2-1942
Area 2-1992
0-5
6-10
11-15
16-20
21-25
26-30
>30
Patch area size class (ha)

Figure 2.7 Patch size class distributions for three vegetation types within the entire study area in 1942 and 1992.

(a) Tall eucalypt woodland (EW)

(b) Mid-high *Melaleuca* woodlands (MW)

(c) Tall eucalypt open woodlands (EOW)

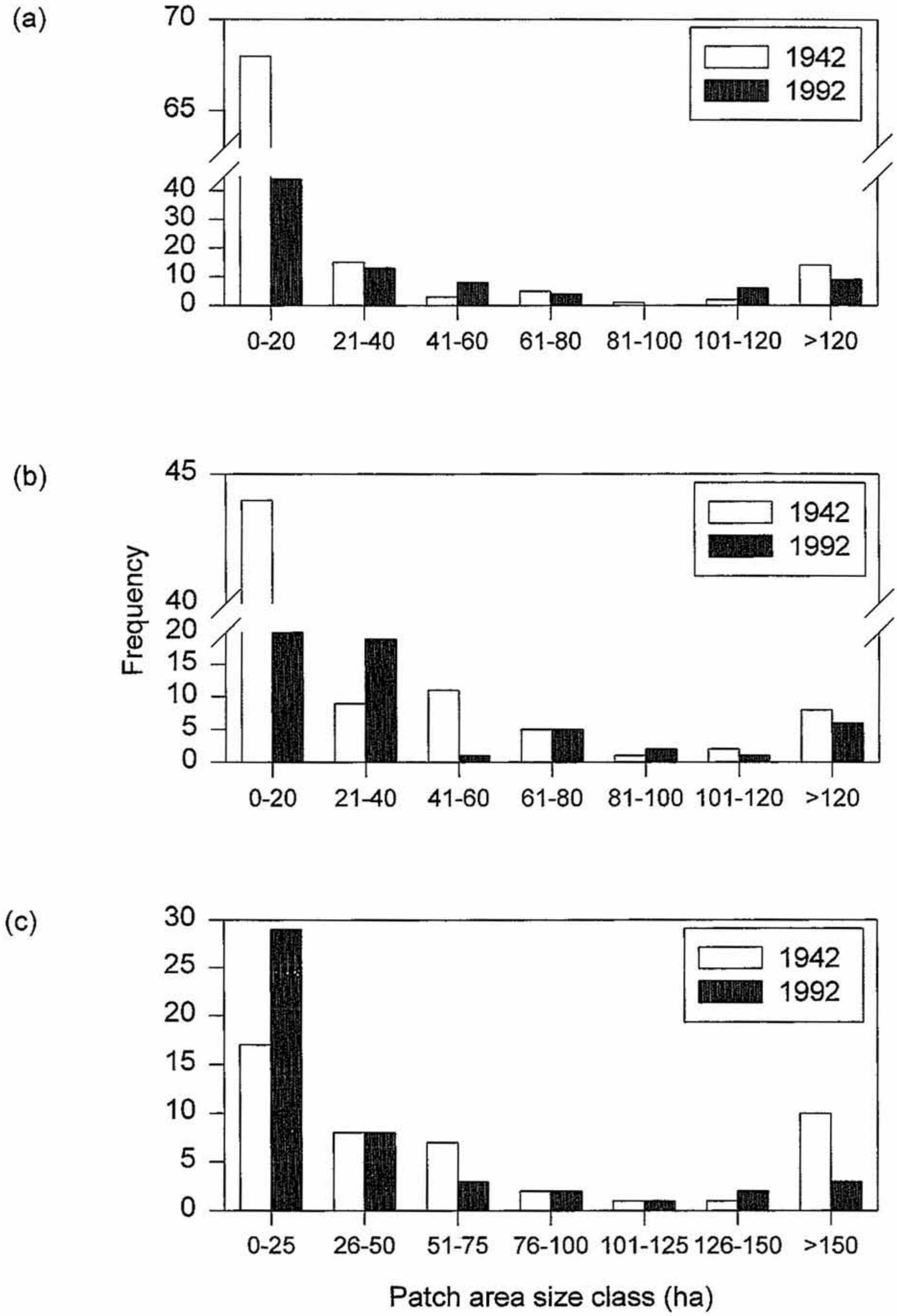

(a)
70
65
40
30
20
10
0
1942
1992
0-20
21-40
41-60
61-80
81-100
101-120
>120
(b)
45
40
20
15
10
5
0
Frequency
1942
1992
0-20
21-40
41-60
61-80
81-100
101-120
>120
(c)
30
25
20
15
10
5
0
1942
1992
0-25
26-50
51-75
76-100
101-125
126-150
>150
Patch area size class (ha)

Table 2.5 **Results of chi-squared (χ^2) analyses for different vegetation types in the study areas.** Data also includes analysis for total number of patches in the three areas examined. All significant differences ($p \leq 0.05$) are highlighted in bold. The critical χ^2 values for these tests are as follows: df=5, 11.07; df=6, 12.59; and df=8, 16.91.

Vegetation type/Area	χ^2	df	p value
MOW - Area 1	-	-	-
MOW - Area 2	12.93	6	**0.025<p<0.05**
MOW - Entire area	45.19	6	**<0.001**
EW - Entire area	21.39	6	**0.001<p<0.005**
MW - Entire area	33.79	6	**<0.001**
EOW - Entire area	15.91	6	**0.01<p<0.025**
Total number of patches			
Area 1	15.38	8	**<0.001**
Area 2	32.75	5	**<0.001**
Entire area	155.76	8	**<0.001**

2.3.5 Vegetation types and patch areas

The mean patch areas of each vegetation type in 1942 and 1992, within the three different areas, are presented in Figures 2.8a-c. Of all the vegetation types in the entire study area, CFM and EOW (although only in 1942) had relatively high patch areas (Figure 2.8a). In Area 1, the patch area of cleared areas (CA) had increased to greater than that of the MOW that were present in the area in 1942 (Figure 2.8b). *Melaleuca* woodlands (MW) retained relatively high patch areas here. In Area 2, mangroves (CFM) again had the highest patch area, similar to those of the entire area recorded for PP, MOW, EW and MW (Figure 2.8c).

Figure 2.8 **Mean patch area (ha) for each vegetation type in 1942 and 1992.** Significant differences ($p \leq 0.05$) are indicated with an asterisk * and "near significant" results ($0.05 < p < 0.10$) are indicated by an asterisk in parentheses (*).

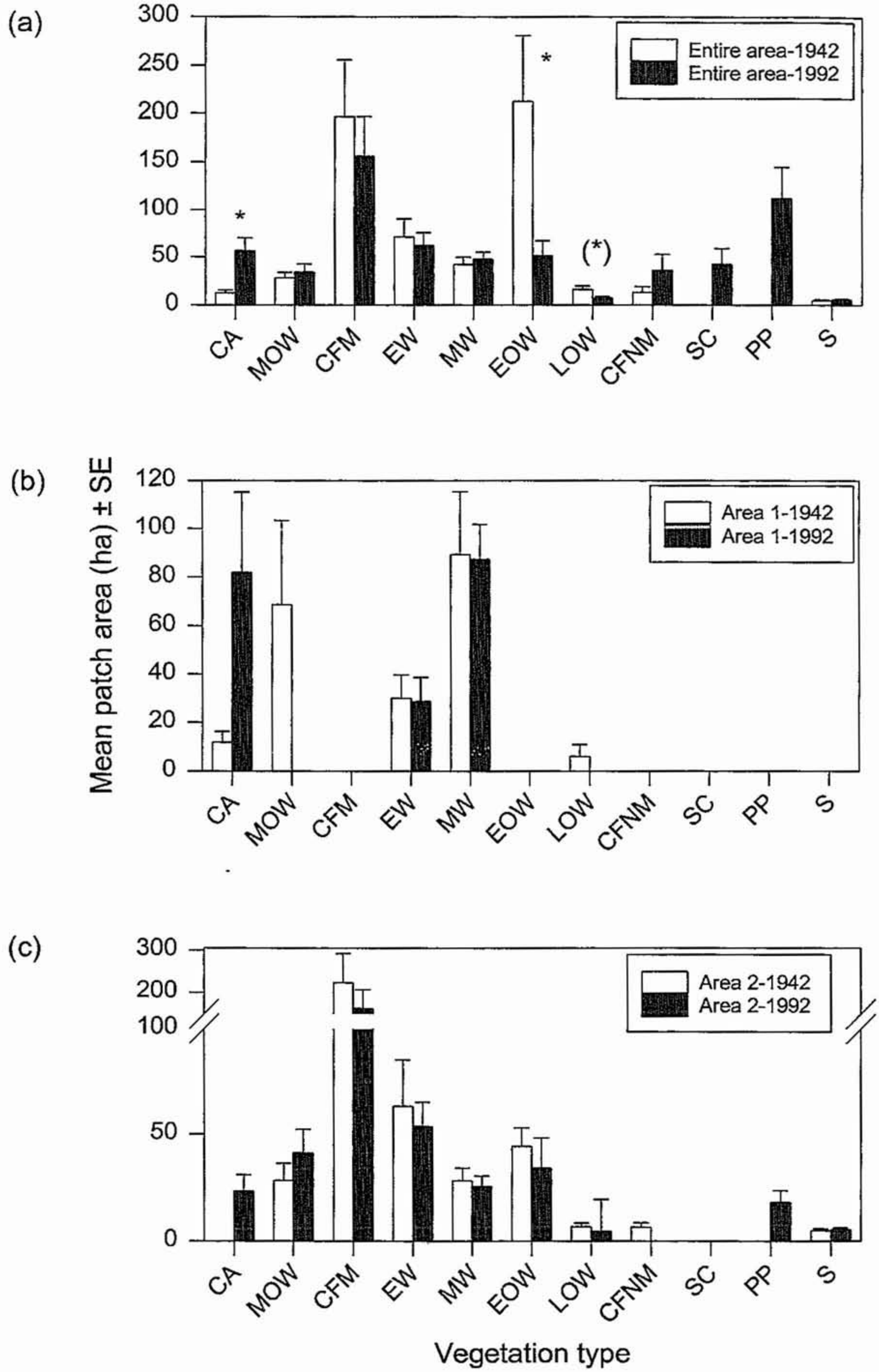
(a) **Entire area**

(b) **Area 1**

(c) **Area 2**

Legend (from Table 2.3):

CA	Cleared areas
MOW	*Melaleuca* mid-high open woodlands
CFM	Tall closed forests (mangroves)
EW	Tall eucalypt woodlands
MW	Mid-high *Melaleuca* woodlands
EOW	Tall eucalypt open woodlands
LOW	Low open woodlands
CFNM	Tall closed forests (non-mangroves)
SC	Sugar cane
PP	Plantation pine
S	Saltmarsh


(a)
Entire area-1942
Entire area-1992
(b)
Area 1-1942
Area 1-1992
(c)
Area 2-1942
Area 2-1992
Mean patch area (ha) ± SE
CA
MOW
CFM
EW
MW
EOW
LOW
CFNM
SC
PP
S
Vegetation type

Results of the unbalanced 3-way ANOVA indicate that for all records, patch area was significantly different between areas and vegetation types but not time period (Table 2.6). Where vegetation types were present in both time periods, a majority (68%) recorded a decrease in patch area across all areas examined (Figures 2.8a-c).

Table 2.6 **Results of unbalanced 3-way ANOVA's of each landscape variable and area, vegetation type and time period.** F ratios and their significance (in parentheses) are given. Significant results ($p \leq 0.05$) are highlighted in bold and near significant results ($0.05 < p < 0.10$) are highlighted in italics.

Variable	Area	Vegetation type	Time
df	2	10	1
Patch area	3.06 **(0.05)**	17.17 **(0.00)**	1.29 (0.26)
Perimeter length	2.49 *(0.08)*	21.21 **(0.00)**	0.13 (0.71)
Shape index	1.10 (0.33)	26.05 **(0.00)**	6.03 **(0.01)**
Fragmentation index	3.30 **(0.04)**	17.38 **(0.00)**	7.41 **(0.01)**
Dispersion index	4.31 **(0.01)**	6.14 **(0.00)**	1.57 (0.21)

For individual areas and time periods (1-way ANOVA), patch area was significantly different between vegetation types, except in Area 1 in 1992 (Table 2.7). In Area 1 in 1942, although a significant result was obtained for all vegetation types ($p=0.02$), no significant difference was recorded between any two individual types. In Area 2, a majority of groups were significantly different to CFM for both time periods (Tables A1 and A2, Appendix A). For the entire area in 1942, Tukey's HSD test indicated many types were significantly different to CFM and EOW (Table A3, Appendix A). In 1992, however, all types except CFNM and sugar cane (SC) were significantly different to CFM and, in addition, S and LOW were significantly different to PP (Table A4, Appendix A).

The only significant differences for individual vegetation types over the 50 year time period (t-test results) were recorded within the entire study area for CA and EOW, whilst LOW recorded a "near significant" result with $p=0.06$ (Table 2.8a). Area 1 and 2 recorded no significant differences (Tables 2.8b and 2.8c).

Table 2.7 **Results of one-way ANOVA's of each landscape variable and vegetation type.** F ratios and their significance (in parentheses) are given. For significant results ($p \leq 0.05$ and highlighted in bold), multiple comparison test data (Tukey's-HSD) are presented in Appendix A. Near significant results ($0.05 < p < 0.10$) are highlighted in italics.

Variable	Area 1		Area 2		Entire area	
	1942	1992	1942	1992	1942	1992
df	4	3	7	8	8	10
Patch area	3.15 **(0.02)**	1.46 (0.24)	9.78 **(0.00)**	9.89 **(0.00)**	8.95 **(0.00)**	4.74 **(0.00)**
Perimeter length	2.15 *(0.09)*	2.22 (0.12)	8.91 **(0.00)**	10.97 **(0.00)**	8.22 **(0.00)**	6.01 **(0.00)**
Shape index	0.75 (0.56)	0.80 (0.50)	4.97 **(0.00)**	16.92 **(0.00)**	8.26 **(0.00)**	10.27 **(0.00)**
Fragmentation index	2.31 *(0.07)*	0.31 (0.74)	2.63 **(0.01)**	10.53 **(0.00)**	23.51 **(0.00)**	3.03 **(0.00)**
Dispersion index	1.26 (0.30)	2.11 (0.14)	2.94 **(0.01)**	4.35 **(0.00)**	4.66 **(0.00)**	2.25 **(0.01)**

2.3.6 Vegetation types and patch perimeter lengths

The average perimeter lengths for each vegetation type over the 50 year time period in each of the study areas are presented in Figures 2.9a-c. As with patch area, CFM and EOW initially had high mean perimeter lengths for the entire area. Closed forest (mangroves) retained this high value in 1992 whereas EOW changed significantly (Figure 2.9a, Table 2.8a). Other significant differences were recorded for both CA and LOW (Table 2.8a). No significant differences were recorded for any individual vegetation type in either Area 1 or

Figure 2.9 **Mean perimeter length (km) for each vegetation type in 1942 and 1992.** Significant differences ($p \leq 0.05$) are indicated with an asterisk * and "near significant" results ($0.05 < p < 0.10$) are indicated by an asterisk in parentheses (*).

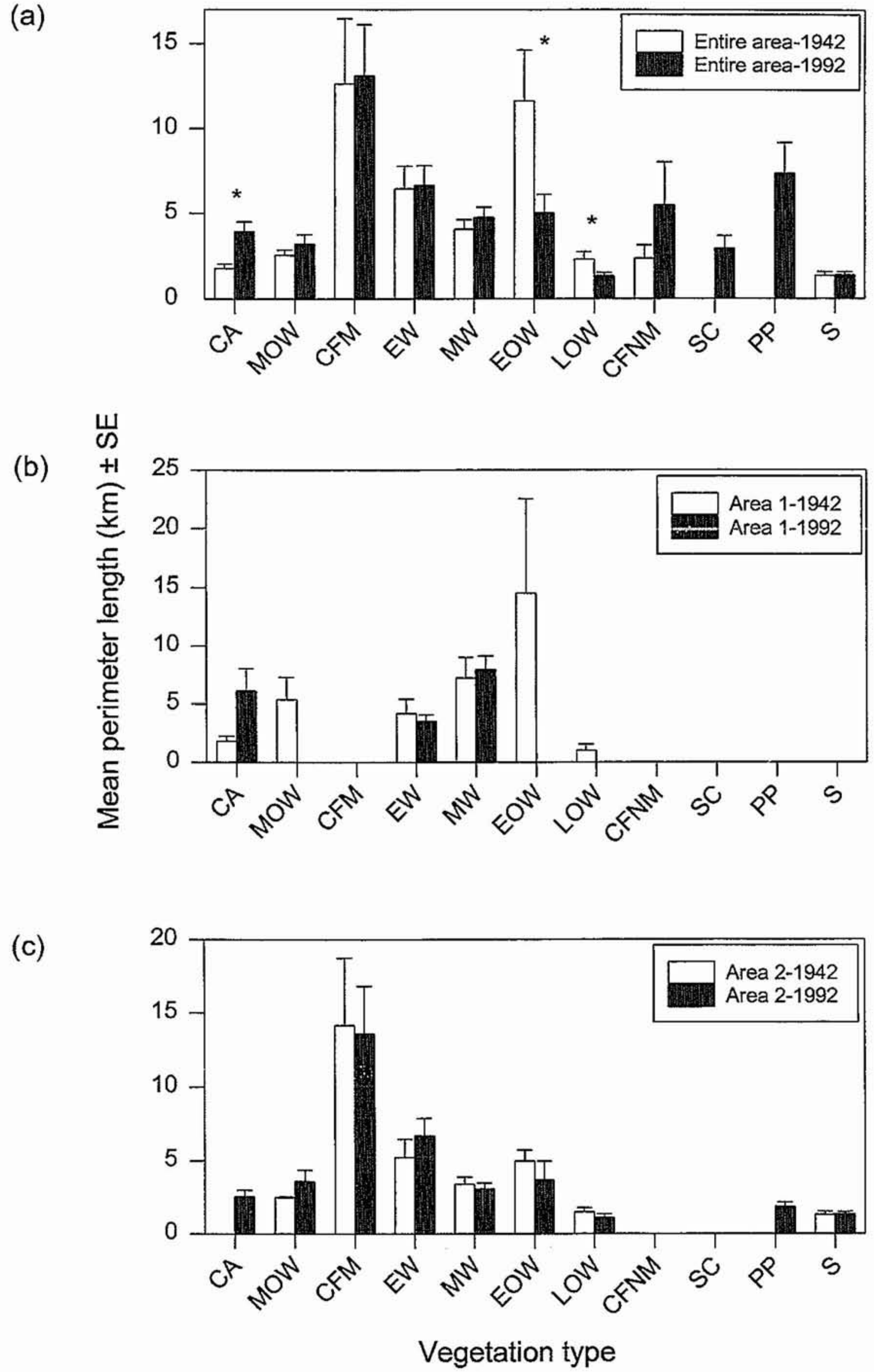
(a) **Entire area**

(b) **Area 1**

(c) **Area 2**

Legend (from Table 2.3):

CA	Cleared areas
MOW	*Melaleuca* mid-high open woodlands
CFM	Tall closed forests (mangroves)
EW	Tall eucalypt woodlands
MW	Mid-high *Melaleuca* woodlands
EOW	Tall eucalypt open woodlands
LOW	Low open woodlands
CFNM	Tall closed forests (non-mangroves)
SC	Sugar cane
PP	Plantation pine
S	Saltmarsh


(a)
Entire area-1942
Entire area-1992
(b)
Area 1-1942
Area 1-1992
(c)
Area 2-1942
Area 2-1992
Mean perimeter length (km) ± SE
CA
MOW
CFM
EW
MW
EOW
LOW
CFNM
SC
PP
S
Vegetation type

2 over the 50 year time period (Tables 2.8b and 2.8c). In Area 1, mean perimeter lengths were generally higher for each vegetation type compared with the entire area (particularly for PP). In Area 2, mean perimeter lengths reflected the trends observed over the entire area.

Table 2.8 (a) **Results of the t-tests conducted on individual vegetation types within the entire area at the two times examined.** Significant results ($p \leq 0.05$) are highlighted in bold and near significant results ($0.05 < p < 0.10$) are highlighted in italics. Vegetation types SC and PP were excluded as they were only recorded in 1992.

Vegetation type	Patch area	Perimeter length	Shape index	Fragmentation index	Dispersion index
CA	**0.02**	**0.01**	**0.02**	**0.02**	*0.09*
MOW	0.57	0.29	0.28	**0.05**	**0.01**
CFM	0.56	0.92	**0.05**	0.68	0.45
EW	0.71	0.91	0.23	0.67	0.84
MW	0.63	0.43	0.21	0.12	0.35
EOW	**0.02**	**0.05**	*0.09*	0.85	**0.02**
LOW	*0.06*	**0.05**	**0.02**	0.86	0.18
CFNM	0.21	0.25	0.37	0.63	0.53
S	0.62	0.92	0.60	**0.01**	0.90

For all areas, times and vegetation types combined, however, the results of the 3-way ANOVA indicated perimeter length was significantly different between vegetation types, near significant ($0.05 < p < 0.10$) for area and not significant for time (Table 2.6). Where vegetation types were present at both times, a majority (63%) exhibited an increase in perimeter length across all the areas studied (Figures 2.9a-c).

Table 2.8 (b) **Results of the t-tests conducted on individual vegetation types within Area 1 at the two times examined.** Significant results ($p \leq 0.05$) are highlighted in bold and near significant results ($0.05 < p < 0.10$) are highlighted in italics. Vegetation types not recorded in the area for both sampling times were excluded from the analysis.

Vegetation type	Patch area	Perimeter length	Shape index	Fragmentation index	Dispersion index
CA	0.15	0.13	0.21	0.21	0.16
EW	0.94	0.74	0.87	0.87	0.16
MW	0.95	0.75	0.29	0.29	0.74

Table 2.8 (c) **Results of the t-tests conducted on individual vegetation types within Area 2 at the two times examined.** Significant results ($p \leq 0.05$) are highlighted in bold and near significant results ($0.05 < p < 0.10$) are highlighted in italics. Vegetation types CA, CFNM, SC and PP were excluded as they were only recorded in 1992.

Vegetation type	Patch area	Perimeter length	Shape index	Fragmentation index	Dispersion index
MOW	0.36	0.15	0.45	**0.04**	0.19
CFM	0.44	0.92	*0.08*	0.79	0.16
EW	0.74	0.42	**0.02**	0.46	0.64
MW	0.74	0.63	0.50	0.31	0.40
EOW	0.57	0.41	**0.04**	0.52	*0.08*
LOW	0.36	0.32	0.27	0.20	0.95
S	0.62	0.92	0.60	**0.01**	0.62

Perimeter lengths were significantly different between vegetation types for individual areas and times, except in Area 1 (Table 2.7). Within Area 2, CFM and EW exhibited significant differences from most other vegetation types, particularly S (Tables A5 and A6, Appendix A). For the entire area, most types were significantly different from both CFM and EOW in 1942, with S also significantly different from EW (Table A7, Appendix A). In 1992, however, although most types remained statistically different from CFM (but not EOW), LOW and EW were also statistically different from PP and EW (Table A8, Appendix A).

2.3.7 Vegetation types and patch shape index

Plots of mean shape index for each vegetation type, area and time are presented in Figures 2.10a-c. Within the entire area, vegetation types with shapes closest to circularity (SI=1) included CA, MOW, LOW and S (Figure 2.9a). Other types recorded more irregular shapes, with CA, CFM and LOW exhibiting significant differences between times (Table 2.8a). The latter recorded a decrease whilst CA and CFM registered an increase. Additionally, EOW recorded a near significant decrease ($p<0.10$) (Table 2.8a).

Within Area 1, nearly all vegetation types had a shape index close to two at both times. Unlike other variables examined thus far, the shape index also produced some statistically significant results for individual vegetation types in Area 2 (Table 2.8c). These included an increase in the irregularity of EW and a decrease in EOW. A near significant result was recorded for the increase in the irregularity of CFM in this area (Table 2.8c). Where particular vegetation types were present at both times, a majority (63%) across all areas recorded an increase in the shape index (Figure 2.10a-c).

For all data combined, the 3-way ANOVA indicated a significant difference for the shape index between vegetation types and times, but not areas (Table 2.6). In specific areas and times, Area 2 and the entire area recorded significant differences between vegetation types in both times (Table 2.7). Within Area 1 for 1942, CFM, EW and EOW were significantly different from other vegetation types, particularly MOW and S (Table A9, Appendix A). A larger difference was recorded for CFM and EW in 1992 (Table A10, Appendix A). These patterns were essentially repeated over the entire study area (Tables A11 and A12, Appendix A). As with the perimeter length data, no significant results were obtained for Area 1.

Figure 2.10 **Mean shape index for each vegetation type in 1942 and 1992.** Significant differences ($p \leq 0.05$) are indicated with an asterisk * and "near significant" results ($0.05 < p < 0.10$) are indicated by an asterisk in parentheses (*).

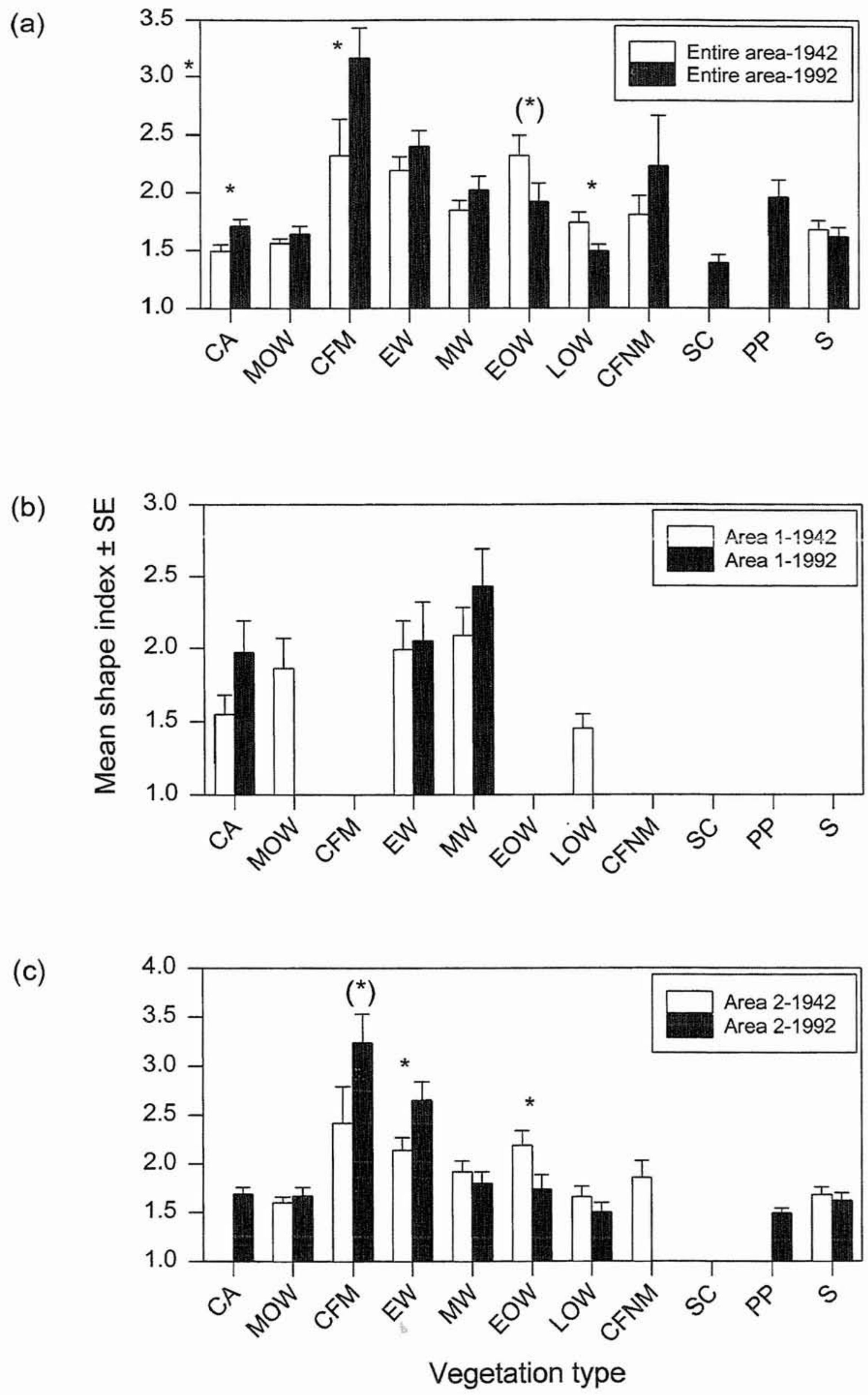
(a) **Entire area**

(b) **Area 1**

(c) **Area 2**

Legend (from Table 2.3):

CA	Cleared areas
MOW	*Melaleuca* mid-high open woodlands
CFM	Tall closed forests (mangroves)
EW	Tall eucalypt woodlands
MW	Mid-high *Melaleuca* woodlands
EOW	Tall eucalypt open woodlands
LOW	Low open woodlands
CFNM	Tall closed forests (non-mangroves)
SC	Sugar cane
PP	Plantation pine
S	Saltmarsh



(a)
Entire area-1942
Entire area-1992
(b)
Area 1-1942
Area 1-1992
(c)
Area 2-1942
Area 2-1992
Mean shape index ± SE
CA
MOW
CFM
EW
MW
EOW
LOW
CFNM
SC
PP
S
Vegetation type

2.3.8 Vegetation types and the fragmentation index

Results for each vegetation type, time period and area are presented in Figures 2.11a-c. This index exhibited marked differences from the shape index. In a majority of cases (67%) a slight decrease in this index was recorded. Five statistically significant differences were recorded for individual vegetation types, including CA, MOW and S over the entire area (Table 2.8a) and MOW and S in Area 2 (Table 2.8c). The largest single decrease for the entire area was recorded for PP. For all data combined, this index is the only variable that recorded a significant difference across areas, vegetation types and times (Table 2.6).

From the 1-way ANOVA results, the dissimilarity of Area 1 (compared with both Area 2 and the entire study area) is again apparent (Table 2.7). For Area 2 in 1942, the only pairwise significant difference was between S and MOW (Table A13, Appendix A). In 1992, however, many vegetation types had significant differences to PP and S, and two statistically significant differences were recorded between LOW and both MOW and MW (Table A14, Appendix A). In the entire area the patterns were more similar across the two time periods, with S most significantly different (Tables A15 and A16, Appendix A). In 1942, LOW recorded significant differences with three other vegetation types (Table A16, Appendix A).

2.3.9 Vegetation types and the dispersion index

The mean dispersion index for each vegetation type, time period and area is presented in Figures 2.12a-c. For data combined, this index produced similar patterns to those recorded for patch areas in that it was significantly different across both area and vegetation type, but not time (Table 2.6).

Results from the 1-way ANOVA were, however, similar to the majority of other variables examined, i.e. vegetation types in Area 2 and the entire area recorded significant differences for both times, whereas the vegetation types in Area 1 did not (Table 2.7). Some statistically significant results were found in Area 2 and the entire area for pairs of vegetation types. In Area 2 S was most significantly different from both EOW and CFM in 1942 (Table A17, Appendix A). Fifty years later, however, saltmarsh recorded significant differences from EW

Figure 2.11 **Mean fragmentation index for each vegetation type in 1942 and 1992.** Significant differences ($p \leq 0.05$) are indicated with an asterisk * and "near significant" results ($0.05 < p < 0.10$) are indicated by an asterisk in parentheses (*).

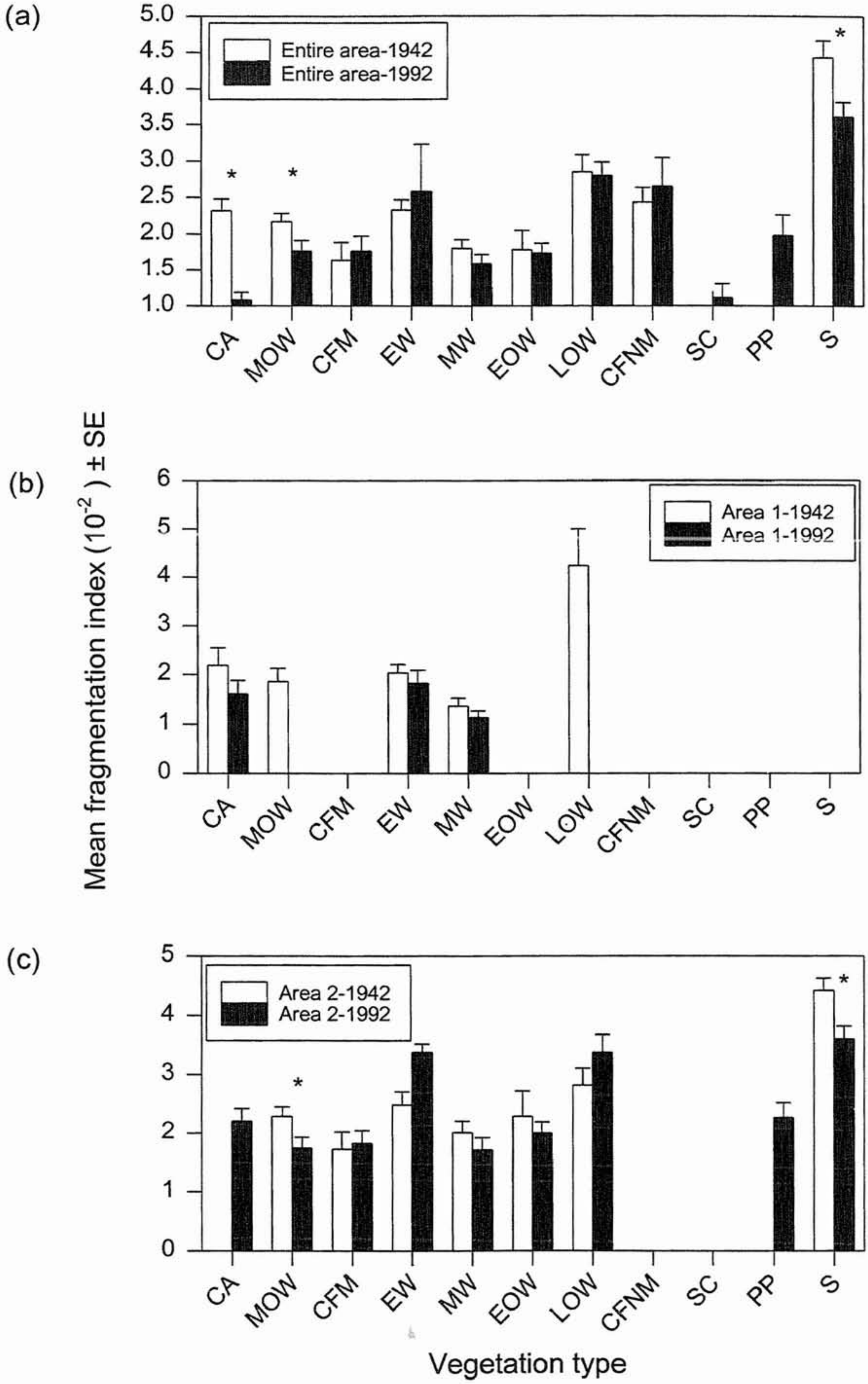
(a) **Entire area**

(b) **Area 1**

(c) **Area 2**

Legend (from Table 2.3):

CA	Cleared areas
MOW	*Melaleuca* mid-high open woodlands
CFM	Tall closed forests (mangroves)
EW	Tall eucalypt woodlands
MW	Mid-high *Melaleuca* woodlands
EOW	Tall eucalypt open woodlands
LOW	Low open woodlands
CFNM	Tall closed forests (non-mangroves)
SC	Sugar cane
PP	Plantation pine
S	Saltmarsh



(a)
Entire area-1942
Entire area-1992
5.0
4.5
4.0
3.5
3.0
2.5
2.0
1.5
1.0
CA
MOW
CFM
EW
MW
EOW
LOW
CFNM
SC
PP
S
(b)
Area 1-1942
Area 1-1992
6
5
4
3
2
1
0
(c)
Area 2-1942
Area 2-1992
5
4
3
2
1
0
Mean fragmentation index (10^{-2}) ± SE
Vegetation type

Figure 2.12 **Mean dispersion index for each vegetation type in 1942 and 1992.** Significant differences ($p \leq 0.05$) are indicated with an asterisk * and "near significant" results ($0.05 < p < 0.10$) are indicated by an asterisk in parentheses (*). Values above the dotted line indicate a random distribution, while those below the line indicate a clumped distribution.

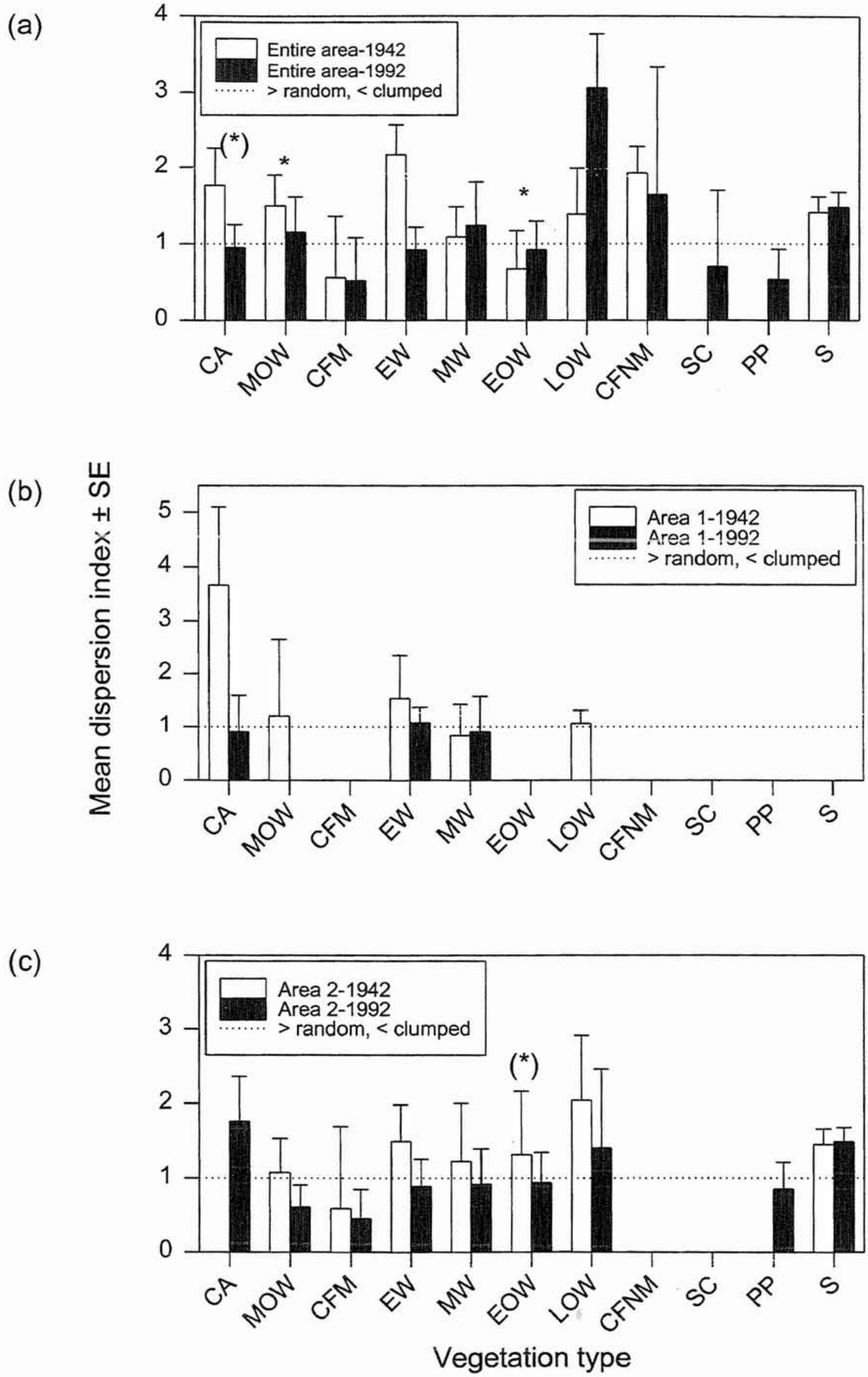
(a) **Entire area**

(b) **Area 1**

(c) **Area 2**

Legend (from Table 2.3):

CA	Cleared areas
MOW	*Melaleuca* mid-high open woodlands
CFM	Tall closed forests (mangroves)
EW	Tall eucalypt woodlands
MW	Mid-high *Melaleuca* woodlands
EOW	Tall eucalypt open woodlands
LOW	Low open woodlands
CFNM	Tall closed forests (non-mangroves)
SC	Sugar cane
PP	Plantation pine
S	Saltmarsh



(a)
Entire area-1942
Entire area-1992
> random, < clumped
(*)
*
*
CA
MOW
CFM
EW
MW
EOW
LOW
CFNM
SC
PP
S
(b)
Area 1-1942
Area 1-1992
> random, < clumped
Mean dispersion index ± SE
(c)
Area 2-1942
Area 2-1992
> random, < clumped
(*)
Vegetation type

and CA, and CA were also different from MOW and PP (Table A18, Appendix A). For the entire area, differences were restricted to S, and these differences were most distinct in 1942 (Tables A19 and A20, Appendix A).

The overriding trend for most vegetation types (72% of cases where a type was present in both times) was a decrease in 1992. Most individual vegetation types retained values close to the critical value of the dispersion index (DI=1). In the entire area PP changed from a random to a clumped distribution, whereas EOW recorded the opposite trend (Figure 2.12a). Both these results were statistically significant at the $p \leq 0.05$ level (Table 2.8a). Cleared areas (CA) and EOW recorded near significant results (both decreases) in the entire area and Area 2 respectively (Figures 2.12a and 2.12b, Tables 2.8a and Table 2.8c).

2.4 DISCUSSION

2.4.1 The broad picture

The annual clearing rate of remnant native vegetation recorded during this study is more than double that found in the only other study available for lowlands in the WTBR (Bianco 1994). The figures for the Mulgrave Shire study did, however, include upland vegetation types on relatively steep slopes, which are characteristically cleared at much lower rates than lowland vegetation (Catterall & Kingston 1993). Both tropical studies fall well short (< 2%) of even the long-term annual clearing rates calculated for south-eastern Queensland. This region continues to exhibit faster population growth than any other in Australia, which has resulted in extreme clearing rates, particularly in recent times (Catterall & Kingston 1993).

The changes in the total numbers of patches for each vegetation type across the landscape suggest that broad-scale clearing has taken place (Figures 2.5a-c). Most types except the disturbed classes, exhibited decreases in the numbers of patches, which indicated patches have predominantly been lost rather than split into several new ones. In fact, it is likely that both processes have occurred, but without a detailed assessment of the fate of individual patches, the relative contribution of these processes cannot be determined.

For all remnant vegetation types considered collectively across the entire study area, the size class distribution of patches has altered dramatically. Decreases in the numbers of smaller size class categories were the most pronounced (Figure 2.3). Most other studies (e.g. Catterall & Kingston 1993; Mladenoff *et al.* 1993; Bianco 1994) have recorded an increase in the smaller patch size classes which may be a result of the type of clearing that has occurred. Broad-scale clearing for urbanisation or plantation pine (mean patch areas >50 and 100 ha respectively, Figure 2.8a) has been the most common method of clearing in the area studied during this investigation, which tends to eliminate small patches rather than create them. The insignificant changes in most patch areas and perimeter lengths for remnant vegetation types support the broad-scale clearing argument. The main exceptions to this are Tall eucalypt woodlands (EW) which are discussed in more detail in Section 2.4.3, and low open woodlands (LOW). Low open woodlands, like *Melaleuca* open woodlands (MOW), were completely removed from Area 2, which has left much smaller patches (p=0.06, Table 2.8a) restricted to the southern section of the study area (Figure 2.2).

2.4.2 The landscape indices

The landscape indices for all remnant vegetation combined indicate a reduced number of smaller patches with slightly higher perimeter lengths and significantly more irregular shapes (Table 2.4). This is likely to render them more susceptible to edge effects which include increases in weed invasion (Laurance 1987; Noss & Csuti 1994; Fox 1995), altered micro-climates and changes in a wide range of other ecological processes (Saunders *et al.* 1991).

The perceived difference in the three separate areas chosen for closer analysis appear to have been validated by the results of the chi-squared analysis (Table 2.5), 3-way ANOVA and 1-way ANOVA (Tables 2.6 and 2.7). Differences in landscape variables between vegetation types (for the same time and area) are much more likely to be significant in an undisturbed landscape (Area 2), or in a very large area exhibiting both extremes of landscape disturbance (entire area) (Table 2.7).

The shape index produced the greatest number of significant differences between individual vegetation types over the 50 year study period. Although a variety of shape indices have been developed, when used in conjunction with patch size these indices are particularly useful for studies which seek to investigate edge effects (Ripple *et al.* 1991). Furthermore, to improve the ecological interpretation of shape indices other authors have called for the development of a shape index that takes into account both the edge to interior ratio and the complexity of the patch perimeter (Hulshoff 1995). Different trends (and strengths of the trends) were recorded for different vegetation types in this study with respect to the shape index (a significant increase for CFM and CA yet a decrease for LOW and EOW over the same area). Increases are of most concern for remnant vegetation as they indicate a further move from circularity, and thus an increased likelihood of management problems posed by edge effects.

Saunders *et al.* (1991) argued that a circular shape is only really important for small areas, so it may be that for the preservation of remnants, decision making should be focused on small areas that have exhibited an increase in the shape index. The suitability of these types of remnants for conservation purposes would also have to be based on ecological data (i.e. the integrity of the remnant) and weighed against resource availability as such shapes will require more energy, time and effort to maintain their internal characteristics (Meffe & Carroll 1994).

Of all the indices examined, only the fragmentation index (a basic area/perimeter ratio) was significantly different over the entire area for times, area and vegetation type (Table 2.6). Interpretation of the results for this index need to be considered carefully, as it is scale invariant (Ripple *et al.* 1991). Other authors have also indicated that simple perimeter/area ratios should be interpreted with caution, as under some conditions different shapes can produce similar ratios (Laurance & Yensen 1991). This index produced results very different to those of the shape index, and was the only variable to record a significant result for changes observed to S vegetation (Tables 2.8a and 2.8b). All other variables examined indicate that this vegetation type has remained relatively unchanged over the 50 year time period. In future studies this index could probably be excluded so long as some attempt to describe patch shape is undertaken.

Changes in the dispersion index, which were significant in different directions for MOW and EOW, could have significant ecological ramifications for these plant communities in the landscape. Alterations to patch dispersion patterns within a landscape could feasibly affect seed dispersal, an effect of fragmentation that is well documented, particularly for wind-dispersed species (Hopkins 1990). Flow-on effects from this change could include changes to population heterozygosity, eventual changes in population demographics, and the rates and types of local extinctions (Wilcox & Murphy 1985). Effects such as these have, however, generally been subject to less research compared with the short-term effects of habitat fragmentation documented for animal populations.

Other indices used to assess landscape pattern include patch complexity, landscape diversity, contagion (the positive or negative association between patches) and a range of others based on fractal geometry. Although these indices have not been investigated in this thesis, they may provide useful insights into the data set in the future. Relatively simple indices, however, have repeatedly been shown to adequately describe changes in landscape pattern (e.g. Hulshoff 1995).

2.4.3 Specific vegetation types of interest

This study has provided the first detailed examination of habitat loss and fragmentation for the tropical lowlands of north-eastern Queensland. Specific figures for each vegetation type have enabled the most threatened ecosystems in the study area (in terms of area and shape alterations) to be identified. Several of these are considered in more detail below.

Tall eucalypt open woodlands (EOW)

In terms of the total area of each vegetation type, EOW were the most affected by clearing across the entire study area. The loss within this region of tropical lowlands (78%) far exceeds the 14% estimated by QDPI (1993) for the entire catchment and even the overall national estimate of 65% (DEST 1995b). The loss is even higher than that calculated for south-eastern Queensland (36%), where losses of dry eucalypt communities have been second only to *Melaleuca* forests (Catterall & Kingston 1993).

Approximately half of the loss (48%) recorded in this study was not actual loss *per se* via clearing, but rather conversion of this open woodland to a more mesic type of vegetation. This occurs through the invasion of vine and rainforest species, and results in a community that is structurally similar to tall eucalypt woodlands (EW). Tall eucalypt woodlands, although reduced in terms of total area in Area 1, exhibited an overall increase for the entire study area, particularly in Area 2 (Figures 2.4a-c).

The process of conversion, which is believed to be the result of altered fire regimes, has long been of concern to national park management in the WTBR (Stanton 1989). Harrington and Sanderson (1994) have recently quantified similar reductions in wet sclerophyll forest in the wet tropics of Queensland following rainforest invasion. They also concluded that altered fire regimes are the primary cause of the invasion and raise concerns regarding loss of biodiversity from wet sclerophyll ecosystems should the process remain unchecked. No figures for habitat conversion of lowland plant communities have been documented prior to this study, despite the process having been observed widely in the field and identified from vegetation pattern changes from aerial photographs (Mr P. Stanton, pers. comm.). Stanton (1989) claims that the rapidity (<30 years) of this process (which is usually irreversible in management terms) can have significant impacts on habitat diversity in the tropical lowland habitat mosaic. It therefore follows that overall biodiversity within the region will also be affected.

The habitat conversion process, coupled with actual habitat loss and fragmentation (shape index decreased significantly) has resulted in an increase in the number of smaller patches (Figure 2.7c) and a significant decrease in patch area (Figure 2.8a). Even in a relatively undisturbed section of the landscape, significant differences have occurred with respect to the dispersion of this vegetation type (changing from clumped to random in Area 2, Figure 2.12c), presumably as a result of the loss of a majority of large remnant patches (Figure 2.7c). Unfortunately, no figures are available to compare the recorded extent of the conversion process with that occurring prior 1942. In addition, it may be possible that this conversion was exacerbated by logging activities (secondary regrowth).

Within the WTBR, the communities being lost as a result of the conversion process are poorly represented in the existing conservation reserve system, and have recently been classified as vulnerable ecosystems of the region (QDEH 1995a). This assessment, although based largely on field survey data, estimates that for the entire WTBR only 5-10% of the pre-European extent remains in an intact condition. This is despite the fact that three national parks are located within the study area. Edmund Kennedy National Park in the north-eastern section largely protects habitats not covered by this study (coastal beach scrubs, wetlands, vine forests), except for MOW (Walsh 1994). Two other parks occur in the south-eastern corner of the study area. Hinchinbrook Channel National Park largely contains mangrove communities, and although some protection for tall open woodlands is afforded by Lumholtz National Park, the majority of this park is not located in the coastal lowlands.

Melaleuca open woodlands (MOW) and woodlands (MW)

These two vegetation types have also been severely affected over the last 50 years. In Area 1 the complete loss of MOW was the only total reduction of a vegetation type recorded. This localised reduction increases the conservation significance of remnants in other areas, and highlights the value of considering sections of the landscape with high and low levels of disturbance separately. This type of approach has not been adopted in previous studies, which have traditionally focused on large areas of landscape, quite often at scales well above the one chosen for use here. In contrast, MW were one of the least affected vegetation types in Area 1.

In terms of patch size class distributions, MOW are again unique in that they had the highest loss of patches in the smallest size class (0-5 ha), the complete loss of a size class in Area 2 and the second largest loss of large patches for the entire area (Figures 2.6a-c). Despite these massive changes, the only landscape indices to alter significantly were the dispersion index for the entire area (changing from random to clumped) and a decrease in the fragmentation index in Area 2. This indicates that apart from distribution across the landscape, the shape characteristics of this vegetation type have remained essentially unchanged. No significant changes were recorded for MW for any of the indices used.

Melaleuca open woodlands have recently been classified as a vulnerable ecosystem (QDEH 1995a). Despite preservation in two mainland national parks in the study area (Lumholtz and Edmund Kennedy National Parks), regional representation of this vegetation type within the WTBR remains limited (Walsh 1994). These woodlands are the focus of the research presented in Chapters 3-6 of this thesis.

2.4.4 Management implications for fragmented landscapes

Fragmented landscapes pose perhaps the greatest challenge to conservation management in Australia. Without active management small, irregularly shaped remnants prone to edge effects and external influences have questionable long-term viabilities (House & Moritz 1991), and may not continue to support native plant and animal communities (Saunders *et al.* 1993). In fact, the decline and localised extinction of many plant and animal species from small, isolated remnants has been repeatedly demonstrated both in Australia (e.g. Saunders

& Hobbs 1995) and overseas (e.g. Myers 1994).

The relevance of landscape pattern information derived from indices to conservation questions has not been fully demonstrated (Noss & Csuti 1994). Indeed, it could be argued that this type of information is only relevant for the design of future nature reserves in relatively undisturbed landscapes. The ability of indices to facilitate patch selection and edge design in variously disturbed environments has, however, been demonstrated (Buechner 1987; Fensham submitted). Results from this investigation also suggest that critical communities for conservation initiatives can be identified in highly disturbed sections of the landscape. The powerful nature of the time series approach to landscape analysis for use as a monitoring tool has also been observed (Noss 1990).

Most conservation planning exercises have been based primarily on the present distribution of plant communities, with little information on historical patterns available. This is particularly true in the lowland vegetation mosaic of the north-eastern tropics. Information gained from studies such as these is essential so that priorities for conservation can be formulated based on a knowledge of past clearing patterns, and therefore an appreciation of which communities are truly threatened. For example, as a result of this study, arguments for increased representation of the three communities discussed in Section 2.4.3 within the conservation reserve system can be formulated. In fact, an increasing number of studies are now illustrating the disappointingly poor results achieved in terms of overall biodiversity conservation when reserve selection (as it has traditionally been) is essentially *ad hoc* (Lambeck 1994). Ever increasing recognition is also being given to a combination of on- and off-park conservation measures (Taylor 1993; Wells *et al.* 1994; Fensham submitted), with the emphasis on planning in an integrated fashion at the landscape scale (Saunders *et al.* 1993).

Following a detailed study of remnant bushland in south-eastern Queensland, Catterall and Kingston (1993) proposed a framework and priorities for conservation of vegetation in fragmented landscapes. Many management issues are equally important, including altered hydrological patterns (particularly drainage), changes in fire regimes and the invasion of exotic species. The effects of the latter two problems are considered in more detail for MOW in Chapters 3/5 and 6 respectively. These effects may be difficult to differentiate from those of habitat loss and fragmentation.

The mitigation of these external influences and the control of internal patch dynamics is critical if the long-term viabilities of remnants are to be maintained (Saunders *et al.* 1991). Indeed, studying the internal dynamics of isolated remnants can help identify processes that can act as early warning indicators for the management of larger remnants in the landscape as they continue to fragment (Given 1994). Management also has to address the various scales at which these effects operate, from the level of meta-populations right through to the level of individual species. At the same time, whilst trying to appreciate that the responses of specific taxa to the many and varied effects of fragmentation can differ considerably, modern day conservation managers also have to contend with a wide range of constraining realities including funding (or the lack thereof) for land acquisition, land tenure, development pressures, clearing guidelines (currently being developed for leasehold land in Queensland, DoL 1995) and other legislative issues.