CONSEQUENCES OF WEED INVASION AND CONTROL ON PLANT-BIRD INTERACTIONS AND BIRD COMMUNITIES

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CERTIFICATION

I, Carl R. Gosper, declare that this thesis, submitted in fulfilment of the requirements for the award of Doctor of Philosophy, in the Department of Biological Sciences, University of Wollongong, is wholly my own work unless otherwise referenced or acknowledged. The document has not been submitted for qualifications at any other academic institution.

Carl R. Gosper

10 September 2004

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Abbreviations and conventions

Chrysanthemoides monilifera (or *C. monilifera*) – After the general introduction, this refers to Bitou Bush (*C. monilifera* ssp. *rotundata*) unless otherwise specified. The other subspecies invasive in Australia is Boneseed (*C. m.* ssp. *monilifera*).

Invasive plant – Throughout this thesis, alien plant species that are established and expanding in natural habitats, such as *C. monilifera*, are referred to as invasive plants. Other terminology exists that is equally applicable to *C. monilifera* in coastal NSW, such as environmental weed (Humphries *et al.* 1991) and ecosystem transformer (Richardson *et al.* 2000a).

Study site location abbreviations:
ML – Myall Lakes National Park
PB – Perkins Beach Recreation Reserve
ENP – Eurobodalla National Park

NSW – The state of New South Wales, Australia

NPWS – The NSW National Parks and Wildlife Service

Nomenclatural conventions vary through the text. For the Chapters (2-5) that have been published, published in part or prepared for publication, the conventions of the relevant journals have been followed. Otherwise:

- Australian bird species nomenclature follows Christidis and Boles (1994). Common names are used throughout the text, with scientific names in the text on first mention and in Appendix A4 (for species recorded in this study). The scientific names for other fauna are given in the text with common name if available.
- Australian plant species nomenclature follows Harden (1990-1993). Scientific names are used in the text, with common names on first mention if available.

Abstract

Introduced plants can have a variety of impacts in ecosystems in which they become invasive. These impacts can include the disruption of interactions between native plant and animal species, such as seed dispersal and pollination. Subsequently, other interactions and mutualisms can be affected, both at the site of the plant invasion and elsewhere. Interactions can also become established between the invasive plant and native and invasive animal species. The removal of an invasive plant has the potential to disrupt these newly formed interactions, thus disadvantaging some fauna and potentially affecting subsequent interactions involving these species. While control of invasive plants is typically a goal of conservation management, the consequences of control for other species are rarely fully considered or investigated. In this study, I have tested several hypotheses regarding the interaction of an invasive plant with fauna, and the effects of the plant's invasion and control on plant-animal interactions.

Bitou Bush (*Chrysanthemoides monilifera* ssp. *rotundata*) is an invasive plant in coastal New South Wales (NSW), Australia. South African in origin, it has now spread to occupy at least 80% of the NSW coastline. Fruit production in *C. monilifera* is prolific and fruits are consumed and dispersed by vertebrates, especially birds. In addition to other effects of *C. monilifera*, this plant-animal interaction has the potential to affect the seed dispersal of other vertebrate-dispersed plants and also the composition of the bird community, which may subsequently affect other plant-bird interactions. In order to quantify the magnitude of these possible effects, I designed this study with the following five major components:

(i) comparison of the fruiting characteristics of *C. monilifera* with those of co-occurring bird-dispersed native plant species;

(ii) description of the plant-bird interactions that involve flowers and fruits in vegetation that has been invaded or is at risk of invasion by *C. monilifera*;

(iii) measurement of the rate of removal of *C. monilifera* fruits and those of some co-occurring bird-dispersed plant species in: habitat dominated by *C. monilifera*, where *C. monilifera* had been eliminated by the application of herbicide, and uninvaded vegetation;

(iv) assessment of the effect of *C. monilifera* removal by herbicide application on the species composition and abundance of the bird community; and

(v) assessment of the effect of dominance of the vegetation community by *C. monilifera* on the species composition and abundance of the bird community.

I found that the fruits of *C. monilifera* are within the range of physical dimensions and nutrient composition of those of co-occurring native species. The greatest distinction in fruit

characteristics is in phenology and the combination of phenology and morphology, as peak production of *C. monilifera* fruits occurs when native fruits are scarce. Consequently, *C. monilifera* fruits are attractive to vertebrate dispersers, especially birds. At least 25 species of birds feed on *C. monilifera* fruits in NSW, most of which are indigenous and are likely to disperse viable seeds.

In an experimental study using feeding stations, I found that the rate of removal of fruits of native plant species was unaffected by either dense infestation of *C. monilifera*, or its elimination. This is likely to be due to highly facultative relationships between frugivorous birds and plants, combined with differences in phenology and, in some cases, the morphology of fruits of native plants and *C. monilifera*. Consequently, in this system there has been little impact of an invasive plant on this plant-bird interaction. The rate of removal of *C. monilifera* fruits, however, was less in herbicide-treated habitat. This has implications for long-term *C. monilifera* control, because herbicide treatment often leaves scattered individual plants alive, which would have poorer seed dispersal than plants in dense *C. monilifera* stands.

The removal of *C. monilifera* affected the bird community, with the overall abundance of birds declining in herbicide-treated areas after the *C. monilifera* died. Only those birds that consume *C. monilifera* fruits were affected, while other groups of birds that do not directly use *C. monilifera* resources were unaffected. Although this impact was minor, it illustrates that removal of an invasive plant can affect bird communities, and these impacts should be considered before control programs are implemented. Dominance of the vegetation by *C. monilifera* also affected bird communities, with overall bird abundance, specifically that of insectivorous birds, and at some locations nectarivorous birds, being lower in *C. monilifera* than native habitat. The quantity of remaining native vegetation, particularly of nectar-producing plants widely used by birds, appears to be important in determining bird community composition in invaded areas.

In this study, I have demonstrated that both dominance of the vegetation by an invasive plant, and the control of an invasive plant, can induce change in fauna communities, and disrupt some plant-animal interactions. These changes will need to be considered carefully in planning management actions to conserve coastal bird communities and their interactions with plants. While continued efforts to control *C. monilifera* are clearly justified, these should form part of a broad strategy for coastal community conservation, including consideration of other threats to native communities that act independently or in concert with *C. monilifera* invasion. These considerations should include the potential impacts of other invasive plant species, targeted sites and species for control efforts, and other forms of habitat loss and degradation.

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CHAPTER 1. GENERAL INTRODUCTION

A Bitou Bush (*Chrysanthemoides monilifera* ssp. *rotundata*) plant in coastal vegetation near Wollongong, Australia. Over 80% of the NSW coastline has been invaded by *C. monilifera*.

Preface

In this section, I introduce the key questions that underpin this research. I include background information on the history, status, distribution, ecology and ecological impacts of *Chrysanthemoides monilifera* in NSW, which clearly justify the need for this research and illustrate the scale and severity of the problem. I have written this thesis with the aim of each Chapter being a stand-alone manuscript. Each chapter, therefore, has a detailed introduction that expands on the ecological principles and literature relevant to that experiment or series of observations.

1.1 Plant-animal interactions

Interactions between animals and plants form an essential component of ecosystem functioning. For example, many plants rely on animals for pollination and seed dispersal and many animal species rely on plants for food and shelter. In some cases, these relationships are obligate and specific (Janzen 1979). More frequently, the relationships are facultative and more general, involving sets of species (Jordano 1987). Some of these interactions are of such importance that their disruption is predicted to trigger a cascade of subsequent effects (including extinction) throughout the community (the keystone mutualist hypothesis) (Gilbert 1980). It has been argued that these crucial interactions are fundamental to maintaining the structure and diversity of communities and may, therefore, be essential for their maintenance (Christian 2001).

Interactions between plants and animals have been compromised by global declines in animal species, with consequent disruption of plant-animal interactions. This is the case for pollinators (Kearns *et al*. 1998; Cox and Elmqvist 2000) and seed dispersers (Cox *et al.* 1991; Meehan *et al*. 2002). Disruption of plant-animal interactions has the potential to have far-reaching effects. For mobile species of fauna, these effects may extend well beyond the system in question.

1.2 Invasive species and their ecological impacts

The rate at which organisms have been transported beyond their natural distribution by deliberate or accidental movement by humans has increased greatly in recent times (Vitousek *et al.* 1996). Many of these translocated organisms have become established and spread in their new environments. The change on ecosystems induced by these species may form a significant component of global environmental change (Vitousek *et al.* 1996). As an illustration of the scope of biological invasions, Usher (1988) suggests that no conservation reserves (outside Antarctica) are likely to be free of invasive species.

Invasive plants, for example, are widely known to have a variety of impacts, including alteration of the structure and functioning of ecosystems (Vitousek *et al.* 1996). These impacts include changes in nutrient cycling (Vitousek and Walker 1989), soil nutrient and physical properties (Hamilton 1965; Witkowski and Mitchell 1987), fire regimes (van Wilgen and Richardson 1985; D'Antonio and Vitousek 1992), vegetation structure and plant communities (Braithwaite *et al.* 1989; Ogle *et al.* 2000), geomorphological processes (Lane 1992), hydrology (Loope *et al.* 1988), and recruitment patterns of plants (Waterhouse 1986; Smith 1994). At the species level, invasive plants have adversely affected a number of plant species (see Adair and Groves 1998 for a review of published studies). Most studies of invasive plants and their impacts on biodiversity, both within Australia and overseas, have been directed at plants. Comparatively, effects on fauna have been largely ignored (Adair and Groves 1998).

1.2.1 Disruption of plant-animal interactions by biological invasions

Biological invasions can disrupt mutualisms. For example, in South Africa, the invasive Argentine Ant (*Linepithema humile*) has displaced indigenous ant species that are important dispersal agents for seeds of indigenous plants (Bond and Sligsby 1984). This change in the ant community has induced community-level consequences through reduced dispersal of seeds of some plant species and a subsequent change in plant community composition (Christian 2001). Similarly, the loss of indigenous seed dispersers in the Balearic Islands and their replacement with an invasive species has altered the propagule distribution and dispersal rate, plant distribution and fruit size, of the indigenous plant *Cneorum tricoccon* (Riera *et al.* 2002).

1.2.2 Effects of invasive plants on plant-animal interactions and fauna communities

In this study, I am interested in the changes invasive plants induce on plant-animal interactions, and how these changes further alter ecosystems. Biological invasions can alter mutualistic relationships (Christian 2001). Hence, it follows that if an invasive plant displaces a native plant species or changes the composition or behaviour of the fauna community (see Table 1.1A), subsequent changes on plant-animal interactions are possible and indeed likely. The loss or reduction of resources provided by any displaced plants, such as nectar or fruit, may adversely affect animal species that rely on them. Further plant species may then be affected by a decline in a pollinator or seed dispersal agent, both within the invaded system and elsewhere, illustrating a flow-on effect of the disruption of a plant-animal interaction. Likewise, altered abundance or behaviour of fauna species may affect pollination or seed dispersal of plants at the invaded locality and elsewhere. Whether these changes eventuate depends on the level of redundancy in the plant-animal interaction system: for example, whether other species in the community substitute for those lost or reduced in abundance (Westman 1990; Christian 2001). Howard-Williams and Davies (1988), for example, reported an increase in the abundance of a crayfish and the introduced Black Swan (*Cygnus atratus*) in wetlands following invasion by the aquatic plant *Lagarosiphon major*. These increases then compounded the impact of the invasive plant by adversely affecting remaining native plants at this and nearby wetlands, through increased herbivory. Similarly, changes in the abundance, diversity or behaviour of frugivores or pollinators could impact on the seed dispersal and pollination of other plant species. Knight (1986), for example, observed that avian use (and presumably subsequent dispersal) of the fruits of native *Rhus* spp. in South Africa was reduced by avian use of invasive *Acacia cyclops* seed.

Table 1.1 - Examples of the effects of invasive plants and their management on fauna species and communities

A. Effects of dominance of the vegetation by invasive plant(s) on bird species and communities [a]

Authors	Location	Overall Abundance	Species richness	Abundance of guilds	Abundance of individual species
Scheiman *et al.* 2003	USA				↓↔
Dean *et al.* 2002	South Africa	↓	↓	↓↔	
French and Zubovic 1997	Australia	↔	(↓?)	↓(↑?)	
Pakeman and Marrs 1992	Britain				↓↑
Braithwaite *et al.* 1989	Australia				↓↑
Griffin *et al.* 1989	Australia	↓	↔	↓↑↔	
Wilson and Belcher 1989	Canada	↔	(↓?)		↓(↑?)
Howard-Williams and Davies 1988[c]	New Zealand				↑
Knoph and Olson 1984	USA			↓↑	

B. Effects of invasive plant removal on fauna species and communities [b]

Authors	Fauna group	Location	Overall Abundance	Species richness	Abundance of orders	Abundance of individual species
Homan *et al.* 2003 [c]	Birds	USA				↔
Ailstock *et al.* 2001	Invert.	USA	↔	↔		
Linz *et al.* 1999	Invert.	USA	↑		↓↑↔	
Linz *et al.* 1996a; 1996b; 1997; Linz and Blixt 1997	Birds	USA				↓↑
Olaleye and Akinyemiju 1996	Fish	Nigeria	↑			

↑ = increase in parameter in invaded vegetation (A) or after invasive plant removal (B)
↓ = decrease in parameter in invaded vegetation (A) or after invasive plant removal (B)
↔ = no change in parameter recorded
? = some evidence of change, but either not statistically significant or not uniform (*e.g.* invasion x location interaction)
'Invasive plants' include some indigenous species that are considered undesirable for fauna conservation and are managed as such (*e.g.* Ailstock *et al.* 2001).
[a] Only examples of studies on birds are tabulated here, although a similar range of responses has been reported for other fauna. [b] Examples of studies on all types of fauna are tabulated. [c] study of effects on a single species of bird

Invasive plants often provide resources directly used by fauna, and therefore interactions can develop between invasive plants and fauna species (Richardson *et al.* 1992; Richardson *et al.* 2000b). There are many examples of the use of nectar or fruit resources of invasive plants, by both native and exotic fauna (Woodward *et al.* 1990; Loyn and French 1991; Richardson *et al.* 1992; Williams and Karl 1996). In some cases, the resources provided by the invasive plant have been as least as well used as native alternatives (Knight 1986; Waring *et al.* 1993; French and Major 2001).

Disruption and establishment of plant-animal interactions are one component of the change in fauna communities induced by invasive plants. Invasive plants have been recorded having a variety of effects on indigenous fauna, at both the species and community level (Table 1.1A).

Although only studies of birds are shown in this Table, a similar range of responses to invasive plants has been recorded for other fauna. More often than not, invasive plants have caused reductions in the abundance of some fauna species or fauna overall, or changed the composition of the fauna community. In many cases, some indigenous fauna increased in abundance in habitat dominated by the invasive plant. Invasive plants do not appear to induce a general response, but rather the impact(s) may depend on specific characters of the invasive plant and the invaded ecosystem.

1.3 Management of invasive plants and its effects on plant-animal interactions and fauna communities

All the studies in Table 1.1A illustrate that habitats dominated by invasive plants support native fauna, although often not of the same composition and/or abundance of natural habitats. Redundancy in plant function allows invasive plants to provide resources that partly substitute for those provided by indigenous species (Westman 1990). Consequently, fauna species use habitat and resources provided by invasive plants, and, in some cases, species or assemblages of species prefer them (Knopf and Olsen 1984; Braithwaite *et al.* 1989).

Managing invasive plants is a goal of conservation management (*e.g.* Westman 1990; NSW NPWS 1999). The potential arises, especially in degraded landscapes, for an invasive plant to provide essential resources for some fauna. For example, a number of native butterflies in Davis, California, appear to be entirely dependent for breeding on alien plants (Shapiro 2002). In northern NSW, extensive infestations of Camphor Laurel (*Cinnamomum camphora*) and Large-leaved Privet (*Ligustrum lucidum*) have been identified as providing important habitat and food for birds (Date *et al.* 1991; Ekert and Bucher 1999). Similarly, in South Africa, expansion of the Rameron Pigeon (*Columba arquatrix*) has been linked to its' use of fruits of the invasive *Solanum mauritianum* for food (Knight 1986). Control of invasive plants, by whatever means, has the potential to have impacts on fauna species that now use the resources these invasive plants provide. These resources will be lost, and if the control program is one that causes rapid change (*e.g.* herbicide treatment or physical removal), it is likely that, in the short-term, no replacement resources would be available to substitute for those provided by the invasive plant. Furthermore, other species may be affected by disruption of the existing interactions of the invasive plant and fauna. Whether these predictions are realised when invasive species are controlled has only been tested rarely (Pakeman and Marrs 1992). Where studies have been done, fauna species displayed a diversity of responses to invasive plant removal, from increasing in abundance to no change to decreasing (Olaleye and Akinyemiju 1996; Linz *et al.* 1996a; 1996b; Homan *et al.* 2003) (Table 1.1B). This illustrates the need for either case-by-case assessment or more empirical studies before generalisations can be made of the responses of fauna to habitat change, whether invasive plant expansion or control induces it.

1.4 Invasive plants in Australia

Deliberate or accidental introductions of plants to Australia began before European invasion (Macknight 1976). Afterwards, the number of naturalised species has increased linearly (Specht 1981) at an average rate of 11 species per annum (Csurhes and Edwards 1998), but this rate has possibly increased in recent decades (Carr 1993; Groves 1997). A substantial number of plant species may be introduced into Australia each year (Panetta *et al.* 1984). The majority of recently naturalised plants were introduced deliberately, mainly as ornamentals (used in horticulture) (65%) (Groves 1997). Plants were also introduced deliberately for agricultural, medicinal, culinary, forestry and land rehabilitation purposes.

Approximately 2200 plant species had become naturalised across Australia by the early 1990s, representing up to 15% of the total vascular flora (Hnatiuk 1990; Humphries *et al.* 1991). Only about half of these have been recorded invading native vegetation (Swarbrick and Skarratt 1994). The typical places of origin of Australian invasive plants have changed over time. In the initial decades following European settlement, plants introduced reflected the countries of origin of the settlers: northern European. Later, American and Mediterranean-region plants became more significant. Recently (since 1971), new introductions originate relatively evenly from the Americas, Africa and Europe (Groves 1997).

1.4.1 Interactions and effects of invasive plants on birds in Australia

Several studies have investigated the impact of invasive plants on bird species or communities in Australia (Table 1.1A). Additionally, plant invasions have been identified as a process affecting 16 species of extinct, threatened or near-threatened birds (Garnett and Crowley 2000), although this assessment probably underestimates the true impact of invasive plants.

Australian birds use a variety of foods provided by invasive plants (Holmes 1987; Buchanan 1989; Loyn and French 1991; Gosper and Gosper 1996; Spennemann and Allen 2000; Stansbury 2001). In terms of interactions between invasive plants and fauna, fauna consuming, and probably dispersing, the fruit of invasive species seems disproportionately high compared to pollination. In coastal NSW, and temperate Australia more broadly, invasive trees, shrubs and vines with seeds dispersed by vertebrates appear particularly well represented. The majority of these plant species have black and/or red fruit, which reflects the predominance of birds among vertebrate seed dispersers in southern Australia (Clifford and Drake 1981; Floyd 1990; French

1991). Birds are well known to select fruit of black and/or red colouration (Ridley 1930). Invasive plants with black or red fruit and with seed dispersed by birds in southern Australia include *Chrysanthemoides monilifera*, *Cinnamomum camphora*, Lantana (*Lantana camara*), *Solanum* spp., *Protasparagus* spp., Bridal Creeper (*Myrsiphyllum asparagoides*), Privets (*Ligustrum* spp.), Hawthorn (*Crataegus monogyna*), Cotoneaster (*Cotoneaster* spp.), Inkweed (*Phytolacca octandra*) and Olives *Olea europaea* (Dodkin and Gilmore 1984; Holmes 1987; Buchanan 1989; Loyn and French 1991; Gosper 1994; Spennemann and Allen 2000; Stansbury 2001). This readily illustrates the ability of many native and exotic bird species to widely use novel fruit sources. In contrast, invasive plants pollinated predominantly by vertebrates in Australia are scarce, despite the relatively high frequency of vertebrate pollination in Australian ecosystems (Ford *et al.* 1979). This may be largely a reflection of the historical bias of northern hemisphere origin of many Australian weeds, from where vertebrate pollination is not common (Ford 1985), in contrast to vertebrate seed dispersal (Herrera 1984).

1.5 Study species – Bitou Bush (*Chrysanthemoides monilifera* spp. *rotundata*)

Chrysanthemoides monilifera (L.) Norlindh (Asteraceae) (syn. *Osteospermum moniliferum* L.) grows naturally in a variety of habitats along the southern, south-western and south-eastern coasts of South Africa and on inland ranges extending north to tropical Africa (Neser and Morris 1984). There are six subspecies, two of which are invasive in Australia: *C. m.* ssp *monilifera* (DC.) Norlindh (Boneseed); and *C. m.* spp. *rotundata* (DC.) Norlindh (Bitou Bush). In South Africa, *C. m. monilifera* occurs along the south-eastern to south-western coasts and adjacent ranges, and *C. m. rotundata* has a largely coastal distribution along the eastern coast (Neser and Morris 1984). In addition to Australia, *C. monilifera* is naturalised in California, Sicily, southern France and New Zealand (Parsons and Cuthbertson 1992).

1.5.1 Invasive *C. monilifera* in Australia

Chrysanthemoides monilifera is now naturalised in all states and territories of mainland Australia and Tasmania except the Northern Territory. Humphries *et al.* (1991) include it among the country's 17 most significant environmental weeds and it is listed as a Weed of National Significance (ARMCANZ, ANZECC and Forestry Ministers 2000). In NSW, 'Invasion of native plant communities by Bitou Bush' has been listed as a key threatening process under the NSW *Threatened Species Conservation Act* (1995) (Dickman 1999).

Chrysanthemoides monilifera ssp. *rotundata* (hereafter *C. monilifera*) was first recorded in Australia in 1908, just north of Newcastle, NSW. This occurrence may have originated from seed in the ballast of a ship arriving from South Africa (Humphries *et al.* 1991). Until the 1950s, *C. monilifera* was restricted to the Newcastle district (Mort and Hewitt 1953). The main period of expansion of *C. monilifera* was from 1946 to 1970 when *C. monilifera* seed and seedlings were sown for stabilisation of coastal sand dunes and revegetation following sand mining (Hewitt 1954; Barr 1965). The potential for *C. monilifera* to become invasive can be readily seen in the descriptions of the 'suitability' of the plant for coastal erosion control works made at the time. Mort and Hewitt (1953) described *C. monilifera* as "free flowering and free seeding in habit", "(G)ermination of seed is usually profuse in the lee of the plant and natural regeneration appears to be comparatively rapid following maturity", and the fruit being "much sought after by birds". Unfortunately, these characteristics were regarded as good qualities at the time, and the use of *C. monilifera* in dune stabilisation continued. *Chrysanthemoides monilifera* was at one time proclaimed a noxious weed in the Newcastle District, but this classification was removed (before 1953) when the usefulness of this species in sand-dune stabilisation became apparent (Mort and Hewitt 1953). The recommendation for the use of *C. monilifera* in coastal planting was withdrawn in the early 1970s (Love 1984) but it was already well established along the NSW coastline by this time (Gray 1976). An intense eradication program implemented in Queensland over the past 10-15 years has generally restricted the distribution of *C. monilifera* in that state (Csurhes and Edwards 1998).

A systematic survey of the distribution of *C. monilifera* along the northern NSW coastline was instigated in 1981, and extended to the remainder of the NSW coastline in 1982. The presence of *C. monilifera* on frontal dunes and the seaward aspect of headlands were mapped to determine the density and distribution of infestations, and their relation to areas of conservation interest. *Chrysanthemoides monilifera* was present along nearly 60% of the total NSW coastline, with continuous infestation along 21% (Love 1984). Love (1984) also extrapolated to predict the distribution of *C. monilifera* in 2010, suggesting nearly 100% infestation of the coastline of NSW from the Queensland border south to near Narooma. As recognised by Love (1984), *C. monilifera* spread is likely to be somewhat less than this, due to the combined impacts of the many control programs active along the NSW coastline.

The NSW coastline was re-surveyed in 2000, with *C. monilifera* recorded on over 80%, and with up to 95% of the coast infested in some regions (NSW NPWS 2001). In NSW, the area of heavy infestation by *C. monilifera* is 6,700 ha, 9,000 ha have moderate infestation and 20,100 ha light infestation. Two-thirds of the NSW coastline has near continuous infestations of *C. monilifera* (NSW NPWS 2001).

1.5.2 Ecology of *C. monilifera*

Chrysanthemoides monilifera is an evergreen scrambling to decumbent shrub, producing yellow daisy-like inflorescences, that develop approximately spherical, black, drupe-like fruits containing a single seed (Harden 1992; Csurhes and Edwards 1998). Up to 13 fruit are produced per infructescence (Simmons and Flint 1986). Fruits are consumed and seeds spread by a variety of birds, mammals (Dodkin and Gilmore 1984; Gosper 1999a) and ants (Parsons and Cuthbertson 1992). Other mechanisms of seed dispersal include the movement of seeds in running water, contaminated soil or garden refuse (Parsons and Cuthbertson 1992) and in clothing and on vehicles (pers. obs.). Production of seed can be prolific, with up to 50,000 seeds per plant per year (Parsons and Cuthbertson 1992). This produces a high quantity of soil-stored seed, with up to 9500 seeds/m^3 (Parsons and Cuthbertson 1992). Soil-stored seed can remain dormant for an unknown period (Vranjic 2000), although its viability declines rapidly, to 2% after three years (Weiss 1986).

Germination of *C. monilifera* is maximised by removal of the pericarp or weathering of the seeds (Weiss 1983). Seeds germinate throughout the year, with a flush in autumn. *Chrysanthemoides monilifera* can spread vegetatively, with prostrate stems in contact with soil, or buried, able to develop roots (Parsons and Cuthbertson 1992). Plants can occasionally flower in their first year, but more typically take 18 months to three years (Vranjic 2000). *Chrysanthemoides monilifera* produces flowers and fruit throughout the year but with an autumn flowering peak (Parsons and Cuthbertson 1992) and an early winter peak in fruit production (Weiss 1984).

Chrysanthemoides monilifera responds positively to a variety of natural and human induced disturbances. Following wildfire, some *C. monilifera* shrubs re-sprout from adventitious buds at the base of the plant or along the stems (Weiss 1983). Fire also stimulates prolific germination of seed from the soil-stored seed bank (Vranjic 2000). Post-fire seedlings reach maturity more quickly than those germinating otherwise do, often flowering and setting seed within one year of germination (Weiss 1986). Mechanical disturbance, such as slashing, results in rapid regeneration from adventitious buds, and soil disturbance results in germination from the soil-stored seed bank. Grazing and trampling by stock, and cultivation, however, remove existing plants (Parsons and Cuthbertson 1992) and, therefore, *C. monilifera* does not tend to be problematical on agricultural lands. In Australia, *C. monilifera* prefers sandy to medium-textured soils.

1.5.3 Effects of *C. monilifera* invasion on biodiversity

A number of impacts of *C. monilifera* invasion have been demonstrated on native species and communities. Coastal Wattle (*Acacia sophorae*) is displaced, and may have lower seed production (Weiss and Noble 1984a), seed germination and seedling growth (Weiss and Noble 1984b; Vranjic *et al.* 2000) when co-occurring with *C. monilifera*. Vegetation dominated by *C. monilifera* has lower abundance and diversity of plant-feeding birds (French and Zubovic 1997) and some orders of litter invertebrates (French and Eardley 1997) than native vegetation, although has no effect on either overall abundance or species richness of these fauna. *Chrysanthemoides monilifera* is also thought to displace many other coastal plant species (NSW NPWS 2001), although this has not been substantiated by published research.

1.5.4 Control of *C. monilifera*

Much research has been conducted on the control of *C. monilifera* in Australia, with the current state of knowledge outlined in Vranjic (2000), and will not be repeated here. I will, however, give a brief account of aerial herbicide application used to control *C. monilifera*, as this technique was used in the removal of *C. monilifera* for the experiments in this study. NSW Agriculture commenced research in 1985 on the control of *C. monilifera* using herbicides to establish a protocol for wide-scale application. Glyphosate and metsulfuron methyl achieved both adequate control of *C. monilifera* and sufficient selectivity when tested on several common native coastal plants (Toth *et al.* 1996). An aerial herbicide application protocol to control *C. monilifera* was developed, typically using glyphosate applied at low concentration in winter (Toth *et al.* 1996). This has several advantages over other control techniques: the level of control is typically high (90-95% mortality); inaccessible *C. monilifera* infestations can be treated; and aerial spraying is relatively cost efficient (Toth *et al.* 1994; NSW NPWS 2001).

A limited amount of research has been conducted into the off-target effects of this control methodology. Kohler *et al.* (1995) investigated the impact of aerial glyphosate application on fauna at Hawks Nest, central NSW, concluding that the impact on fauna was minimal but recognising the scale and replication of the experiment was inadequate to draw meaningful conclusions. My research project has arisen largely out of the need for this type of knowledge. Recent research (Elizabeth Lindsay and Kris French, University of Wollongong, unpublished) revealed little impact of spraying on litter invertebrates (K. French pers. comm.). In contrast, several projects have researched and monitored plant species exposed to herbicide application for *C. monilifera* control (Cooney *et al.* 1982; Kohler *et al.* 1995; Toth *et al.* 1996; Matarczyk *et al.* 2002).

1.6 This study

Adair and Groves (1998) identified *C. monilifera* as one of few species of invasive plants that have weed control strategies based on detailed environmental impact studies where the effect of invasion on biodiversity values have been evaluated and documented. Although more research has been conducted on the impacts of *C. monilifera* invasion than on most invasive plants, our knowledge really only covers a small fraction of potential impacts. Detailed scientific studies (*c.f.* informed opinion) have only been completed on the impacts of *C. monilifera* on a select few plant species (Weiss and Noble 1984a; 1984b; Vranjic *et al*. 2000), litter invertebrates (French and Eardley 1997) and a short-term study on birds (French and Zubovic 1997). Some information is also available on bird use of *C. monilifera* fruits (Dodkin and Gilmore 1984; Gosper 1999a).

Much more needs to be known, however, before coastal habitats, including those invaded by *C. monilifera*, can be adequately managed for biodiversity conservation. Although birds use *C. monilifera* fruits for food (Dodkin and Gilmore 1984; Gosper 1999a), it is not known if there are particular characteristics of *C. monilifera* fruits, relative to indigenous fruits, that make them particularly attractive to birds and enhance the invasiveness of *C. monilifera.* Similarly, it is not known if birds actually prefer *C. monilifera* fruits to native alternatives or which species of birds are effective dispersal agents. Furthermore, alternative foods available to birds, either before or after *C. monilifera* invasion, are also poorly known. *Chrysanthemoides monilifera* invasion alters the composition of the bird community, particularly of plant-feeding birds (French and Zubovic 1997). It is not known what impact this change has on plant-animal interactions, such as the rate of removal of fruits or pollination. Additionally, impacts on bird communities from *C. monilifera* invasion may differ depending on resource availability. The French and Zubovic (1997) study was conducted outside of the main fruiting period of *C. monilifera*. Presumably, the impact of *C. monilifera* on birds may differ over periods when its fruits are a major food source for birds. Minimal research has been conducted on the impacts of *C. monilifera* control methods on biota, particularly fauna. This study largely arose out of the need for this information for aerial herbicide application for *C. monilifera* control. Without such research, the impacts of control methods on biota (positive or negative) cannot be known, and hence whether the resources allocated to such control measures are justified.

In order to rectify several of these important gaps in knowledge, I investigated the following questions in this study:

(1) What are the characteristics of the fruits of invasive *C. monilifera* in coastal NSW, and how do these compare with co-occurring native plant species? Fruit characteristics affect fruit selection by birds (and other vertebrates). Hence, comparing these characteristics might identify native plant species at risk of competition with *C. monilifera* for dispersal agents, the relative attractiveness of *C. monilifera* fruits to dispersers and changes in fruit availability in coastal vegetation that invasion by *C. monilifera* may cause.

(2) What plant foods (fruits, seeds, nectar and flowers) are used by bird species in coastal vegetation invaded by *C. monilifera* or at risk of invasion? Identifying some of the interactions between plant and bird species will assist in identifying if and where disruptions to these interactions may occur, and the consequences for ecosystem functioning and individual plant and bird species.

(3) Does the presence of *C. monilifera*, or its elimination (through herbicide treatment), affect the rate of removal of fruits of vertebrate dispersed plants? Rates of fruit removal may be affected because invasion and elimination of *C. monilifera* changes the availability of fruit resources, which may affect frugivorous species and subsequently the removal of fruits of other plant species.

(4) Does the elimination of *C. monilifera* affect the abundance, species richness or composition of the coastal bird community? Removal of *C. monilifera* has the potential to impact the bird community in the short term through the method of removal and/or the loss of resources provided by *C. monilifera*. If an impact is detected, identifying at what stage of the control process it occurs and the birds affected will assist in describing the changes to birds induced by invasive plant removal and identifying ameliorative measures.

(5) Does dominance of coastal vegetation by *C. monilifera* affect the abundance, species richness or composition of the coastal bird community? This assessment will assist in identifying species, groups of species and plant-bird interactions at risk from further *C. monilifera* invasion, and additionally, place the impacts induced by *C. monilifera* removal in context. More broadly, this research will provide additional information on the impacts of an invasive plant on native communities and interactions and assist in the interpretation of the previous experiments.

I chose birds as the group of fauna most suited for the study of interactions with *C. monilifera* for the following reasons:

(i) many bird species feed upon the fruits of *C. monilifera* and native coastal plant species (Dodkin and Gilmore 1984; Gosper 1999a). These plant species presumably rely on these birds for the dispersal of their seeds (Clifford and Drake 1981; Gosper 1999a). Fruit is a food type likely to be substantially altered in abundance and composition by *C. monilifera* invasion and control and, hence, birds may be more responsive to these habitat changes than other fauna as they use fruit resources directly. Any change in the abundance or composition of the bird community due to *C. monilifera* invasion may have further effects on interactions of birds with native bird-dispersed plants. Birds as a community, in groups or as individual species, may therefore be more responsive to the invasion or removal of *C. monilifera*;

(ii) invasion of native vegetation by *C. monilifera* has been shown to affect the bird community (French and Zubovic 1997), hence birds are sensitive to this type of habitat change;

(iii) few other vertebrate fauna were recorded in preliminary studies of *C. monilifera* stands (Kohler *et al.* 1995); and

(iv) birds are readily surveyed in coastal vegetation.

1.6.1 Thesis outline

In Chapters 2 to 7, I describe and interpret self-contained experiments or a series of observations. Chapter 2 contains a comparison of fruit characteristics of *C. monilifera* and native plant species – Question 1 above. In Chapter 3, I present the information collected on the plant foods of birds in coastal vegetation, and Chapter 4 I focus on observations of frugivory of *C. monilifera* fruits (Question 2). The experiment assessing the rate of removal of fruits is presented in Chapter 5 (Question 3); the effects of *C. monilifera* removal on bird communities in Chapter 6 (Question 4); and the effects of dominance of the vegetation by *C. monilifera* on bird communities in Chapter 7 (Question 5). In Chapter 8, I bring together the results and interpretations of the experiments and observations into an overall assessment of the effects of *C. monilifera* on plant-bird interactions and bird communities. Additionally, I describe some management measures and further research required that might assist in ameliorating identified impacts. In the Appendices, I have included a reprint of a paper already published from this research and more detailed descriptions of the survey sites than appear in any of the Chapters.

CHAPTER 2. FRUIT CHARACTERISTICS OF *CHRYSANTHEMOIDES MONILIFERA* AND A COMPARISON WITH CO-OCCURRING NATIVE PLANT SPECIES

Endocarp size of *C. monilifera* (C.m.) and of several of the native plant species (*Elaeocarpus reticulatus* – E.r.; *Monotoca elliptica* – M.e.; and *Melia azedarach* – M.a.) that co-occur with *C. monilifera* in coastal NSW. The fruits and endocarps of *C. monilifera* were intermediate in size compared to native species.

Preface

In this chapter, I assess several aspects of *C. monilifera* fruits and fruit production. These characters may be important in the invasiveness of *C. monilifera* in NSW. A manuscript from this work, slightly modified from this chapter, has been published in the *Australian Journal of Botany*. The referencing, species nomenclature, figure and table formatting conventions used in this journal are followed in this chapter.

Publication:

Gosper, C.R. (2004) Fruit characteristics of invasive Bitou Bush, *Chrysanthemoides monilifera* (Asteraceae), and a comparison with co-occurring native plant species. *Australian Journal of Botany* **52**, 223-230.

2.1 Introduction

Bitou Bush, *Chrysanthemoides monilifera* (L.) Norlindh ssp. *rotundata* (DC.) Norlindh (Asteraceae), is an exotic shrub that has been introduced to Australia from South Africa. It is now naturalised along more than 80% of the NSW coastline (NSW NPWS 2001). *Chrysanthemoides monilifera* invasion has a number of impacts on native species and communities. Coastal Wattle, *Acacia sophorae* (Labill.) R. Br. (Mimosaceae), is displaced, and may have lower seed production (Weiss and Noble 1984a), seed germination and seedling growth (Weiss and Noble 1984b; Vranjic *et al.* 2000) when co-occurring with *C. monilifera*. Vegetation dominated by *C. monilifera* has lower abundance and diversity of plant-feeding birds (French and Zubovic 1997) and some orders of litter invertebrates (French and Eardley 1997) than native vegetation. 'Invasion of native plant communities by Bitou Bush' has been listed as a key threatening process under the NSW *Threatened Species Conservation Act 1995* (Dickman 1999), recognising the detrimental impact of *C. monilifera* on coastal plant communities and threatened species.

Chrysanthemoides monilifera produces fleshy fruit, and birds and mammals constitute the primary seed-dispersal vectors (Dodkin and Gilmore 1984; Gosper 1999a). Fruit and endocarp morphology, the nutrient content of the fruit pulp and phenology are among the factors that may influence fruit choice by dispersal agents (Sorensen 1981; Sorensen 1984; Murray *et al.* 1993; Stiles 1993). A number of native plant species with which *C. monilifera* co-occurs also have fleshy fruit dispersed by vertebrates. *Chrysanthemoides monilifera* could potentially compete with these species for dispersal agents. Differences in fruit characteristics may also provide some indication as to whether *C. monilifera* fruits are particularly attractive to vertebrate dispersers, and hence have substantially contributed to its invasiveness.

The aim of this study was to measure a number of inflorescence and fruit characteristics (morphology, nutrient content and phenology) of naturalised *C. monilifera* and compare these to native plant species that co-occur with *C. monilifera* in coastal NSW.

2.2 Methods

A sample of 50 fruits was collected from each of five plant species: *C. monilifera*, Tree Heath (*Monotoca elliptica* (Smith) R.Br. (Epacridaceae)), Blueberry Ash (*Elaeocarpus reticulatus* Smith (Elaeocarpaceae)), White Cedar (*Melia azedarach* L. (Meliaceae)) and Red Ash (*Alphitonia excelsa* (Fenzl) Reisseck ex Benth. (Rhamnaceae)). Most fruit were collected from several plants at Perkins Beach Reserve, Wollongong (34°36’S, 150°53’E). Exceptions were fruits of *M. azedarach*, which were collected at Wollongong, and *Monotoca elliptica*, which were collected at Myall Lakes National Park (NP) (32°32’S, 152°18’E). The native plant species were selected based on three criteria: they co-occur with *C. monilifera* in coastal vegetation in NSW; they have fruits dispersed by vertebrates (determined from records in Barker and Vestjens 1989; Floyd 1989; Barker and Vestjens 1990; Eby and Palmer 1991; Green 1993; Gosper 1999a); and they produce fruits in sufficient quantity and accessibility for collection. The native plant species all differed from *C. monilifera* in growth form, however. They are all tall shrubs to trees, while *C. monilifera* has a variable shrubby habit.

The following variables of each fruit were measured: (i) length, width and weight of the whole fruit; (ii) the length, width and wet weight of the hard endocarp; (iii) wet weight of the combined pulp and pericarp; and, from these measurements, (iv) the wet weight ratio of endocarp to pulp was calculated. Water and nutritional content of the combined pulp and pericarp (the expected digestible portion of the fruit for vertebrates acting as dispersers of seed) was measured for *C. monilifera, M. elliptica* and *E. reticulatus*. Nutritional content was analysed by Weston Bioproducts, Enfield, NSW. For each species, the pulp and pericarp from all fruits were combined to obtain a sufficiently large sample for analysis of crude protein, phosphorus, fat content and sugar (glucose, fructose and sucrose). The protein measures are only a rough guide to true protein content; as the method used to estimate protein (total nitrogen x 6.25) does not account for other nitrogen-based compounds that are sometimes present (Izhaki 2002). Fermentation was detected in the sample of pulp and pericarp of *M. elliptica* used for sugar analysis, rendering the output unreliable. Consequently, sugar content values are not included for this species.

Over three periods of ten months, the numbers of *C. monilifera* inflorescences and ripe (black) fruit per plant were counted monthly. Ten plants were selected at random along each of three transects and one count per month was completed. The first sample was monitored from September 1995 to June 1996 (when plants were sprayed with herbicide) and the second and third sample from May 1999 to February 2000 (when several plants were run over by recreational 4WD vehicles and some died seemingly of natural causes). All samples were at Perkins Beach.

Monthly records were made of the plant species with vertebrate-dispersed fruits or seeds (determined from records in Barker and Vestjens 1989; Floyd 1989; Barker and Vestjens 1990; Eby and Palmer 1991; Green 1993; Gosper 1999a) in fruit. Records were collated separately for Myall Lakes NP (May 1996 to June 1997), Perkins Beach (June 1995 to June 1996 and May

1999 to February 2000) and Eurobodalla NP near Moruya Heads (35º56'S, 150°10'E) (June 1996 to May 1997).

2.3 Results

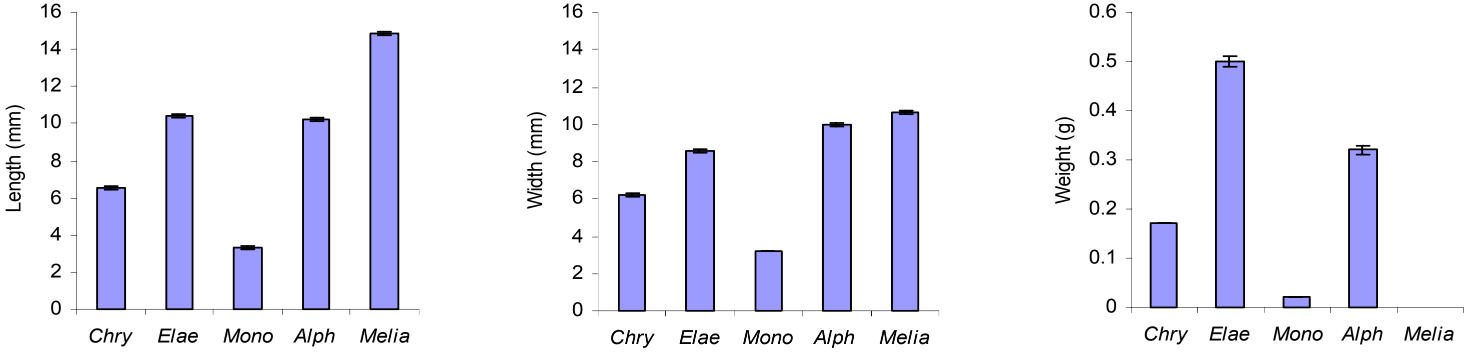
2.3.1 Morphology

The sizes of fruits, hard endocarps and pulps of *C. monilifera* were within the range of the other fruits sampled (Figure 2.1, Table 2.1). Fruits averaged 6.5 mm in length, 6.2 mm in width and 0.17 g in weight, with an endocarp 6.0 mm in length, 3.8 mm in width and 0.06 g in wet weight, and a wet weight of pulp and pericarp of 0.11 g. *Monotoca elliptica* fruits, endocarps and pulps were smaller than *C. monilifera* in all measurements, and those of *E. reticulatus*, *A. excelsa* and *Melia azedarach* larger in all measurements.

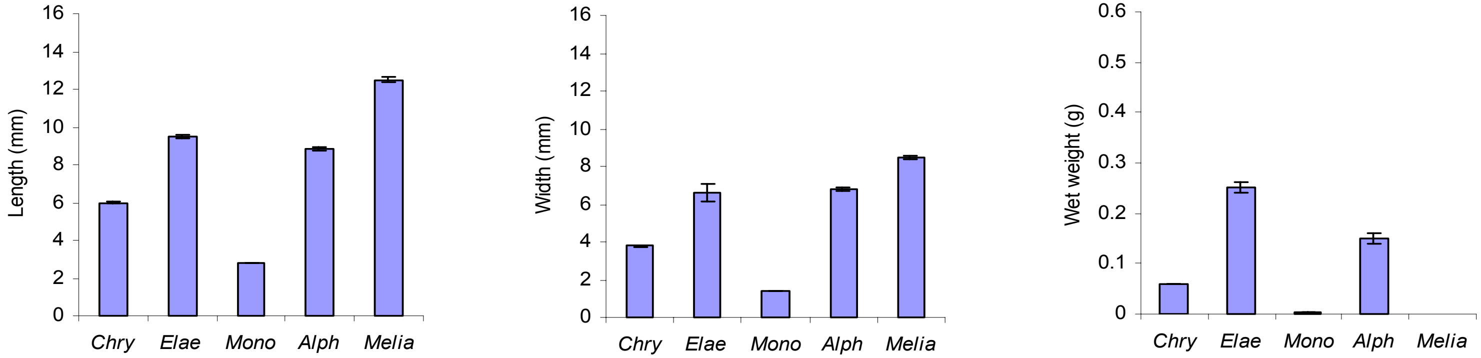
Monotoca elliptica had the lowest ratio of indigestible (for dispersers of seed) endocarp to pulp, making it the most profitable fruit for dispersers to consume per unit weight of seed (1:4.75) (Table 2.1). *Chrysanthemoides monilifera* fruits were the next most profitable for dispersers by this measure (1:1.83). Both *E. reticulatus* and *A. excelsa* fruits have a comparatively poor ratio of endocarp to pulp (approximately 1:1).

2.3.2 Nutrient content

Chrysanthemoides monilifera pulp and pericarp contained a substantially higher proportion of dry weight of sugars than *E. reticulatus* (Table 2.1). All of the sugar of *C. monilifera* fruits was fructose, while *E. reticulatus* fruits also contained small amounts of glucose and sucrose. *Elaeocarpus reticulatus* fruit pulp, however, contained slightly more protein and fat than *C. monilifera* and three times more phosphorus. *Monotoca elliptica* pulp contained more protein per dry weight than *C. monilifera* and nearly as much as *E. reticulatus*, and the greatest fat content of the three fruit pulps measured. *Chrysanthemoides monilifera* pulp contained the greatest proportion of water of the species sampled, and *E. reticulatus* the least by a substantial margin.



(a) Fruit length **(b) Fruit width** **(c) Fruit weight** ^{A}



(d) Endocarp length **(e) Endocarp width** **(f) Endocarp wet weight** ^{A}

Figure 2.1 - Dimensions and weights of fruits and endocarps of *Chrysanthemoides monilifera* and selected native plant species

Key: *Chry* = *Chrysanthemoides monilifera ssp. rotundata*; *Elae* = *Elaeocarpus reticulatus*; *Mono* = *Monotoca elliptica*; *Alph* = *Alphitonia excelsa*; *Melia* = *Melia azedarach*. Error bars are standard error, which are too small to be shown in some cases. Endocarp dimensions are those of the hard structure within the fruit that contained the seed(s). For the non-oval structure of *A. excelsa*, endocarp width is the maximum measurable. ^{A} weights were not measured for *M. azedarach*.

Table 2.1 - Fruit morphology and nutrient content of *Chrysanthemoides monilifera* and selected native plant species

Wet weight is mean ± s.e. from a sample of 50 fruit. Endocarp refers to the hard inner structure of the fruit encasing the seed(s). NM - Not measured

	Plant species	*Chrysanthemoides monilifera* ssp. *rotundata*	*Elaeocarpus reticulatus*	*Monotoca elliptica*	*Alphitonia excelsa*
Pulp	Wet weight (g)	0.11 ± 0.00	0.25 ± 0.01	0.02 ± 0.00	0.16 ± 0.01
	Water content (%)	83.7	59.8	81.8	NM
Nutrients [A]	Protein	0.85	1.45	1.30	NM
	Fat	0.50	0.80	3.30	NM
	Phosphorus (ppm)	146	503	140	NM
	Total sugars	22.80	2.19	NM	NM
	Glucose	<0.01	0.72	NM	NM
	Fructose	22.80	1.31	NM	NM
	Sucrose	<0.01	0.16	NM	NM
Ratio of endocarp : pulp (by wet weight)		1 : 1.83	1:1	1 : 4.75	1 : 1.06

[A] Nutrient content is expressed as a percentage of dry weight (except for phosphorus content).

2.3.3 Periods of inflorescence and fruit production

Chrysanthemoides monilifera produces inflorescences throughout the year (Parsons and Cuthbertson 1992; Figure 2.2). At Perkins Beach, the months of greatest inflorescence production were March through to May (Figure 2.2). Inflorescence production was relatively low throughout the remainder of the year.

Ripe fruits are produced throughout the year (Weiss 1984; Figure 2.3), with a peak in production in May and June at Perkins Beach (Figure 2.3*b*). Fruit production peaked in May and June in two of the three samples, and was high in the other sample over this period. A secondary peak in fruit production occurred in December. In one sample, more fruits were produced at this time than in May or June, and the other samples also exhibited substantial fruit production at this time. Fruit availability was consistently low in all samples over the period January to April and August to October.

At all three locations, late spring to early summer (November or December) was the period when ripe fruits were produced by the maximum number of native vertebrate-dispersed plant species (Figure 2.3*c*; Table 3.5 - Chapter 3). Relatively high numbers of native species also produced ripe fruits throughout summer and into early autumn. Winter and early spring (June to September) were periods of limited availability of native fruit types at all locations. Most native plant species had short, well-defined fruiting seasons (Table 3.5 – Chapter 3). *Chrysanthemoides monilifera*, and several other invasive species, however, fruited over longer periods of the year.

The number of plant species with vertebrate-dispersed fruit occurring in coastal vegetation declined from north (Myall Lakes) to south (Eurobodalla) (Table 3.5 – Chapter 3). Additionally, the temporal pattern of fruit availability became more restricted north to south. At Myall Lakes, fruits of native plant species were available throughout the year. This was not the case at either Perkins Beach or Eurobodalla. Furthermore, Eurobodalla had the fewest months in which vertebrate-dispersed fruits (other than *C. monilifera*) were available.

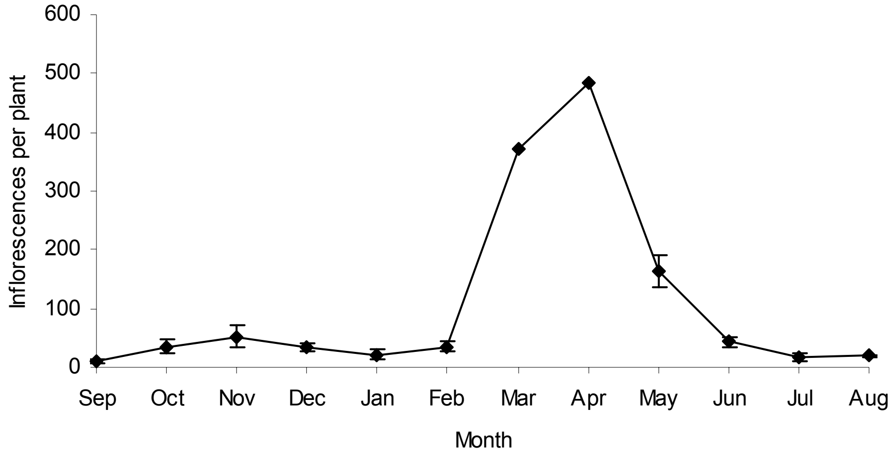


Figure 2.2 - Monthly inflorescence production of *C. monilifera*

Mean number of inflorescences per plant with standard error, calculated from three samples of ten plants, except July and August (two samples) and March and April (one sample).

2.4 Discussion

2.4.1 Morphology

Among the species sampled from coastal vegetation, *C. monilifera* fruits were intermediate in size. Compared to a larger sample of fruit and seed masses from plants occurring in temperate wet sclerophyll forest in south-eastern Australia (French 1991, several of which do co-occur with *C. monilifera*), *C. monilifera* fruit mass was lower than average. *Chrysanthemoides monilifera* was at the lower end of profitability for dispersers in terms of the endocarp to pulp ratio compared to these species (French 1991).

A consequence of the lesser dimensions of *C. monilifera* fruits and endocarps, compared with a range of native vertebrate-dispersed fruits, is that they would be physically suitable for ingestion by a wide diversity of native frugivores. Smaller fruits often have more dispersal agents than large fruits (Herrera 1985). The small fruit and endocarp size may have contributed to the large

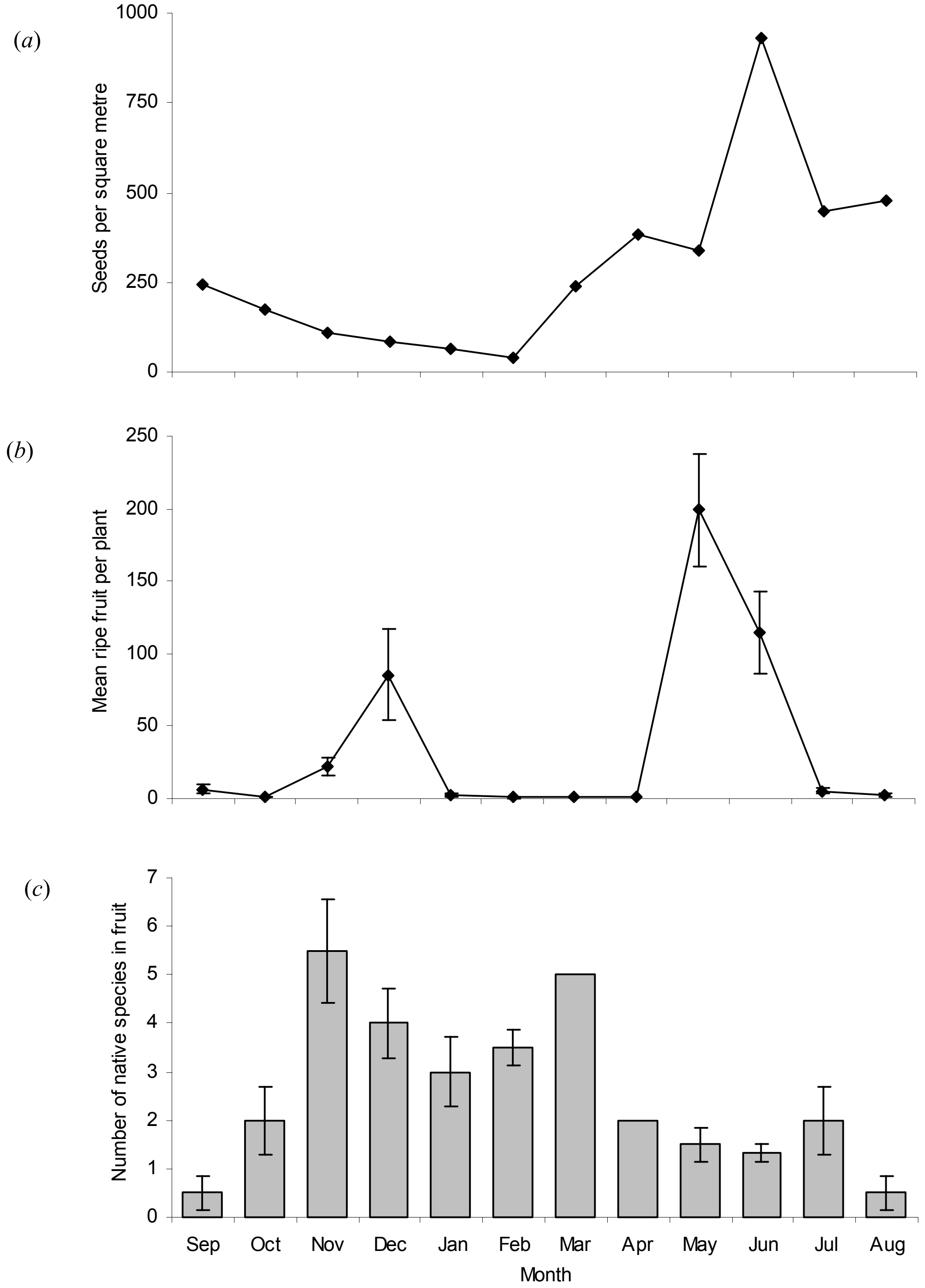


Figure 2.3 - Monthly fruit production of *C. monilifera* and co-occurring native plant species

(*a*) *Chrysanthemoides monilifera* from Weiss (1984). Mean of two years of seeds collected in seed traps at Moruya, NSW; (*b*) *C. monilifera* from this study. Ripe fruit counted on the plant at Wollongong, NSW. Mean number of ripe fruit per plant with standard error calculated from three samples of ten plants, except July and August (two samples) and March and April (one sample); (*c*) number of native vertebrate-dispersed plant species in fruit per month at Wollongong (mean with standard error calculated from two years, except June (three years) and March and April (one year)).

number of bird species that consume *C. monilifera* fruits (Dodkin and Gilmore 1984; Gosper 1999a). In contrast, ingestion of the larger fruits and endocarps of *E. reticulatus, A. excelsa* and *Melia azedarach* are likely to be limited by the gape width of dispersal agents; consequently, fewer species are likely to be able to disperse them.

2.4.2 Nutrient content

Elaeocarpus reticulatus pulp and pericarp contained relatively little sugar per dry weight compared to *C. monilifera*. Per fruit this difference is lessened, because *E. reticulatus* fruits have over twice the wet weight of pulp and pericarp and less watery fruit. From an energetic perspective, *E. reticulatus* fruits may also be more advantageous due to higher protein and fat contents. From the perspective of vertebrate dispersers, however, these nutritional benefits of *E. reticulatus* may be offset by the bulk of the indigestible endocarp (reflected in the poor endocarp to pulp ratio), allowing less room in the guts of dispersers for other food. The types of sugars in the fruit pulps of these two species may also be important. Fructose was the only sugar present in measurable quantities in *C. monilifera* pulp and pericarp. Fructose (and glucose) is preferred over sucrose by many frugivorous birds, as some lineages of birds are unable to digest sucrose (Martínez del Rio and Restrepo 1993) (although little is known of the digestive physiology of groups of birds of Australian origin). This may render *C. monilifera* fruits more attractive to frugivorous birds than total sugar figures alone would suggest.

Compared to a range of vertebrate-dispersed fruits in south-eastern Australia (French 1991), *C. monilifera* was roughly average in sugar and water content per unit mass, while *E. reticulatus* was substantially less than average in both variables. *Chrysanthemoides monilifera* fruits would appear to be moderately profitable for vertebrate dispersers in terms of sugar content compared to a range of native species. The pulp of many native species, however, was more rewarding in crude protein, fat and phosphorus (French 1991; Table 2.1). Each of these nutritional factors may influence bird food choice (Sorensen 1981; Sorensen 1984; Murray *et al.* 1993; Stiles 1993). There is little indication from these results that the nutritional value of *C. monilifera* fruits should cause them to be either highly favoured or substantially less desirable than native fruits.

2.4.3 Periods of inflorescence and fruit production

The peak period of inflorescence production recorded for *C. monilifera* in this study may be slightly earlier than that reported in Weiss (1986), when flowering was mainly between April and July. Patterns of *C. monilifera* inflorescence production appear similar in NSW to its native range in South Africa. In South Africa, *C. monilifera* also flowers throughout the year, has highly seasonal peak production, and there is some indication of multiple annual peaks in inflorescence production (Scott 1996).

The pattern of *C. monilifera* fruit production was broadly in accordance with the available information on fruit production in eastern Australia (Weiss 1984; NSW National Parks and Wildlife Service 2001) and South Africa (Liversidge 1972; Scott 1996). The May peak in production in this study, however, occurred a month earlier than recorded by Weiss (1984) (Figure 2.3*a v.* 2.3*b*). This may reflect either different timing of peak fruiting at these different locations along the NSW coastline (Wollongong versus Moruya), or differences in the survey technique. As both studies were an average of samples from more than one year, inter-year variation, including the influence of recent climatic conditions, is possible but is less likely to explain the observed differences. Weiss (1984) measured seed production by the capture of seeds in seed traps, which may exhibit a lag time from the fruits being ripe on the plant (as measured in this study). However, peak periods of inflorescence production as reported in Weiss (1986) were also a month later than peak flowering in this study, suggesting that the difference at these coastal locations or in these years may be real.

The peak in fruit production followed roughly one month after the peak in inflorescence production (Figures 2.2; 2.3*b*). High inflorescence production did not always precede proportionally high levels of fruit production, as also found by Liversidge (1972). This may be due to any number of factors, such as differences in pollinator efficiency or changes in resource availability. Variation in both inflorescence and fruit production between plants was substantial, probably influenced by the methodology of randomly selecting whole plants for survey. This resulted in a range of possible contributors to high variation, including differences in plant size and resource availability. Variability in the timing and gross differences in the quantity of fruit produced (some *v.* none) were recorded for a number of native species between the years of sampling at Perkins Beach (Table 3.5 – Chapter 3). This is not atypical among vertebrate-dispersed plant species (Herrera 1998b).

The peak late autumn-early winter period of fruit production in *C. monilifera* occurs when few native species are in fruit in coastal vegetation (Figure 2.3*b v.* 2.3*c*). Additionally, of those that are in fruit at this time, several differ substantially from *C. monilifera* in fruit morphology. For example, *E. reticulatus* fruits are substantially larger (Figure 2.1), while Sweet Pittosporum (*Pittosporum undulatum* Vent. (Pittosporaceae)) presents seeds for dispersal within a viscous fluid. These characters would substantially influence the disperser assemblage. The secondary peak of *C. monilifera* fruit production occurs partially within the late spring to autumn period when many native plant species are in fruit. In South Africa, *C. monilifera* spp. *monilifera* also fruits over a period when there is little competition with other indigenous plant species for dispersal agents (Knight 1986). It seems that *C. monilifera* in NSW has filled a similar ecological role.

Only one native species, Coastal Fan-flower (*Scaevola calendulacea* (Andrews) Druce (Goodeniaceae)), produced fruit in every month of the year, like *C. monilifera*. Most native species produced fruit over relatively short periods, unlike *C. monilifera* and unlike two other invasive plants in coastal vegetation, Lantana (*Lantana camara* L. (Verbenaceae)) and Black Nightshade (*Solanum nigrum* L. (Solanaceae)) (Table 3.5 – Chapter 3). These invasive species with extended fruiting seasons substantially alter temporal patterns of fruit availability in coastal vegetation. Areas with these invasive species now have fleshy fruits readily available throughout the year, and the periods of fruit shortage characteristic of native coastal vegetation have been lessened.

No measure was made of the quantity of fruit produced per month for native species. This is likely to be greatest over late spring-early summer, however, as several of the species that fruit at this time are abundant in coastal vegetation, produce prolific quantities of fruit and appear to fruit relatively reliably (pers. obs.). It is not known if the amount of fruit produced by naturalised *C. monilifera* results in late autumn-early winter now eclipsing late spring-early summer as the period of greatest total fleshy fruit availability, but it undoubtedly alters the temporal availability of fruit in coastal vegetation substantially.

The consequences for frugivore populations of the alteration of temporal fruit availability have yet to be determined. French and Zubovic (1997) found that bird species that rely more heavily on plant resources (including fruits) were less abundant and diverse in *C. monilifera* stands than in native vegetation. That study, however, was conducted over spring and summer, when fruits from native plants were likely to have been abundant, but when *C. monilifera* fruits were likely to have been relatively less important to frugivores.

2.4.4 Conclusion

Chrysanthemoides monilifera fruits are of suitable physical dimensions for being ingested by a wide diversity of bird dispersers. The pulp and pericarp are moderately rewarding in terms of sugar content. Other valuable nutrients, such as phosphorus, protein and fat, would be more profitably sourced by frugivores from other fruits, however, including those of the native species tested here. Each of these fruit characteristics of *C. monilifera* is similar to those of some naturally occurring species in coastal vegetation, as would be expected given the generality and universality of frugivore-plant interactions (Howe and Smallwood 1982;

Wheelwright and Orians 1982; French 1991). The main distinction between *C. monilifera* and native species in terms of fruiting characteristics is their phenology. *Chrysanthemoides monilifera* produces fruits throughout the year and particularly over periods when few species of native plants are in fruit, especially those of similar morphology. This exposes the fruit of *C. monilifera* to the greatest potential range of dispersers and allows dispersers to use fruits over periods of the year in coastal vegetation that would naturally be poor in fruit resources.

These fruit and crop characteristics are likely to substantially influence fruit-consuming species. They also explain why *C. monilifera* fruit are taken by such a wide diversity of native birds (Dodkin and Gilmore 1984; Gosper 1999a) and why native birds quickly adapted to using this novel fruit source. This, in turn, has undoubtedly contributed to the successful dispersal of the propagules of this invasive species in coastal NSW.

In the following three chapters, I investigate bird use and rates of removal of native and *C. monilifera* fruits in the field. The results of the work in this chapter will be important in interpreting these subsequent observations and experiments.

CHAPTER 3. PLANT FOODS OF BIRDS IN COASTAL NSW AND PREDICTED IMPACTS OF *CHRYSANTHEMOIDES MONILIFERA* ON PLANT-BIRD INTERACTIONS

The New Holland Honeyeater (*Phylidonyris novaehollandiae*) (left) is common in coastal vegetation, feeding on a variety of nectar foods, and occasionally on fruits. Coastal Banksia (*Banksia integrifolia*) (right) nectar is an important winter food of New Holland Honeyeaters and other coastal birds.

Preface

In this chapter, I describe bird usage of plant-based foods in coastal vegetation. From this assessment, I identify bird groups, individual species and bird-plant interactions likely to be substantially affected by either further *C. monilifera* expansion or control. The data on which this Chapter are based have been published (Gosper 1999a). These data are reworked here to improve their contribution to the overall project objectives. Gosper (1999a) contains notes on interesting or unusual observations of food type and/or foraging technique for particular bird species, which are not repeated here.

Publication (included as Appendix A1):

Gosper, C.R. (1999) Plant food resources of birds in coastal dune communities in New South Wales. *Corella* **23**, 53-62.

3.1 Introduction

Many species of birds rely directly on plants for food. They use a range of resources, including nectar, parts of flowers, pollen, leaves, exudates, seeds and fruits. Likewise, some plants depend on birds to achieve pollination and/or seed dispersal. Extensive lists of the food sources of Australian birds (particularly plant foods) have been published by several authors, most notably several chapters in Ford and Paton (1986), Floyd (1989), Barker and Vestjens (1989; 1990) and Lepschi (1993; 1997). These accounts cover broad geographic areas and a variety of habitat types by collating either a range of published sources and/or opportunistic personal records. Detailed studies of herbivory by birds within a defined region and/or habitat include Crome (1975), Paton and Ford (1977), Holmes (1987), French (1990) and Green (1993). Overall, however, there is still much to learn about the diets of most species of birds, and of the food sources of suites of birds at particular localities.

Few studies have investigated the use of invasive plants by native fauna, as habitat and/or as food (see Table 1.1). Invasive plants provide food for native animals (Buchanan 1989; Loyn and French 1991; Gosper and Gosper 1996), and in some cases (Knight 1986; Waring *et al.* 1993), the exotic food has been no less favoured than native alternatives. Many invasive plants in eastern NSW produce vertebrate-dispersed fruits, and potentially provide a large food source for native fauna, particularly in areas where much of the native vegetation has been lost. For example, Date *et al.* (1991) and Gosper (1994) both identified the importance of the fruit of the invasive weed *Cinnamomum camphora* for fruit-eating birds in northern NSW, particularly in sustaining populations of several species of pigeon.

Invasive plants can have a variety of impacts on ecosystem processes (Vitousek *et al.* 1996) and cause changes in floristic communities (for example Braithwaite *et al.* 1989). These floristic changes would subsequently alter the type, variety and abundance of plant food resources, and presumably, birds would respond to these changes (such as found by French and Zubovic 1997). Describing the resources used by birds in habitats dominated by invasive plants, and at risk of invasion, will be important in understanding how invasive plants affect birds (French and Zubovic 1997). Although food resources are undoubtedly important for birds, other changes subsequent to weed invasion, such as in vegetation structure, may be equally important in determining the impact of weed invasion on native species.

Bitou Bush (*Chrysanthemoides monilifera* ssp. *rotundata*) is invasive in coastal NSW and produces fruits consumed by a number of species of birds and mammals (Dodkin and Gilmore

1984; Gosper 1999a). An essential component of an assessment of the importance of *C. monilifera* (relative to other plant species) to birds is a survey of the plant food sources used by each bird species. The plant species that provide food resources for a large number of bird species, and the bird species that have the most diverse diets (on plant foods), can be identified. Subsequently, the generality of the relationships between plants and birds can be ascertained and likely specialised plant-bird interactions (for either pollination or seed dispersal) identified. Although bird use of plant foods from particular coastal plant species have been opportunistically recorded, notably of *C. monilifera* fruits and Coastal Banksia (*Banksia integrifolia*) nectar (Dodkin and Gilmore 1984), there have been no assessments of the diets of birds in coastal vegetation communities as a whole. The aim of this part of the study was to:

(i) document the plant foods used by birds in coastal dune communities in NSW;
(ii) identify the plant species with floral and fruit resources used by a wide variety of bird species and those with specialised interactions;
(iii) identify the diversity and types of plant food used by different bird species; and
(iv) use this information to predict the impact of *C. monilifera* invasion on the ecological functioning of these communities; including identifying bird species likely to change in abundance following *C. monilifera* invasion, and consequently which bird-plant interactions are likely to be disrupted. Identifying the food sources of birds will also assist in identifying those birds and interactions likely to be affected by *C. monilifera* removal. This can then be compared to the observed outcomes from these community changes (Chapters 6 and 7).

3.2 Methods

Over the period March 1995 to January 2000, I recorded plant foods of birds in coastal vegetation through opportunistic, direct observation of feeding birds and analysis of pellet content. Pellet records were confined to those species whose pellets could be readily recognised, namely Pied Currawongs (*Strepera graculina*) and Ravens (*Corvus* spp.). I paid particular attention to birds feeding at flowers to determine if insects, nectar, and/or flower parts were taken. Birds inserting their bills into flowers, and holding them steady for prolonged periods, were interpreted as taking nectar. In contrast, birds that ‘snatched’ at items in and around flowers were assessed to be foraging for invertebrates. Birds feeding on flower parts ripped off whole flowers or flower parts in their bill and chewed them. Even if flower components were then discarded (indicating that nectar may have been the food ingested), this foraging technique was assessed as obtaining flower part foods, which probably included nectar.

Observations were made at a range of coastal sites from Lennox Head in northern NSW to Moruya Heads in southern NSW. The bulk of observations were from the four locations used in the experiments of Chapters 5 to 7; Bundjalung/Yuraygir National Park (NP), Myall Lakes NP, Perkins Beach Recreation Reserve and Eurobodalla NP (sites described in section 5.2). For the purposes of this study, coastal dune communities were defined to include all vegetation on fore-dunes and hind-dunes.

I kept a monthly record of those plant species producing vertebrate-attracting flowers and/or ripe fruiting bodies as I traversed set paths at three survey locations (Myall Lakes, Perkins Beach and Eurobodalla). This included a subjective assessment of the overall quantity of the resource available as a ranked score, relative to periods of peak production for that particular species: - = no flowers/fruits present; x = small numbers of flowers/fruits, or low proportion of plants in flower/fruit; X = moderate to high quantity of flowers/fruits and most plants in flower/fruit.

To allow a visual assessment of the diversity of food types used by birds, I have arranged each bird species for which use of a plant food item was recorded into foraging guilds (Tables 3.1 and 3.2). Bird species were arranged according to how other researchers within Australia have classified the species, or closely related species, on the basis of field observations. For this process, I consulted Recher *et al.* (1985), Ford *et al.* (1986), Woinarski and Tidemann (1991) and MacNally (1994). The generality in the diet of birds that consume plant products in coastal habitats will be assessed by examining the number of plant species for which a food type (nectar, fruit *etc*) was used by bird species that was not expected based on a narrow interpretation of their foraging guild classification.

3.3 Results

Fifty-three species of birds were observed using 83 plant foods, from 65 plant species. Seventy-five percent of these food items for particular bird species have not been previously documented. The plant foods used by each bird species are shown in Table 3.1 (fruits, seeds and elaiosomes) and Table 3.2 (nectar, flowers, sap and leaves). Table 3.3 details the number of plant species of each food type (nectar, fruit *etc*) utilised by each bird species.

3.3.1 Plant visitors

Leaves were rarely used as a food source by birds. Only a single bird species in coastal vegetation, the Crimson Rosella (*Platycercus elegans*) ate leaves, of Bangalay (*Eucalyptus botryoides*) and Coastal Beard-Heath (*Leucopogon parviflorus*) (Table 3.2). New Holland

Honeyeaters (*Phylidonyris novaehollandiae*) and Silvereyes were the only species to feed on plant exudates. Both fed on the exudate (manna) of a Coastal Wattle (*Acacia sophorae*) from a large wound probably caused by wind damage (Table 3.2). Both bird species fed at the same wound on at least two occasions (Gosper 1999a).

Birds used fruiting bodies and flower resources much more frequently. For fruits, seeds and other fruiting appendages, the plant species used by the greatest number of bird species were *C. monilifera* (18 species of birds), *L. parviflorus* (16 species), Tree-heath (*Monotoca elliptica*) (14 species) and *A. sophorae* (14 species) (Table 3.1). Both species of Epacridaceae (*M. elliptica* and *L. parviflorus*) common on coastal dunes and *A. sophorae* had fruits frequently taken by several of the larger honeyeaters (wattlebirds, *Anthochaera* spp., and in the case of the epacrids, friarbirds, *Philemon* spp.) and lorikeets (*Trichoglossus* spp.). These birds are typically regarded as being mostly reliant on nectar.

Several plant species were visited by a large number of bird species that were harvesting nectar. These were *B. integrifolia* (visited by 21 species of birds, with two additional species feeding destructively on flower parts), Swamp Mahogany (*E. robusta*) (21 species), Saw Banksia (*B. serrata*) (12 species), Broad-leaved Paperbark (*Melaleuca quinquenervia*) (10 species) and Coral Tree (*Erythrina X sykesii*, invasive) (10 species) (Table 3.2). Several bird species not noted for usually using nectar resources, such as the Regent Bowerbird (*Sericulus chrysocephalus*), Satin Bowerbird (*Ptilonorhynchus violaceus*), Brown Thornbill (*Acanthiza pusilla*), Variegated Fairy-wren (*Malurus lamberti*) and Forest Raven (*Corvus tasmanicus*), were observed visiting *B. integrifolia* flowers for nectar. Likewise, Red-whiskered Bulbul (*Pycnotus jocosus*, invasive), White-throated Treecreeper (*Cormobates leucophaeus*) and Common Myna (*Acridotheres tristis*, invasive) visited *Eucalyptus robusta* flowers unexpectedly for nectar (Gosper 1999a).

3.3.2 Bird diets

Approximately 50% of the bird species recorded in coastal dune environments over the several years of this study consumed at least one plant-derived food. Thirty-five percent took a fruit, seed or elaiosome food, and 30% used floral resources. Twenty percent of all birds fed on both fruit and floral resources (Table 3.3). Lewin's Honeyeaters (*Meliphaga lewinii*) used the greatest number of plant species for food (24 food items from 23 plant species, including fruits, seeds, arillate seeds and nectar). Silvereyes (22 food items from 21 plant species; fruit, seed, arillate seed, nectar and exudate foods), New Holland Honeyeaters (15 food items from 15 plant species; nectar and a single fruit and single exudate food) and Eastern Spinebills (*Acanthorhynchus tenuirostris*) (14 nectar foods) also used large numbers of plant species for food. Silvereyes utilised the greatest diversity of plant food types (five), followed by Crimson Rosellas, Lewin's Honeyeaters, Pied Currawongs and Red-whiskered Bulbuls (four).

Approximately 20% of the individual bird-plant foraging interactions involved a bird species using a food type not expected from a narrow interpretation of their foraging guild classification (Tables 3.1 and 3.2). For example, where a putative nectarivore ate fruit, or a putative carnivore ate either fruit or nectar. Omnivores were expected to consume all types of plant food for the purposes of this exercise, even though several have been rarely previously recorded using nectar foods, such as the ravens. Of the five bird species putatively classed as frugivores, three were recorded feeding on nectar or flower parts. For one of them, the Green Catbird (*Ailuroedus crassirostris*), a flower was the only food item recorded (this species was rare at the study areas) (Table 3.2). All other so-called 'frugivores' used a wider variety of fruits, seeds and arils than nectar or flowers. Destructive feeding on flower parts was observed on each occasion a putative granivore fed at flowers.

Of 14 species of putative nectarivores, 50% took at least one fruit or seed in coastal vegetation (Table 3.1). Three of the 'nectarivores' not recorded taking fruits were rare in coastal vegetation, and recorded only using a single plant food (the nectar of *E. robusta*). The remaining species used, on average, fruits or seeds from fewer plant species than for nectar. Interestingly, putative nectarivores/carnivores (regarded as more reliant on invertebrate foods than the nectarivores *i.e.* honeyeaters with shorter bills) used less fruits and seeds, with only the Yellow-faced Honeyeater (*Lichenostomus chrysops*) (out of five species) taking any foods of these types.

Twelve putative insectivorous/carnivorous species (including ground, shrub, bark and foliage feeders) took at least one plant food; fruits, nectar or both. Plant food items were recorded for eight omnivorous species, with all taking fruits or seeds, with five also using floral resources.

3.3.3 Temporal patterns of plant resource availability

The plant-derived food types most widely used by birds in coastal vegetation, flowers and fruiting bodies, were available throughout the year (Tables 3.4 and 3.5). At some locations or in some years, however, the only vertebrate-dispersed fruits available were from invasive *C. monilifera*. The period when the most vertebrate-dispersed plant species produced fruits was over late spring and early summer (discussed in more detail in Chapter 2). This was also when the indigenous plant species used by the most frugivores were in fruit. Several plant species with nectar resources used by birds were in flower throughout the year. There was no glaringly obvious season of peak number of species in flower. The plants visited by the most bird species, with one exception, flowered over autumn-winter.

3.4 Discussion

In this study, bird use of foods produced by plants was assessed, namely nectar, flower parts, leaves, exudates, fruiting bodies and seeds. Investigating the usage of these resources provides information on the plant species that appear to be particularly important sources of food for birds, the bird species that use a greater variety of plant foods, and how disruption of natural plant-animal interactions by an invasive plant may affect future community composition.

3.4.1 Study biases

The information in this chapter is based on unstructured observations. As such, there are sampling biases that warrant discussion before the results of the study are interpreted further. The observations may be biased towards the more abundant bird species or those more easily observed, common or larger plant species (Wheelwright *et al.* 1984), plant species that flowered or fruited regularly or for an extended period, and to situations in which I suspected particular birds to be foraging on particular plant foods. Greater attention was allocated to fruiting *C. monilifera, M. elliptica*, Blueberry Ash (*Elaeocarpus reticulatus*), Hard Quandong (*E. obovatus*) and Red Ash (*Alphitonia excelsa*), species in which I had a particular research interest (Chapter 5). No effort was made to account for these plant or bird characteristics in the observations. Hence, I can identify interactions between plants and birds that appear to be quantitatively important in coastal habitats, but am unable to investigate their importance in further detail. For example, I cannot indicate whether the observation that some plant resources are used by more bird species than others is because the plants or resources are more abundant, or if they are preferred food sources for birds. Similarly, the seasonal assessment of plant resource availability does not explicitly account for plant abundance or the quantity of resources produced, so it does not measure overall resource availability for birds.

Furthermore, inter-annual variability (which was not explicitly examined in this study) cautions against assuming the annual repetition the plant-bird interactions described. The composition of bird diets (Herrera 1998a) and plant phenology (Herrera 1998b) are known to vary over time. The list of plant-bird interactions presented here is not definitive; undoubtedly more plant food types are used by all bird species; and plants receive other visitors for floral and fruit resources.

Logically, the abundance, diversity and spatial and temporal distribution of other food resources, such as invertebrates, has a major influence on birds, but it was beyond the scope of this study to attempt to identify these resources in coastal vegetation. An exception was the use of cicadas (Hemiptera: Cicadidae) by birds at Perkins Beach Reserve, Primbee (Gosper 1999b). The impact of invasive plants or their control on invertebrate communities has been little studied (see Table 1.1). However, French and Eardley (1997) found that *C. monilifera* invasion made no difference to litter invertebrate species richness and abundance. Some taxa, however, were more abundant in either native or *C. monilifera* vegetation. The subsequent impact on ground-foraging insectivorous birds and other components of community structure and functioning remain unstudied. Birds, of course, require a range of resources other than food, such as shelter from predators and for breeding and roosting, and sites to interact with other birds. No measurement of the effect of *C. monilifera* invasion or removal on these resources has been made, or if changes in these resources may have interacting effects with changes induced by altered plant food resources.

As a result of these study biases, I am limited in this discussion to describing how changes, due to *C. monilifera* management, in the availability of plant-produced foods used by birds will affect the bird community and plant-bird interactions.

3.4.2 Plant-bird interactions

Plant visitors

The food resources of some plant species were used by more species of birds than others. Use by a greater diversity of birds is inferred to indicate one aspect of higher importance as a food source for the bird community as a whole, in conjunction with the season of visitation, the amount of visitation and the quality of the resource provided (which were not assessed).

Floral visitors - *Banksia integrifolia, E. robusta, B. serrata, Melaleuca quinquenervia* and *Erythrina X sykesii* were visited for nectar and other floral resources by the greatest number of bird species. Some received visits by a range of specialised (*e.g.* honeyeaters, lorikeets and Silvereyes, which have brush-tipped tongues adapted for the extraction of nectar), and unspecialised (*e.g.* thornbills (*Acanthiza* spp.) and rosellas (*Platycercus* spp.)) bird species. They may also receive flower visits from a range of invertebrates and mammals. Visits to flowers, however, do not always result in successful pollination (examples described in Gosper 1999a). The ability of particular bird species to act as pollinators probably depends both on the frequency of flower visits, which is unknown, and behaviour at the flowers. Floral visits by unspecialised birds were particularly characteristic of winter in coastal vegetation. It is likely that these plants widely used by nectarivorous birds in coastal habitats have fairly diffuse relationships with individual pollinator species, and that they are important in supporting a diverse bird community.

Banksia integrifolia and *Eucalyptus robusta* were the plant species visited for flower resources by the greatest number of unspecialised bird species (eight and four respectively). In monsoonal woodland in the Northern Territory, another member of the Proteaceae, the Fern-leaved Grevillea (*Grevillea pteridifolia*), and another eucalypt, Darwin Woollybutt (*E. miniata*), were the most visited plant species by opportunistic nectarivores (Franklin 1999), also mainly in winter. This perhaps reflects the possibilities of a generality of inflorescence/flower structure facilitating visits by unspecialised birds among some species within these plant groups, greater nectar production, and/or that visits to plants for nectar may be more rewarding in winter. *Banksia serrata* flowers, in contrast to those of *B. integrifolia*, were not visited by any unspecialised bird species. *Banksia serrata* differs from *B. integrifolia* in flowering mainly over summer (*c.f.* winter, Table 3.4), and having flowers with a longer perianth and pistil (George 1996), which may limit the ability of birds with short bills to access nectar (such as many of the unspecialised floral visitors). Eucalypts have relatively unspecialised flowers (Paton 1986a), in that nectar can be accessed from a variety of positions and directions by a variety of floral visitors. This contrasts with more specialised flowers, such as the tubular flowers of *Styphelia viridis*, which only permit access to the nectaries (without destructive foraging) among birds by the most long-billed species. Consequently, Eastern Spinebills, the honeyeaters with the longest bills (Paton 1986b), were the only visitors to *S. viridis* flowers.

The possibility that use of nectar resources may be greater over winter may arise due to lower invertebrate activity with the cooler temperatures reducing food availability for birds, and/or that low invertebrate activity may allow nectar resources to accumulate unused at flowers (Paton 1986a). For example, unspecialised thornbills (including the Brown Thornbill, a visitor to *B. integrifolia* flowers) mainly use nectar resources when insect resources are scarce, in winter (Paton 1986a). Brown Thornbills took other plant foods in late spring, however. This suggests that suitable plant foods may be taken by this species whenever they are readily available.

Birds visited several plant species for floral resources that were unexpected due to the small size of the flowers and their presumed adaptation for insect pollination. Rare visits to the flowers of *Leucopogon parviflorus* (by Lewin's Honeyeater), *Pimelea linifolia* and *Hardenbergia violacea* (Eastern Spinebill) were made for nectar. All of these observations were made in winter, at the beginning of the flowering season of the plant (Table 3.4). Suggested reasons why floral visits may be more rewarding in winter have been outlined above, or alternatively, these observations may reflect birds experimenting with new food types when they initially become available.

Fruit and seed removal - The fruits/seeds of *Chrysanthemoides monilifera*, the two epacrids (*Monotoca elliptica* and *L. parviflorus*) and *Acacia sophorae* were used by the most species of birds. These species share a number of fruit characteristics that influence food choice by birds. All have small fruit size and small seeds or endocarps (<5 mm in width) (Harden 1990-1993; Chapter 2); thus are capable of being taken, and usually ingested, by most (if not all) species of frugivorous birds in coastal vegetation, including unspecialised foragers. Ingestion of fruits is limited by the gape size of birds, which in turn limits the range of potential dispersers. Small fruits are typically taken by a greater number of bird species than large fruits (Herrera 1985). All these plant species are shrubs, from 0.5 to 5 m in height, and present fruits in the outer foliage in quite conspicuous displays. With the exception of the invasive *C. monilifera*, these plants produce fruits over late spring and early summer.

Few bird species specialise in consuming fruits, seeds and elaiosomes in coastal vegetation, which has implications for plant-bird interactions. Birds that are able to use other food resources in lieu of fruits, such as invertebrates and nectar, may not be particularly responsive to changes in fruit abundance, and less likely to develop obligate relationships with plant species for seed dispersal. Many plants also have seeds dispersed by mammals, reptiles or invertebrates. Among unspecialised frugivorous birds, those that feed primarily on insects, such as the fairy-wrens, Golden Whistler (*Pachycephala pectoralis*) and Brown Thornbill, are likely to pass any ingested seeds of fleshy-fruited plants intact, as are species that feed mainly on nectar (Loyn and French 1991). When feeding on fruits, some birds do not ingest the seeds, either dropping them or leaving them attached to the plant (examples described in Gosper 1999a). Birds that use such foraging strategies have been termed “pulp consumers” (Jordano 1994), or “mashers” (referring to the crushing, or mashing of the fruit, separating the seed and pulp externally) (Levey 1987), and are generally regarded as not dispersing seed. The viability of seeds that are discarded by birds after the removal of the attractant is unknown, but is very unlikely to be reduced by the foraging activity in many cases, such as thornbills taking the elaiosomes off *Acacia sophorae* seeds (Gosper 1999a). Discarded seeds that remain viable are then available for dispersal by other mechanisms, such as by wind, water and other biotic agents (although with the loss of the attractant, further biotic dispersal may be unlikely). Seeds may then contribute to the soil-stored seed bank of the plant species.

Bird use of plant resources

Few bird species used only a single plant food type. Most bird species, for instance, did not only use nectar (in addition to unquantified invertebrate foods), but also took fruits, exudates or seeds on occasion. Of the 22 bird species for which four or more dietary items were recorded, only four used a single plant food type (Table 3.3). These were the Eastern Spinebill, Scarlet

Honeyeater (*Myzomela sanguinolenta*) and White-cheeked Honeyeater (*Phylidonyris nigra*), which were recorded solely using nectar resources (White-cheeked Honeyeaters feed on *C. monilifera* fruits, however; Dodkin and Gilmore 1984), and the Red-browed Finch (*Neochmia temporalis*), which took only seeds.

Several bird species were observed using plant food types that had not been previously recorded, or noted only on few occasions (see Gosper 1999a). It appears that many bird species are able and willing to take a variety of food types opportunistically when they are readily available. Similarly, Clifford and Drake (1981) identified as generalist many of the bird genera associated with the dispersal of diaspores and pollen in heath communities in eastern Australia (which share many species and genera of plants and birds with coastal habitats).

Dietary shifts

The diets of a number of bird species shifted according to season. Several honeyeaters (such as Yellow-faced Honeyeaters, Red Wattlebirds (*Anthochaera carunculata*) and Noisy Friarbirds (*Philemon corniculatus*)) and rosellas tended to use predominantly nectar and flower foods in winter, such as that of *B. integrifolia* and *Eucalyptus robusta*, but fruit foods over spring and summer, such as *Monotoca elliptica* and *L. parviflorus*. Seasonal frugivory is well documented among migratory Northern Hemisphere birds (Thompson and Willson 1979; Herrera 1984). The pattern observed here, however, differs in that the dietary shift occurs among species that are typically present in coastal vegetation throughout the year, with dietary change possibly a response to the abundance of different plant-produced foods. No frugivorous bird species occurred in coastal vegetation solely over the period of maximum fruit availability, although some increase in the abundance of particular species from local movements may have occurred. Similarly, minimal seasonal correlation between bird abundance and fruit abundance occurred in forests in southern Australia (French 1992) and New Zealand (Williams and Karl 1996). Alternatively, there is often a coincident peak of frugivore and fruit abundance in Northern Hemisphere ecosystems (Thompson and Willson 1979; Herrera 1984).

Few frugivorous bird species of coastal vegetation are regular migrants, with exceptions including one subspecies of the Silvereye and the Common Koel (*Eudynamys scolopacea*). Local nomadic movements, local wandering and supplementation of populations by partial migrants (altitudinal movements of Pied Currawongs, for example; Ford 1989) are more common. For many Australian birds, however, movement patterns are unpredictable, complex and poorly understood, or involve only portions of the population (Ford 1989). Such variability in bird movements, both between species and between years, would not facilitate the evolution of peak native fruit production coinciding with peak bird abundance and a critical period in the annual calendar of birds, as occurs in the Mediterranean, for example (Herrera 1984; 1995). The most common migratory bird in coastal vegetation (Chapter 7) is a subspecies of the Silvereye (*Z. l. lateralis*), of which some of the population winters in coastal vegetation. This subspecies apparently departs for Tasmania in September (Ford 1989) prior to the spring peak of fruit abundance in October-November (Table 3.5). Ford (1989) asserts that Australian landbirds typically migrate in short hops rather than in long journeys, which has implications for the development of narrow fruiting seasons based on bird movements. The period of peak fruit abundance, however, does coincide with the breeding period for many bird species.

Differences in the diets of birds spatially were also noted. In central NSW, for example, Noisy Friarbirds were not recorded foraging on the large quantities of *L. parviflorus* fruits available, while being present in the immediate area. In contrast, in southern NSW in the same season, the bulk of foraging records of Noisy Friarbirds were of *L. parviflorus* fruits. Differences in the availability of nectar resources between the two areas may have contributed to these dietary differences. At the northern area (Myall Lakes), *B. serrata*, a regularly used nectar source by Noisy Friarbirds, is more abundant and flowers over a longer period (Table 3.4). Reduced availability of this resource (at least over the survey period) at Eurobodalla NP (southern NSW) may have led to Noisy Friarbirds extensively using a less preferred food resource.

Periods of resource availability

The generalist foraging strategy of many birds, as described above, has obvious benefits where the availability of fruit and nectar resources peaks at different times, as in NSW coastal habitats. While floral and fruit resources used by birds become available sequentially throughout the year, the total resource available at any one time probably varies substantially. Two of the dominant tree species in coastal habitats, *B. integrifolia* and *Eucalyptus robusta*, flower in winter, and are used by a variety of birds. Similarly, the abundance and reliability of *M. elliptica, L. parviflorus* and *Acacia sophorae* fruiting in spring and summer probably outweighs native fruit crops at other times and may structure the native frugivore community. This pattern of fruit availability is not unique, indeed substantial seasonal differences in fruit abundance may be typical (Herrera 1985). Frugivore response to such patterns of fruit availability can include dietary shifts, as seems to be occurring for many bird species of coastal vegetation, or movement to other habitats.

Consequently, in coastal vegetation, nectar resources appear consistently greatest in late autumn and winter, and fruits and seeds most abundant in late spring and early summer. Franklin (1999), and Franklin and Noske (1999), describe a regular, seasonal, opportunistic use of floral resources, by a wide group of birds, in monsoonal woodlands in northern Australia. I consider that a similar response to seasonally prolific floral and fruit/seed resources occurs in coastal vegetation in NSW, in autumn/winter for flowers and spring/summer for fruits and seeds.

3.4.3 Predicted effects of *C. monilifera* invasion and control

Several important implications of *C. monilifera* invasion on plant-bird interactions in coastal communities can be drawn from this survey of plant foods of birds. Firstly, natural coastal plant communities provide extensive floral resources used by birds in autumn through to early spring and large quantities of fleshy fruits and seeds from late spring through to summer, although some of both these food types are available throughout the year. Invasion by *C. monilifera*, and consequent displacement of at least some native plants (Weiss and Noble 1984a), is likely to substantially alter this natural temporal pattern of food availability. *Chrysanthemoides monilifera* produces fruit throughout the year, although with a peak over autumn and early winter when few native plant species carry ripe fruit (Chapter 2), but plant species that attract many vertebrates with nectar resources are flowering. With *C. monilifera* invasion and partial displacement of native plants, fruits remain available throughout the year, but now has a skewed distribution to have an additional peak in abundance in late autumn and early winter, as opposed to a single peak in spring and early summer in uninvaded vegetation. Whether the lesser amounts of *C. monilifera* fruits produced over spring and summer would adequately replace a long-term decline in native fruits available for frugivorous birds over this period is unknown. The fruits of most native species that ripen in autumn and winter are comparatively large (Harden 1990-1993; Chapter 2), and are used by fewer bird species (Table 3.1). Other invasive plants also contribute to altered temporal availability of fruits in coastal vegetation (Chapter 2).

Frugivores

Most frugivores tended to be adaptable in terms of fruits consumed. Nearly 65% of all frugivores in this study use *C. monilifera* fruits (Dodkin and Gilmore 1984; Gosper 1999a). Therefore, the majority of frugivorous species are able and willing to use this new food source, although little information is available on the relative importance of *C. monilifera* in their diet, or if other specific resources at particular times are critical.

Invasive plants have extended the period of fleshy fruit availability into times when few native fruits were available in both North America and New Zealand (White and Stiles 1991; Williams and Karl 1996). In New Zealand, two bird species, the Silvereye (a recent colonist), and the invasive Blackbird (*Turdus merula*) benefited from these changes (Williams and Karl 1996). Examples of species that may respond positively to changes in fruit resources with *C. monilifera* invasion are indigenous Silvereyes, Lewin's Honeyeaters and Pied Currawongs and invasive Common Starlings (*Sturnus vulgaris*). These species have broad diets, take fruits throughout the year (including that of *C. monilifera*) and are mobile. However, these species may be affected by *C. monilifera* removal, such as through broad-scale herbicide treatment. In this situation, few native plants usually pre-exist in the treated area to provide an alternative fruit resource after removal (as the most severely infested areas are typically treated first).

A few bird species that take moderate quantities of fruits may be disadvantaged in the long-term if native plants are displaced by *C. monilifera*. These have either only been recorded using the fruit resources of native plant species, or appear to be seasonally frugivorous (in spring/summer) in coastal habitats. Nearly all of the bird species recorded feeding only on native fruits, such as Scaly-breasted Lorikeets (*Trichoglossus chlorolepidotus*) and Brown Thornbills, were only recorded using a very limited variety of plant species. The future availability of other foods, such as nectar in the case of the lorikeets and invertebrates for the thornbills, may be more important in determining the response of these species to *C. monilifera* invasion. It is unknown if these fruit resources may be critical for these bird species in certain seasons or years, or whether they are opportunistically taken when available. Some other bird species predominantly took fruit over the spring and summer period, such as Yellow-faced Honeyeaters, which used *C. monilifera* fruit in addition to those of native species, and the friarbirds, which were only observed feeding on the fruits of epacrids.

Bird species that are primarily seed predators that can use either the fruits or seeds of *C. monilifera* and/or of other invasive plants in coastal vegetation (such as the rosellas and Red-whiskered Bulbuls) are less likely to be affected by native vegetation decline than birds that use little or no exotic plant food, such as Yellow-tailed Black Cockatoos (*Calyptorhynchus funereus*). Unlike most coastal birds, Yellow-tailed Black Cockatoos appear to be relatively specialised in terms of their foraging requirements. They feed on a minimal diversity of plant-derived foods; *B. integrifolia* and *B. serrata* seeds (Barker and Vestjens 1989) and/or weevils in their infructescences (Scott and Black 1981), borers in *Acacia* spp. stems and, if available, the seeds of Pine Trees (*Pinus* spp.) in adjoining urban areas (pers. obs.). Coastal vegetation may become progressively more marginal for this species if *B. integrifolia* (Dodkin and Gilmore 1984) and *A. sophorae* (Weiss and Noble 1984a) are displaced by *Chrysanthemoides monilifera*.

Nectarivores

Chrysanthemoides monilifera may displace plants that provide widely used floral resources for birds (Dodkin and Gilmore 1984). Floral resources are likely to be critical for a number of coastal bird species, for example New Holland Honeyeaters (Paton 1982). Energetic limitations are likely to be greatest in winter, which is when the native nectar resources used by the most bird species are available. Bird species that rely on floral resources are, on the whole, likely to be at a greater disadvantage following *C. monilifera* invasion than mainly frugivorous or granivorous species. The quantity of floral resources available would be expected to decline in peak periods, and no new resources will become available at other times of year (*C. monilifera* flowers do not attract bird visits). Bird species unable to adapt to habitat that is fruit-rich, but nectar-poor, are predicted to be adversely affected by *C. monilifera* invasion.

Twenty-three bird species were recorded foraging at *B. integrifolia* flowers, one of the most-used nectar sources for birds in coastal habitats. Of these, seven were not recorded feeding on any fruit or seed foods, and 11 are not known to use *C. monilifera* fruits (Dodkin and Gilmore 1984; Table 3.1). At present, *B. integrifolia* exists in many coastal habitats as scattered mature trees above a dense *C. monilifera* shrub layer. This vegetation pattern may currently mask the actual long-term impact of *C. monilifera* invasion on nectarivorous birds, as bird species that rely on *B. integrifolia* nectar resources may be able to persist in many areas infested by *C. monilifera* as mature *B. integrifolia* remain. Some of these birds may make limited use of *C. monilifera* fruit resources, giving the impression that they may be able to persist in a changed community without, or with less, *B. integrifolia*. Little Wattlebirds, for example, were recorded on a single occasion consuming *C. monilifera* fruit, although other plant food items were mainly from native plants, including the extensive use of *B. integrifolia* nectar. Over time if, as expected, the mature *B. integrifolia* die without replacement (Dodkin and Gilmore 1984), the nectar resources for Little Wattlebirds and other species will be reduced substantially.

The nectar resource used by the next greatest number of bird species in coastal habitats was that of *Eucalyptus robusta*, which also flowers in autumn and winter. This species is less at risk from *C. monilifera* invasion, as *C. monilifera* does not establish well or grow as vigorously in the occasionally waterlogged habitat (Vranjic 2000) preferred by *E. robusta*. This habitat is naturally of limited extent, being confined to occasionally waterlogged near-coastal sites, and is becoming increasingly fragmented by development along the NSW coastline (Brereton 1996). Consequently, the nectar resources of this community would not be expected to provide for the large number of nectarivorous animals currently using the more widespread *B. integrifolia* scrub on coastal dunes, if this habitat continued to decline.

The high floral resource availability in autumn and winter in coastal habitats does not extend to other nearby forests. Law *et al.* (2000) identified late winter and spring as periods of regional blossom shortage for myrtaceous trees (*Eucalyptus, Corymbia, Lophostemon, Melaleuca, Angophora* and *Syncarpia*) on the mid-north coast of NSW. Significantly, *E. robusta* and *B. integrifolia* were identified as species that flower prolifically and reliably (on an almost annual basis) over much of this period. The data collected on flowering times in coastal vegetation in this study supports this conclusion. *Eucalyptus robusta* flowered every winter at each site, in contrast to the other Myrtaceous species. Blackbutt (*Eucalyptus pilularis*) flowered irregularly (though often from autumn to spring, as found by Law *et al.* 2000), while *C. gummifera, A. floribunda* and *E. botryoides* only flowered in some years. *Banksia integrifolia* also flowered regularly from autumn through to spring in each year at each site (Table 3.4).

Law *et al.* (2000) regard tree species with regular flower production, including *E. robusta* and *E. tereticornis* in coastal communities, as key species for mobile, nectarivorous fauna. This concentration of nectar resources in spatially limited patches (on a regional scale) of coastal vegetation in winter is likely to be highly significant for nectarivorous vertebrates. It may be somewhat analogous to the concentration of nectar resources in riparian *Melaleuca* forests during periods of regional nectar shortage in the Northern Territory (Woinarski *et al.* 2000). This case differs from the Northern Territory situation, however, in that the critical period of nectar production occurs when invertebrate abundance would be expected to be at a minimum. It also partially covers the period that fruit resources are also naturally most regionally scarce (late winter-spring in rainforests (Holmes 1987; Floyd 1990) and temperate forests (French 1991); and winter in coastal vegetation; Table 3.5). Recent plant invasions have modified this natural pattern of food resources somewhat, with a number of species (*e.g. Cinnamomum camphora, Chrysanthemoides monilifera, Ligustrum* spp. and *Morus alba*; Holmes 1987; Table 3.5) producing vertebrate-dispersed fruit through winter and/or early spring.

3.4.4 Conclusion

The results of this study support others (such as Forde 1986; Ford 1989; French 1990; French *et al.* 1992; Franklin 1999) which suggested that nectar, fruits and/or seeds are taken by many species of Australian birds that are not widely recognised as using those food types. Ford (1989) also stated that most Australian bird species are generalists. At times of high nectar, fruit or seed abundance, these foods constitute an easily obtained source of energy. In coastal dune communities in NSW, generalist birds could obtain the more widely used plant foods. Thus, most bird species visited a range of plant species for food, and similarly most plant species (for either or both pollination and seed dispersal) were visited by several bird species. Paton and Ford (1977) and Brown and Hopkins (1996) reached similar conclusions in the use of flower resources by birds in South Australia and Papua New Guinea respectively.

The available plant food resources for birds, and other vertebrates, in coastal vegetation vary throughout the year. A fruit-rich spring and summer and nectar-rich autumn and winter structures the native bird community. The invasion of coastal communities by *C. monilifera* alters this temporal pattern of food availability. Fruit production probably increases overall, and the temporal pattern is skewed to a new period of high abundance in late autumn and early winter. *Chrysanthemoides monilifera* fills a period of natural fruit scarcity, allowing year-round frugivory to become a more viable strategy for sedentary birds in coastal habitats. Nectar resources for birds are likely to decrease overall following *C. monilifera* invasion as native plants are displaced (Dodkin and Gilmore 1984; Weiss and Noble 1984a). Subsequent *C. monilifera* control will then remove these new fruit resources and, in the short term, have little impact on nectar resources. As a result of these resource changes, the following impacts on bird communities may be expected:

- On the whole, frugivores and granivores may exhibit inconsistent responses to *C. monilifera* invasion. Some species, that exhibit flexible foraging strategies, are likely to adapt well to the changed resource patterns and may increase in abundance or may spend longer periods of the year in coastal vegetation. These birds may, however, be vulnerable to *C. monilifera* removal. A few species with more specialised fruit requirements, and a number of other species that are highly facultatively frugivorous or only use small quantities of fruit, may be adversely affected by *C. monilifera* invasion, but this would depend on other vegetation and resource changes associated with *C. monilifera* invasion, which are currently unknown. The use of *C. monilifera* fruits by birds is examined in more detail in Chapter 4.
- Nectarivores are likely to be, on the whole, negatively affected by *C. monilifera* invasion, as *C. monilifera* itself provides no floral resources used by birds but it probably displaces nectar-producing native plants (Dodkin and Gilmore 1984). The degree to which individual species will be affected is likely to vary according to their specialisation of resource use: those that can supplement their diet with fruits are less likely to be adversely affected. Additionally, impacts on nectarivores are likely to increase with time since invasion, as more native vegetation elements are lost. The short-term impacts of *C. monilifera* removal of nectarivores are likely to be negligible, as plant resources used heavily by these birds are not likely to change in abundance.
- The response of mainly insectivorous species to *C. monilifera* invasion cannot be predicted from the information collected in this study. However, the work of French and Eardley (1997) indicated shifts in species composition, but not abundance or species richness in litter invertebrates with *C. monilifera* invasion. This may have subsequent impacts on insectivorous birds.

The responses of birds to the habitat changes induced by *C. monilifera* management are complicated by the diversity of resource use by many species. Few birds, for example, are wholly frugivorous or nectarivorous, but have a diverse diet. Some species may be mainly frugivorous over parts of the year or at some localities, but not at others, or always have a mixed diet. The response of these species to resource changes cannot be accurately predicted with the current qualitative information on resource use. However, the opportunity exists to test whether these predictions are borne out by the changes in coastal vegetation, as will be examined in Chapter 7 (invasion of *C. monilifera*) and 6 (removal). This may broadly provide some indication as to the extent to which birds appear to be responding to changes in the plant community and subsequent changes in plant food resources.

Plant-animal interactions such as pollination and seed dispersal are also potentially disrupted if the predicted bird community changes eventuate. Declines in nectarivores may have impacts on the pollination success of remaining native plants (Cox and Elmqvist 2000). Outcrossing appears to be particularly important for some *Banksia* species (Carthew *et al.* 1996), for example. Changes in fruit availability or disperser populations have the potential to affect the seed dispersal of native plant species (Christian 2001; Meehan *et al.* 2002). As impacts of *C. monilifera* invasion on frugivorous birds are predicted to be variable, consequences for the seed dispersal of bird-dispersed plants are difficult to predict. This interaction is examined experimentally in Chapter 5. Changes in floral and fruit resources will also potentially impact on other pollinators (such as fruit-bats *Pteropus* spp., arboreal mammals and invertebrates) and frugivores (*Pteropus* spp. and terrestrial mammals), which may have subsequent, though unpredictable, implications for birds and plant-bird interactions. Coastal *B. integrifolia* and *E. robusta* woodlands appear to be regionally important nectar resources for vertebrates (this study; Law *et al.* 2000). Consequently, the effects of *C. monilifera* invasion on native plants and birds may be felt more broadly, with the potential for disruption of pollination and seed dispersal in other ecosystems in eastern Australia. This may occur if *C. monilifera* invasion reduces a critical winter resource for mobile pollinators or through the provision of an alternative winter food source for mobile frugivores.

Table 3.1 - Fruit, seed and elaiosome foods of birds in coastal NSW

Table 3.1A Putatively frugivorous, granivorous and omnivorous birds

Codes following plant species identify food type: F - fruit; Sd - seed; A - arillate seed; El - elaiosome. [1] = Introduced taxa. Scientific names for birds in Table 3.3. Rationale for classification of birds in section 3.2.

Plant Species #	**Frugivores**	White-headed Pigeon	Regent Bowerbird	Mistletoebird	Red-whiskered Bulbul [1]	**Granivores**	Brown Quail	Bar-shouldered Dove	Yellow-tailed Black Cockatoo	Galah	Crimson Rosella	Eastern Rosella	House Sparrow [1]	Red-browed Finch	European Goldfinch [1]	**Omnivores**	Lewin's Honeyeater	Olive-backed Oriole	Figbird	Pied Currawong	Australian Raven	Forest Raven	Satin Bowerbird	Silvereye
Phytolaccaceae																								
Phytolacca octandra [1] *F*																								x
Chenopodiaceae																								
Chenopodium album [1] *Sd*					x								x											
Dilleniaceae																								
Hibbertia scandens A																	x	x			x			
Elaeocarpaceae																								
Elaeocarpus obovatus F																	x							
E. reticulatus F																				x				
Moraceae																								
Ficus watkinsiana F			x																					
F. obliqua F																	x							
Morus alba [1] *F*					x																			x
Rhamnaceae																								
Alphitonia excelsa F																	x							
Euphorbiaceae																								
Breynia oblongifolia F																								x
Flacourtiaceae																								
Scolopia braunii F					x																			
Cucurbitaceae																								
Diplocyclos palmatus F																	x							
Casuarinaceae																								
Casuarina equisetifolia Sd														x										
C. glauca Sd														x										
Proteaceae																								
Banksia integrifolia Sd									x															
B. serrata Sd									x															
Myrtaceae																								
Eucalyptus botryoides Sd											x													
Syzygium oleosum F																	x			x				
Acmena smithii F																	x							
Leptospermum laevigatum F											x													
Onagraceae																								
Oenothera stricta [1] *Sd*															x									

Table 3.1A cont.

Plant Species #	**Frugivores**	White-headed Pigeon	Regent Bowerbird	Mistletoebird	Red-whiskered Bulbul [1]	**Granivores**	Brown Quail	Bar-shouldered Dove	Yellow-tailed Black Cockatoo	Galah	Crimson Rosella	Eastern Rosella	House Sparrow [1]	Red-browed Finch	European Goldfinch [1]	**Omnivores**	Lewin's Honeyeater	Olive-backed Oriole	Figbird	Pied Currawong	Australian Raven	Forest Raven	Satin Bowerbird	Silvereye
Sapindaceae																								
Alectryon subcinereus A					x																			x
Guioa semiglauca A					x												x							x
Cupaniopsis anacardioides A																			x					
Fabaceae																								
Acacia sophorae Sd					x						x		x							x	x			x
A. sophorae El																								
Loranthaceae																								
Muellerina celastroides F				x																				
Santalaceae																								
Exocarpos cupressiformis A																								
Pittosporaceae																								
Pittosporum undulatum Sd																	x							x
Araliaceae																								
Polyscias elegans F			x														x							
Asteraceae																								
Taraxacum officinale [1] *Sd*														x										
Sonchus oleraceus [1] *Sd*											x				x									
Chrysanthemoides monilifera ssp. *rotundata* [1] *F*				x	x						x	x	x				x	x		x	x	x	x	x
Solanaceae																								
Duboisia myoporoides F																			x					
Solanum nigrum [1] *F*					x																			x
S. seaforthianum [1] *F*																	x							
S. stelligerum F											x													
Epacridaceae																								
Monotoca elliptica F					x												x	x	x					x
Leucopogon parviflorus F				x							x						x	x	x	x	x		x	x
Goodeniaceae																								
Scaevola calendulacea F							x										x			x		x		
Oleaceae																								
Notelaea longifolia F		x	x																					
Verbenaceae																								
Lantana camara [1] *F*					x																			x
Cyperaceae																								
Gahnia clarkei Sd																								x
Poaceae																								
Setaria gracilis [1] *Sd*								x						x										
Spinifex sericeus Sd										x														
Ammophila arenaria [1] *Sd*												x												

Table 3.1B Putatively nectarivorous and insectivorous/carnivorous species

Plant Species #	**Nectarivores**	Rainbow Lorikeet	Scaly-breasted Lorikeet	Red Wattlebird	Little Wattlebird	Noisy Friarbird	Little Friarbird	New Holland Honeyeater	**Nectarivore/carnivores**	Yellow-faced Honeyeater	**Insectivores/carnivores**	Superb Fairy-wren	Variegated Fairy-wren	Brown Thornbill	Striated Thornbill	Golden Whistler	Grey Shrike-thrush	Black-faced Cuckoo-shrike	Australian Magpie	Common Starling [1]	**Total bird species per plant**
Phytolaccaceae																					
Phytolacca octandra [1] *F*																					1
Chenopodiaceae																					
Chenopodium album [1] *Sd*																					2
Dilleniaceae																					
Hibbertia scandens A																					3
Elaeocarpaceae																					
Elaeocarpus obovatus F																					1
E. reticulatus F																					1
Moraceae																					
Ficus watkinsiana F																					1
F. obliqua F																					1
Morus alba [1] *F*																					2
Rhamnaceae																					
Alphitonia excelsa F																					1
Euphorbiaceae																					
Breynia oblongifolia F																					1
Flacourtiaceae																					
Scolopia braunii F																					1
Cucurbitaceae																					
Diplocyclos palmatus F																					1
Casuarinaceae																					
Casuarina equisetifolia Sd		x	x																		3
C. glauca Sd																					1
Proteaceae																					
Banksia integrifolia Sd																	x				2
B. serrata Sd																					1
Myrtaceae																					
Eucalyptus botryoides Sd																					1
Syzygium oleosum F																					2
Acmena smithii F																					1
Leptospermum laevigatum F																					1
Onagraceae																					
Oenothera stricta [1] *Sd*																					1
Sapindaceae																					
Alectryon subcinereus A																					2
Guioa semiglauca A																					3
Cupaniopsis anacardioides A																					1

Table 3.1B cont.

Plant Species #	**Nectarivores**	Rainbow Lorikeet	Scaly-breasted Lorikeet	Red Wattlebird	Little Wattlebird	Noisy Friarbird	Little Friarbird	New Holland Honeyeater	**Nectarivore/carnivores**	Yellow-faced Honeyeater	**Insectivores/carnivores**	Superb Fairy-wren	Variegated Fairy-wren	Brown Thornbill	Striated Thornbill	Golden Whistler	Grey Shrike-thrush	Black-faced Cuckoo-shrike	Australian Magpie	Common Starling [1]	**Total bird species per plant**
Fabaceae																					
Acacia sophorae Sd		x		x						x		x							x	x	12
A. sophorae El														x	x						2
Loranthaceae																					
Muellerina celastroides F																					1
Santalaceae																					
Exocarpos cupressiformis A																		x			1
Pittosporaceae																					
Pittosporum undulatum Sd																					2
Araliaceae																					
Polyscias elegans F																					2
Asteraceae																					
Taraxacum officinale [1] *Sd*																					1
Sonchus oleraceus [1] *Sd*																					2
Chrysanthemoides monilifera ssp. *rotundata* [1] *F*				x	x					x		x						x		x	18
Solanaceae																					
Duboisia myoporoides F																					1
Solanum nigrum [1] *F*																					2
S. seaforthianum [1] *F*																					1
S. stelligerum F																					1
Epacridaceae																					
Monotoca elliptica F			x		x		x	x		x		x	x	x		x					14
Leucopogon parviflorus F		x		x	x	x				x								x		x	16
Goodeniaceae																					
Scaevola calendulacea F																					4
Oleaceae																					
Notelaea longifolia F																					2
Verbenaceae																					
Lantana camara [1] *F*																					2
Cyperaceae																					
Gahnia clarkei Sd																					1
Poaceae																					
Setaria gracilis [1] *Sd*																					2
Spinifex sericeus Sd																					1
Ammophila arenaria [1] *Sd*																					1

Table 3.2 - Nectar, flower part, leaf and sap foods of birds in coastal NSW

Table 3.2A Putatively frugivorous, granivorous, insectivorous/carnivorous and omnivorous species

Codes following plant species: Fl - Whole flower or flower parts; N - nectar; Ex - stem exudate; L - leaves. [1] = Introduced taxa. Scientific names for birds in Table 3.3. Rationale for classification of birds in section 3.2.

Plant Species #	**Frugivores**	Green Catbird	Regent Bowerbird	Red-whiskered Bulbul [1]	**Granivores**	Crimson Rosella	Eastern Rosella	European Goldfinch [1]	**Insectivores/carnivores**	White-throated Treecreeper	Variegated Fairy-wren	Brown Thornbill	Spangled Drongo	Common Myna [1]	**Omnivores**	Lewin's Honeyeater	Pied Currawong	Forest Raven	Satin Bowerbird	Silvereye
Thymelaeaceae																				
Pimelea linifolia N																				
Rosaceae																				
Rubus sp. [1] *N*																				
Proteaceae																				
Persoonia stradbrokensis N																				
P. lanceolata N																				x
Banksia integrifolia N			x								x	x				x	x	x	x	x
B. integrifolia Fl						x	x										x			
B. serrata N																x				
B. aemula N																				
Myrtaceae																				
Corymbia gummifera N																				x
Corymbia maculata N																				
Eucalyptus botryoides L						x														
E. botryoides N																				
E. robusta N				x						x			x	x		x				x
E. tereticornis N																				
E. pilularis N																x				x
Leptospermum laevigatum N																x				x
Callistemon citrinus N																				
Melaleuca quinquenervia N																x				
Rutaceae																				
Correa alba var. alba N																				
Sapindaceae																				
Cupaniopsis anacardioides N																x				
Fabaceae																				
Acacia sophorae Ex																				x
Erythrina X sykesii [1] *N*																x				x
Kennedia rubicunda N																				x
Kennedia rubicunda Fl		x																		
Hardenbergia violacea N																				
Loranthaceae																				
Muellerina celastroides N																				

Table 3.2A cont.

Plant Species #	**Frugivores**	Green Catbird	Regent Bowerbird	Red-whiskered Bulbul [1]	**Granivores**	Crimson Rosella	Eastern Rosella	European Goldfinch [1]	**Insectivores/carnivores**	White-throated Treecreeper	Variegated Fairy-wren	Brown Thornbill	Spangled Drongo	Common Myna [1]	**Omnivores**	Lewin's Honeyeater	Pied Currawong	Forest Raven	Satin Bowerbird	Silvereye
Pittosporaceae																				
Billardiera scandens N																				
Pittosporum undulatum N																				x
Asteraceae																				
Bidens pilosa [1] *Fl*						x		x												
Hypochaeris radicata [1] *Fl*						x		x												
Solanaceae																				
Duboisia myoporoides N																				
Epacridaceae																				
Styphelia viridis N																				
Leucopogon parviflorus L						x														
L. parviflorus N																x				
L. parviflorus Fl						x														
Goodeniaceae																				
Scaevola calendulacea Fl																		x		
Verbenaceae																				
Lantana camara [1] *N*																				

Table 3.2B Putatively nectarivorous species

Plant Species #	**Nectarivores**	Rainbow Lorikeet	Scaly-breasted Lorikeet	Musk Lorikeet	Little Lorikeet	Swift Parrot	Red Wattlebird	Little Wattlebird	Noisy Friarbird	Little Friarbird	Brown Honeyeater	New Holland Honeyeater	White-cheeked Honeyeater	Eastern Spinebill	Scarlet Honeyeater	**Nectarivores/carnivores**	Blue-faced Honeyeater	Noisy Miner	Yellow-faced Honeyeater	Brown-headed Honeyeater	White-naped Honeyeater	**Total bird species per plant**
Thymelaeaceae																						
Pimelea linifolia N														x								1
Rosaceae																						
Rubus sp. ^{1} *N*												x										1
Proteaceae																						
Persoonia stradbrokensis N											x		x									2
P. lanceolata N																						1
Banksia integrifolia N		x	x				x	x			x	x	x	x	x		x		x	x	x	21
B. integrifolia Fl																						3
B. serrata N		x	x				x	x	x			x	x	x			x	x	x			12
B. aemula N		x	x																			2
Myrtaceae																						
Corymbia gummifera N												x										2
Corymbia maculata N		x					x															2
Eucalyptus botryoides L																						1
E. botryoides N												x										1
E. robusta N		x	x	x	x	x	x	x	x			x	x	x	x			x	x	x		21
E. tereticornis N																			x			1
E. pilularis N								x				x		x	x				x		x	8
Leptospermum laevigatum N								x				x	x						x			6
Callistemon citrinus N								x														1
Melaleuca quinquenervia N		x	x					x		x	x		x	x	x				x			10
Rutaceae																						
Correa alba var. alba N												x		x								2
Sapindaceae																						
Cupaniopsis anacardioides N								x						x								3
Fabaceae																						
Acacia sophorae Ex												x										2
Erythrina X sykesii ^{1} *N*		x					x	x	x			x		x			x	x				10
Kennedia rubicunda N												x		x	x							4
Kennedia rubicunda Fl																						1
Hardenbergia violacea N														x								1
Loranthaceae																						
Muellerina celastroides N												x			x							2
Pittosporaceae																						
Billardiera scandens N														x								1
Pittosporum undulatum N																						1
Asteraceae																						
Bidens pilosa ^{1} *Fl*																						2
Hypochaeris radicata ^{1} *Fl*																						2

Table 3.2B cont.

Plant Species #	**Nectarivores**	Rainbow Lorikeet	Scaly-breasted Lorikeet	Musk Lorikeet	Little Lorikeet	Swift Parrot	Red Wattlebird	Little Wattlebird	Noisy Friarbird	Little Friarbird	Brown Honeyeater	New Holland Honeyeater	White-cheeked Honeyeater	Eastern Spinebill	Scarlet Honeyeater	**Nectarivores/carnivores**	Blue-faced Honeyeater	Noisy Miner	Yellow-faced Honeyeater	Brown-headed Honeyeater	White-naped Honeyeater	**Total bird species per plant**
Solanaceae																						
Duboisia myoporoides N												x										1
Epacridaceae																						
Styphelia viridis N														x								1
Leucopogon parviflorus L																						1
L. parviflorus N																						1
L. parviflorus Fl																						1
Goodeniaceae																						
Scaevola calendulacea Fl																						1
Verbenaceae																						
Lantana camara [1] *N*														x								1

Table 3.3 - Food types used by each bird species in coastal NSW

[1] Introduced species. F - fruit; S - seed; A - arillate seed; El - elaiosome; N - nectar; Fl - whole flower or flower parts; Ex - exudate; L - leaves.

		Number of plant species of each food type used								
Common name	**Scientific name**	**F**	**S**	**A**	**El**	**N**	**Fl**	**Ex**	**L**	**Sum**
Brown Quail	*Coturnix ypsilophora*	1								1
White-headed Pigeon	*Columba leucomela*	1								1
Bar-shouldered Dove	*Geopelia humeralis*		1							1
Yellow-tailed Black-Cockatoo	*Calyptorhynchus funereus*		2							2
Galah	*Cacatua roseicapilla*		1							1
Rainbow Lorikeet	*Trichoglossus haematodus*	1	2			7				10
Scaly-breasted Lorikeet	*T. chlorolepidotus*	1	1			5				7
Musk Lorikeet	*Glossopsitta concinna*					1				1
Little Lorikeet	*G. pusilla*					1				1
Crimson Rosella	*Platycercus elegans*	4	3				4		2	13
Eastern Rosella	*P. eximius*	1	1				1			3
Swift Parrot	*Lathamus discolor*					1				1
White-throated Treecreeper	*Cormobates leucophaeus*					1				1
Superb Fairy-wren	*Malurus cyaneus*	2	1							3
Variegated Fairy-wren	*M. lamberti*	1				1				2
Brown Thornbill	*Acanthiza pusilla*	1			1	1				3
Striated Thornbill	*A. lineata*				1					1
Red Wattlebird	*Anthochaera carunculata*	2	1			5				8
Little Wattlebird	*A. chrysoptera*	3				9				12
Noisy Friarbird	*Philemon corniculatus*	1				3				4
Little Friarbird	*P. citreogularis*	1				1				2
Blue-faced Honeyeater	*Entomyzon cyanotis*					3				3
Noisy Miner	*Manorina melanocephala*					3				3
Lewin's Honeyeater	*Meliphaga lewinii*	12	1	2		9				24
Yellow-faced Honeyeater	*Lichenostomus chrysops*	3	1			7				11
Brown-headed Honeyeater	*Melithreptus brevirostris*					2				2
White-naped Honeyeater	*M. lunatus*					2				2
Brown Honeyeater	*Lichmera indistincta*					3				3
White-cheeked Honeyeater	*Phylidonyris nigra*					6				6
New Holland Honeyeater	*P. novaehollandiae*	1				13		1		15
Eastern Spinebill	*Acanthorhynchus tenuirostris*					14				14
Scarlet Honeyeater	*Myzomela sanguinolenta*					6				6
Golden Whistler	*Pachycephala pectoralis*	1								1
Grey Shrike-thrush	*Colluricincla harmonica*		1							1
Spangled Drongo	*Dicrurus bracteatus*					1				1
Black-faced Cuckoo-shrike	*Coracina novaehollandiae*	2		1						3
Olive-backed Oriole	*Oriolus sagittatus*	3		1						4
Figbird	*Sphecotheres viridis*	3		1						4
Australian Magpie	*Gymnorhina tibicen*		1							1
Pied Currawong	*Strepera graculina*	5	1			1	1			8
Australian Raven	*Corvus coronoides*	2	1	1						4
Forest Raven	*C. tasmanicus*	2				1	1			4
Green Catbird	*Ailuroedus crassirostris*						1			1
Regent Bowerbird	*Sericulus chrysocephalus*	3				1				4
Satin Bowerbird	*Ptilonorhynchus violaceus*	2				1				3
House Sparrow [1]	*Passer domesticus*	1	2							3
Red-browed Finch	*Neochmia temporalis*		4							4
European Goldfinch [1]	*Carduelis carduelis*		2				1			3
Mistletoebird	*Dicaeum hirundinaceum*	3								3
Red-whiskered Bulbul [1]	*Pycnonotus jocosus*	5	2	2		1				10
Silvereye	*Zosterops lateralis*	8	3	2		9		1		23
Common Starling [1]	*Sturnus vulgaris*	2	1							3
Common Myna [1]	*Acridotheres tristis*					1				1

Table 3.4 - Monthly availability of flower resources used by birds in coastal NSW

X = large crop of open flowers present
x = small crop of open flowers present
[1] = introduced plant species
- = no flowers present
Months in which location was not visited are left blank.

Table 3.4A Perkins Beach Recreation Reserve, Primbee

Year/month	1995							1996						1999								2000	
Plant species	J	J	A	S	O	N	D	J	F	M	A	M	J	M	J	J	A	S	O	N	D	J	F
Leptospermum laevigatum	-	-	X	X	X	x	-	-	-	-	-	-	-	-	x	x	X	X	x	-	-	-	-
Kennedia rubicunda	-	-	X	X	X	X	X	X	X	X	X	x	x	x	-	x	X	X	x	x	x	x	-
Pimelea linifolia	-	x	x	X	X	X	X	X	X	X	X	x	x	x	X	X	X	X	x	x	x	-	-
Billardiera scandens	-	-	-	-	X	X	X	X	-	-	-	-	x	x	-	x	x	x	x	x	x	x	x
Duboisia myoporoides	-	-	-	-	-	X	-	-	-	-	-	-	-	-	-	-	X	X	X	-	-	-	-
Banksia serrata	-	-	-	-	x	X	X	X	x	x	x	x	x	x	-	x	x	x	x	x	X	X	x
Lantana camara [1]	-	-	-	-	-	X	X	X	X	X	X	X	X	x	x	x	x	X	X	X	X	X	x
Corymbia (Euc.) gummifera	-	-	-	-	-	x	x	x	X	x	x	-	-	-	-	-	-	-	-	-	-	-	-
Rubus sp. [1]	-	-	-	-	-	-	X	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Persoonia lanceolata	-	-	-	-	-	-	X	X	X	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Angophora floribunda	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	X	x	x
Eucalyptus botryoides	-	-	-	-	-	-	-	X	X	x	x	-	-	-	-	-	-	-	-	-	-	-	-
Muellerina celastroides	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	x	-
Eucalyptus pilularis	-	-	-	-	-	-	-	-	-	-	X	X	X	X	X	X	X	x	x	x	-	-	-
Banksia integrifolia	X	X	X	X	x	-	-	-	x	X	X	X	X	X	X	X	X	x	-	-	-	x	x
Eucalyptus robusta	X	X	X	X	x	-	-	-	-	x	X	X	X	X	X	X	x	-	-	-	-	-	-
Total native species flowering	2	3	5	5	7	7	6	7	7	7	8	6	7	7	5	8	9	8	7	5	5	6	4
Total species flowering	2	3	5	5	7	8	8	8	8	8	9	7	8	8	6	9	10	9	8	6	6	7	5

Table 3.4 cont.

Table 3.4B Myall Lakes National Park, Hawk's Nest

Year/month	1996								1997					
Plant species	M	J	J	A	S	O	N	D	J	F	M	A	M	J
Leucopogon parviflorus	-	X	X	X	-	-	-		-			-	-	-
Cupaniopsis anacardioides	-	X	X	X	-	-	-		-			-	-	-
Leptospermum laevigatum	-	X	X	X	X	x	-		-			-	-	-
Eucalyptus tereticornis	-	-	X	-	-	-	-		-			-	-	-
Hardenbergia violacea	-	-	X	X	X	x	-		-			-	-	-
Kennedia rubicunda	-	-	-	X	X	x	-		-			-	-	-
Correa reflexa	-	-	-	X	x	-	-		-			-	-	-
Banksia serrata	x	x	X	X	x	x	X		X			X	x	x
Eucalyptus pilularis	-	-	-	-	-	-	-		X			x	x	-
Muellerina celastroides	-	-	-	-	-	-	-		X			-	-	-
Eucalyptus robusta	X	X	X	x	-	-	-		-			X	X	X
Banksia integrifolia	X	X	X	X	X	x	x		-			X	X	X
Melaleuca quinquenervia	-	X	X	-	-	-	-		-			-	X	X
Styphelia viridis	X	X	X	x	-	-	-		-			-	X	X
Correa alba	X	X	X	X	x	-	-		-			-	X	X
Total native species flowering	5	9	11	11	7	5	2		3			4	7	6
Total species flowering	5	9	11	11	7	5	2		3			4	7	6

Table 3.4C Eurobodalla National Park, Moruya Heads

Year/month	1996								1997					
Plant species	M	J	J	A	S	O	N	D	J	F	M	A	M	J
Leucopogon parviflorus		-	X		X	X		-	-			-	-	
Banksia serrata		-	-		-	-		x	X			-	-	
Eucalyptus botryoides		-	-		-	-		-	X			-	-	
Banksia integrifolia		X	X		x	x		-	-			X	X	
Correa alba		X	X		X	-		-	-			X	-	
Total native species flowering		2	3		3	2		1	2			2	1	
Total species flowering		2	3		3	2		1	2			2	1	

Table 3.5 - Monthly availability of fruit and seed resources used by birds in coastal NSW

X = large crop of ripe fruit or seed present
x = small crop of ripe fruit or seed present
- = no fruit or seed present
[1] = introduced plant species
Months in which location was not visited are left blank.

Table 3.5A Perkins Beach Recreation Reserve, Primbee

Year/month	1995							1996						1999								2000	
Plant species	J	J	A	S	O	N	D	J	F	M	A	M	J	M	J	J	A	S	O	N	D	J	F
Chrysanthemoides monilifera [1]	X	X	x	x	x	x	X	x	x	x	x	X	X	X	X	x	x	x	x	x	X	x	x
Pittosporum undulatum	X	X	-	-	-	-	-	-	-	-	-	-	-	-	X	x	-	-	-	-	-	-	-
Breynia oblongifolia	-	-	-	-	X	X	X	X	x	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Monotoca elliptica	-	-	-	-	X	X	X	X	x	x	-	-	-	-	-	-	-	-	X	X	-	-	-
Exocarpos cupressiformis	-	-	-	-	x	x	-	x	-	x	-	-	-	-	-	-	-	-	-	-	-	-	-
Morus alba [1]	-	-	-	-	X	X	-	-	-	-	-	-	-	-	-	-	-	X	X	x	-	-	-
Duboisia myoporoides	-	-	-	-	-	X	X	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Alphitonia excelsa	-	-	-	-	-	X	-	-	-	-	-	-	-	-	-	-	-	-	-	X	X	-	-
Acacia sophorae	-	-	-	-	-	X	X	x	-	-	-	-	-	-	-	-	-	-	-	X	x	-	-
Persoonia lanceolata	-	-	-	-	-	X	X	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Solanum nigrum [1]	-	-	-	-	-	X	X	X	X	X	x	-	-	X	x	x	x	x	-	X	X	x	x
Lantana camara [1]	x	-	-	-	-	X	X	X	X	X	x	x	x	x	x	x	x	-	x	X	X	X	X
Gahnia clarkei	-	-	-	-	-	-	-	-	X	X	-	-	-	x	-	-	-	-	-	-	-	X	X
Einadia hastata	-	-	-	-	-	-	-	-	X	X	x	-	-	X	x	x	-	-	-	X	X	X	X
Guioa semiglauca	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	X
Muellerina celastroides	-	-	-	-	-	-	-	-	-	x	-	-	-	-	-	-	-	-	-	-	-	-	-
Chenopodium album [1]	-	-	-	-	-	-	-	-	-	-	-	-	-	X	x	-	-	-	-	-	-	-	-
Elaeocarpus reticulatus	-	-	-	-	-	-	-	-	-	-	x	x	x	-	-	x	X	X	-	-	-	-	-
Total native species in fruit	1	1	0	0	3	7	5	4	4	5	2	1	1	2	2	3	1	1	1	4	3	2	3
Total species in fruit	3	2	1	1	5	11	8	7	7	8	5	3	3	6	6	6	4	4	4	8	6	5	6

Table 3.5 cont.

Table 3.5B Myall Lakes National Park, Hawk’s Nest

Year/month	1996								1997					
Plant species	M	J	J	A	S	O	N	D	J	F	M	A	M	J
Scaevola calendulacea	X	X	X	X	X	X	X		X			X	X	X
Polyscias elegans	X	X	X	x	-	-	-		-			-	-	X
Acmena smithii	-	-	X	X	-	-	-		-			-	-	-
Notelaea longifolia	-	-	X	-	-	-	x		-			-	-	-
Muellerina celastroides	-	x	-	-	X	-	-		-			x	x	-
Leucopogon parviflorus	-	-	-	-	X	X	X		x			-	-	-
Cassytha spp.	-	-	-	-	X	X	X		x			-	-	-
Monotoca elliptica	-	-	-	-	-	X	X		-			-	-	-
Persoonia lanceolata	-	-		-	-	X	X		-			-	-	-
Acacia sophorae	-	-	-	-	-	-	X		-			-	-	-
Leucopogon lanceolatus	-	-	-	-	-	-	X		-			-	-	-
Exocarpos cupressiformis	-	-	-	-	-	-	x		-			-	-	-
Alphitonia excelsa	-	-	-	-	-	-	-		X			-	-	-
Hibbertia scandens	x	x	-	-	-	-	-		X			X	x	-
Pittosporum undulatum	-	-	-	-	-	-	-		-			X	-	-
Cupaniopsis anacardioides	-	-	-	-	-	-	-		-			X	x	-
Chrysanthemoides monilifera [1]	X	X	X	x	x	x	x		x			X	X	X
Elaeocarpus reticulatus	X	X	-	-	-	-	x		-			-	X	X
Pittosporum revolutum	-	-	-	-	-	-	-		-			-	x	-
Total native species in fruit	4	5	4	3	4	5	10		5			5	6	3
Total species in fruit	5	6	5	4	5	6	11		6			6	7	4

Table 3.5C Eurobodalla National Park, Moruya Heads

Year/month	1996								1997					
Plant species	M	J	J	A	S	O	N	D	J	F	M	A	M	J
Chrysanthemoides monilifera [1]		X	X		x	x		x	x			X	X	
Einadia hastata		-	x		-	-		-	-			x	-	
Monotoca elliptica		-	-		-	X		X	x			-	-	
Leucopogon parviflorus		-	-		-	x		X	X			x	-	
Acacia sophorae		-	-		-	-		X	X			-	-	
Solanum stelligerum		-	-		-	-		X	X			-	-	
Total native species in fruit		0	1		1	2		4	4			2	0	
Total species in fruit		1	2		1	3		5	5			3	1	

CHAPTER 4. INVASIVE BITOU BUSH (*CHRYSANTHEMOIDES MONILIFERA* SSP. *ROTUNDATA*) FRUITS AS FOOD FOR BIRDS IN EASTERN AUSTRALIA

Ripe (black) and developing (green, tinged red) fruits of *Chrysanthemoides monilifera*. Fruits of *C. monilifera* were fed upon by many species of birds in coastal vegetation.

Preface

In this Chapter, I describe avian frugivory of *C. monilifera* in detail, identifying all the Australian birds known to feed on *C. monilifera* fruits, their fruit handling techniques and their abilities to act as dispersers of *C. monilifera* seeds. *Chrysanthemoides monilifera* fruits comprise an important resource for birds in *C. monilifera* stands, and assessing bird usage of this resource will be essential for interpreting experimental manipulations in forthcoming Chapters.

4.1 Introduction

Plant invasions are a significant conservation problem throughout the world. Among the diverse impacts of invasive plants on ecosystems can be reductions in abundance of native fauna or altered composition of the fauna community (Braithwaite *et al.* 1989; Griffin *et al.* 1989; Table 1.1). Invasive plants, however, can provide food and habitat for a range of native fauna (Knopf and Olson 1984; Loyn and French 1991; Westman 1990). In some cases, food provided by invasive plants has been at least as widely used as native alternatives (Knight 1986; Waring *et al.* 1993; French and Major 2001). Additionally, extensive use of food provided by invasive plants has contributed to population recovery, maintenance, or increase, of some indigenous bird species (Knight 1986; Date *et al.* 1991; Fulton and Ford 2001). Avian use of the fruits of invasive plants influences the ability of these plant species to colonise new areas and determines the distribution of propagules.

Following observations of widespread use of food resources of invasive plants, management recommendations have occasionally urged the retention of areas of invasive plants, at least in the short-term, to cater for fauna that may otherwise be unable to persist in their current abundances (Date *et al.* 1991; Wood 1993; Gosper 1994; Ekert and Bucher 1999). In these cases, removal of the invasive plant has been presumed to be detrimental to some fauna species, although this has rarely been experimentally tested.

Bitou Bush (*Chrysanthemoides monilifera* ssp. *rotundata*) is an exotic shrub that has been introduced to Australia from South Africa. It is now naturalised along more than 80% of the NSW coastline (NSW NPWS 2001). Invasion has impacted on native fauna, with French and Zubovic (1997) describing an effect on plant feeding birds, although not on overall bird abundance. ‘Invasion of native communities by Bitou Bush' has been listed as a threatening process under the NSW *Threatened Species Conservation Act 1995* (Dickman 1999), recognising the detrimental impact of *C. monilifera* on coastal vegetation communities and threatened species. *Chrysanthemoides monilifera* produces fleshy fruits, and birds and mammals constitute the primary seed-dispersal vectors (Dodkin and Gilmore 1984; Gosper 1999a).

The aim of this study was to document the species of birds that consume *C. monilifera* fruits in eastern Australia. Identifying these species should form a component of any assessment of the habitat value provided by *C. monilifera*, and be considered prior to any extensive control program. The method used by birds to handle fruits, their behaviour when feeding and seed deposition was also recorded where possible. Likely dispersers, predators and pulp consumers of *C. monilifera* fruits can thus be identified. This Chapter provides important natural history information useful for the interpretation of the impacts of *C. monilifera* invasion (and its removal) on plant-animal interactions and on bird species and communities.

4.2 Methods

I collected records of birds foraging on the fruits of *C. monilifera* opportunistically on visits to coastal vegetation from 1995 to 2000. This study formed part of a larger project aimed at examining plant resource use in coastal dune vegetation (Chapter 3). Locations were visited along much of the NSW coastline, although the bulk of the observations came from three regularly visited sites: Myall Lakes National Park (NP), Hawk's Nest (32°32'S, 152°18'E); Perkins Beach Recreation Reserve, Wollongong (34°49'S, 150°53'E); and Eurobodalla NP, Moruya Heads (35°56'S, 150°10'E). Foraging records included the direct observation of feeding birds (with the aid of binoculars) and the analysis of pellets. For only a few species could pellets be identified with accuracy: *Corvus* spp. and Pied Currawongs (*Strepera graculina*).

Previously published observations have been included to increase the comprehensiveness of this assessment of bird use of *C. monilifera* fruits. However, in some cases little additional information was available on the method of fruit-handling and foraging behaviour. Therefore, few conclusions can be drawn on the importance of these birds as consumers of *C. monilifera* fruits, and if the bird is a disperser or predator of seeds or consumer of the pulp of fruits. The classification of bird species into dispersers, predators or pulp consumers is based on observations of fruit handling by the birds (if available), or the classification of Loyn and French (1991). No testing of the viability of passed seeds was conducted.

4.3 Results

The fruits of *C. monilifera* were very attractive to birds. Eighteen species were observed during this study consuming *C. monilifera* fruits or parts thereof and an additional seven species have been recorded by other observers in Australia (Table 4.1). The majority of these bird species have been indigenous.

In this study, the bird species most often observed feeding on *C. monilifera* fruits at Myall Lakes NP were Pied Currawongs, Lewin's Honeyeaters (*Meliphaga lewinii*) and Silvereyes (*Zosterops lateralis*). At Perkins Beach, Silvereyes, Red-whiskered Bulbuls (*Pycnonotus jocosus*) and House Sparrows (*Passer domesticus*) fed most frequently on the fruits, while at Eurobodalla NP it was Silvereyes, Pied Currawongs and Yellow-faced Honeyeaters

(*Lichenostomus chrysops*). Few records were kept on the total number of fruit consumed from plants in foraging bouts, although one Olive-backed Oriole (*Oriolus sagittatus*) consumed between 30 and 40 fruit from a single *C. monilifera* plant in approximately two minutes.

Table 4.1 - Bird species that consume fruits of *Chrysanthemoides monilifera* in Australia

Species	Common name	Fate of seed	Source[1]
Dromaius novaehollandiae	Emu	Dispersed	B
Larus novaehollandiae	Silver Gull	Dispersed	C
Platycercus elegans	Crimson Rosella	Predated or pulp consumed	A; D
P. eximius	Eastern Rosella	Predated	D
Eudynamys taitensis	Common Koel	Dispersed?	B
Malurus cyaneus	Superb Fairy-wren	Dispersed	D
Anthochaera corniculata	Red Wattlebird	Dispersed	B; D
A. chrysoptera	Little Wattlebird	Dispersed	D
Meliphaga lewinii	Lewin's Honeyeater	Dispersed	B; D
Lichenostomus chrysops	Yellow-faced Honeyeater	Dispersed	D
Phylidonyris nigra	White-cheeked Honeyeater	Dispersed?	B
Coracina novaehollandiae	Black-faced Cuckoo-shrike	Dispersed	D
Oriolus sagittatus	Olive-backed Oriole	Dispersed	B; D
Sphectotheres viridis	Figbird	Dispersed?	B
Strepera graculina	Pied Currawong	Dispersed	A; B; D
Corvus coronoides	Australian Raven	Dispersed	D
C. tasmanicus	Forest Raven	Dispersed	D
Sericulus chrysocephalus	Regent Bowerbird	Dispersed?	B
Ptilonorhynchus violaceus	Satin Bowerbird	Dispersed	D
Passer domesticus#	House Sparrow	Predated	D
Dicaeum hirundinaceum	Mistletoebird	Dispersed	B; D
Pycnonotus jocosus#	Red-whiskered Bulbul	Dispersed	D
Zosterops lateralis	Silvereye	Dispersed	B; D
Sturnus vulgaris#	Common Starling	Dispersed	D
Turdus merula#	Common Blackbird	Dispersed	B

Notes: For each bird species the usual fate of the seed was determined from observations of fruit handling (including ingestion) by the species or records of undamaged seeds in pellets. No testing of the viability of passed seeds was completed. "?" indicates where there was no information recorded on fruit handling to determine the usual fate of the seed. The classification given is that of Loyn and French (1991). Species of birds only recorded feeding on Boneseed (*C. m.* ssp. *monilifera*) have not been included.# = Invasive bird species.[1] A = Weiss (1983); B = Dodkin and Gilmore (1984); C = Rose (2000); D = this study.

4.3.1 Dispersers

The majority of bird species that consume *C. monilifera* fruits are also likely to successfully disperse seeds. These species have been observed ingesting entire *C. monilifera* fruits and not undertaking any crushing or chewing on the seed likely to damage the hard endocarp. For only a single species, the Emu (*Dromaius novaehollandiae*), has actual germination of passed seeds been recorded (Dodkin and Gilmore 1984) or seed viability tested.

The size of bird dispersers ranged from small (Silvereyes, Superb Fairy-wrens, Mistletoebirds (*Dicaeum hirundinaceum*) and Yellow-faced Honeyeaters) to large (Emus). Several of the smaller species had difficulty at times ingesting *C. monilifera* fruits, or engaged in behaviour that may have manipulated fruit size. Yellow-faced Honeyeaters often dropped fruits in attempting to swallow them and fruits were regularly manoeuvred in the bill prior to swallowing. Superb Fairy-wrens manipulated fruits by hitting them against a branch several times prior to swallowing. This may have served to alter the fruits into a more manageable shape for swallowing, as practiced by Silvereyes when taking large fruits (Gosper 1999a).

Sites of deposition of *C. monilifera* seeds were recorded for few species; only those with pellets that could be identified. At Perkins Beach, Australian Raven (*Corvus coronoides*) pellets contained a variety of seeds (including *Chrysanthemoides monilifera*) and invertebrate and vertebrate remains (Gosper 1999a). Pellets were found (although this may reflect ease of discovery) amongst grassland on the foredune, underneath perches on the foredune and nearby scrub and along access paths to the beach. Corvid pellets (of Australian or Forest Raven (*Corvus tasmanicus*) or Torresian Crow (*Corvus orru*)) examined at Myall Lakes NP also contained *Chrysanthemoides monilifera* seeds. Ravens (including Forest Ravens; Secomb 1997) appeared to preferentially forage along paths through coastal vegetation, on the grassland of the foredune, and along the high-tide mark of the beach. Ravens may thus be important in the movement of *C. monilifera* seeds onto the foredune.

Regurgitated pellets from Pied Currawongs containing *C. monilifera* seeds were located on dunes, along beach access trails, over the bonnet of my car (when parked beneath large Bangalay (*Eucalyptus botryoides*) trees near bird feeding trays), and next to taps in a camping area. Pied Currawongs often regurgitate pellets after drinking (Robertson 1969), and may disproportionately move seeds to sites close to water, such as dripping taps, creeks and soaks. This may not be optimal for *C. monilifera* establishment and expansion, as occasionally waterlogged soils are not as suitable for *C. monilifera* growth (Vranjic 2000).

Rose (2000) recorded *C. monilifera* seeds in regurgitated Silver Gull (*Larus novaehollandiae*) pellets at resting sites at the edge of the water. From these sites of deposition, ocean currents, wind and wave action could move seeds to suitable locations for germination. Although the seeds were described as being from 'Boneseed' (usually meaning *C. m. monilifera*), the locality of this record (Forster) suggests that they were from ‘Bitou Bush’ (*C. m. rotundata*).

For other species, the fate of taken seeds can only be presumed based on bird behaviour. Common Starlings regularly feed on *C. monilifera* near urban areas, and typically fly large distances (hundreds of metres) when disturbed or following the completion of foraging. They may be important dispersal agents moving *C. monilifera* from the coast inland into urban areas. Several of the larger frugivores also tended to fly inland when disturbed, presumably to shelter in taller vegetation (*c.f.* the low scrub of *C. monilifera*) and view the area from prominent perches. These species, such as Black-faced Cuckoo-shrikes (*Coracina novaehollandiae*) and Olive-backed Orioles, may effectively disperse seed to the coastal edge of taller scrub.

4.3.2 Predators

Of the 25 species recorded consuming *C. monilifera* fruits, only three are probable seed predators (Table 4.1). Two of these, the rosellas (*Platycercus* spp.), chew and crush green fruits into many pieces in their bill, and presumably irreparably damage the seeds. The House Sparrow is presumed to be a seed predator, as although it ingested ripe fruits intact, their grinding gizzard is likely to destroy the seeds (Loyn and French 1991). Sparrows could destroy significant quantities of *C. monilifera* seeds from coastal vegetation adjacent to urban areas.

During this study, Crimson Rosellas (*Platycercus elegans*) were recorded utilising two techniques in handling fruits. Commonly, they acted as seed predators, picking and crushing immature (green) *C. monilifera* fruits in their bill before swallowing parts of the fruits. Alternatively, when feeding on ripe fruits, Crimson Rosellas acted as pulp consumers (see below). One rosella was observed to forage on green and ripe fruits on the one plant with the two techniques within a five-minute period. In Weiss (1983), Crimson Rosellas consumed 15% of the total *C. monilifera* seed source, feeding on the seeds of the plant and discarding the outer material. Crimson Rosellas may thus destroy a significant amount of *C. monilifera* seeds.

4.3.3 Pulp consumers

Pulp consumers are those species that consume parts of the flesh and pericarp of fruits but discard the seeds (without ingestion). Seeds are often discarded intact beneath the parent plant, and thus foraging in this way contributes little to seed dispersal but may not damage the seed. A single species acted as a pulp consumer of *C. monilifera,* the Crimson Rosella. Rosellas removed the flesh and juice of ripe fruit with their bill and then dropped the seed.

4.4 Discussion

4.4.1 Bird species that use *C. monilifera* fruits

Silvereyes, Lewin's Honeyeaters and Pied Currawongs are probably the most significant consumers (and probably dispersers) of *C. monilifera* fruits in NSW, although more quantitative measurements would be necessary to demonstrate this conclusively. Crimson Rosellas, and at a few locations close to urban areas, House Sparrows, are probably the only major avian seed predators. These rankings may well be derived from records simply in proportion with bird abundance or ease of observation, but they nevertheless provide a guide as to the bird species likely to be consuming the greatest quantities of *C. monilifera* fruits in central NSW. Weiss (1983), for comparison, recorded Pied Currawongs consuming 20% and Crimson Rosellas 15% of the fruit crop in a study in southern NSW. *Chrysanthemoides monilifera* fruits appeared to form a significant component of the diet of several bird species. Most of these were indigenous, although the invasive Red-whiskered Bulbul, House Sparrow and Common Starling all consumed substantial quantities of fruits at some locations. Each of these invasive birds was observed more frequently or exclusively at Perkins Beach, which is adjacent to urban and industrial areas. It is likely that the establishment of *C. monilifera* is rendering coastal vegetation more suitable for these species and facilitating their expansion into coastal habitats.

A high proportion of the bird species that consume *C. monilifera* fruits have increased in range, abundance, and/or have altered behavioural patterns in recent decades. Such changes should be regarded as similar indicators of community disarray as declines in species' range or abundance (Recher 1999). 'Increaser' species that consume *C. monilifera* fruits include Pied Currawongs, Silver Gulls, Red Wattlebirds, Common Koels, Eastern Rosellas and Australian Ravens (Smith *et al.* 1989; Higgins and Davies 1996; Higgins 1999; Reid 1999; Higgins *et al.* 2001). While the impact of *C. monilifera* expansion has almost certainly been trivial for some of these 'increasers' (*e.g.* Silver Gull; Higgins and Davies 1996), for others it may have been significant. For example, much of the cause of the population increase and changed migratory behaviour in the Pied Currawong has been attributed to increased abundance of exotic winter-fruiting plants providing food over a period of natural food stress, reducing juvenile mortality (Fulton and Ford 2001). *Chrysanthemoides monilifera* has not been explicitly identified as one of these exotic species, but it has the main characteristic: an autumn-winter peak in fruit production (Weiss 1984; Chapter 2). The fruit are also widely used by Pied Currawongs (Weiss 1983; this study).

'Increasing' bird species form a higher proportion of the species that consume *C. monilifera* fruits than among Australian birds as a whole. This indicates that birds that feed on *C. monilifera* fruits are, as a group, coping better with human-induced landscape change in Australia than other species. Of course, this is expected, as using a novel food source such as *C. monilifera* fruits illustrates flexibility in diet and use of habitat that is likely to reflect a greater ability to adapt to changing circumstances generally. Most of these bird species are unspecialised in diet, using a diversity of plant species and food types (nectar, fruits, invertebrates; Chapter 3). Species that include a significant component of fruits in their diet also appear to be more highly represented among 'increaser' species, compared to nectarivorous and insectivorous birds (Recher 1999). The generalised nature of fruit dispersal systems throughout the world (Wheelwright and Orians 1982) has facilitated the rapid adoption of exotic fruits by native frugivores (French 1991). Consequently, food provided by invasive plants in other systems has contributed to population increases in indigenous bird species (Knight 1986; Date *et al.* 1991). The rapid spread of *C. monilifera* (NSW NPWS 2001) and other invasive vertebrate-dispersed plants is likely to advantage some of these 'increaser' species.

Increases in the populations of species can have subsequent impacts on others. For example, increases in the abundance, range and changed movements of omnivorous nest-predators, such as Pied Currrawongs and Australian Ravens, may have caused declines in small birds in habitats as diverse as urban areas, rural woodland remnants and seabird breeding colonies (Priddel and Carlile 1995; Major *et al.* 1996; Fulton and Ford 2001). It is possible that the ability of these birds to widely use *C. monilifera* fruits has altered their occurrence in coastal vegetation, including near to where their predation effects have been significant, such as the breeding colony of Gould's Petrel (*Pterodroma leucoptera*) offshore from Myall Lakes.

4.4.2 Fruit and crop characteristics of *C. monilifera*

Chrysanthemoides monilifera fruits are an attractive food source for a diverse range of bird species, which is not surprising given its fruit and crop characteristics (Chapter 2). Morphological and nutritional measures of *C. monilifera* fruits fall within the range of native fruits, although there are distinctions in phenology. In appearance and size, fruits of *C. monilifera* bear a superficial resemblance to the native fruits of Soft Corkwood (*Duboisia myoporoides*) and Boobiallas (*Myoporum* spp.). Black fruits are frequent among bird-dispersed fruits across the world (Willson *et al.* 1989) and among invasive plants in Australia. It would therefore require little novel behaviour by birds to try *C. monilifera* fruits. These similarities in fruit characteristics are expected given the generality and universality of plant-frugivore interactions (Howe and Smallwood 1982; Wheelwright and Orians 1982; French 1991).

4.4.3 Conclusion

The ability and willingness of many bird species to feed on the fruits of *C. monilifera* have undoubtedly contributed to its invasiveness in Australia. Most of these bird species are indigenous and are probable dispersers of seed. These types of new mutualisms have been important in the invasion of numerous plant species worldwide (Richardson *et al.* 2000b) and possibly also in changing the abundance of indigenous frugivores (Fulton and Ford 2001). The bird species that consume large quantities of *C. monilifera* fruits, such as Silvereyes, Lewin's Honeyeaters and Pied Currawongs, are likely to be able to cope with habitat and resource changes associated with *C. monilifera* invasion of coastal vegetation. I predict they will increase in abundance compared to in uninvaded vegetation. However, when these new mutualisms are disrupted by *C. monilifera* removal, I expect these birds may decline in abundance relative to *C. monilifera*-dominated vegetation. I will test these predictions in Chapters 6 and 7. Changes in the abundance of frugivores could also affect the removal of fruits and dispersal of seeds of indigenous plants co-occurring with *C. monilifera* – this I will examine in the next Chapter.

CHAPTER 5. THE EFFECT OF *CHRYSANTHEMOIDES MONILIFERA* MANAGEMENT ON THE RATE OF REMOVAL OF VERTEBRATE-DISPERSED FRUITS

An experimental station used in this experiment, showing the end of the bamboo tomato stake, manuscript clips and the attached fruits of *Elaeocarpus reticulatus*. Green (unripe) *Chrysanthemoides monilifera* fruits are in the background. I measured the rate that fruits were removed from these stations.

Preface

In this chapter, I describe the first experimental component of this study. I investigate how the presence, and removal, of an invasive plant with vertebrate-dispersed fruits affects the rate that fruits are removed of several plant species.

5.1 Introduction

Biological invasions constitute a significant component of global environmental change (Vitousek *et al.* 1996). Among the changes to ecosystems invasive plants may cause are alterations to the composition of the faunal community, for example birds (Griffin *et al.* 1989) and invertebrates (Slobodchikoff and Doyen 1977; French and Major 2001). Plant invasions have occasionally benefited some native fauna by providing increased food and/or cover (Knopf and Olson 1984; Westman 1990). In some cases, the food from the invasive plant has been at least as favoured as native alternatives (*e.g.* Waring *et al.* 1993).

Any impacts of invasive plants on either flora or fauna may be magnified by flow-on effects to other organisms through the disruption of plant-animal interactions, such as pollination, seed dispersal and herbivory. For example, Howard-Williams and Davies (1988) reported an increase in the abundance of a crayfish and the introduced Black Swan (*Cygnus atratus*) in wetlands following invasion by the aquatic plant *Lagarosiphon major*. These increases then compounded the impact of the invasive plant by adversely affecting remaining native plants at this and nearby wetlands, through increased herbivory. The impact of invasions by plants that have fleshy fruits may be compounded by alteration of the foraging patterns and diets of local frugivorous vertebrates (Knight 1986). Vegetation with a large relative abundance of a fleshy-fruited invasive plant may provide a greater quantity (although perhaps reduced diversity) of fleshy fruits for consumption by vertebrates than natural vegetation, at least over some seasons. Increased quantities of fleshy fruits could affect the dispersal of native species with similar dispersal strategies in one of three ways:

(1) *Increase consumption and dispersal of native fruits*

This may occur if the vegetation supports a more diverse or abundant frugivore assemblage. Greater numbers of fruits of natives could be taken, in addition to the consumption of invasive plant fruits. In northern NSW, for example, the invasive Camphor Laurel (*Cinnamomum camphora*) provides a seasonally important food source for several species of frugivorous rainforest birds, which would otherwise be less abundant (Date *et al.* 1991; Gosper 1994). This could conceivably improve the dispersal of propagules of native rainforest plants, particularly those fruiting in other seasons. I can find no studies, however, that have investigated this.

(2) *Reduce consumption and dispersal of native fruits*

Greater fruit availability could reduce consumption and dispersal of native fruits through competition for a limited supply of seed dispersers. In southern Africa, the availability of large fruit crops of the invasive *Acacia cyclops* may have reduced avian frugivore use of native *Rhus* spp. (Knight 1986). Intra-specific competition between Olives (*Olea europaea*) for avian dispersers increases with lower frugivore diversity (Alcántara *et al.* 1997), and disperser populations can become satiated with large fruit crops (Herrera 1995). Dispersers are thought to select fruits according to certain fruit and crop characteristics (Sorensen 1981; Sorensen 1984; Murray *et al.* 1993; Stiles 1993), thus in communities with abundant fruits, dispersers can select those fruits most favoured. If the fruits of native plants were less favoured than fruits of the invasive plant, avian dispersers may take proportionally less of the native fruits.

(3) *Not affect the consumption and dispersal of native fruits*
This outcome could occur if there were sufficient differences in the dispersal agents of each species, if fruit production were temporally separated, or if dispersers were overabundant and not a limiting resource.

Removal of an invasive plant could also plausibly affect the fruit dispersal of native plant species and the remaining individuals of the invasive species. After removal, the total quantity of fleshy fruits available may decline dramatically, as little or no invasive plant fruits remain and there may be insufficient pre-existing native plants to provide a significant alternative fruit source. In habitats with invasive plants removed, fruit consumption by vertebrates of the native and invasive species could: (i) increase, as there would be less fruits present, thus reducing the options of frugivores; (ii) decrease, if frugivore abundance or diversity declined with the lower quantity of fruits available; or (iii) in the case of native plant species, not change if the fruits of native plants and those of the invasive plant were taken by different frugivores, at different times or by a group of generalist frugivores that consume fruits when available. The ability of the invasive plant to have fruits dispersed after most individuals have been removed has important implications for the success of control efforts, particularly in the ability of the invasive plant to re-colonise areas from where it has been largely eliminated.

Bitou Bush (*Chrysanthemoides monilifera* ssp. *rotundata*) is an exotic shrub introduced to Australia from South Africa. It is now widespread and at high density along more than 80% of the coastline of NSW (NSW NPWS 2001). *Chrysanthemoides monilifera* produces fleshy fruits, and birds and mammals constitute the primary seed-dispersal vectors (Dodkin and Gilmore 1984; Gosper 1999a).

Chrysanthemoides monilifera has the ability to invade coastal areas that have not obviously been disturbed by human activity (French and Eardley 1997), eventually displacing native plants and forming a monoculture. A number of impacts of *C. monilifera* invasion have been demonstrated on native species or communities. Weiss and Noble (1984a) found that it displaced the native plant Coastal Wattle (*Acacia sophorae*) and may cause decreased seed production in this species. French and Zubovic (1997) described an effect of *C. monilifera* invasion on plant-feeding birds, but there was no effect on overall bird abundance or on species richness. Likewise, no impacts on overall abundance of litter invertebrates were found in areas invaded by *C. monilifera* (French and Eardley 1997), but the abundances of particular taxa did differ between invaded and uninvaded areas. No studies have investigated the subsequent impact of the changes induced by *C. monilifera* invasion on plant-animal interactions.

In this study, I investigated whether the presence or removal of this invasive plant affected the specific plant-animal interaction of fruit-removal, for two native plant species and for *C. monilifera* itself. I investigated the following questions:

(1) Are fruits of different plant species removed at similar rates?
(2) Does the presence of *C. monilifera*, or its elimination, affect the rate of removal of fruits of native vertebrate-dispersed plants, compared to removal rates in native vegetation?
(3) Does the elimination of *C. monilifera* affect the rate of removal of *C. monilifera* fruits?
(4) Are the observed patterns of fruit-removal correlated with the occurrence of frugivorous bird species or certain fruit characteristics (including season, nutrient content and morphology)?

5.2 Methods

5.2.1 Study areas

I conducted the study at four locations along the NSW coastline:
(1) the adjoining Bundjalung National Park (NP), Iluka Nature Reserve and Yuraygir NP, near the Clarence River (29°24'S, 153°53'E) in northern NSW (hereafter BNP);
(2) Myall Lakes NP, north of Hawks Nest (32°32'S, 152°18'E) in northern NSW (ML);
(3) Perkins Beach Reserve, south of Wollongong (34°36'S, 150°53'E) in central NSW (PB); and
(4) Eurobodalla NP, near Moruya Heads (35°56'S, 150°10'E) in southern NSW (ENP).

All locations included areas of three habitat types: (i) natural habitat that was uninfested with *C. monilifera*, (ii) habitat now dominated by *C. monilifera*, and (iii) habitat previously dominated by *C. monilifera* but treated with herbicide within the preceding two years. The dominant plant species in each habitat type varied at the different locations, although this difference was substantially less than that between the habitats.

In habitat dominated by *C. monilifera*, the cover of *C. monilifera* exceeded 30% of the shrub layer. Amongst the *C. monilifera* were a variety of native plant species (most of which persisted following herbicide treatment in the *C. monilifera*-removed sites), most commonly *Banksia integrifolia* trees and *A. sophorae* shrubs. Native vegetation sites lacked *C. monilifera*, had a canopy of *B. integrifolia* and/or *B. serrata* (sometimes with *Eucalyptus* spp.) over either a heath understorey or littoral rainforest shrub layer. When applied to dense infestations, aerial herbicide application for *C. monilifera* control (see Toth *et al.* 1996) produces an open scrub of surviving native plants (such as *B. integrifolia, B. serrata* and *A. sophorae*), rare *C. monilifera* plants and large areas of the 'skeletons' of dead *C. monilifera* plants.

5.2.2 Study species

I selected plant species for use in this study according to the following four criteria:

(1) They occur in coastal vegetation types subject to *C. monilifera* invasion in NSW;

(2) They produce fleshy fruits dispersed by vertebrates, in particular birds; this being determined from published records (such as Barker and Vestjens 1989; Floyd 1989; Barker and Vestjens 1990; Eby and Palmer 1991; Green 1993; Gosper 1999a);

(3) They are representative of some of the range of fruit types available in coastal vegetation. For example, of the plants listed below, two have small fruits (<7 mm diameter) and three have large fruits (>8 mm diameter), they are a range of colours, and are available in a range of months; and

(4) They produce fruit in sufficient quantity and accessibility for collection.

I chose the following species:

Chrysanthemoides monilifera ssp. *rotundata* (Asteraceae): a sprawling shrub with small, drupe-like, black, approximately spherical fruits, containing a small endocarp. Fruits are ripe throughout the year but with a major peak in late autumn and early winter and a lesser peak in early summer. The pulp and pericarp contains little protein or fat, but a moderate content of sugars (22.8% of dry weight) (Chapter 2).

Monotoca elliptica (Smith) R. Br. (Epacridaceae): a shrub that produces small ovoid orange to red drupes, with a small endocarp. Fruits ripen in late spring and early summer. *Monotoca elliptica* pulp and pericarp contains a comparatively high amount of protein (1.3% of dry weight) and fat (3.3%) (Chapter 2).

Blueberry Ash (*Elaeocarpus reticulatus* Smith) (Elaeocarpacae): a small tree with comparatively large, blue, ovoid to globose drupes with a single large endocarp. Fruits ripen over many months in autumn and winter. Sugar content of the pulp and pericarp is low (<3% dry weight), but protein (1.45%) and phosphorus content is high (Chapter 2).

Hard Quandong (*E. obovatus* G. Don.) (Elaeocarpaceae): a tree with similar fruits to *E. reticulatus*, but they are slightly smaller (Harden 1990) and ripen in autumn.

Red Ash (*Alphitonia excelsa* (Fenzl) Reisseck ex Benth.) (Rhamnaceae): a tree producing comparatively large, black, ovoid to globose drupes that contain powdery orange 'flesh' over a large endocarp. The endocarp splits upon drying to expose 2 to 3 arillate seeds (Harden 1990). Fruits are available over summer (Chapter 2).

White Cedar (*Melia azedarach* L.) (Meliaceae): a deciduous tree with large, elliptical, yellow drupes containing a large bony endocarp. Fruits ripen in autumn and winter.

5.2.3 Experimental stations

Measuring differences in frugivore activity through the removal rates of fruits from actual plants may be biased by the familiarity of some individual animals with particular fruit sources. Frugivorous vertebrates would be expected to have learned the locations of existing food sources in their local area, and the same individuals may be expected to return to the same food sources on many occasions. Additionally, animals may visit food sources known historically in a habitat type now altered, and in other respects little used, and where new fruit sources may remain undiscovered. To avoid these potential biases, I measured fruit-removal rates from experimental (artificial) stations, modelled on the approach of French and Westoby (1992). The use of experimental stations reflects the appearance of a new location of a food source, which has to be discovered by fruit-eating vertebrates before any fruits can be consumed.

Experimental stations consisted of 30-60 cm bamboo tomato stakes, to which I attached two plastic 19mm manuscript clips to one end. I placed fruits, attached to their stalks, in these clips (see photo on Chapter front piece). I collected whole infructescences from plants growing naturally at each of the locations. Regular vertebrate dispersers or seed predators should, therefore, have been familiar with each fruit type, although some fruit types may have been more abundant in some habitats and locations than others.

I presented fruits in a standard fashion in each habitat, with stations tied to vegetation 0.7 to 2m from the ground. In *C. monilifera*-dominated and herbicide-treated habitats, this was typically on the upper branches of the shrub layer. In native coastal vegetation a more complex canopy of tall shrubs and trees generally occurred (Appendix A2), although stations were set out in the understorey in such a way as to mimic those in the other habitats. To investigate if the patterns of fruit-removal at experimental stations and at actual plants were similar, I undertook a pilot study in which branches of several *C. monilifera* and *E. reticulatus* plants were tagged and I monitored removal rates of fruits over a two-week period at PB.

5.2.4 Experimental design

I conducted initial trials at PB and BNP to test and refine the fruit-presentation technique and examine the suitability of particular fruits. At PB in November and December 1995, I tested *Monotoca elliptica* and *A. excelsa* fruits concurrently, one caché of 3-7 fruits of one species per station. In April 1996 at BNP, I tested *E. obovatus* and *Melia azedarach* concurrently, one caché of three fruits of one species per station. In these trials, I placed ten stations for each species at 25m intervals in each habitat type. I checked stations at dusk each day and noted any removal of fruits or damage to fruits.

I undertook more extensive trials of *Monotoca elliptica*, *E. reticulatus* and *C. monilifera* at PB, ML and ENP. At ML and ENP, I selected two sites in each habitat in which I set out transects of ten stations. At PB, the limited size of the reserve restricted our experiment to one transect of ten stations for each species in each habitat. I randomly located stations along a 100m transect, and placed them to prevent visibility from the adjoining station to avoid fruit consumers moving from one station directly to another. After each trial, I examined the pattern of removal of fruits from stations relative to station location, with no obvious trends apparent for any fruit type. I attached six ripe fruits (sometimes along with some green unripe fruits and leaves in the case of *M. elliptica*) to the clips on the stations by their pedicel.

As *C. monilifera* and *E. reticulatus* fruits ripen over the same period, I conducted the fruit-removal experiments for these species concurrently on the same transects, in May to July 1998 (with one species per station). Exceptions were the native vegetation transect(s) at each location, at which I did not present *C. monilifera* fruits (to avoid introducing propagules into vegetation currently free of *C. monilifera*). I conducted the *M. elliptica* experiment in November and December 1998.

I set out fruits at dawn of the first day, and checked them at dusk for 5-7 days. I recorded the number and location of damaged and removed fruits. Fruit condition was also monitored, and I replaced fruits when they began to show signs of desiccation or damage.

To investigate why removal rates might differ between locations and habitats, I kept records of the frugivorous bird species present at each site over the trial period. I did not attempt to monitor individual stations to identify the agent(s) responsible for removing fruit. However, I observed fruiting plants of each species to identify the species of birds responsible for fruit-removal in-situ (see Gosper 1999a).

5.2.5 Statistical analysis

I set the critical parameters for the analyses as: (i) the time to removal of the first fruit from a station; and (ii) the time to removal of all fruits from a station; both to be measured as the number of days. The first parameter corresponds to a measure of frugivore foraging activity, by quantifying the time taken to discover the fruit source. The second represents a measure of a combination of foraging activity and foraging pressure, by measuring the time taken to exhaust the fruit source. I pooled data at locations that had two transects per habitat.

I analysed the rates of removal of fruits using survival analysis programs on SAS and JMP, which are designed to utilise ‘censored’ data. As I monitored stations for a maximum of seven days, at some stations, the critical level of fruit-removal was not reached, or no fruits were removed at all. These stations are termed ‘censored’, but I included them in the analyses as they still provide useful information on foraging activity. Large amounts of censoring, however, make interpretation of the observed removal rates difficult. Consequently, due to the low numbers of fruits removed, I did not analyse removal rates of *A. excelsa* and *Melia azedarach* statistically.

I used the *Lifereg* procedure on SAS to fit parametric models to the censored data. Models for the response variable have a random disturbance term that can take the form of a range of distributions. In all cases, I used a Weibull survival distribution (which includes log transformation) in the first instance. I tested the validity of using this distribution by plotting the generated models against a standard Weibull model in JMP. In all cases, this proved to be acceptable. In addition to normal parametric analysis assumptions, the *Lifereg* analysis assumes that lifetimes and censoring times are independent, and that the survival function has specific distributional properties (Klein and Moeschberger 1997). Klein and Moeschberger (1997) describe the χ^2 statistic I used in these analyses.

I present fruit-removal rates graphically in two ways. Where I had few censored data, I use histograms to illustrate the average survival time (number of days until the removal of *x* fruit(s)) for each habitat within each location. Where data are censored, average survival time and standard errors are only estimates, and underestimate the true average survival time (as the survival time is set at the last day of sampling for censored data). Consequently, average survival time only provides an acceptable representation of the data when there is little censoring. Where censoring is high, I present data graphically only in the form of partial ‘survival’ functions. These plot the proportion of stations with no fruits removed (or without 100% of fruits removed) against the number of days lapsed. To reduce the complexity of the survival functions, each location has been plotted separately, and contrary to traditional practice, I present them as line graphs rather than step functions for ease of interpretation.

5.3 Results

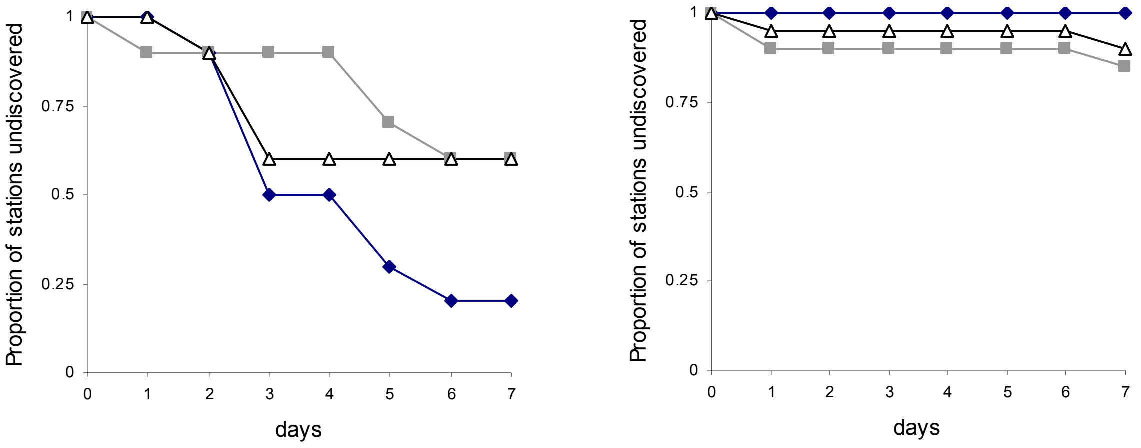
Fruits of all the plant species were removed from at least some experimental stations. However, for two fruit types, *A. excelsa* and *M. azedarach*, only minimal removal occurred at the single location I tested them. Consequently, the interpretation of the effect of *C. monilifera* management on the rate of fruit-removal of these species was not possible within the time frame of the experiment. In the five-day trial of *A. excelsa* fruits at PB, only four fruits out of 166 (2.4%) were removed from two stations in native vegetation and two out of 178 (1.1%) were removed from two stations in herbicide-treated habitat. Many *A. excelsa* fruits were damaged, however, such as by removal of part of the pericarp and the powdery flesh, with up to half the volume of a fruit sometimes removed. In native vegetation, 14.8% of available fruits (excluding fruits removed) were damaged and 13.1% in herbicide-treated vegetation. By comparison, 69% of *Monotoca elliptica* fruits were removed from stations in herbicide-treated habitat and 44% in native vegetation over two days within the same survey period at the same sites.

In the five-day trial of *Melia azedarach* fruits at BNP, no fruits were removed from any of the stations in *C. monilifera*-dominated or herbicide-treated vegetation, and only one fruit out of 30 available (3.3%) in native vegetation. Damage of fruits was more widespread, consistent with the removal of pieces of the fruit by vertebrates unable, or unwilling, to swallow fruits whole. Two fruits were damaged in *C. monilifera*-dominated vegetation (6.7% of the fruit available), one in native vegetation (3.4%), and three in herbicide-treated vegetation (10%). Removal of *E. obovatus* fruits over the same period and sites was substantially greater, with 50%, 20% and 13.3% of the available fruits removed over the first five days of the trial in *C. monilifera*-dominated, herbicide-treated and native vegetation habitats respectively.

After seven days of the BNP trial, one or more *E. obovatus* fruits had been removed from approximately 50% of stations across all habitats, although there were no instances of all fruits being removed from a station by this time. Consequently, I analysed statistically only the number of days until station discovery (and removal of the first fruit). Habitat type did not have a significant effect on the time to station discovery ($\chi^2 = 4.395$, df = 2, P = 0.1111), although looking at the results graphically, more stations seemed to be discovered in *C. monilifera*-

(a) Bundjalung, *E. obovatus*

(c) Perkins Beach, *E. reticulatus*



(b) Myall Lakes, *E. reticulatus*

(d) Eurobodalla, *E. reticulatus*

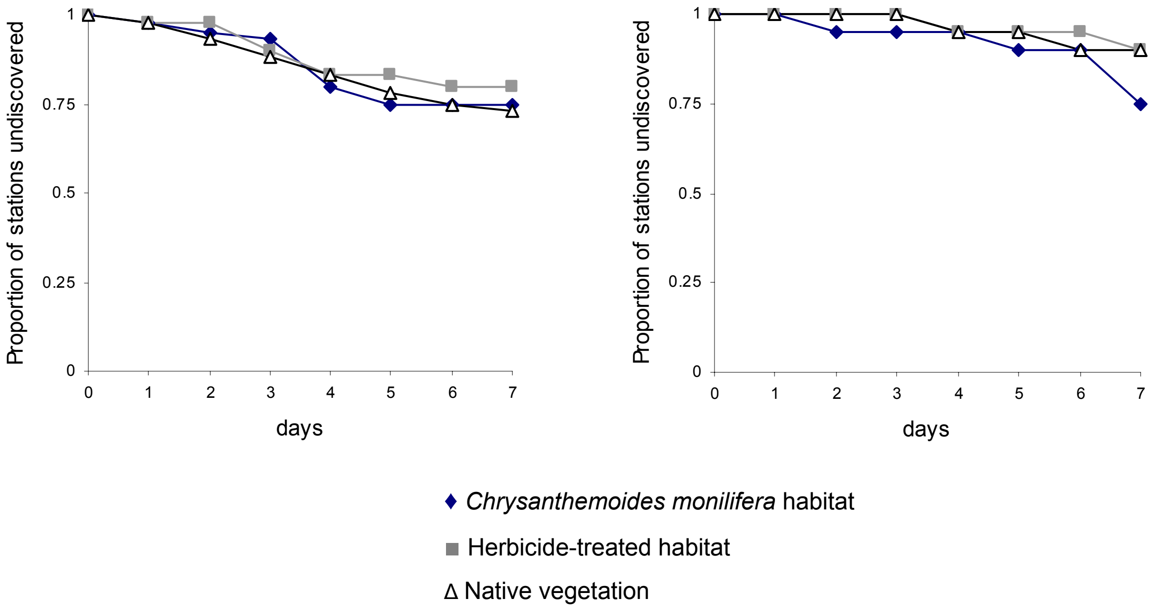


Figure 5.1 - Survival curves for the discovery of *Elaeocarpus* fruits at experimental stations

dominated vegetation (Figure 5.1a). Several fruits were also damaged over the course of the study, with part of the pericarp and the thin layer of flesh removed.

Fruits of *Monotoca elliptica, E. reticulatus* and *C. monilifera* were removed much more frequently than those of either *A. excelsa* or *Melia azedarach*. In all habitats at all locations, experimental stations with *Monotoca elliptica* fruits were discovered (and the first fruit removed) after about two days (Figure 5.2). Daily rates of removal of *M. elliptica* fruits were consistently high compared to the other species. Discovery time and subsequently time to exhaustion of *M. elliptica* fruits at stations did not differ significantly with habitat ($\chi^2 = 2.233$, df = 2, P = 0.3274 for station discovery; and $\chi^2 = 7.646$, df = 2, P = 0.6174 for total fruit-removal) (Figures 5.2 and 5.3). Location had no effect on the rate of discovery of stations (Figures 5.2 and 5.3) ($\chi^2 = 0.843$, df = 2, P = 0.6560). However, exhaustion of fruits at stations was more rapid at PB that at the two other locations (Figure 5.3). This location effect was significant overall ($\chi^2 = 7.646$, df = 2, P = 0.022), with exhaustion at PB significantly more rapid than at ML ($\chi^2 = 6.801$, df = 1, P = 0.009) and ENP ($\chi^2 = 5.3248$, df = 1, P = 0.021). The rate of exhaustion of stations at ML and ENP were not significantly different ($\chi^2 = 0.0266$, df = 1, P = 0.871).

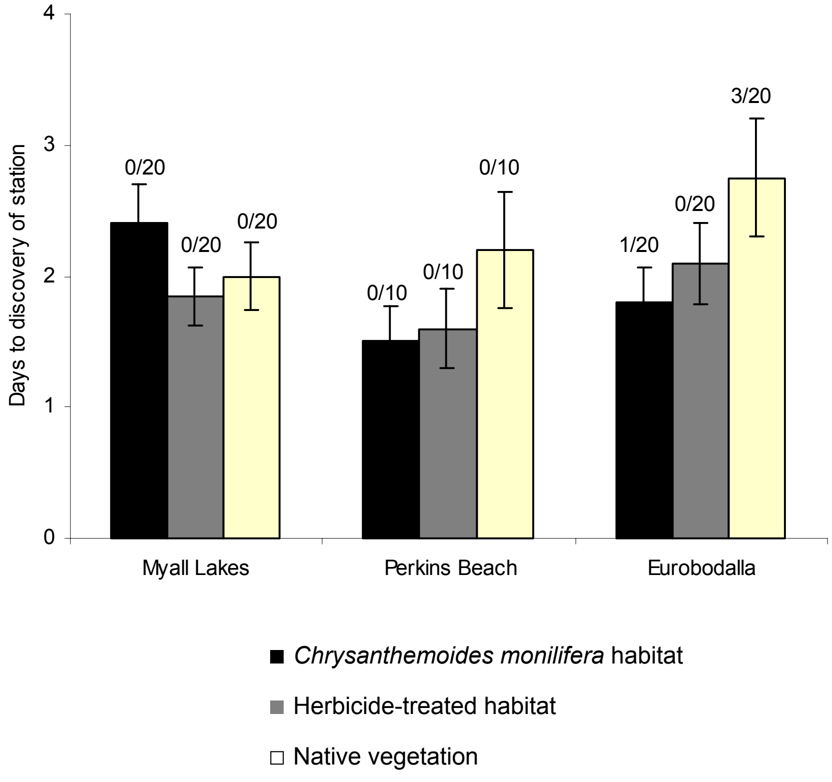
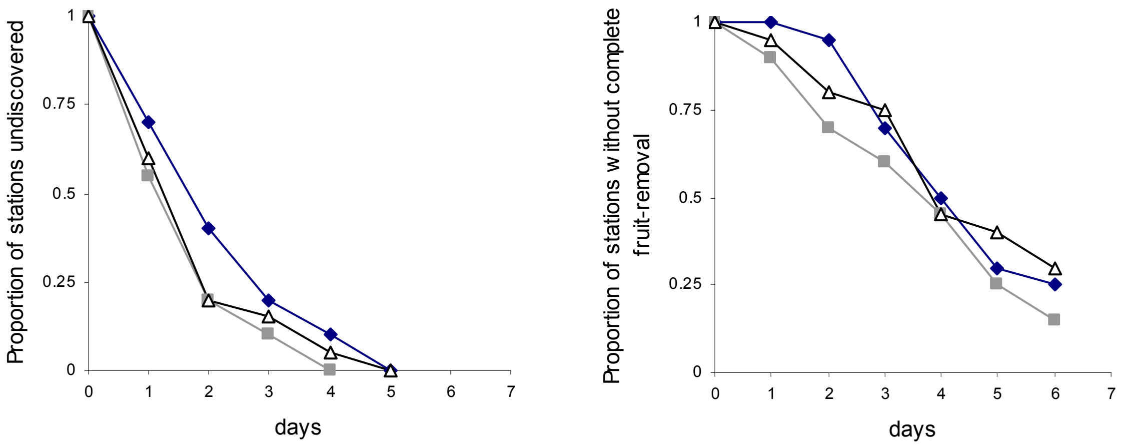


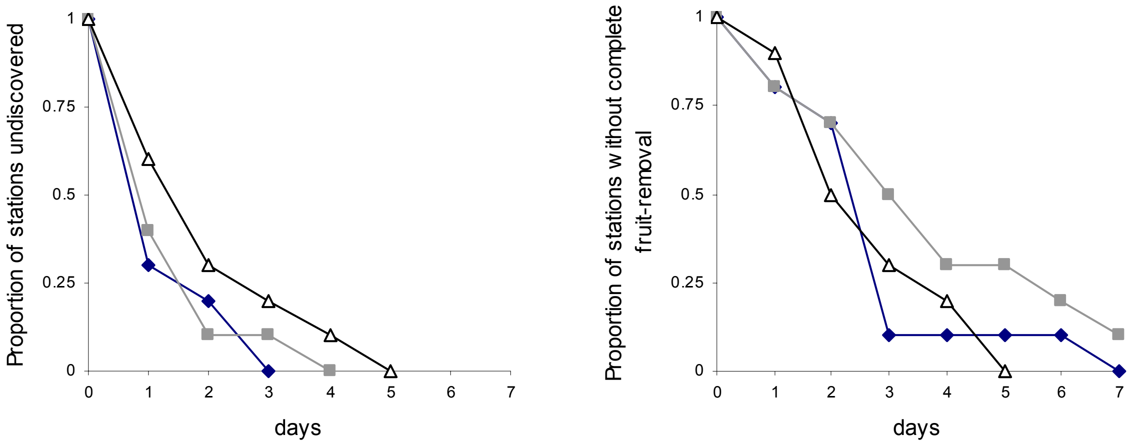
Figure 5.2 - Time to discovery of *Monotoca elliptica* fruits at experimental stations

Note that the mean number of days to the discovery of the station (and removal of the first fruit) is the mean survival time, which can include censored data. Censored data points are set at five days (the day monitoring was completed at ENP – the only location with censored data) and means are thus approximate. The number of censored stations is shown above each column, over n. Error bars are SEM, with the censored data treated as above.

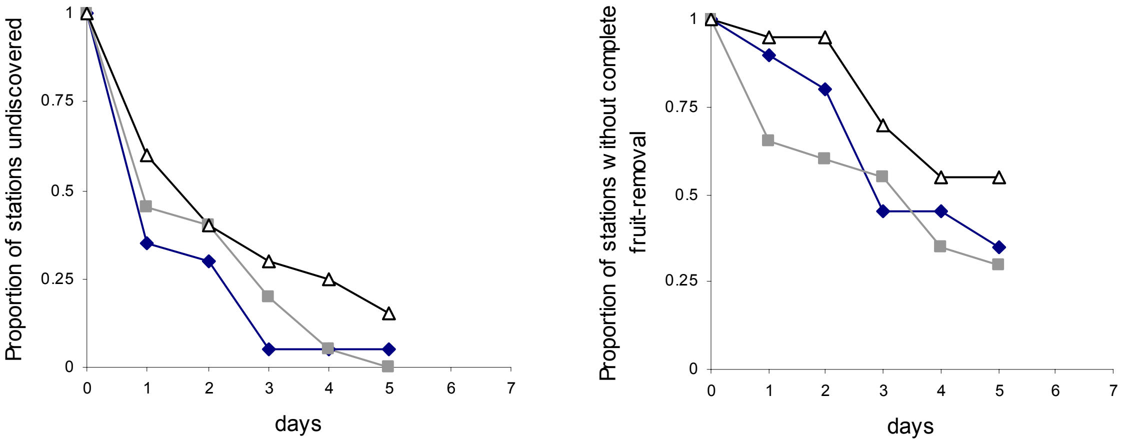
(a) Myall Lakes



(b) Perkins Beach



(c) Eurobodalla



♦ *Chrysanthemoides monilifera* habitat
■ Herbicide-treated habitat
Δ Native vegetation

Figure 5.3 - Survival curves for the discovery and exhaustion of *Monotoca elliptica* fruits at experimental stations

The amount and rate of *E. reticulatus* fruit-removal was substantially lower than for *M. elliptica,* the congeneric *E. obovatus* and *C. monilifera*. At no stations were all fruits removed by the end of the monitoring period and at only 17.9% had at least one fruit been taken. Consequently, I only analysed statistically the time until station discovery. The rate of discovery of stations with *E. reticulatus* fruits did not differ significantly between habitats (Figure 5.1b to d) (χ^2 = 0.237, df = 2, P = 0.8881), however, there was a location effect (χ^2 = 6.2925, df = 2, P = 0.0430). In pairwise location comparisons, discovery of stations was more rapid at ML than at PB (χ^2 = 5.302, df = 1, P = 0.0213), but not than at ENP (χ^2 = 2.290, df = 1, P = 0.1303); nor was discovery at ENP significantly more rapid than that at PB (χ^2 = 1.081, df = 1, P = 0.2983). In addition to removal of fruits from stations, parts of the pericarp and flesh had been taken from fruits left at stations.

Removal of *C. monilifera* fruits from experimental stations was comparatively rapid. At each location, stations were discovered sooner in the *C. monilifera*-dominated vegetation (Figure 5.4). For both habitats tested (*C. monilifera*-dominated and herbicide-treated), I found time to discovery and removal of the first fruit was about three times longer at ENP than ML, with PB

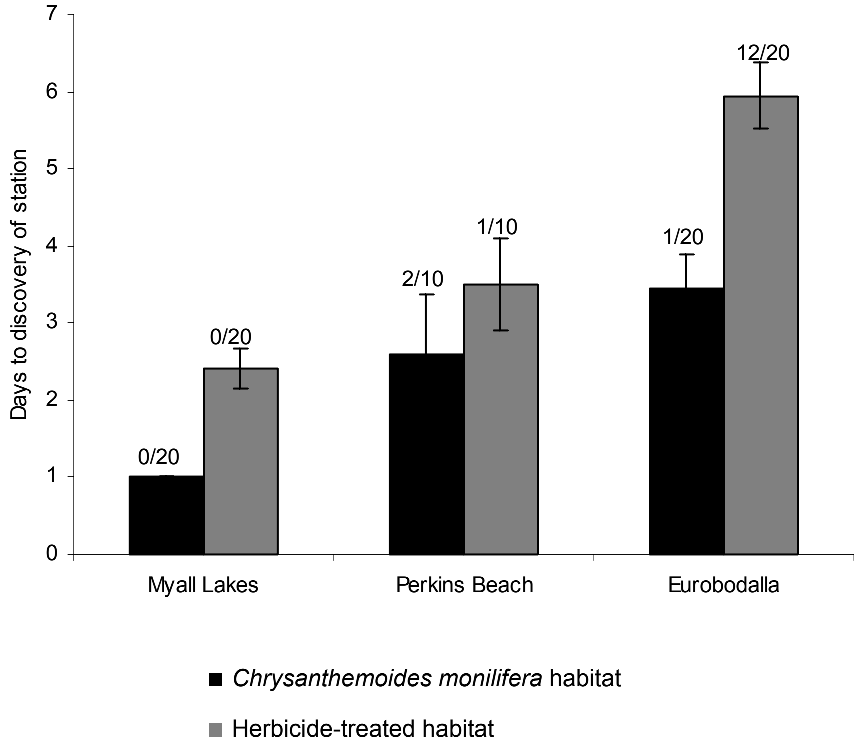


Figure 5.4 - Time to discovery of *C. monilifera* fruits at experimental stations

Note that the mean number of days to the discovery of the station (and removal of the first fruit) is the mean survival time, which can include censored data. Censored data points are set at seven days (the day monitoring was completed) and means are thus approximate. The number of censored stations is shown above each column, over n. Error bars are SEM, with the censored data treated as above.

intermediate. This represents a pattern of increasing discovery time from north (ML) to south (ENP). These effects of habitat and location were statistically significant overall (Figures 5.4 and 5.5) ($\chi^2 = 41.455$, df = 1, $P < 0.001$ for habitat) ($\chi^2 = 106.41$, df = 2, $P < 0.001$ for location). ML experienced more rapid discovery of stations than at either PB ($\chi^2 = 37.67$, df = 1, $P < 0.001$) or ENP ($\chi^2 = 99.10$, df = 1, $P < 0.001$) and more rapid discovery of stations occurred at PB than at ENP ($\chi^2 = 4.65$, df =1, $P = 0.031$).

Time to the exhaustion of *C. monilifera* fruits at stations was also significantly different between the two habitats and three locations tested (Figure 5.5). Stations in *C. monilifera*-dominated vegetation were exhausted more rapidly than in herbicide-treated habitat ($\chi^2 = 18.46$, df = 1, $P < 0.001$). Comparing locations, time to exhaustion of fruits at stations again increased from north (ML) to south (ENP) ($\chi^2 = 108.69$, df = 2, $P < 0.001$). Exhaustion of fruits was more rapid at ML than PB ($\chi^2 = 55.51$, df = 1, $P < 0.001$) and ENP ($\chi^2 = 90.06$, df = 1, $P < 0.001$); and PB experienced more rapid exhaustion than ENP ($\chi^2 = 5.87$, df = 1, $P = 0.015$).

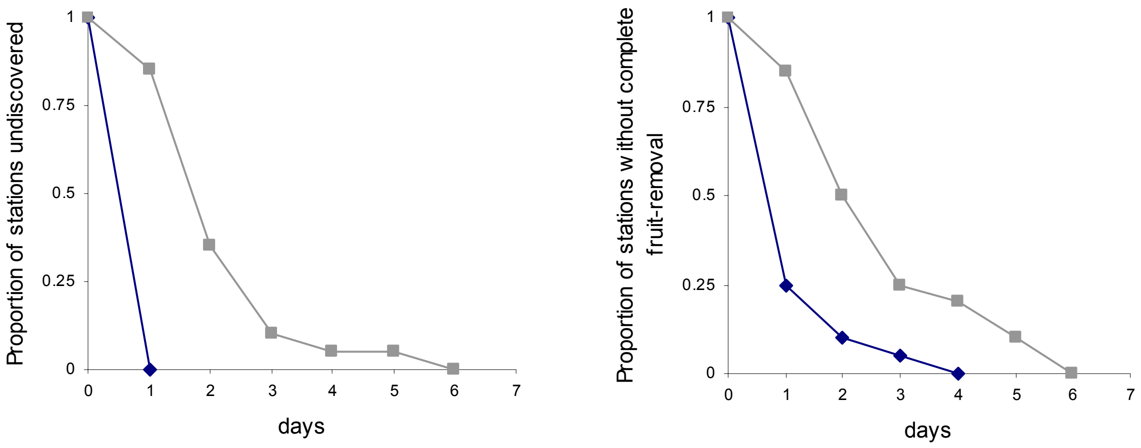
The rate of removal of *C. monilifera* fruits from actual plants was also greater than that of *E. reticulatus*. Within a week of tagging branches, 89% of *C. monilifera* fruits had been removed, while the remainder became desiccated. In contrast, only 13% of ripe fruits were removed from branches of *E. reticulatus* over two weeks.

5.4 Discussion

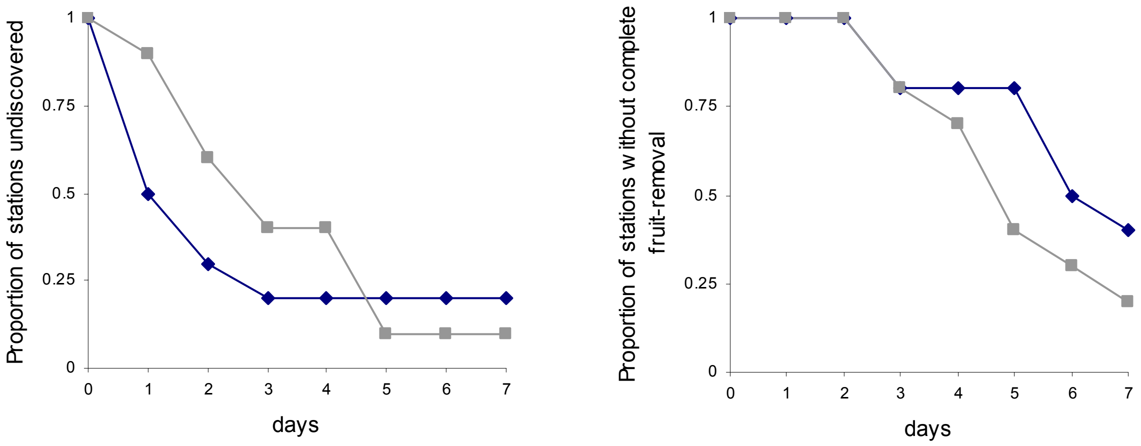
5.4.1 Effects of habitat

I did not detect any significant differences between habitats in the time to discovery of experimental stations (and removal of the first fruit), or exhaustion of all fruits at stations, for any of the native fruit types tested. This suggests that the invasion of *C. monilifera* is not having a substantial impact on frugivory in these native plant species. Even if *C. monilifera* is effecting change in the suite of bird species that consume fleshy fruits and other plant resources (as suggested by French and Zubovic 1997), this impact has not been transferred to the removal of fruits of the native plant species investigated here. However, the implications of this minimal effect on fruit-removal on the actual dispersal of native seeds, and establishment in a *C. monilifera*-dominated environment, are unknown. Several studies, however, have demonstrated the ability of some invasive plants to inhibit the germination and establishment of native plants (Kelly and Skipworth 1984; Lee *et al.* 1986; Braithwaite *et al.* 1989).

(a) Myall Lakes



(b) Perkins Beach



(c) Eurobodalla

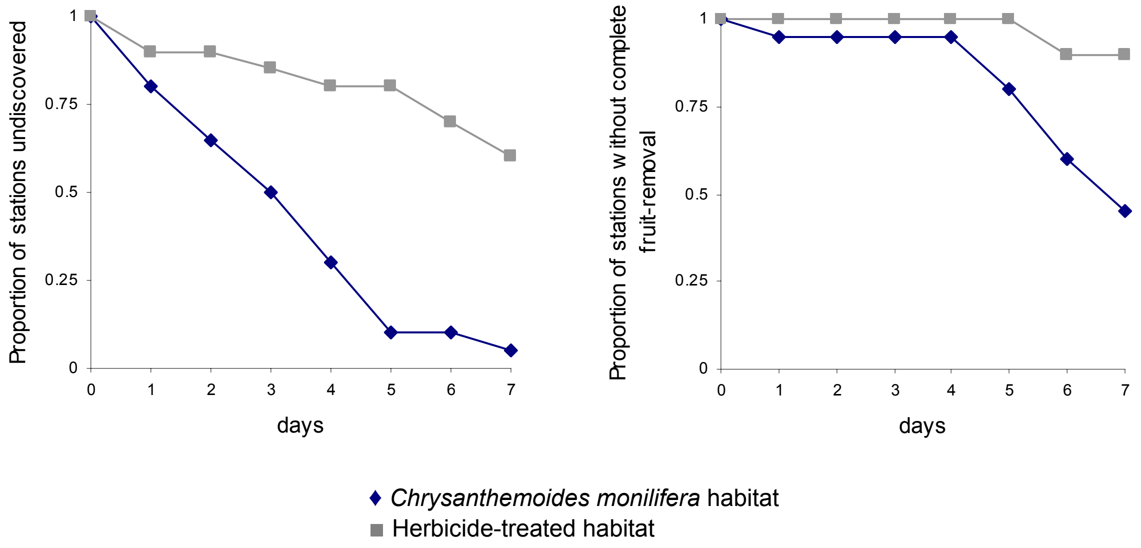


Figure 5.5 - Survival curves for the discovery and exhaustion of *C. monilifera* fruits at experimental stations

The large amount of censored data (stations where a specified level of removal was not reached) in the case of *E. reticulatus* and *E. obovatus* could potentially mask trends in removal rates and limit my ability to detect differences. However, in the case of *E. reticulatus,* all treatments were remarkably similar, and for *E. obovatus*, there was only a slight trend towards more rapid discovery in *C. monilifera*-dominated habitat (Figure 5.1a), although this was not statistically significant.

These findings indicate that the native species and *C. monilifera* may not be competing for a limited fruit-remover resource. The *E. reticulatus* and *E. obovatus* trials were conducted during the period of maximum *C. monilifera* fruit production (Chapter 2), when the impact of *C. monilifera* fruits on plant-bird interactions would be expected to be greatest. For these two plant species, the absence of any detected impact of *C. monilifera* invasion is likely to be at least partly a result of differences in fruit morphology and subsequent differences in the species of birds that feed on them. Fruits of *E. reticulatus* and the similar *E. obovatus* have comparatively large endocarps (Chapter 2), which would restrict the numbers of birds able to ingest them relative to *C. monilifera* (Herrera 1984). Thus, the relative importance of particular bird species involved in fruit-removal would differ between these plant species, with the *Elaeocarpus* spp. utilising a more limited range of dispersers.

Temporal separation of periods of peak fruit production means that *M. elliptica*, and many other native plant species, avoid maximum competition for seed dispersal vectors with *C. monilifera*. Few vertebrate-dispersed native plant species produce fruits over the period of maximum *C. monilifera* fruit production, and those that do mostly differ greatly in fruit morphology (such as having larger fruits *e.g. Elaeocarpus* spp.) (Chapter 2). These temporal differences in periods of maximum fruit production presumably contributed to the similar rates of removal of *M. elliptica* fruits recorded from all habitats. In this situation, frugivores would be unlikely to be being satiated by the smaller *C. monilifera* fruit crop, even if *C. monilifera* fruits were preferred (which is not known).

Of potential importance for the long-term control of *C. monilifera* is my finding that the rate of removal of *C. monilifera* fruits from experimental stations in herbicide-treated habitat was significantly lower than removal rates in habitats dominated by *C. monilifera*. Lower fruit-removal rates are likely to be reflected in poorer seed dispersal. Ripe *C. monilifera* fruits become desiccated within one to two weeks, so any decrease in the rate that fruits are removed is likely to result in less fruits being dispersed in total. Hence, this would retard the ability of *C. monilifera* to re-colonise treated areas. Aerial *C. monilifera* control, as carried out in the herbicide-treated habitat, typically kills more than 95% of existing plants (Toth *et al.* 1996). I have demonstrated that surviving plants will have lower rates of fruit-removal than would occur in *C. monilifera*-dominated habitat, providing an additional advantage from the control program. Quantification of seed dispersal distances from remaining *C. monilifera* plants in herbicide-treated areas would provide additional information on the ability of *C. monilifera* to re-colonise by dispersed seed. A large seed bank can occur beneath *C. monilifera* stands (9500 seeds/m^3 have been recorded) and some seed remain viable for at least 10 years (Parsons and Cuthbertson 1992). Consequently, it would take several years of reduced seed input to substantially reduce the ability of *C. monilifera* to regenerate at sites at which it had been well established and control work had been undertaken.

5.4.2 Effects of location

For each of the plant species I tested at multiple locations (*C. monilifera, M. elliptica* and *E. reticulatus*), the rate of removal of fruits from experimental stations differed significantly between at least some locations. The predominant trend was a decrease in the rate of removal of fruits from north to south. The location effects detected could be related to the presence and abundance of particular frugivores. These frugivores, in turn, may be responding to a gradient in vegetation composition. French and Westoby (1992), for example, recorded greater fruit-removal from experimental stations in sites that had greater abundance and diversity of fruit-eating birds.

Factors likely to influence the occurrence of frugivorous birds include the abundance and diversity of fleshy-fruited plants and the temporal pattern of fruit availability. I did not measure overall fruit abundance at the different locations. The number of plant species with vertebrate-dispersed fruits occurring in coastal vegetation, however, declines from north (ML) to south (ENP) (Chapter 3). Additionally, the temporal pattern of fruit availability becomes more restricted from north to south (Chapter 2).

Both removal measures indicated that fruits of *C. monilifera* were more rapidly removed in the north (ML) than the south (ENP). This north-south decrease in fruit consumption may have contributed to the observed pattern of invasion of *C. monilifera* in NSW. Lane (1984) recorded greater expansion of *C. monilifera* from sites of initial introduction north of the Hunter River (which is to the south of ML) than from those in the south. The bird species that are both important dispersal agents for *C. monilifera* fruits (Chapter 4) and have foraging strategies I consider render them more likely to discover experimental stations are Silvereyes (*Zosterops lateralis*), Lewin's Honeyeaters (*Meliphaga lewinii*) and Red-whiskered Bulbuls (*Pycnonotus jocosus*). Silvereyes were abundant at all sites, Lewin’s Honeyeaters abundant only at ML (they were absent from PB, and very rarely recorded at ENP) and Red-whiskered Bulbuls occurred only at PB (Chapter 7). The relative abundance of these three frugivores broadly reflects the observed change in the rate of fruit-removal from north to south.

Discovery and removal of the first *E. reticulatus* fruit from experimental stations was significantly more rapid at ML than at PB, fitting the north-south pattern in frugivore foraging pressure. This is likely to be a consequence of the abundance of Lewin's Honeyeaters at ML, and the absence of this species from PB. Lewin's Honeyeaters were one of the few consumers of *E. reticulatus* fruits (Holmes 1990) common in coastal vegetation at some locations in this study. However, time to removal of all *M. elliptica* fruits from stations was significantly less at PB than at either ML or ENP, an exception to the north-south trend. However, I found no difference between locations for time to removal of the first *M. elliptica* fruit.

5.4.3 Differences in removal rates between species

The rate of removal of fruits from experimental stations differed greatly between the species I tested. *Monotoca elliptica* and *C. monilifera* fruits were removed comparatively rapidly, while *E. reticulatus, E. obovatus, A. excelsa* and *Melia azedarach* fruits were removed more slowly. These differences in removal rates between plant species may be a consequence of fruit characteristics and their influence on frugivore choice and/or be an artefact of my experimental technique.

The relationships between fruit and crop characteristics and frugivore food choice appear highly complex. Choice may be influenced by factors such as: fruit colour, size, nutritional content, fruiting display size, crop size, seed to pulp ratio, mass of pulp, secondary chemicals, accessibility, palatability, seed passage rates, availability and digestibility (Sorensen 1981; Sorensen 1984; Murray *et al.* 1993; Stiles 1993). Such a diversity of factors is not easily examined in a study of the removal of a few natural fruits from experimental stations. Studies investigating choice of fruits by birds have often produced ambiguous results, including in comparative aviary and field trials using the same species of plants and birds (Whelan and Willson 1994). Consequently, Whelan and Willson (1994) suggested that fruit choice by frugivores depends heavily on environmental context.

In comparing the experimental fruit types (see Chapter 2), *C. monilifera* fruits provide a substantially higher content of sugars per unit of dry mass than *E. reticulatus*. *Elaeocarpus reticulatus* also rated poorly for frugivores in having a less attractive endocarp to pulp ratio. *Elaeocarpus reticulatus* fruits contain three times the phosphorus and more protein and fat than *C. monilifera*, however. I tested the rates of fruit-removal of these two species concurrently, and the sugar and endocarp to pulp ratio advantages of *C. monilifera* may have contributed to its greater removal. *Monotoca elliptica* pulp contained more protein per dry weight than *C. monilifera*, and nearly as much as *E. reticulatus*, and the greatest fat content of the three fruit pulps. *Monotoca elliptica* was the most profitable fruit in terms of endocarp to pulp ratio (Chapter 2). Fruits of this species were consistently removed rapidly from experimental stations in all locations and treatments.

Fruit size, and particularly endocarp size, determines the range of bird species able to ingest the fruit. Herrera (1984) recorded higher fruit-removal from plant species with fruit sizes smaller than the gape width of the abundant dispersers, compared to those with larger fruits (but see Johnson *et al.* 1985). The two smallest fruits (with the smallest endocarps), *C. monilifera* and *M. elliptica*, were more rapidly removed from our experimental stations than the larger fruits. These small fruits can be ingested by most, if not all, frugivorous bird species of coastal habitats, and this undoubtedly contributed to the high rates of fruit-removal observed. Interestingly, French and Westoby (1996) also proposed that plant species with small fleshy fruits would be favoured by the dispersal and burial of propagules by ants in fire-prone environments in eastern Australia. I observed ants feeding on the flesh of *M. elliptica* fruits *in-situ*, and ant dispersal is consequently another possible evolutionary influence on the fruit characteristics of this species.

In contrast, the size of the hard endocarps and fruits of *E. reticulatus*, *E. obovatus*, *A. excelsa* and *Melia azedarach* probably exceeds the gape width of many of the small fruit-eating bird species, preventing them from swallowing fruits whole. Silvereyes, the most abundant dispersers at all my study locations, have a gape width of 5 mm (Williams and Karl 1996), which is less than the mean endocarp width of *E. reticulatus, M. azedarach* and *A. excelsa* (Chapter 2). Silvereyes have, however, been previously recorded feeding on several of these larger-fruited plant species (*E. obovatus* - Holmes 1987; Barker and Vestjens 1990; *A. excelsa* - Barker and Vestjens 1990; *M. azedarach* - Green 1993). This is likely to have been by removing part of the pericarp and flesh and not swallowing the seed (as described by Green 1993; Gosper 1994; Gosper 1999a) or possibly, by selecting fruits of lower than average size (Williams and Karl 1996). For each of the plant species with large endocarps, fruits were recorded at experimental stations with part of the pericarp and flesh removed, but without the fruit being removed from the station and contributing to observed fruit-removal. For both *M. azedarach* and *A. excelsa*, more fruits were damaged in this way than actually removed from stations. Pulp-consuming birds that pick fruits, consume the flesh, then drop the seeds, however, would contribute to observed fruit-removal in this study.

Slower rates of removal of the larger fruits are also supported by the limited data on removal rates from actual plants, where *E. reticulatus* removal was substantially slower than that of *C. monilifera*. Differences in removal rates from experimental stations therefore show the same trends as occur at actual plants.

Whilst the rate of fruit-removal of *E. reticulatus, E. obovatus, M. azedarach* and *A. excelsa* is substantially slower than that of *C. monilifera* and *M. elliptica*, this may not necessarily correspond to a lower proportion of the total fruit crop being removed over the entire fruiting season. *Elaeocarpus reticulatus*, and probably the other large-fruited species, bear less watery fruits (Chapter 2), which remain in a state suitable for removal on the canopy of the parent plant for a longer period. *Chrysanthemoides monilifera* fruits, once ripe, last less than two weeks before becoming desiccated. Fruits of *E. reticulatus* last at least four weeks ripe on the plant before falling. *Alphitonia excelsa* also presents fruits for frugivores for an extended period, with the endocarp of the fruit eventually splitting upon drying to reveal arillate seeds that then remain attached to the pedicel until dispersal (Harden 1990). This probably represents a duel strategy to seed dispersal. Bird species with smaller gapes may take the arillate seeds after larger frugivores have had the opportunity to take whole drupes.

5.4.4 Assessment of the experimental technique

In assessing the effects of experimental factors on patterns of fruit-removal, it is essential to consider how well my methodology represents the actual functioning of the plant-animal dispersal system. The range of daily removal rates of fruits, with the exception of *C. monilifera* at one locality, lie within the range of previously observed daily rates of removal of fruits in a wide range of systems (Eastern Australia - French and Westoby 1992; Central America - Jordano 1983; North America - Morden-Moore and Willson 1982). At ML, I recorded up to 100% of the available *C. monilifera* fruits in *C. monilifera*-dominated vegetation being removed per day, which exceeds previously recorded daily removal rates. Although calculated from only a few days of observation (as all fruit were rapidly removed from stations), consumption of *C. monilifera* fruits at this locality and habitat type appears extremely high.

I employed experimental stations as a technique for measuring fruit-removal to avoid the biases inherent in using existing plants, such as prior knowledge of fruit location by birds, and defence of such resources. The experimental stations, therefore, represented the appearance of small quantities of fruits at irregular intervals throughout the habitat. This methodology mimics the natural distribution of fruits of some plant species better than others. Of our study species, *C. monilifera* and *M. elliptica* were more widespread, abundant and evenly distributed than the other fruit sources on the coastal dunes where the study was undertaken.

Small, dispersed fruit displays such as in this experiment favour fruits being taken by generalist and opportunistic dispersers, which use a variety of foods as they move throughout the habitat, rather than being specifically attracted to a fruit display. Lewin's Honeyeaters, Silvereyes, fairy-wrens (*Malurus* spp.), thornbills (*Acanthiza* spp.) and the larger honeyeaters (*Anthochaera* and *Philemon* spp.) feed on fruits, nectar and insects (Barker and Vestjens 1990; Gosper 1999a). They would be expected to chance upon my experimental fruit displays regularly in general foraging activity. Generalist and opportunistically foraging birds form a substantial component of the bird community of coastal vegetation and they readily use fruits with small endocarps, such as *C. monilifera* and *M. elliptica* (Chapter 3), in addition to some primarily frugivorous species. The experimental technique used may magnify, therefore, any effect of fruit characteristics on frugivore choice. However, as I recorded broadly similar patterns of fruit-removal from actual plants and experimental stations, the differences in removal rates are likely to be an accurate reflection of fruit-removal from both new fruit sources and existing plants.

5.4.5 Conclusion

I did not detect any impact of the presence, absence or removal of *C. monilifera* on the rate of removal of fruits of any of the native plant species tested. These native plants, therefore, do not appear to be at substantial risk of dispersal failure as a result of *C. monilifera* invasion. The elimination of *C. monilifera*, however, did reduce the rate of discovery and exhaustion of *C. monilifera* fruits compared to untreated vegetation. Thus, *C. monilifera* fruits on surviving plants in herbicide-treated areas would not be removed as rapidly as fruits on plants in already invaded vegetation, perhaps providing some assistance to *C. monilifera* control efforts.

Examination of the fruit-removal of additional native species would be beneficial, from either experimental stations or existing plants. In particular, those species that fruit over the period of maximum *C. monilifera* fruit production, and presumably maximum impact on plant-bird interactions, should be tested. Examining native plant species that have a similar seed dispersal syndrome, including being able to be dispersed by small, generalist-foraging bird species, would also target those plants most likely to be impacted by *C. monilifera* invasion. Native species that fit these criteria are limited, but include *Polyscias elegans* and *Einadia hastata*. Few native plant species do fruit over this period in coastal NSW (Chapter 2), and of those, most differ from *C. monilifera* in having large endocarps and consequently are likely to have a different disperser assemblage. For most plant species, competition with *C. monilifera* for dispersal agents is therefore minimised through temporal separation of maximum fruit production and/or having a different seed disperser assemblage. For those plant species with small fruits, the abundance of generalist frugivores may also result in dispersal agents not being limiting.

CHAPTER 6. AN EXPERIMENTAL TEST OF THE IMPACT ON BIRDS OF CONTROL OF THE INVASIVE PLANT *CHRYSANTHEMOIDES MONILIFERA*

This photo depicts coastal vegetation after aerial herbicide application to control *Chrysanthemoides monilifera*, showing surviving native plants (*Banksia integrifolia* in the foreground) and dead *C. monilifera*.

Preface

In this chapter, I investigate what impacts, if any, removal of an invasive plant has on bird communities. While the adverse impacts of plant invasion on fauna are widely known, or at least supposed, the consequences of removal have rarely been experimentally investigated. Could some components of the bird community now rely on resources provided by the invasive plant?

6.1 Introduction

Invasive plant species have a variety of impacts in ecosystems in which they become established. These include changes to fauna such as birds (*e.g.* Braithwaite *et al.* 1989; Griffin *et al.* 1989) and invertebrates (*e.g.* Slobodchikoff and Doyen 1977; French and Major 2001). However, many native animal species are known to use vegetation dominated by invasive species for habitat and/or food. In some cases, the resources provided by invasive plants have been used by some native fauna species at least as much as the native alternatives (Knight 1986; Knopf and Olson 1984;Westman 1990; Waring *et al.* 1993; French and Major 2001).

Managing invasive plants is a goal of conservation management (*e.g.* Westman 1990; NSW NPWS 1999), because negative impacts of invasive plants on a variety of ecosystems and species are widely known (Adair and Groves 1998). Implicit in the arguments for control of invasive plants are the assumptions that the invasive plant has deleterious impacts on native species and/or ecosystems and that these impacts can, at least partly, be ameliorated by control of the invasive species. It is unusual for either of these assumptions to be investigated prior to control programs being implemented (Pakeman and Marrs 1992). In fact, the consequences of control of invasive plants on other biota have rarely been investigated experimentally in any system. In these rare cases, control of an invasive plant has benefited ecosystem function (D'Antonio *et al.* 1998) and native flora (Hester and Hobbs 1992; Ailstock *et al.* 2001). Control has had varying effects (benefit, disadvantage or no effect) on fauna (Olaleye and Akinyemiju 1996; Linz *et al.* 1996a; 1996b; Linz *et al.* 1999; Homan *et al.* 2003). These divergent responses suggest that research specific to each case of invasive plant removal may be necessary. It is more typical for the emphasis in the management of invasive plants to be on the control of the infestations, rather than on the long-term effects of control on native biodiversity (Williams and West 2000).

It is conceivable that the removal of invasive plants could have unintended adverse impacts. These may include direct impacts of the method of invasive plant removal, such as the use of herbicides (*e.g.* Matarczyk *et al.* 2002). They may also include the loss of resources for fauna, loss of vegetation cover facilitating the invasion of more problematic invasive species, or disruption of ecosystem function *e.g.* increased erosion or changed rates of nutrient cycling (Westman 1990). These potential problems are likely to be greatest over the period of transition to a replacement vegetation community (Westman 1990), and should be most apparent shortly after the removal of the invasive species. In an invasive species-specific removal program, the resources provided by co-occurring plant species will be available to fauna before and after removal. Although remaining species will provide resources for fauna, some resources provided by the invasive species, such as food and shelter, will have been lost. Investigating the composition and status of the faunal community before and after removal could highlight the immediate short-term changes to fauna of the loss of resources provided by the invasive species.

Experimental investigation of the impacts of the removal of invasive species on fauna has important management applications, such as determining priorities for control of invasive species, conserving native fauna in disturbed environments and in the design of habitat restoration programs. An adverse impact on fauna of removal of invasive plants has occasionally been proposed (Date *et al.* 1991; Gosper 1994; Ekert and Bucher 1999), but rarely experimentally tested. Nevertheless, predictions of adverse impacts have been used to recommend against, or place limits on, invasive plant control in particular circumstances (*e.g.* Date *et al.* 1991; Wood 1993; Gosper 1994; Ekert and Bucher 1999). Williams and West (2000) also cautioned generally that management strategies for invasive plants should consider the supply of alternative shelter and food resources for native species that use resources of the invasive plants.

Bitou Bush (*Chrysanthemoides monilifera* ssp. *rotundata*) is an invasive plant that has become established along an estimated 80% of the NSW coastline (NSW NPWS 2001). Invasion of native communities by *C. monilifera* affects the bird (French and Zubovic 1997) and invertebrate (French and Eardley 1997) faunas of coastal ecological communities. In both these studies, total abundance and species richness of fauna were similar between invaded and non-invaded vegetation, although the composition of the fauna community was different. *Chrysanthemoides monilifera* produces fleshy fruit that are attractive to a range of fauna that act as important dispersal vectors (Dodkin and Gilmore 1984; Gosper 1999a), both of *C. monilifera* and of native plant species. Each of these studies illustrates that habitat with a significant component of *C. monilifera* supports a range of fauna.

Removal of *C. monilifera* has the potential to affect fauna species through the removal of important resources, such as food and shelter. In this study, the removal of *C. monilifera* was achieved with the application of herbicide under a specific methodology designed to minimise off-target impacts (Toth *et al.* 1996). Consequently, the impact of the loss of some *C. monilifera* resources should be distinguishable as native plant species are largely unaffected by the control methodology (Toth *et al.* 1996).

I selected birds as the faunal group of interest in this study for the following reasons: (i) many bird species use *C. monilifera* fruits as food (Chapter 3), hence are known users of *C. monilifera* resources; (ii) invasion of native vegetation by *C. monilifera* has been shown to affect the bird community (French and Zubovic 1997), hence birds of coastal habitats are sensitive to habitat change; and (iii) birds are readily surveyed in coastal vegetation.

The following null hypotheses were tested:

(1) herbicide application has no immediate impact on the abundance or species richness of birds or composition of the bird community; and

(2) the death of *C. monilifera* has no impact on the abundance or species richness of birds or composition of the bird community.

The first of these hypotheses was advanced as an equivalent of a 'procedural control'. The process of application of herbicide (involving flying low over the area in a helicopter and applying herbicide) may be, in itself, a significant impact on birds. If the death of *C. monilifera* had direct effects on birds, it would have been impossible to distinguish this as a cause from the physical effects of applying the control measure. The second hypothesis was posed because it is generalisable to a variety of methods of *C. monilifera* control. In this study, the two treatments could be distinguished because the aerial herbicide application takes about one month to kill *C. monilifera*.

If an impact of *C. monilifera* removal on bird abundance, species richness or community composition occurs, it is unlikely to be uniform across all species (or groups of species) and areas. Birds that use food resources provided by *C. monilifera* are expected to be most disadvantaged, because they may rely directly on resources to be removed by herbicide treatment. A substantial number of bird species consume *C. monilifera* fruits at the study locations (Chapter 3). These species are predicted to be the most susceptible to *C. monilifera* removal. In contrast, removal of *C. monilifera* is not expected to substantially reduce the food resources of species that do not use *C. monilifera* fruits for food: these include most of the nectarivorous and insectivorous bird species. Furthermore, any impact would be expected to be greater at sites with a higher cover of *C. monilifera*, and consequently with a proportionally higher loss of resources following *C. monilifera* control.

If the second null hypothesis were to be rejected, the following would be tested:

(3) bird species or groups of bird species that differ in resource requirements, would not differ in their response to *C. monilifera* removal.

6.2 Methods

6.2.1 Study areas and *C. monilifera* control

This study was conducted at three locations along the coastline of eastern Australia:
(1) Myall Lakes National Park (NP) (44,172 ha) north of Hawks Nest (32°32'S, 152°18'E) on the mid-north coast of NSW (location: ML);
(2) Perkins Beach Reserve (200 ha) (34°36'S, 150°53'E) south-east of Wollongong, on the central coast of NSW (location: PB); and
(3) Eurobodalla NP (2,220 ha) on the south coast of NSW, near Moruya Heads (35°56', 150°10') (location: ENP).

All study areas (defined as ‘locations’ in the analyses) contained extensive infestations of *C. monilifera*. One patch at each location was selected for *C. monilifera* removal by the land management agency, which, in all cases, was completed by aerial application of herbicide (defined as ‘treatments’ in the analyses: sprayed) (see Toth *et al.* 1996). Other patches of *C. monilifera* infestation were left untreated (treatment: unsprayed). Prior to herbicide application, all sites consisted of open scrub in which the cover of *C. monilifera* exceeded 30% of the shrub layer. Amongst the *C. monilifera* were a variety of co-occurring tree and shrub species, most of which persist following herbicide treatment (Toth *et al.* 1996). These species were mostly indigenous but varied between locations, with *Banksia integrifolia* and *Acacia sophorae* common at all of them, *Leucopogon parviflorus, B. serrata* and *Leptospermum laevigatum* frequent at ML, *A. saligna* (naturalised) and *L. laevigatum* at PB, and *Leucopogon parviflorus* and *B. serrata* at ENP.

When applied to dense infestations, aerial herbicide application for *C. monilifera* control (described in Toth *et al.* 1996) produces an open scrub of surviving other plants, sporadic *C. monilifera* plants that escape the spray, and large areas of the ‘skeletons’ of dead *C. monilifera* plants. *Chrysanthemoides monilifera* take one to three years to reach reproductive maturity (Vranjic 2000), so shortly after control, the only *C. monilifera* fruits available are from plants that survived the initial treatment. The application procedure involves flying low over the vegetation in a helicopter and applying a Roundup® mix out of a boom spray. It takes approximately four weeks from the time of application for readily identifiable signs of *C. monilifera* senescence to become apparent (pers. obs.).

The study was conducted from June to August 1995 at PB, and June to September 1996 at ML and ENP. After the completion of the study, vegetation measures were compiled for each survey area. These included the percentage cover of dominant plant species and structural composition of the vegetation. The proportion of the shrub layer covered by *C. monilifera* at unsprayed sites ranged from an average of 70% at PB to 59% at ENP and 34% at ML (Appendix A2). The percentage cover of *C. monilifera* at sprayed sites before herbicide application was not measured, but it is presumed to be similar to that of the unsprayed sites at the same location.

6.2.2 Bird surveys

Movement through the habitat was highly restricted by dense *C. monilifera* and considered unduly disruptive to birds, so the bird survey technique selected as most likely to facilitate a reasonable estimation of bird abundance and species composition was a point-count procedure. Birds were identified and counted with the observer standing at a fixed point, rather than having to also concentrate on minimising noise (and staying upright!) while moving through the vegetation (Reynolds *et al.* 1980). Paths through the vegetation were established at all sites and survey points (ten at each site at ML and ENP, seven at PB) located at 50m intervals.

A further complication for the estimation of bird abundance and composition was that the structure of the vegetation at sprayed sites changed during the experiment coincident with death of *C. monilifera*. The change in structure would alter the relative detectability of birds in these two treatments (Dawson 1981), with obvious consequences for treatment comparisons. With the death of *C. monilifera*, the vegetation becomes progressively more open, as the leaves on dead plants shrivel and drop. Consequently, there would be a higher probability of detecting birds at greater distances from the observer in the sprayed sites after *C. monilifera* death. To counter this, the size of the survey area sampled per point count was restricted to an area that I could be confident of detection of all birds in the denser vegetation (unsprayed sites) over the specified length of time. A pilot study demonstrated that the frequency of detection of birds declined rapidly at distances exceeding 10m from the observer in *C. monilifera* vegetation, and in surveys greater than 10 minutes in length (appropriate survey length determined as per Pyke and Recher (1984)). Thus, the point census procedure used in the study was 10 minute counts of all birds within a 10m radius of a central point.

In common with all other bird survey techniques, this methodology has assumptions, the most significant of which is likely to be that rates of movement of birds (*i.e.* into the census area) are similar between experimental factors. I did not test this assumption, and it applies in most published studies using bird census to compare areas. The census procedure involved counting both the birds initially present at the site at the start of the census period and new birds arriving during the census. Unlike many point-count procedures (Pyke and Recher 1984), I did not set an "equilibration time", during which no observations were made, at the start of the time at each census point. This is often done to allow birds to become accustomed to the observer and resume their usual activities. I chose not to do this for several reasons: (i) I believed that doing so would underestimate the birds present at the sprayed sites after *C. monilifera* death, because birds would be disturbed from the census area on the approach of the observer and would be unlikely to return due to their presence, especially in more open habitats; and (ii) I observed that my arrival at a census point often stimulated responses from birds (*e.g.* alarm calls, movements) that facilitated their detection in dense vegetation.

6.2.3 Experimental design

The *C. monilifera* control program was implemented according to the management needs and available resources of land management agencies. Consequently, I was not able to establish an experimental design that would maximise the power of detecting an impact of the control program. Rather, I had to use areas in which *C. monilifera* control was being undertaken for other reasons. This placed several limitations of the design of the experiment, analyses of the data collected, and its interpretation. While these limitations do not undermine the study overall, they warrant more detailed discussion here.

Chrysanthemoides monilifera control by aerial herbicide application was conducted at only a single patch of vegetation at each location. Thus, it was only possible to have a single independent sprayed site per location. In contrast, there were often more options for unsprayed sites. At ML, three unsprayed sites were established in *C. monilifera* stands, each site independent and separated by sections of coast with either natural vegetation (without *C. monilifera*) or where past control activities had been undertaken. Similarly, two unsprayed sites at ENP were established. At PB, the small size of the reserve and its isolation, coupled with the continuous cover of *C. monilifera*, meant that only a single unsprayed site could be established. This unequal sampling of treatments is not typically recommended in experimental design, as the mean and variance of treatments sampled more should be estimated with greater accuracy and precision. In this case, however, to detect changes induced by a perturbation occurring at an impact site, it is beneficial to compare any change to a wide spread of unimpacted sites. The rationale for the use of multiple unimpacted sites compared to single impact sites is outlined in detail in Underwood (1993). Unlike many environmental impact studies, however, the same treatment was applied to several, widely spaced, locations, allowing for generalisation of the interpretation of the responses of the bird community.

Sprayed and unsprayed sites were sampled before and after specified perturbations (aerial herbicide application and then *C. monilifera* death). The aim of this was to ensure that any differences detected between the treatments are due to the treatments themselves, rather than unknown prior differences between sites. In this study, I wished to distinguish the impact on birds of two components of the *C. monilifera* control method. These were, firstly, the impact of the disturbance and possible reactions to chemicals during and immediately after the aerial application of herbicide and, secondly, the impact of the loss of resources (such as food and shelter) from the death of *C. monilifera* itself. Thus, surveys at sites were conducted over three periods: before the herbicide was applied at sprayed sites (defined as 'periods' in the analyses: before spraying); after herbicide application but prior to the death of *C. monilifera* (period: post-spraying); and after the death of *C. monilifera* (period: post-death). This allowed only a four-week window of opportunity between the application of herbicide and death of *C. monilifera* to assess the impact of the application procedure. Bird surveys were conducted on multiple occasions at all sites over this period (four at ML and ENP, six at PB). Given the short time period in which they were spaced, however, and the limited size of sprayed (and thereby unsprayed) sites, these were deemed to not be sufficiently independent to be used as replicate times for composition of the bird community within a period of the experiment. Consequently, the average of all survey points and all surveys within a period was taken and used in the analyses.

6.2.4 Statistical analysis

The factors in a three-factor ANOVA were location (fixed, 3 levels – ML, PB and ENP), treatment (fixed, 2 levels – unsprayed and sprayed) and period (fixed, 3 levels – before spraying, post-spraying and post-death). Sites (random) were nested within treatment and location. In this study, the primary interest was in the interaction of period and treatment. A significant interaction between these two factors indicates that the sprayed sites are responding differently, during a specified period, to the unsprayed sites. SNK tests were used in multiple comparisons where significant differences were detected. In the design of the analysis, it is recognised that it was not possible to test for an interaction between period and site nested in treatment and location. This interaction is only relevant to the unsprayed sites, as the sprayed site was not replicated within locations. While it is possible that different unsprayed sites within each location will behave differently over time, there was no evidence that this occurred from a *post-hoc* examination of the data (Figure 6.1).

Average bird abundance, species richness per point count, and cumulative species richness were the univariate parameters analysed statistically. Cumulative species richness was the total number of species recorded among all point counts within each treatment/period combination of the site. To examine any impact of *C. monilifera* spraying or death on birds in more detail, birds were split into groups based on broad resource use patterns (Appendix A3). One group consisted of species recorded consuming *C. monilifera* fruits at the study locations (Chapter 3).

This group is an obvious candidate for an assessment of collective impact of *C. monilifera* removal as they all use they same *C. monilifera* resource directly. Other broad foraging groups that were analysed were: (i) strongly nectarivorous species; and (ii) mainly insectivorous species. Membership of these two groups was exclusive of species in the first group. There were too few records of frugivorous species that did not eat *C. monilifera* fruits or carnivorous species for these to be analysed as groups. The arrangement of species into groups to analyse collective patterns of response to experimental factors is fraught with problems. These include the generality of resource use by many species (Gosper 1999a), lack of knowledge of resource use of others and the most abundant species within a group determining the response of the group as a whole (French and Zubovic 1997). Given the low sample sizes for individual species, however, grouping was the only means to proceed with examining differential impacts of the treatments.

Multivariate procedures (using the PRIMER statistical package; Clarke and Warwick 1994) were used to further investigate changes in the bird community. Mean abundance of each species per point count per site (within each treatment/period combination) and presence/absence data were analysed separately. The Bray-Curtis similarity measure was used in non-metric multi-dimensional scaling ordination. Two-factor analyses of similarity (ANOSIM) with pairwise tests were conducted on each pair of the same factors as in the ANOVAs above. It should be noted, however, that the key interaction of factors in this experiment, of treatment and period, couldn't be directly tested by this procedure. Consequently, to test for changes at sprayed sites after the specified perturbations the factors of period and treatment were combined into a single factor – 'perturbation' (*i.e.* whether the site was unsprayed or sprayed AND whether it was before spraying, post-spraying or post-death).

Throughout the analyses, the α–level specified for a significant departure from the null hypothesis was set at 0.1. This varies from the most common practice in biological studies to set α at 0.05. It is entirely appropriate, however, for experiments of this kind, in which I am investigating whether a particular procedure to manage an invasive plant is having an impact on birds. I would prefer to accept a greater risk of type I error (to conclude there was an impact when there actually was not one) than the lower power associated with a lower probability of type I error (thus possibly missing detecting an impact when there was one). Such an approach is precautionary and appropriate in environmental impact studies (Underwood 1997).

6.3 Results

Changes in the abundance and species richness of birds occurred during the experiment. Interpreting the causes of these changes is complicated by the number of factors involved and the number of significant interactions between factors (one of which, treatment x period, is the key statistic of the experiment). Because of the numerous interactions, little testing of main effects was undertaken, and this was not, in any case, the primary purpose of this experiment. The effects of location as both a main effect and in interaction with period, and treatment, are not considered in detail in this chapter. Determining differences caused by location was not the main thrust of the experiment, and some variation at the spatial scales of locations and sites would be expected. Similarly, I did not pursue which sites showed differences if an effect of site nested in treatment and location was detected.

6.3.1 Bird abundance

The *C. monilifera* control program clearly had an impact on birds. This is demonstrated by the significant change in the mean abundance of birds per survey point between periods across treatments (period x treatment interaction; Table 6.1). A SNK test was not able to detect differences in this interaction, and consequently could not contribute to developing an alternative hypothesis. The SNK test is less powerful than ANOVA (Underwood 1997), hence the inability to detect a difference in this case. In examining the differences graphically, however, there is no evidence for a decline in bird abundance at the sprayed sites (relative to unsprayed sites) immediately following herbicide application and before *C. monilifera* death (the post-spraying period) (Figure 6.1). In contrast, there is strong evidence for a decline in bird abundance at sprayed sites after *C. monilifera* death. The greatest difference in bird abundance among the three periods at sprayed sites (averaged across locations) is between post-death (lowest mean abundance) and post-spraying (greatest abundance). Consequently, it is inferred that the significant change in bird abundance in the period by treatment interaction occurred after the death of *C. monilifera*.

Other interactions of factors produced changes in abundance of birds. For example, the abundance of birds differed between periods across locations (period x location interaction, Table 6.1) and the abundance of birds differed between sites within particular treatment and location combinations (site nested in treatment and location, Table 6.1).

6.3.2 Species richness

There was not a significant interaction between period and treatment for either mean species richness of birds per point or cumulative species richness at the site (Table 6.1). This indicates

Table 6.1 - ANOVA results for the impact on birds of aerial herbicide application to control *C. monilifera*

Key: abun = mean bird abundance per point; rich = mean bird species richness per point; cRich = cumulative species richness; frug = mean abundance of bird species that eat *C. monilifera* fruits per point; nect = mean abundance of primarily nectar-feeding birds per point; ins = mean abundance of primarily insectivorous birds per point.

Source of variation	df	MS abun	F abun	MS rich	F rich	MS cRich	F cRich	MS frug	F frug	MS nect	F nect	MS ins	F ins
Location	2	1.3339	1.68	0.2248	1.03	19.478	0.42	2.5027	4.38	0.4146	8.64*	1.1643	31.45***
Treatment	1	0.0079	0.01	0.0270	0.12	5.5862	0.34	1.3460	2.36	1.5034	31.35**	0.0396	1.07
Period	2	1.1839	7.16**	0.1085	2.63	1.2816	1.21	0.5712	15.54***	0.1835	5.16*	0.0223	0.24
Treatment x period	2	0.9845	6.07**	0.1371	3.33	2.1092	2.00	0.7992	21.75***	0.0298	0.84	0.0382	0.41
Location x treatment	2	0.4860	0.62	0.0308	0.14	3.2464	0.19	0.6336	1.11	0.0641	1.33	0.0067	0.18
Period x location	4	0.6578	4.06*	0.2427	5.89**	1.4710	1.39	0.3708	10.09***	0.2688	7.55**	0.1173	1.24
Treatment x location x period	4	0.3027	1.86	0.0667	1.62	0.7826	0.74	0.0832	2.28	0.0561	1.58	0.0436	0.46
Site(location x treatment)	3	0.7864	4.85**	0.2178	5.29**	16.667	15.79***	0.5707	15.53***	0.0480	1.35	0.0370	0.39
Period x site(location x treatment)	6	0.1622		0.0412		1.0556		0.0367		0.0356		0.0943	

* P<0.1, ** P<0.05, *** P<0.01

Table 6.2 - ANOSIM results for the impact on birds of aerial herbicide application to control *C. monilifera*

Factor 'perturbation' is a combination of the factors treatment and period (see section 6.2.4). Global R statistic shown.

	Factor included in ANOSIM of abundance data				Factor included in ANOSIM of presence/absence data			
Factor tested	**Location**	**Period**	**Treatment**	**Perturbation**	**Location**	**Period**	**Treatment**	**Perturbation**
Location	-	0.959***	0.893***	0.970***	-	0.962***	0.933***	0.939***
Period	0.335***	-	-0.088	-	0.089	-	-0.145	-
Treatment	0.072	-0.074	-	-	-0.031	-0.127	-	-
Perturbation	0.417***	-	-	-	0.025	-	-	-

* P<0.1, ** P<0.05, *** P<0.01

that neither herbicide application or *C. monilifera* death affected the number of bird species at sprayed sites (Figures 6.1). The interaction of other factors did have a significant impact on bird species richness. Mean species richness per point and cumulative species richness differed between sites within the one combination of treatment and location, and over periods across locations for mean species richness per point (Table 6.1).

6.3.3 Groups of species

The impact of the *C. monilifera* control program on bird abundance was not spread evenly across groups of birds. Nectarivores and insectivores were apparently unaffected by the *C. monilifera* control program (no significant period x treatment interactions; Table 6.1 and Figure 6.2). Nectarivore abundance, however, was significantly different over periods across locations (period x location interaction) and between treatments. No interaction terms were significant with insectivorous bird abundance but there were differences between locations, with fewer birds at ENP than the other two locations (by SNK test).

The *C. monilifera* control program adversely affected frugivores, specifically those that consume *C. monilifera* fruits (Table 6.1). As with the abundance of all birds, a SNK test was unable to determine where this difference arose. There was no evidence, however, to suggest a change in frugivore abundance after herbicide application, but dramatic changes following *C. monilifera* death (Figure 6.3). Again, the greatest difference in the abundance of frugivores of *C. monilifera* fruits at sprayed sites between periods (averaged across locations) was from post-death (lowest abundance) to post-spraying (greatest abundance). The abundance of frugivores of *C. monilifera* was also different at sites nested in treatment and location, and between periods across locations.

6.3.4 Community composition

In the multivariate analyses, location was the strongest influence on bird community composition. This is apparent from both the ordination (Figure 6.4, where the three very distinct groups of points comprise those of the different locations - bird abundance shown here, although the same pattern occurred for presence/absence) and the ANOSIM results (Table 6.2). The effects of location are of a similar magnitude for both the abundance and presence/absence data, suggesting that differences in species composition may be the driving influence. In all pairwise comparisons in each test, all locations were different from each other.

Treatment effects were not significant in the ANOSIMs (Table 6.2), but this indicates that unsprayed and sprayed sites did not differ in bird composition across all locations or periods and does not rule out the possibility of a treatment by period interaction. Period had a significant

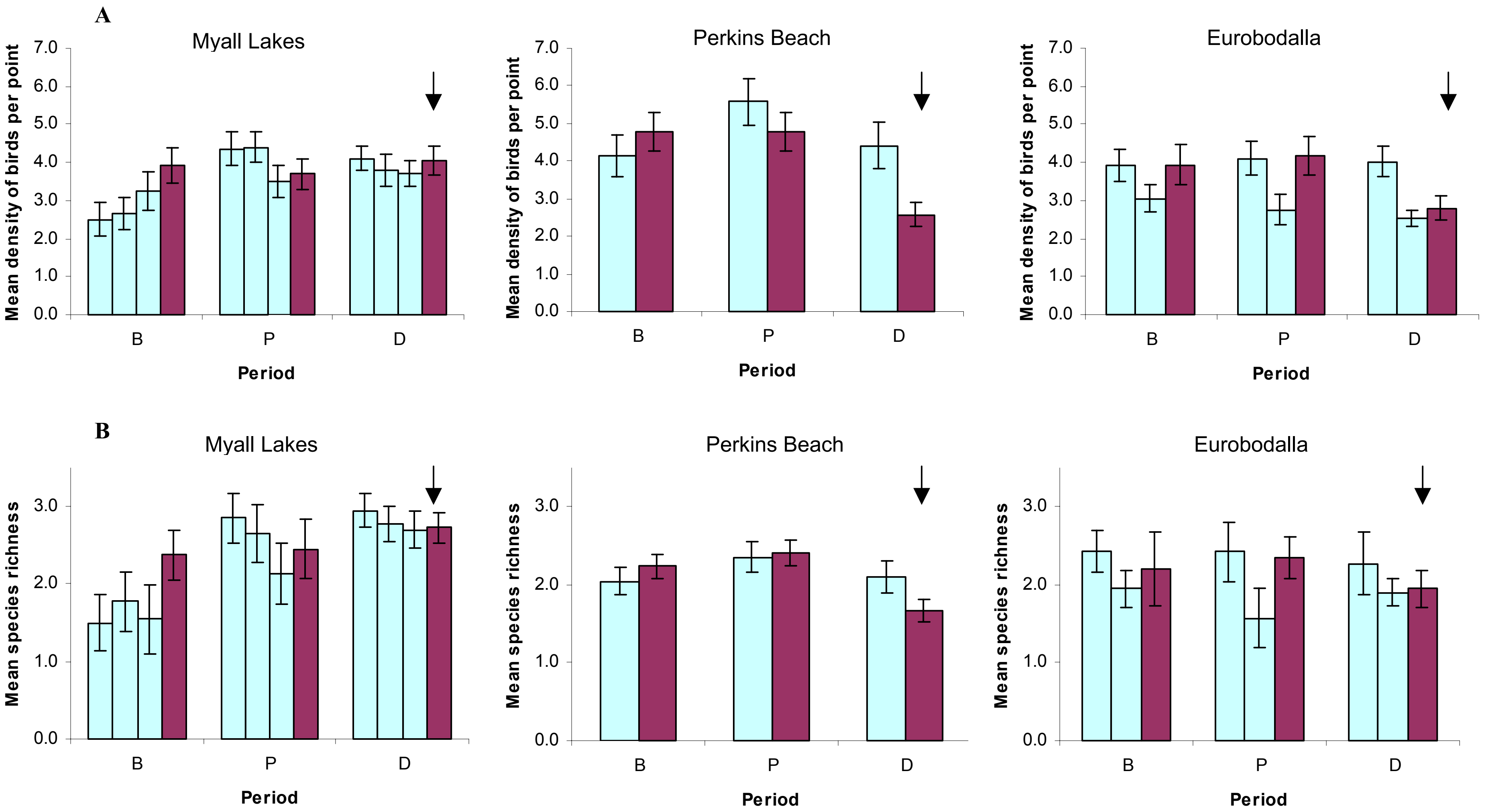


Figure 6.1 – Mean bird density (A) and species richness (B) per survey point over the three experimental periods

Period: B = before herbicide application; P = after aerial herbicide application (but before *C. monilifera* death); and D = after *C. monilifera* death, at the three study locations. 🟦 = unsprayed sites; 🟥 = sprayed sites. The arrow indicates where a change would occur if death of *C. monilifera* impacted bird abundance (A) or species richness (B).

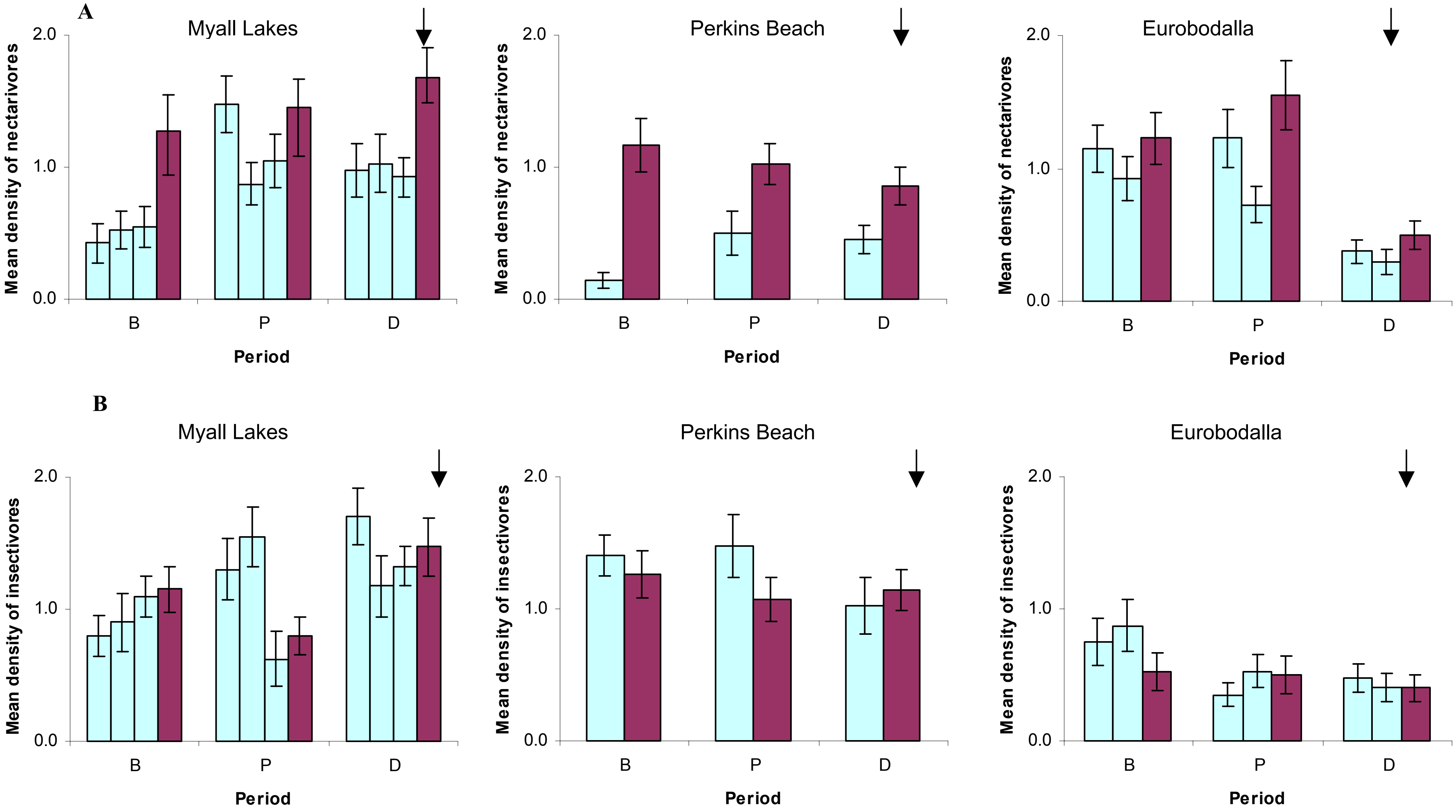


Figure 6.2 - Density of birds that are mainly nectarivorous (A) and insectivorous (B) per survey point over the three experimental periods

Period: B= before herbicide application; P = after aerial herbicide application (but before *C. monilifera* death); and D = after *C. monilifera* death, at the three study locations. ▢ = unsprayed sites; ■ = sprayed sites. The arrow indicates where a change would occur if death of *C. monilifera* impacted bird abundance.

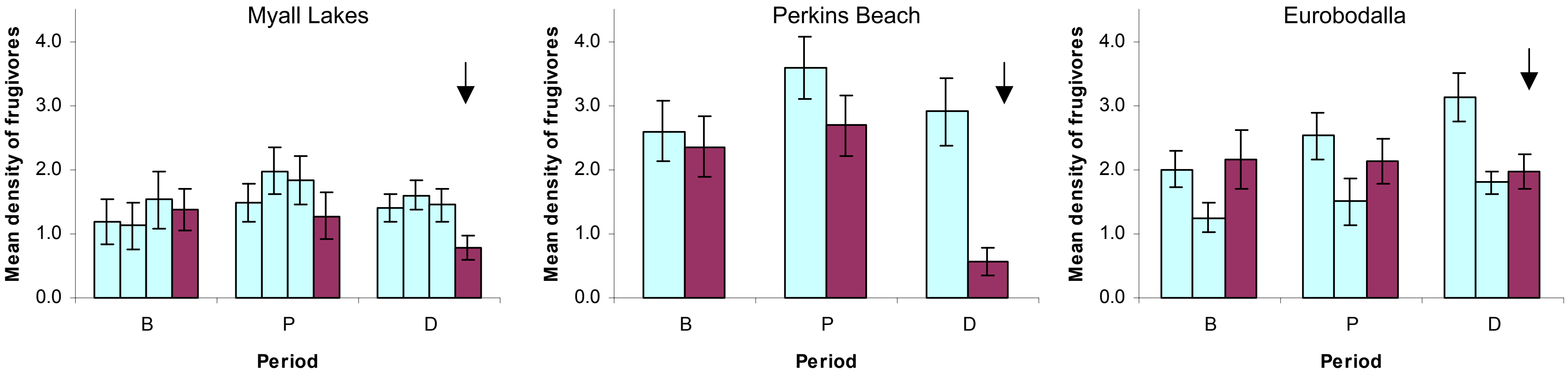


Figure 6.3 - Density of birds that consume *C. monilifera* fruits per survey point over the three experimental periods

Period; B = before herbicide application; P = after aerial herbicide application (but before *C. monilifera* death; D = and after *C. monilifera* death, at the three study locations. □ = unsprayed sites; ■ = sprayed sites. The arrow indicates where a change would occur if death of *C. monilifera* impacted bird abundance.

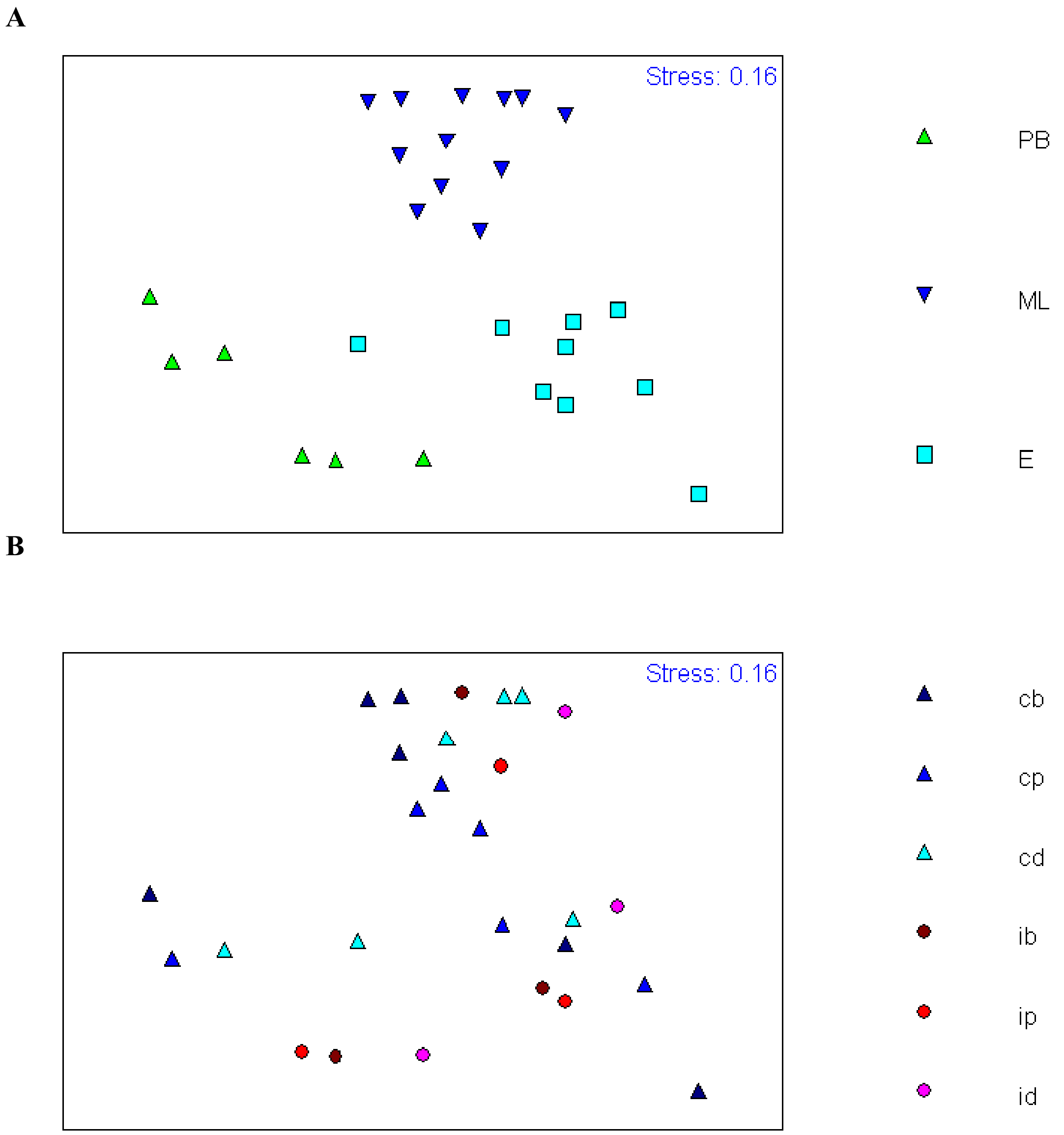


Figure 6.4 - Non-metric multi-dimensional scaling ordination of survey sites by bird abundance, with location (A) or perturbation (B) imposed on the ordination

Key for perturbation: cb = unsprayed, before spraying; cp = unsprayed, post-spraying; cd = unsprayed, post-death; ib = sprayed, before spraying; ip = sprayed, post-spraying; id = sprayed, post-death.

effect on the abundance on birds in the community but only when tested against location (Table 6.2). This is suggestive of a period by location interaction, which was also identified in the ANOVAs above for a number of variables. In pairwise tests (averaged across all locations), there are differences between all periods, although these were of greatest magnitude between before spraying and post-death, and post-spraying and post-death.

In the tests of the combined treatment and period factor (perturbation), a significant difference was found between levels of perturbation when tested against location with bird abundance, but not presence/absence data (Table 6.2). Evidence in the ordination of change in community composition induced by the effects of the *C. monilifera* control program at sprayed sites is weak (Figure 6.4). All sprayed sites within the post-death period appear at the right edge of the clusters of points from each location, though not far from other points. The sample sizes of sprayed sites were insufficient to permit pairwise comparisons between periods.

6.4 Discussion

The statistical procedures used in this study have tested the relationships between factors in the experiment in different ways. In the ANOSIMs, interactions were not tested directly, but the main effects were. In contrast, the identification of significant interactions between factors in most of the ANOVAs resulted in very few cases of the main effects being tested. This was not a problem, because the main test of interest in this experiment was the interaction between period and treatment. These two types of tests reveal that somewhat different factors and combinations of factors are most strongly influencing components of the bird community. These two types of tests emphasise different components of variation in a complex pattern of changes between locations, experimental treatments and periods.

6.4.1 Effects of the *C. monilifera* control program

Determining the effects on the bird fauna of the aerial application of herbicide for *C. monilifera* control was the main aim of this study. A reduction in bird abundance has been demonstrated, particularly the abundance of those species that consume *C. monilifera* fruits. Because of the design of the study, this impact can be attributed with confidence to the death of *C. monilifera* plants, occurring over the slightly longer term, rather than to the immediate disturbance from the application of the herbicide or initial reaction to being exposed to the chemicals. Consequently, the results are generalisable to other methods of *C. monilifera* control that are broad-scale and involve vegetation change over a short period. I did not survey the bird community in the minutes and hours after herbicide application, so there may have been a short-term impact on bird occurrence. If there was, it had disappeared by the time I commenced surveying within a few days of the spraying event.

There is little evidence for changes in bird composition between periods for any treatment (a period by treatment interaction) using multivariate analysis (ANOSIM of perturbation against location, Table 6.2) and the ordination. This difference between the tests may have arisen because of the type of change induced by the *C. monilifera* control program. The abundance of birds, and specifically one group of birds, declined after *C. monilifera* death, however, this change in abundance may have been minor compared to the substantial differences between locations in species composition driving the multivariate techniques.

As predicted, the impact of killing *C. monilifera* plants and thus removing certain resources did not affect all components of the bird community equally. The subset of birds that directly consumes *C. monilifera* fruits was significantly affected, and probably contributed substantially to the changes in the overall abundance of birds. The same factors and interactions of factors were significant in explaining changes in abundance of both all bird species at survey points and the abundance of frugivores of *C. monilifera*. This strongly suggests that frugivores of *C. monilifera* caused the drop in abundance of all bird species following *C. monilifera* death. There is no indication of any other group of birds being affected. Once *C. monilifera* had been removed by herbicide treatment at sprayed sites, it is likely that few fleshy fruit resources would have been available to frugivorous birds. The surveys after death of *C. monilifera* were conducted in late winter, a period when few of the native plant species that are vertebrate-dispersed produce fruits (Chapter 2). Consequently, these frugivorous bird species would have been forced to either use other foods or move from the area. It appears that substantial numbers moved elsewhere. At this stage, it is unknown whether frugivore abundance in *C. monilifera* vegetation is similar, greater (due to greater fleshy fruit resources) or less (possibly less fruit variety) than native vegetation. This will be investigated further in the following chapter, which will allow further assessment of whether the observed change in frugivore abundance is of concern or perhaps the reversion to a more ‘natural’ level of frugivore abundance in coastal vegetation.

Neither primarily nectar-feeding species nor primarily insectivorous species appeared to have been affected at all by the *C. monilifera* control program. Bird species in these groups do not directly use resources provided by *C. monilifera*, although they may shelter in it and presumably consume insects that shelter and/or feed in *C. monilifera*. Over the period after *C. monilifera* death measured in this study, the dead *C. monilifera* plants retain much of their former structure. The leaves die and shrivel but remain attached to the plant for some time, for example. Adverse impacts on bird species that require dense, low shelter may develop as the dead *C. monilifera* plants shed their leaves and the habitat becomes even more open. Monitoring of the sites over such a period was not conducted in this experiment, and in all likelihood, it would be difficult to discriminate changes in the bird community due to the gradual opening up of the habitat from a variety of other factors that might become relevant over this time.

French and Zubovic (1997) detected differential responses of groups of bird species to the invasion of native vegetation by *C. monilifera*. Unfortunately, the criteria for the grouping of species and the final division of species in that study differed greatly from this one, making comparison of the findings difficult. Few species in the group French and Zubovic (1997) classified as “more reliant on plant foods”, that were adversely affected by *C. monilifera* invasion, were detected in this study (perhaps this should be expected if they do not do well in *C. monilifera* vegetation). The frugivores of *C. monilifera* that declined in abundance following *C. monilifera* death in this study were distributed mostly between two groups in French and Zubovic (1997). Both these groups showed weak evidence for preferring *C. monilifera* to native vegetation, although only at some locations (French and Zubovic 1997). It is probable that some of these bird species are taking advantage of the invasion of coastal vegetation by *C. monilifera*, particularly of the fruit resources it provides, and lose this advantage when *C. monilifera* is controlled. Extreme examples of these species include House Sparrow (*Passer domesticus*) and Common Starling (*Sternus vulgaris*). Both of these species consume *C. monilifera* fruits (Chapter 4), but were not recorded in bird surveys in either native vegetation (Chapter 7) or at sprayed sites after *C. monilifera* death.

It may be significant that the location at which birds exhibited the strongest response to *C. monilifera* death at sprayed sites was PB. This site had the most dramatic decline in the abundance of all birds and of species that consume *C. monilifera* fruits (Figures 6.1 and 6.3). There are at least two possible contributing factors for this occurrence. First, the unsprayed site at this location had the greatest percentage cover of *C. monilifera*, and it is likely that the sprayed site was similar. Unsprayed sites at ML were uniformly lower in *C. monilifera* cover, and of all the locations, these sites exhibited the least visible effects of *C. monilifera* death on bird abundance. It is possible that the cover of *C. monilifera* at sites at which control measures are taken may contribute to the impact on the bird community. Greater impact is likely where *C. monilifera* density is greater, and where the density of resources provided by *C. monilifera* to birds are presumably greater. Second, the PB location was distinct in being a smaller remnant of vegetation bounded by urban and industrial areas. The other locations formed part of conservation reserves and were bounded by extensive areas of mostly native vegetation. It is possible that these areas of native vegetation were able to support more birds than the highly modified habitat adjoining PB, and some of these birds may have used the sprayed sites. In this way, the surrounding vegetation may provide a ‘buffer’ to the impact of *C. monilifera* death on the bird community. If this were the case, the matrix of surrounding habitat may be important in determining the consequences of invasive plant removal.

No evidence of an impact of the *C. monilifera* control program on the species composition of the bird community was detected. Neither mean species richness per survey point nor cumulative species richness per site was significantly affected by the interaction of period and location. In the multivariate analyses, ANOSIMs were significant when raw abundance data was analysed in several cases, but not presence/absence data. This included the analysis of the combined perturbation factor (representing the period by treatment interaction). Each of these results points strongly to the *C. monilifera* control program only impacting on the abundance of birds, and not instantaneous or cumulative species richness. It appears that while the abundance of frugivorous bird species declines after *C. monilifera* death, most species do not disappear from sprayed sites completely.

6.4.2 Other factors

Several other factors and their interaction were significant in analyses, complicating the identification of the impact of the *C. monilifera* control program. Different sites within the same location and treatment combination supported different abundances of all bird species and of frugivores of *C. monilifera*, and different mean species richness per point count and cumulative species richness. This was not unexpected, and was a significant reason for the sampling of multiple unsprayed sites wherever possible (sprayed sites could not be nested in this way). Indeed, if there were no variations among unsprayed (and sprayed) sites, then multiple sites would not need to have been sampled.

The interaction of period and location was significant for the abundance of all species, frugivores of *C. monilifera* and nectarivores at survey points, and the species richness at survey points. This interaction was also identified as likely in the ANOSIM of period against location (though only for abundance data). That changes in the bird community over time differed between locations was also not surprising given the distribution of locations hundreds of kilometres apart along the NSW coastline. Any number of factors could be causing this divergence, but may include temporal changes in food availability. The study was conducted over the winter months, including the latter part of the period of maximum fruit production of *C. monilifera*. Over the later stages of the study (such as after *C. monilifera* death), the amount of *C. monilifera* fruit available declined (Chapter 2), although there is no evidence that this resulted in a drop in the abundance of consumers of *C. monilifera* fruits at unsprayed sites (Figure 6.3). The changes this (and other patterns of resource availability) may have on bird communities may vary between locations, as the options for replacing these resources would vary with different vegetation composition. There certainly appeared to be a decline in the abundance of extensively nectarivorous species across all sites at ENP in the final period of the study, post-death (Figure 6.2). As this change occurred irrespective of treatment, it can be presumed to be a response to a more general change across the location as a whole. Perhaps significantly, the period of peak flowering of *Banksia integrifolia*, an important species for nectarivorous birds (Gosper 1999a), had finished at ENP by the end of the study, but had not at the other two locations (Table 3.4, Chapter 3). Similarly, temporal changes in the bird community at sites in *C. monilifera* and native vegetation were detected by French and Zubovic (1997), who also attributed this to phenology of native plants or insect availability.

Location effects appeared to be the dominant influence on community composition detected in the ANOSIMs and were readily apparent in the ordination. That substantial location differences existed was not surprising given the distance separating them. In the univariate analyses, only the abundance of insectivorous bird species and cumulative species richness were no interactions of location identified, allowing for their assessment as a uniform main effect. Insectivorous bird abundance did differ with location, with fewer insectivorous birds at ENP than at the other two locations. As location differences are not particularly important in this experiment, the possible reasons for them are not further explored. Cumulative species richness did not differ with locations, suggesting that the differences in presence/absence analysis of locations was probably due to the composition of species, rather than the number of species.

6.4.3 Future use of aerial herbicide application for control of *C. monilifera*

Now that the program of aerial herbicide application has been shown to affect bird communities, what are the likely consequences for the bird community and the control program? First, it should be emphasised that this experiment has only investigated the short-term impacts of the removal of *C. monilifera*: that is in the weeks following herbicide application and then the weeks following *C. monilifera* death. Further changes in the bird community would be expected over the subsequent months as the vegetation changes at sprayed sites. Over time, regeneration of *C. monilifera* and other species from soil-stored and dispersed seeds would begin to replace the dead vegetation, and begin to replace some of the resources lost to birds. Consequently, the impacts identified here are only expected to be transient, with the ultimate impact of such management determined by the composition of the replacement vegetation community. This is not known and likely to be site specific depending on a number of factors, such as if sufficient future management resources are available to control regeneration of *C. monilifera* and other invasive plants, and if a soil-stored seed bank of native plant species exists.

Second, the particular group of birds impacted by *C. monilifera* control contains species that are not of high conservation concern. None are listed as threatened (Garnett and Crowley 2000), and three are exotic, the House Sparrow, Common Starling and Red-whiskered Bulbul

(*Pycnonotus jocosus*). Additional species, such as Pied Currawongs (*Strepera graculina*), Red Wattlebirds (*Anthochaera carunculata*), Common Koels (*Eudynamys scolopacea*), Australian Ravens (*Corvus coronoides*) and Eastern Rosellas (*Platycercus eximius*), are mobile, omnivorous generalists that appear to have benefited from anthropogenic landscape modification, at least in some areas (Smith *et al.* 1989; Higgins and Davies 1996; Higgins 1999; Reid 1999; Higgins *et al.* 2001). This is not surprising, because species able to use *C. monilifera* fruits have already demonstrated some dietary flexibility and this also extends to other vertebrate-dispersed exotic plants (Loyn and French 1991; Spennemann and Allen 2000). This ability to use novel and expanding food sources is a characteristic of obvious benefit in adapting to changing habitats. These species may be primarily attracted to *C. monilifera* vegetation by the abundant fruit resources, and be able to move to other habitats for other requirements. As such, they may not suffer substantial local population declines after *C. monilifera* control. However, their abundance at the actual sites of control may be reduced. House Sparrows are an extreme example of this type of response. They occurred in large flocks in *C. monilifera* vegetation at PB, yet were not recorded after *C. monilifera* death at sprayed sites. They fed on *C. monilifera* fruits and flew into *C. monilifera* vegetation from adjoining urban areas. Presumably after *C. monilifera* death, they had little to attract them into coastal vegetation from other habitats. That this group of species appear disadvantaged by the *C. monilifera* control program is overall probably of little concern and not sufficient justification to limit the use of the aerial control program on its own.

Third, if the vegetation previously dominated by *C. monilifera* could be returned to a more natural state, this might have positive outcomes far outweighing the short-term impacts of control. Vegetation dominated by *C. monilifera* does differ in the bird community it supports compared to native vegetation (French and Zubovic 1997). Presumably, the bird community in native vegetation would be more desirable from a conservation perspective. The feasibility of returning large areas of degraded vegetation previously dominated by *C. monilifera* to a semi-natural state is questionable, however, and it may be that another invasive plant may largely replace it. For example, Asparagus Ferns (*Protasparagus* spp.), Glory Lily (*Gloriosa superba*), Lantana (*Lantana camara*), Mirror Bush (*Coprosma repens*) and *Ehrharta* spp. have become significant problems at sites at which *Chrysanthemoides monilifera* control has been undertaken (NSW NPWS 2001). Additionally, Mother-of-Millions (*Bryophyllum* spp.), Guinea Grass (*Panicum maximum*), Umbrella Tree (*Schefflera actinophylla*), Brazilian Pepper Tree (*Schinus terebinthifolia*), Easter Cassia (*Senna pendula*) and Singapore Daisy (*Wedelia trilobata*) are very invasive on foredunes in southern Queensland (Batianoff and Franks 1998), and are likely to pose problems in NSW.

Fourth, what other options are there for the broad-scale control of *C. monilifera* and are the consequences of aerial herbicide application acceptable if no other feasible means exists to control *C. monilifera* or otherwise reduce its impacts on native species and ecosystems? Furthermore, are the impacts of alternatives likely to be any different? Aerial herbicide application is one of the main strategies employed to control *C. monilifera* in NSW, along and in conjunction with physical removal, vehicle or hand-based herbicide application (both of which are only feasibly applied over small and accessible areas) and fire (Vranjic 2000). Several biological control agents have been released, but none of these has substantially reduced existing *C. monilifera* infestations. At the current time, aerial herbicide application to control *C. monilifera* is one of the most cost-effective control strategies (NSW NPWS 2001) and consequently is likely to continue to be used. Of the control options, only biological control is likely to change the habitat at a gradual pace allowing the bird community to adapt to the new habitat with fewer consequences. It should be noted, however, that since this study was conducted one of the recently introduced biological control agents (Bitou Seed Fly, *Mesoclanis polana*) has spread throughout the range of *C. monilifera* in NSW and may be substantially reducing fruit production (Vranjic 2000). Consequently, it is possible that vegetation dominated by *C. monilifera* is no longer as favourable to frugivores as it once was.

6.4.4 Conclusion

In this part of my study, I have demonstrated that the process of aerial herbicide application to control *C. monilifera* has had an impact on the bird community. This impact was largely restricted to a drop in the abundance of bird species that feed on *C. monilifera* fruits, and appeared to occur only after the *C. monilifera* died. As far as I am aware, this is one of the first studies of the experimental testing (rather than a comparison of treated and untreated areas) of the impact of the control of an invasive plant on a whole community of terrestrial fauna. In aquatic systems, Olaleye and Akinyemiju (1996) recorded an increase in fish following control of the invasive water hyacinth (*Eichhornia crassipes*), and Linz *et al.* (1996a; 1996b) and Homan *et al*. (2003) have recorded varying responses of individual bird species to control of *Typha* spp. This latter set of studies was conducted over a longer time period than this study. In my study, no short-term positive impacts on the bird community of invasive plant control occurred, and the abundance of some species decreased. The extent of *C. monilifera* invasion at all sites was, however, such that the vegetation community after spraying remained highly degraded, and not at all similar to healthy, natural coastal vegetation (Appendix A2). Consequently, the fact that an adverse impact of *C. monilifera* control on birds has been demonstrated should not necessarily prevent future *C. monilifera* management by this, or similar techniques. It does mean, however, that these impacts (of loss of habitat and resources, as well as the direct impact of the control technique) should be considered in the formulation of management strategies for *C. monilifera* and for other invasive plants (among other considerations, such as in Westman 1990). For example, the use that fauna make of habitat dominated by invasive species, what replacement resources are available for fauna after removal of the invasive species and if additional replacement resources can be provided in habitat reconstruction should be assessed. This supports the cautious approach on invasive plant control taken by, for example, Westman (1990) and Date *et al.* (1991).

In this case, the adverse impacts of the control program on birds are not, in my view, sufficiently great to recommend halting the control program. Of course, in coming to this conclusion, the consequences of stopping what is at the current time the most cost-effective method of controlling *C. monilifera* had to be considered. From a conservation perspective, given the species of birds disadvantaged (discussed above) and the known deleterious impacts of *C. monilifera* invasion (Weiss and Noble 1984a; 1984b; French and Eardley 1997; French and Zubovic 1997; Vranjic *et al.* 2000), it may, in the medium term, be better to accept the changes to bird communities induced by the control program. The impact of the aerial control program on other biota needs to be further considered and experimentally examined, however, as some deleterious impacts have been demonstrated on plants (Toth *et al.* 1996; Matarczyk *et al.* 2002) and may occur to other fauna.

CHAPTER 7. EFFECTS OF VEGETATION DOMINANCE BY *CHRYSANTHEMOIDES MONILIFERA* ON BIRD COMMUNITIES

Invasion of coastal habitats by *Chrysanthemoides monilifera* (left) dramatically alters the structure and composition of the vegetation compared to uninvaded vegetation (right). In invaded vegetation, *C. monilifera* can form a near monoculture in the lower shrub layer, here with some emergent *Banksia integrifolia*. The photo of native vegetation shows a dense upper shrub layer of *B. integrifolia* and *Monotoca elliptica*. How do birds respond to these habitat changes?

Preface

In this Chapter, I investigate what effect dominance of coastal vegetation by *C. monilifera* has on the abundance, species richness and composition of the coastal bird community. In particular, this work extends beyond previous research by examining the generality of the bird community response over different periods of the year and by re-classifying bird species into groups after a detailed assessment of their food resource use in coastal vegetation (Chapter 3 and 4).

7.1 Introduction

Invasive plants can have a variety of impacts on ecosystems (Vitousek *et al.* 1996). Changes to vegetation community composition and structure through plant invasion can change animal abundance and/or species richness across a range of taxonomic groups (*e.g.* Slobodchikoff and Doyen 1977; Griffin *et al.* 1989; French and Major 2001). Invasive plants do not appear to induce a general response in fauna, but rather the effect(s) may depend on specific characters of the invasive plant and the invaded ecosystem. In some cases, where an invasive plant has increased the structural complexity of vegetation in invaded ecosystems, portions of the fauna community have responded with increased total abundance (Knopf and Olsen 1984; Braithwaite *et al.* 1989). In other cases, even where no impacts of plant invasion on total fauna abundance or species richness were recorded, the composition of the fauna community changed (Wilson and Belcher 1989; French and Zubovic 1997).

It would seem unlikely that habitat dominated by an invasive plant would provide resources that are sufficiently similar to natural, uninvaded habitat to support an identical fauna community. Changes in the relative abundance of species could be expected to occur, even if broader parameters such as total abundance and species richness were unaffected. Invasive plants do provide fauna with habitat and a range of other resources, including food (Westman 1990; Loyn and French 1991). In some cases, food from invasive plants has been used at least as much as native alternatives (Knight 1986; Waring *et al.* 1993; French and Major 2001). Thus, habitat dominated by an invasive plant has the potential to provide essential resources for some fauna. This has been suggested to be the case in northern NSW, where extensive infestations of Camphor Laurel (*Cinnamomum camphora*) and Large-leaved Privet (*Ligustrum lucidum*) may provide important habitat and food for birds (Date *et al.* 1991; Ekert and Bucher 1999). The changes to fauna communities induced by invasive plants should be a consideration in prioritising species and locations for control programs, along with a range of other factors. At the current time, it would appear that for most invasive species there has been insufficient research to guide these decisions (Pakemann and Marrs 1992).

Bitou Bush (*Chrysanthemoides monilifera* ssp. *rotundata*) is an invasive plant in coastal habitats in eastern Australia. *Chrysanthemoides monilifera* was extensively planted for sand-dune stabilisation from the 1940s to 1970 (Weiss 1986). Since that time, it has spread to occupy an estimated 80% of the NSW coastline (NSW NPWS 2001). A number of impacts of *C. monilifera* invasion have been demonstrated on native biota. *Acacia sophorae* is displaced and may have lower seed production (Weiss and Noble 1984a) and seed germination and growth rates of seedlings are reduced (Weiss and Noble 1984b; Vranjic *et al.* 2000) when co-occurring with *C. monilifera*. French and Eardley (1997) found changes in the species composition of litter invertebrates in invaded vegetation. ‘Invasion of native plant communities by Bitou Bush' has been listed as a key threatening process under the NSW *Threatened Species Conservation Act 1995* (Dickman 1999), recognising the detrimental impact of *C. monilifera* on coastal plant communities and threatened species.

Chrysanthemoides monilifera produces small black fleshy fruit, which are consumed and dispersed by a range of birds and mammals (Dodkin and Gilmore 1984; Gosper 1999a). French and Zubovic (1997) investigated the changes to bird communities induced by *C. monilifera* invasion. They found little evidence for changes in overall bird abundance or species richness between vegetation dominated by *C. monilifera* and native species, but some changes in species composition. The main effect detected was a reduction in the abundance and species richness in invaded vegetation of a suite of species “that are rarely insectivorous and instead feed on nectar, fruits, flowers and/or seeds in the understorey”. In this study, I aimed to extend knowledge of the response of the bird community to *C. monilifera* invasion beyond the work of French and Zubovic (1997), using the following approaches.

(1) *Investigating the response of the bird community at different locations.* Specifically, I wished to examine more northerly sites in NSW, where the composition of the coastal bird community is likely to be somewhat different from that at sites that had previously been studied;

(2) *Re-visiting the classification of bird species into groupings, following the study of plant foods of birds in coastal habitats (Gosper 1999a).* This allows the grouping of species based on observed resource use in the habitats affected by *C. monilifera*. A different classification of species may allow for the detection of differential impacts of *C. monilifera* invasion on groups that use different resources. Vegetation dominated by *C. monilifera* is comparably fruit-rich over much of the year (Chapter 2). Therefore, invasion of vegetation by *C. monilifera* may have a positive impact on bird species that extensively use *C. monilifera* fruits for food. In contrast, species that are primarily nectarivorous or insectivorous would have no or little additional resources in invaded vegetation and their abundance would be expected to either remain stable or decrease; and

(3) *Investigating the generality of the bird community response, by sampling throughout the year.* The French and Zubovic (1997) study was conducted over spring and summer - a period of generally low *C. monilifera* fruit production but the peak period of fruit production

of several native plant species (Chapter 2). The peak period of *C. monilifera* fruit production is in autumn and early winter (Weiss 1984; Chapter 2) and the period of flowering of the two plant species used by the most bird species for nectar in coastal vegetation is also autumn and winter (Chapter 3). Consequently, I thought it useful to examine the response of bird communities to *C. monilifera* invasion over this period of high availability of plant resources in both native and invaded habitats.

The specific objectives of this part of my overall study were as follows:
(1) to determine if the abundance, species richness or cumulative species richness of birds differed between vegetation now dominated by *C. monilifera* compared to native vegetation;
(2) if differences are detected, to determine which bird species or groups of bird species are responding to habitat differences; and
(3) to determine if the response of the bird community was similar over two periods of the year: autumn-winter, corresponding to the period of greatest *C. monilifera* fruit production and availability of the most-used nectar sources by birds in coastal habitats; and spring-summer, a period of lower nectar resources but higher production of native fleshy fruits.

7.2 Methods

7.2.1 Study areas

This study was conducted at three locations along the coastline of eastern Australia:
(1) Myall Lakes National Park (NP) (44,172 ha) north of Hawks Nest (32°32'S, 152°18'E), on the mid-north coast of NSW (location: ML);
(2) Perkins Beach Reserve (200 ha) (34°36'S, 150°53'E) south-east of Wollongong, on the central coast of NSW (location: PB); and
(3) Eurobodalla NP (2,220 ha) on the south coast of NSW, near Moruya Heads (35°56', 150°10') (location: ENP).

All study areas (defined as ‘locations’ in analyses) contained areas of two vegetation types (defined as ‘habitats’ in analyses). These were extensive infestations of *C. monilifera* and native, uninvaded vegetation. *Chrysanthemoides monilifera* habitat was scrub in which the cover of *C. monilifera* always exceeded 30% of the shrub layer. Amongst the *C. monilifera* were a variety of co-occurring tree, shrub and groundcover species. These species were mostly indigenous but varied between locations, with *Banksia integrifolia* and *Acacia sophorae* common at all of them, *Leucopogon parviflorus, B. serrata, Scaevola calendulacea* and

Leptospermum laevigatum frequent at ML; *A. saligna* (naturalised) and *L. laevigatum* at PB and *Leucopogon parviflorus* and *B. serrata* at ENP.

All locations also had sections of native coastal vegetation free of *C. monilifera* and largely free of other invasive plant species. Like the *C. monilifera* sites, the species composition of native vegetation varied between locations, although some dominant plant species, namely *B. integrifolia, B. serrata, A. sophorae* and *Monotoca elliptica,* were widespread at all. Native vegetation sites at ML had a greater component of littoral rainforests plant species, such as *Cupaniopsis anacardioides*, as well as *Leptospermum laevigatum* and *Leucopogon parviflorus*. A canopy of *Eucalyptus botryoides, E. pilularis, E. robusta* and *Leptospermum laevigatum* occurred at PB, while native vegetation at ENP was less diverse and largely dominated by the widespread species. Greater detail on the floristic and structural composition of the vegetation at each site is contained within Appendix A2.

7.2.2 Bird surveys

The methodology of the bird surveys was the same as used in the previous experiment (described in section 6.2.2). Similar to that experiment, the structure of the vegetation differed between *C. monilifera* and native vegetation habitats. *Chrysanthemoides monilifera* sites typically had a very dense lower shrub layer (1-3m), dominated by *C. monilifera*, while native vegetation was typically denser slightly higher up (3-4m), in the canopy of *Banksia* spp., *Leptospermum laevigatum* and *Monotoca elliptica* (Appendix A2). This could affect the relative detectability of birds between these habitats (Dawson 1981), with obvious consequences for habitat comparisons. To counter this, the size of the survey area per point and length of survey time was restricted to that in which I could be confident that all birds had been detected in the denser habitat (*C. monilifera*), as determined in a pilot study (section 6.2.2).

7.2.3 Experimental design

I assumed that the areas invaded by *C. monilifera* had once been very similar to the areas of native vegetation with which they were being compared in this study. It is, of course, impossible to know this with certainty and the results of the analyses should be interpreted accordingly. I made considerable effort, though, to find areas similar to each other within locations and to place all sites on stretches of sandy coastline, as opposed to rocky coasts. Nevertheless, it is possible that sites within locations once differed slightly in vegetation and bird composition and have different histories, especially in relation to disturbances or proximity to disturbances that may, independently, have facilitated invasion by *C. monilifera.*

Constraining the design of the experiment was the availability of reasonably sized patches of native coastal vegetation without *C. monilifera* (NSW NPWS 2001). For example, at ENP, only one patch of native vegetation could be found in close proximity to dense *C. monilifera* stands on a similar substrate. Large areas, however, were in an intermediate phase of invasion, with a component of *C. monilifera,* but less than the 30% cover used to select *C. monilifera* sites. Consequently, multiple *C. monilifera* sites were selected at this location, although with only a single native vegetation site. At ML, three *C. monilifera* and three native sites were established, each independent and separated by sections of coast with the other habitat, an intermediate stage of invasion or where *C. monilifera* control works had been undertaken (as per Toth *et al.* 1996). At PB, the small size of the reserve and its isolation meant that only a single independent *C. monilifera* and native site could be established.

The sites were sampled on multiple occasions over approximately 12 months: June 1996 to July 1997 at ML, June 1996 to May 1997 at ENP and September 1995 to June 1996 at PB.

7.2.4 Statistical analysis

Average bird abundance and species richness per point count, and cumulative species richness, were the parameters analysed statistically using univariate techniques. The unit of analysis was (with the exception of cumulative species richness) the mean per point count, because points were deemed too close for statistical independence and the same points were re-sampled. Consequently, all survey points and all surveys within the period covered by the analysis were averaged. Cumulative species richness was the total number of species recorded among all point counts for the site. In a two factor ANOVA the factors were location (fixed, 3 levels – ML, PB and ENP) and habitat (fixed, 2 levels – *C. monilifera*-invaded and native). SNK tests were used in multiple comparisons where significant interaction effects or location effects were detected.

To examine any impact of *C. monilifera* on birds in more detail, bird species were split into groups of similar broad food resource requirements, based on the study of bird diets at the locations and habitats in question (Chapters 3 and 4). The rationale for this is that bird species that use different resources may be differentially affected by *C. monilifera* invasion. The groups were:

(i) frugivorous birds that consume substantial quantities of fruits and/or seeds in coastal vegetation, but not the fruits of *C. monilifera*;

(ii) frugivorous birds that consume *C. monilifera* fruits (from Dodkin and Gilmore 1984; Gosper 1999a; Chapter 4). I expected that, between these groups of frugivores, those species that use *C. monilifera* fruits would be more able to inhabit invaded vegetation;

(iii) nectarivorous species that feed at the inflorescences of *B. integrifolia*;

(iv) nectarivorous species that feed at the inflorescences of *Eucalyptus* spp. Both these groups of nectarivorous birds are expected to have reduced food resources in invaded vegetation, but were assessed separately due to somewhat different resource requirements;
(v) primarily insectivorous species that catch their food on the wing above the canopy of the vegetation;
(vi) primarily insectivorous species of the understorey and canopy. Insectivorous species were divided into two groups based on their foraging substrate and behaviour;
(vii) birds that prey on vertebrates (carnivores) and/or scavenge.

Using groupings of species for analysis of collective response patterns to experimental factors is fraught with problems. These include the generality of resource use by many species in coastal habitats (Gosper 1999a), lack of knowledge of resource use of others, and the most abundant species within a group determining the response of the group as a whole (French and Zubovic 1997). Given the low sample sizes for many individual species, however, grouping was an essential means to proceed with examining differential impacts of the study factors. Unlike most conventional strategies for grouping species into ‘assemblages’ or ‘guilds’, membership of the groups I defined was not exclusive. That is, any species could be a member of more than one group. I choose this approach because of the frequency that bird diets changed between seasons (Chapter 3), *i.e.* a species may mainly take nectar resources over one period of the year but fruits over another, or their diet always appears to be very broad (*e.g.* Lewin’s Honeyeaters (*Meliphaga lewinii*) and Silvereyes (*Zosterops lateralis*); Gosper 1999a). This does open the possibility of abundant single species or small numbers of species being responsible for a similar pattern of response in two or more groups. The only alternative, however, was what I considered an arbitrary classification of species into exclusive groups with the problems in interpretation that that brings. The composition of the groups used in this study is shown in Appendix A4.

Mean abundance of birds per point count per month was examined graphically prior to analysis. From this, it was apparent that there was substantially greater abundance of birds over most autumn and winter months in several sites of native vegetation. This period approximately matched that of peak flowering (April to September) of the two plant species used by the most species of nectar feeding birds in coastal vegetation, *B. integrifolia* and *E. robusta* (Chapter 3). Consequently, it was thought possible that the impact of *C. monilifera* invasion could be different over this period compared to spring-summer (October to March), when nectar resources appear to be less important for birds in coastal vegetation. The survey data were then split into these two periods for *post-hoc* analyses of the main variables (abundance, species richness and cumulative species richness).

Multivariate procedures (using the *PATN* statistical package; Belbin 1995) were used to explore changes in the bird community in more detail. Mean abundance of each species per point count per site and presence/absence data were analysed separately. The Kulczynski and Bray-Curtis similarity measures were used for abundance and presence/absence data respectively in semi-strong hybrid multidimensional scaling ordination (three-dimensional). Principal axis correlation was used to fit vectors of species and environmental attributes in the ordination space. The environmental attributes used as confirmatory variables in this process were a range of floristic and structural variables measured at all survey points (see Appendix A2 for full description) and two surrogates for location (latitude and longitude). The flexible UPGMA clustering strategy (beta-value = –0.1) was used in agglomerative hierarchical fusion for the generation of a dendrogram of association of sites. Analyses of similarity (ANOSIMs) were conducted on each factor as in the ANOVAs above, without testing of interactions. The ability of species to act as 'discriminators' between location and habitat groups was examined using the Simper module in *PATN* and the Kruskal-Wallis non-parametric test. The latter should be viewed with caution, as the reliability of the probability value may be lower by having few sites in some groups (Belbin 1995).

7.3 Results

Eighty-eight bird species were recorded at survey points during this study. Many occurred both in sites dominated by *C. monilifera* and native plants. Most of those recorded from only one habitat were recorded infrequently and in low abundance. For these species, there was little indication that their occurrence in only one habitat was due to anything other than chance. A single threatened species under the *Threatened Species Conservation Act* (NSW) (NSW NPWS 2003) and Garnett and Crowley (2000) was recorded. The Endangered Swift Parrot occurred at the native vegetation site at PB during and after this study, and nearby to native vegetation sites at ML (in vegetation free of *C. monilifera*). Additionally, the near-threatened (Garnett and Crowley 2000) Forest Raven (New England subspecies; *Corvus tasmanicus boreus*) was recorded throughout the ML location, although rarely at survey points.

7.3.1 Total and relative abundance

Mean bird abundance per survey point was greater in native vegetation than that dominated by *C. monilifera* (Table 7.1; Figure 7.1). Bird abundance was unaffected by location or the interaction of location and habitat.

The relative abundance of bird species within the bird community showed strong evidence of differences between habitats. In the cluster analysis (Figure 7.2), five of the six sites invaded by *C. monilifera* were within one group of the classification; four of the five native vegetation sites within another group; with the final group being made up of the two sites within the PB location. This dendrogram indicates that for the most geographically separate locations (ML and ENP), the differences between the habitats within locations appear to be driving the clustering of sites. Only for PB were similarities between sites within the same location greater than similarities within habitats. This pattern is also apparent in the ordination (Figure 7.3), with a cluster of native vegetation sites, a cluster of sites invaded by *C. monilifera*, and the two sites at PB separate from these clusters and each other. The ANOSIM of the abundance of bird species also show significant location and habitat effects (Table 7.2).

Table 7.1 - ANOVAs assessing the effects of location and *C. monilifera* on the bird community using all survey data

Source of variation	df	MS abun	F abun	MS SpR	F SpR	MS cSpR	F cSpR
Location	2	0.668	4.136	0.070	1.160	13.162	2.015
Habitat	1	1.163	7.197*	0.258	4.287	72.107	11.037*
Location x habitat	2	0.736	4.557	0.111	1.842	42.662	6.530*
Error	5	0.162		0.060		6.533	

Source of variation	df	MS carn	F carn	MS ins	F ins	MS aer	F aer
Location	2	0.000	0.448	0.369	10.30*	0.019	1.844
Habitat	1	0.000	0.323	0.677	18.9**	0.012	1.168
Location x habitat	2	0.001	2.670	0.025	0.697	0.002	0.202
Error	5	0.001		0.036		0.010	

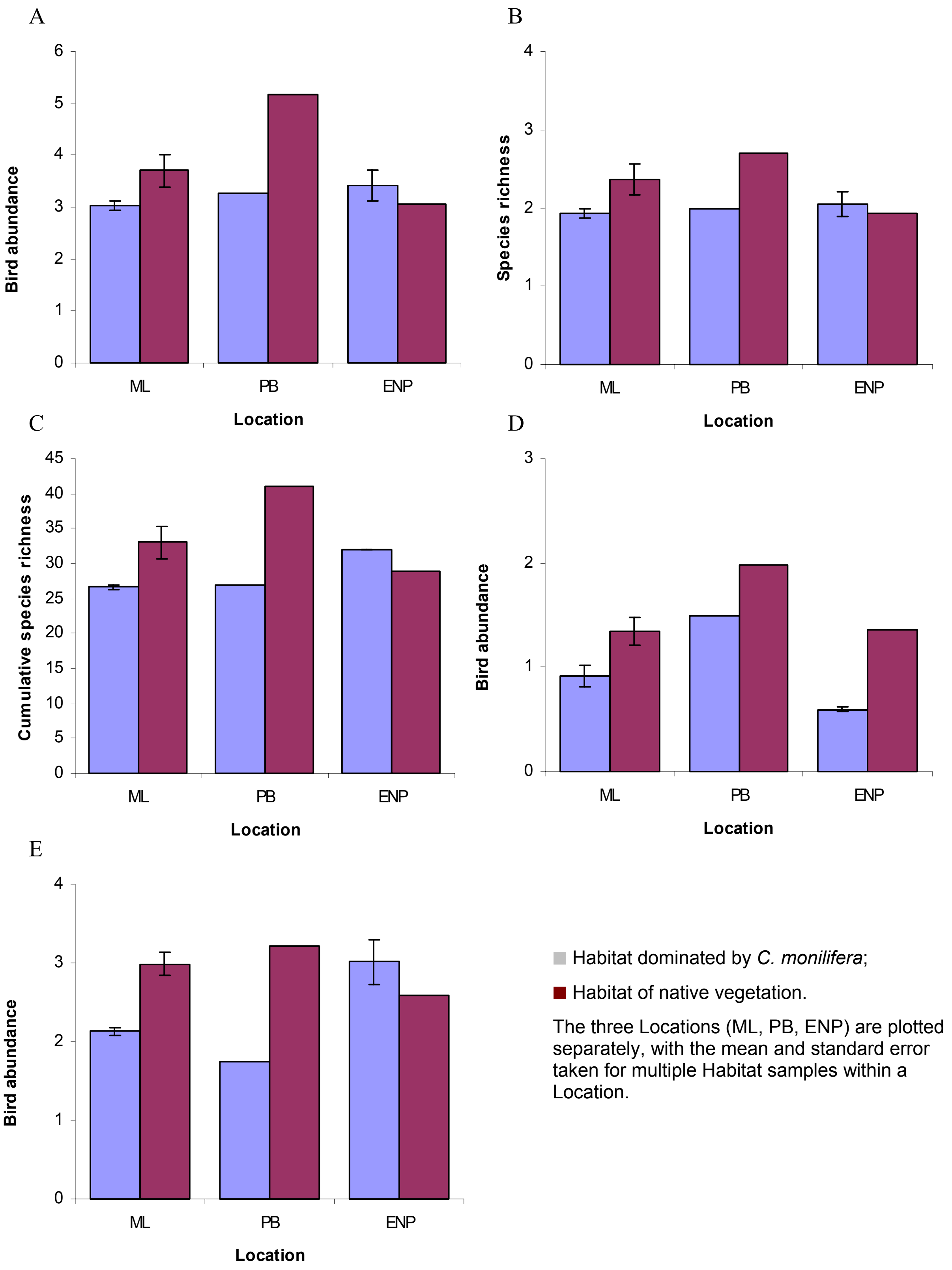
Source of variation	df	MS frug	F frug	MS BBfr	F BBfr	MS B_N	F B_N	MS E_N	F E_N
Location	2	0.012	41.83**	0.079	2.272	0.075	1.274	0.090	2.626
Habitat	1	0.002	8.660*	0.164	4.717	0.872	14.905*	0.161	4.671
Location x habitat	2	0.003	9.707*	0.182	5.259	0.584	9.982*	0.847	24.62**
Error	5	0.000		0.035		0.059		0.034	

* P<0.05; **P<0.01; ***P<0.001. Key: habitat = vegetation dominated by *C. monilifera vs* native vegetation. abun = mean abundance of birds per survey point; SpR = mean species richness of birds per survey point; cSpR = cumulative species richness per site. Mean abundance of groups of bird species per survey point: aer = aerial insectivores; carn = carnivores/scavengers; frug = frugivorous birds (excluding those that eat *C. monilifera* fruits); BBfr = frugivores of *C. monilifera*; ins = canopy and understorey insectivores; B_N = nectar-feeders at *B. integrifolia*; E_N = nectar-feeders at *Eucalyptus* spp.

Table 7.2 - F-ratios in ANOSIMs testing the effects of location and *C. monilifera* on bird community composition using all survey data

Source of variation	Abundance data	Presence/absence data
Location	1.559***	1.835***
Habitat	1.176*	0.970

* P<0.05; **P<0.01; ***P<0.001



▨ Habitat dominated by *C. monilifera*;

■ Habitat of native vegetation.

The three Locations (ML, PB, ENP) are plotted separately, with the mean and standard error taken for multiple Habitat samples within a Location.

Figure 7.1 – Abundance and species richness of birds in *C. monilifera* and native vegetation

(A) Mean bird abundance per point count; (B) Mean species richness per point count; (C) Cumulative species richness; (D) Mean abundance of canopy and understorey insectivores per point count; and (E) Mean abundance of nectarivores of *B. integrifolia* per point count.

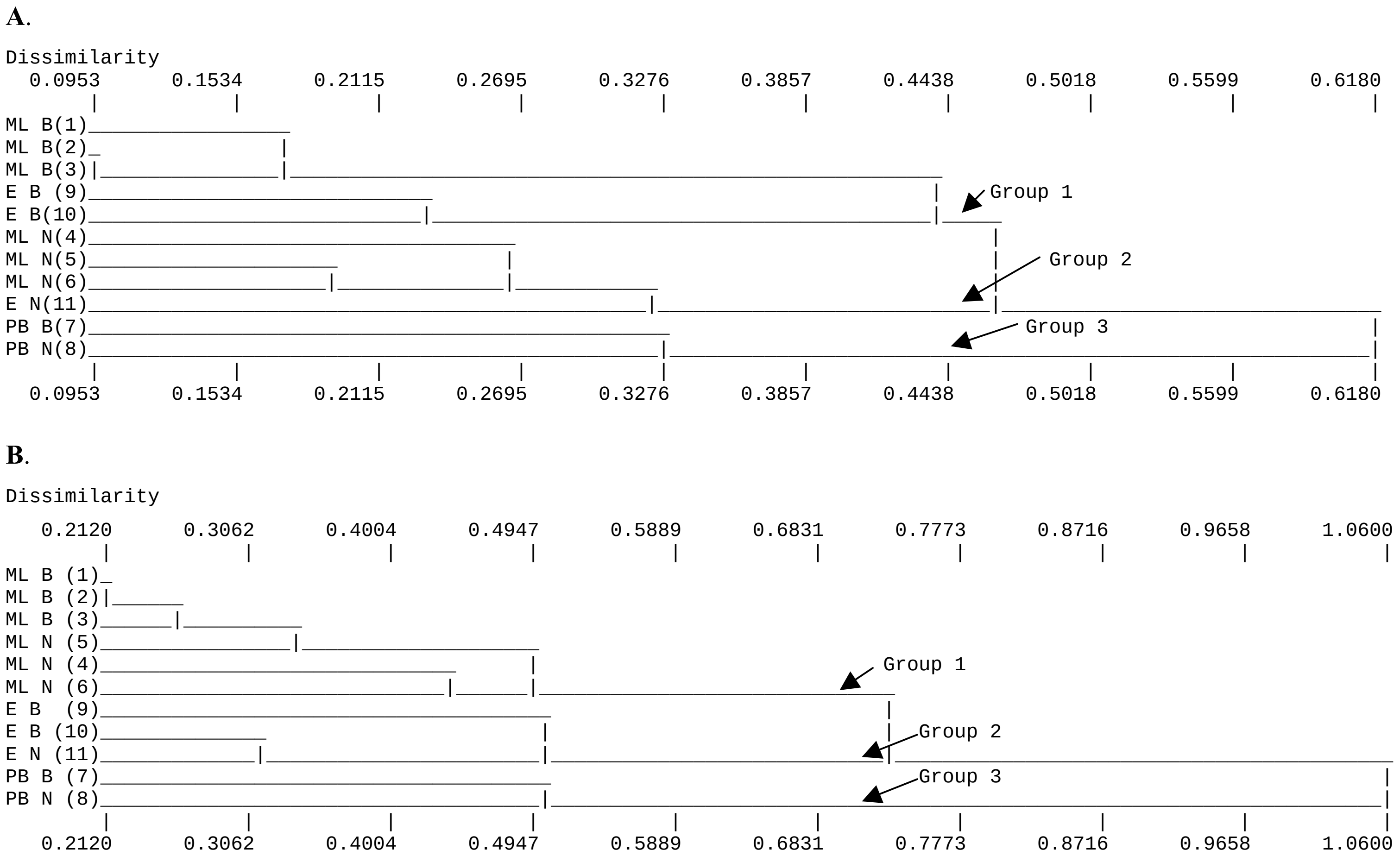


Figure 7.2 - Dendrograms of association of sites by bird community composition: using abundance (A) and presence/absence (B) data

Key for site names – first term for location: ML – Myall Lakes; PB – Perkins Beach; E – Eurobodalla NP. Second term for habitat: B – *C. monilifera*; N – Native

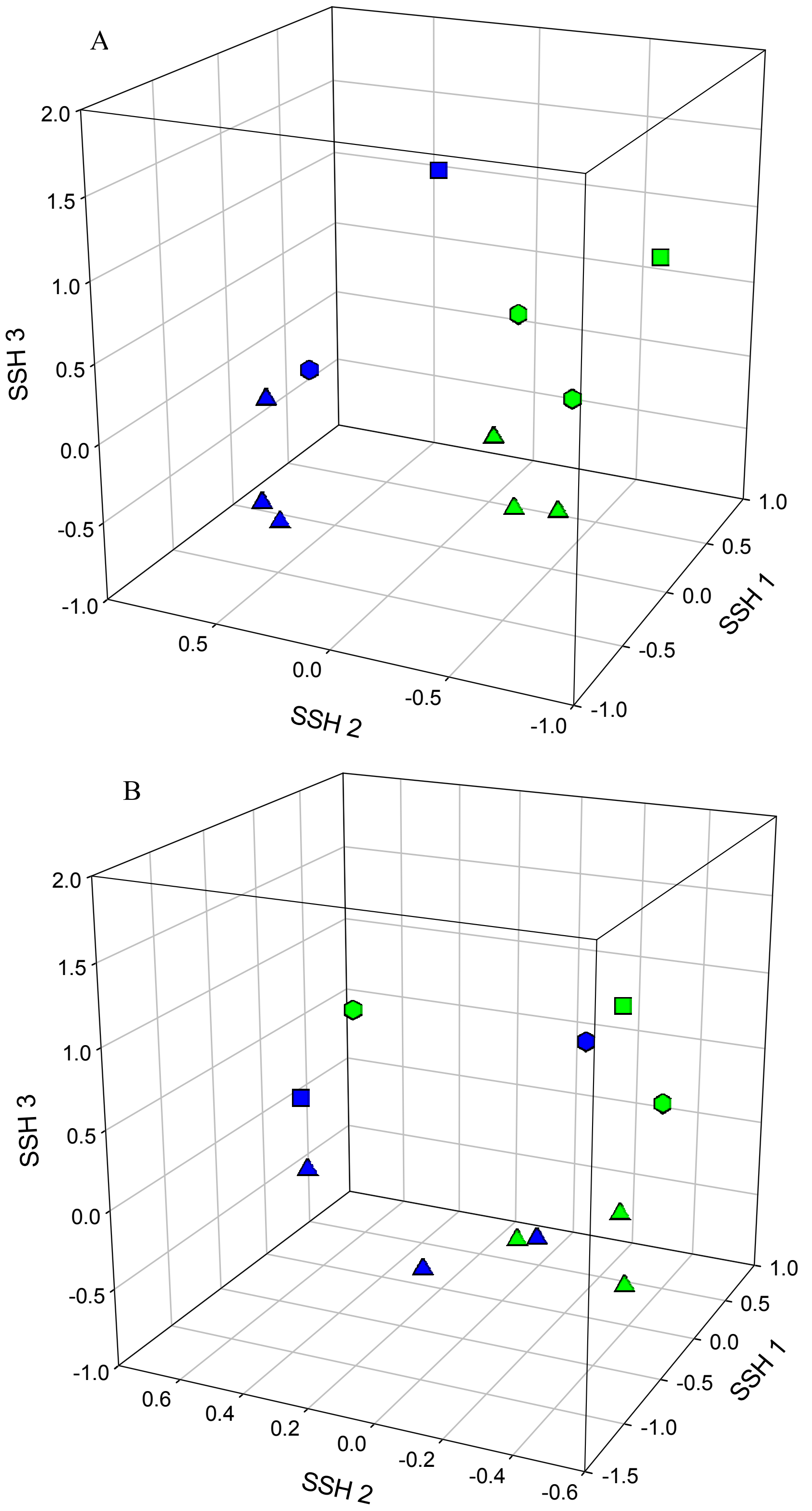


Figure 7.3 - Ordinations of survey sites by bird composition: using bird abundance (A) and presence/absence (B) data

▲ Myall Lakes, *C. monilifera*; ▲ Myall Lakes, native; ■ Perkins Beach, *C. monilifera*; ■ Perkins Beach, native; ● Eurobodalla NP, *C. monilifera*; ● Eurobodalla NP, native.

7.3.2 Abundance of species groups and individual species

The groups of birds defined on resource-use characteristics exhibited different responses to the main study factors. The different habitats, locations, or their interaction did not affect the abundance of several groups: aerially feeding insectivores, carnivores/scavengers and frugivores of *C. monilifera* (Table 7.1). Both aerial insectivores and carnivores were recorded in low densities, however, which may have constrained the identification of differences in abundance. For frugivores/granivores (species other than those that use *C. monilifera* fruits) and nectar-feeders (both groups: of *B. integrifolia* and *Eucalyptus* spp.), a significant location and habitat interaction effect was detected (Table 7.1; Figure 7.1). This made the assessment of main effects inconclusive. More *B. integrifolia* nectar-feeders were recorded in native vegetation at the PB and ML locations, but there was no significant difference in their abundance between habitats at ENP. Nectarivores of *Eucalyptus* spp. were more abundant in native vegetation at PB, *C. monilifera* vegetation at ENP, and not different between the habitats at ML. There was no difference in the abundance of frugivores/granivores (other than species that use *C. monilifera*) at two locations (ML and ENP): only at PB were more found in native vegetation. Both location and habitat significantly affected canopy and understorey insectivore abundance. More insectivores were recorded from native vegetation than that dominated by *C. monilifera* (Table 7.1; Figure 7.1).

In principal axis correlation, the optimal location and orientation of vectors representing individual bird species are placed relative to the same axes as the ordination (Figure 7.4 *c.f.* Figure 7.3). This indicates the bird species contributing substantially to the separation of sites along these axes. The bird species plotted are those with the greatest correlation coefficient (all r>0.9, which is relatively high). Eastern Spinebill (*Acanthorhynchus tenuirostris*) and Brown Thornbill (*Acanthiza pusilla*) are both strongly orientated towards native vegetation sites (except at PB) (group 2 in the cluster analysis, Figure 7.2), indicating that these species appear to have a strong preference for native vegetation over that invaded by *C. monilifera*. In contrast, Silvereye and Yellow-faced Honeyeater (*Lichenostomus chrysops*) are orientated in the opposite direction, showing a strong preference for the sites dominated by *C. monilifera* (group 1). Some of the remaining species on the plot appear to be orientated mainly towards locations, with Grey Fantail (*Rhipidura fuliginosa*) and Red-whiskered Bulbul (*Pycnonotus jocosus*) towards PB (group 3) and White-cheeked (*Phylidonyris nigra*) and Lewin's Honeyeater to ML.

The SIMPER module was used to identify the individual species that contributed most to the dissimilarity among locations and among habitats. Much the same selection of species appear to be highly significant in both cases: these being among the most abundant species overall (Table

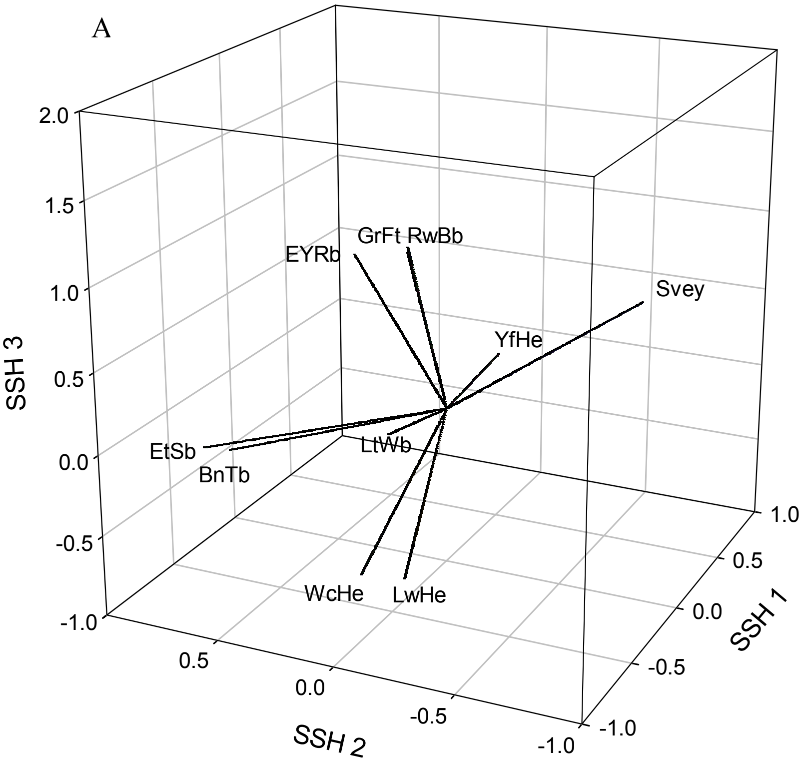
7.3). Changes in the abundance of these highly abundant species appear to be driving the differences between both locations and habitats and explain a large proportion of the dissimilarity (the 15 species most dissimilar between locations and between habitats contribute approximately 85% of the total dissimilarity in each case). Species that strongly differed in mean density between the habitats included Brown Thornbill, Variegated Fairy-wren (*Malurus lamberti*), Eastern Spinebill, Spotted Pardalote (*Pardalotus punctatus*) (all twice as abundant in native vegetation) and Red Wattlebird (*Anthohaera carunculata*) (twice as abundant in *C. monilifera*). Other species also appeared to prefer one or other of the habitats (Table 7.3).

In Kruskal-Wallis tests, Collared Sparrowhawk (*Accipiter cirrhocephalus*), Silvereye (both more abundant in *C. monilifera* vegetation), Eastern Spinebill, Variegated Fairy-wren and Golden Whistler (*Pachycephala pectoralis*) (more abundant in native vegetation) were good discriminators between the habitats ($P<0.05$). Between locations, Lewin's Honeyeater, White-cheeked Honeyeater, Eastern Whipbird (*Psophodes olivaceus*) (common only at Myall Lakes), Yellow Thornbill (*Acanthiza nana*), Red-whiskered Bulbul (only at Perkins Beach), Yellow-faced Honeyeater and Crimson Rosella (*Platycercus elegans*) (most common at ENP) were good discriminating species ($P<0.05$). This test primarily identified species that only occurred in one group of the test (native *vs.* invaded, or at only one location).

7.3.3 Species richness and presence/absence within the bird community

Mean species richness per survey point was not affected by location, habitat, or their interaction (Table 7.1; Figure 7.1). There was, however, an interaction effect on the cumulative species richness at sites. More species were recorded in total in native vegetation at ML and PB, with the reverse at ENP.

Distinct differences were apparent in the clustering of survey sites and in their relationship in ordination in the bird community data between analyses using abundance and presence/absence data. Presence/absence analysis indicated substantial location effects but no effects of habitat. In the cluster analysis (Figure 7.2), sites are divided into groups that correspond perfectly to their location. Even within locations, sites of the same habitat were not always more similar than to sites of the other habitat. In this case, differences between locations appear to be driving the classification. The ordination is less clear, because not all sites from the same location are nearby in ordination space (particularly PB), although there is also no indication at all of a habitat effect (Figure 7.3). The ANOSIM results support the ordination and cluster analysis, with a significant effect of location on the species composition of bird community, but no effect of habitat (Table 7.2).



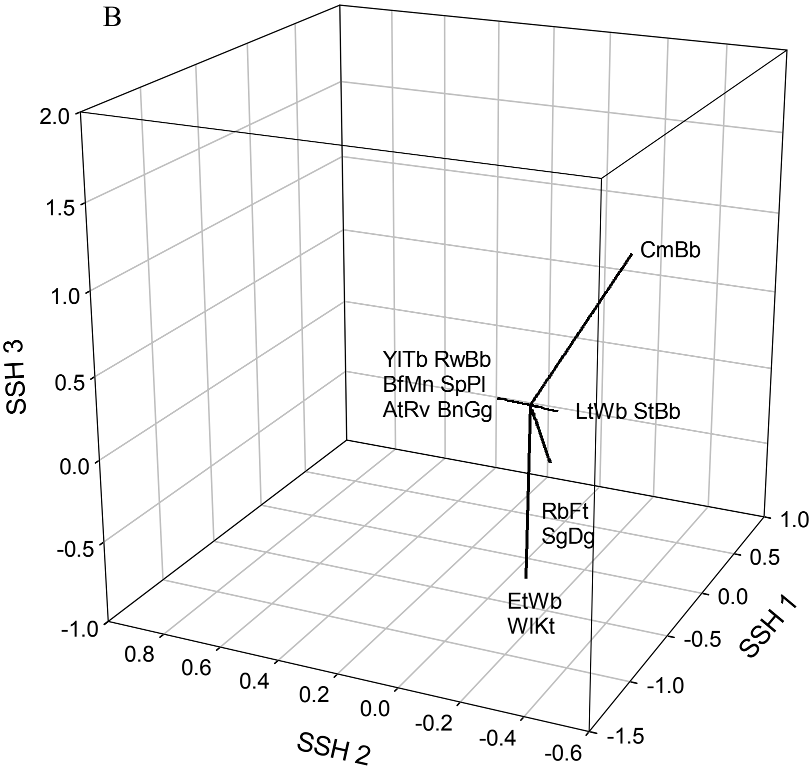


Figure 7.4 - Vectors of species imposed on the ordination of survey sites by bird composition: using bird abundance (A) and presence/absence (B) data

Vectors plotted within the ordinations of Figure 7.3 (A and B respectively), for the species with the ten highest correlation coefficients (r>0.90 for A, 0.91 for B). Key: EtSb = Eastern Spinebill; BnTb = Brown Thornbill; WcHe = White-cheeked Honeyeater; LwHe = Lewin's Honeyeater; LtWb = Little Wattlebird; Svey = Silvereye; YfHe = Yellow-faced Honeyeater; RwBb = Red-whiskered Bulbul; GrFt = Grey Fantail; EYRb = Eastern Yellow Robin; YlTb = Yellow Thornbill; BfMn = Black-faced Monarch; SpPl = Spotted Pardalote; AtRv = Australian Raven; BnGg = Brown Gerygone; EtWb = Eastern Whipbird; WlKt = Whistling Kite; RbFt = Red-browed Finch; SgDg = Spangled Drongo; StBb = Satin Bowerbird; CmBb = Common Blackbird. Scientific names in Appendix A4.

Table 7.3 – Bird species contributing most to dissimilarity between locations and habitats (*C. monilifera* and native vegetation)

Table 7.3A. Using abundance data.

Locations								**Habitats** (*C. monilifera* and native vegetation)						
Species	**ML**[1]	**PB**	**ENP**	**Av. dis**	**Dis/SD**	**Cont. %**	**Cum. %**	**Species**	***C.mon***[1]	**Nat**	**Av. dis**	**Dis/SD**	**Cont. %**	**Cum. %**
New Holland Honeyeater	0.478	0.891	0.411	2.09	49.84	12.50	12.50	Silvereye	0.987$^{\#}$	0.659	1.80	54.11	13.66	13.66
Silvereye	0.696	1.216	0.869	1.93	56.99	11.56	24.06	New Holland Honeyeater	0.403	0.692$^{\#}$	1.75	46.77	13.26	26.92
Brown Thornbill	0.352	0.005	0.334	1.47	57.10	8.79	32.85	Brown Thornbill	0.162	0.430$^{\#\#}$	1.45	54.83	11.01	37.92
Little Wattlebird	0.287	0	0.452	1.38	57.94	8.27	41.13	Little Wattlebird	0.225	0.345$^{\#}$	1.09	52.56	8.31	46.24
Yellow-faced Honeyeater	0.035	0.020	0.301	0.99	55.93	5.94	47.07	Variegated Fairy-wren	0.145	0.342$^{\#\#}$	0.87	51.39	6.58	52.82
White-browed Scrubwren	0.278	0.389	0.178	0.78	57.48	4.68	51.75	White-browed Scrubwren	0.294$^{\#}$	0.242	0.64	48.03	4.83	57.64
White-cheeked Honeyeater	0.178	0	0.001	0.77	58.85	4.64	56.39	Yellow-faced Honeyeater	0.113	0.096	0.53	40.22	4.03	61.67
Variegated Fairy-wren	0.240	0.296	0.182	0.73	56.12	4.37	60.76	Welcome Swallow	0.143$^{\#}$	0.095	0.51	49.89	3.83	65.51
Lewin's Honeyeater	0.161	0	0.001	0.70	55.31	4.17	64.93	White-cheeked Honeyeater	0.081	0.117$^{\#}$	0.48	45.67	3.68	69.19
Yellow Thornbill	0	0.266	0	0.67	40.63	4.00	68.93	Lewin's Honeyeater	0.084	0.093	0.45	45.57	3.45	72.64
Grey Fantail	0.015	0.266	0.025	0.62	47.22	3.68	72.61	Eastern Spinebill	0.026	0.116$^{\#\#}$	0.41	53.80	3.14	75.78
Welcome Swallow	0.172	0.096	0.038	0.61	54.12	3.67	76.28	Grey Fantail	0.048	0.083$^{\#}$	0.34	34.98	2.64	78.42
Superb Fairy-wren	0.153	0.159	0.067	0.41	55.43	2.45	78.73	Yellow Thornbill	0.057$^{\#}$	0.038	0.33	29.12	2.53	80.95
Red Wattlebird	0.001	0.018	0.110	0.39	33.50	2.32	81.05	Superb Fairy-wren	0.122	0.141	0.31	47.44	2.33	83.28
Eastern Spinebill	0.093	0.013	0.052	0.34	52.03	2.04	83.07	Red Wattlebird	0.051$^{\#\#}$	0.014	0.24	27.38	1.80	85.08
Spotted Pardalote	0	0.120	0	0.25	30.31	1.52	84.61	Spotted Pardalote	0.001	0.041$^{\#\#}$	0.17	24.48	1.30	86.37

[1] ML/PB/ENP/*C. mon*/Nat is the mean abundance of that species per that location or habitat. Av. dis = Average dissimilarity; Dis/SD = Av. Diss/standard deviation; Cont.% = Contribution to explaining overall dissimilarity between locations/habitats; Cum. % = Cumulative contribution to explaining overall dissimilarity. Only the 15 most highly ranked species are listed. For the habitats section of the Table: $^{\#\#}$ = species more than twice as abundant in that habitat (>100% difference); $^{\#}$ = species 20-100% more abundant in that habitat; unmarked species had differences in density between habitats of <20%. Scientific names for bird species in Appendix A4.

Table 7.3B. Using presence/absence data.

Locations Species	ML[1]	PB	ENP	Av. dis	Dis/SD	Cont. %	Cum. %
Whistling Kite	0.012	0	0	0.73	60.71	2.79	2.79
Eastern Whipbird	0.048	0	0	0.73	60.71	2.79	5.58
Common Blackbird	0	0.002	0.005	0.67	56.02	2.55	8.13
Lewin's Honeyeater	0.161	0	0.001	0.63	54.23	2.40	10.54
White-cheeked Honeyeater	0.178	0	0.001	0.63	54.23	2.40	12.94
Pied Currawong	0.017	0	0.024	0.63	54.23	2.40	15.35
White-naped Honeyeater	0	0.013	0.003	0.62	54.29	2.37	17.72
Mistletoebird	0.016	0	0.001	0.62	54.29	2.37	20.09
Bar-shouldered Dove	0.013	0	0	0.60	52.79	2.29	22.39
Crimson Rosella	0	0	0.057	0.58	51.47	2.23	24.61
Rufous Whistler	0.001	0.007	0.008	0.57	49.72	2.17	26.78
Grey Shrike-thrush	0.002	0	0.007	0.53	48.28	2.03	28.81
Red Wattlebird	0.001	0.018	0.110	0.51	46.79	1.95	30.76
White-throated Needletail	0.003	0	0.014	0.51	46.66	1.94	32.70
Brown Goshawk	0.001	0.014	0.001	0.50	46.61	1.92	34.62
White-bellied Sea-eagle	0.003	0	0.003	0.50	46.63	1.90	36.52

Species discriminating between habitats are not tabulated, as it was not a significant effect in ANOSIM (Table 7.2). Key as for Table 7.3A.

7.3.4 Presence/absence of individual species

The principal axis correlation of species with the highest correlation coefficients (r>0.91) was dominated by species that were recorded infrequently, although from most or all sites within a single location (Figure 7.4). For example, Eastern Whipbird and Whistling Kite (*Haliastur sphenurus*) were recorded at all sites at ML but not elsewhere, and Yellow Thornbill, Red-whiskered Bulbul, Black-faced Monarch (*Monarcha melanopsis*) and Spotted Pardalote (*Pardalotus punctatus*) similarly at PB. Little Wattlebird (*Anthochaera chrysoptera*) and Satin Bowerbird (*Ptilonorhynchus violaceus*) were orientated away from PB sites, as they were recorded at all or most sites other than those at this location. The species identified in the SIMPER analysis as contributing most to the dissimilarity between locations likewise contained a number of species infrequently-recorded, but uniquely at a single location or completely absent from one location (Table 7.3). Several of the same species as in principle axis correlation have been identified as explaining most of the dissimilarity between locations, although fewer of the species uniquely occurring at PB were among the highest 15. The higher ranked species contributed less of the total dissimilarity in this case (~37%) than when dissimilarity was calculated using abundance data.

7.3.5 Environmental attributes

A selection of environmental attributes was used as confirmatory variables in principal axis correlation to identify which were most strongly correlated with the ordination of sites. As with other analyses, substantial differences existed between ordinations using bird abundance and presence/absence data (Figure 7.5). In the ordination based on abundances, environmental attributes associated with both location and habitat effects had high correlation coefficients (r>0.85). The highest correlation coefficients were for two variables acting as surrogates for location effects: Latitude and Longitude. Next most highly correlated were a range of variables associated with differences between the habitats: cover of survey points by *C. monilifera*, *B. integrifolia*, tree and shrubs >3m; shrubs < 3m; and plant species richness. In the ordination based on presence/absence data, variables representing location effects were most highly correlated: Longitude, Latitude, and vegetation variables distinct between locations (such as cover of *Eucalyptus* spp. and other weeds, which were only widespread at PB; and *Hibbertia scandens* and twiners, which only occurred at ML) (r>0.76).

7.3.6 Time of year

Within the two periods of the year analysed separately, differences in bird abundance between the two habitats were only apparent over the April-September period (hereafter referred to as autumn-winter). The mean abundance of birds per survey point over this period was less at *C. monilifera* sites. There were no differences between habitats in the abundance of birds over the October-March (hereafter referred to as spring-summer) period (Table 7.4; Figure 7.6).

No differences between locations, habitats, or their interaction were detected for species richness at survey points in either period of the year. However, *C. monilifera* habitat had lower cumulative species richness than native vegetation over autumn-winter. In spring-summer, there was a significant interaction of habitat with location, which was the case when all survey data were combined (Table 7.4; Figure 7.6).

Table 7.4 - ANOVAs assessing the effects of location and *C. monilifera* on the bird community over two periods: autumn-winter (A) and spring-summer (B)

Source of variation	df	MS abun	F abun	MS SpR	F SpR	MS cSpR	F cSpR
Autumn-winter period							
Location	2	1.712	4.422	0.079	0.725	32.412	4.675
Habitat	1	2.386	7.557*	0.391	3.592	64.027	9.235*
Location x habitat	2	1.695	5.368	0.182	1.669	17.412	2.511
Error	5	0.316		0.109		6.933	
Spring-summer period							
Location	2	0.510	2.243	0.108	1.195	14.528	3.528
Habitat	1	0.631	2.777	0.190	2.114	16.667	4.032
Location x habitat	2	0.353	1.551	0.079	0.878	82.646	20.00**
Error	5	0.227		0.090		4.133	

Key as for Table 7.1

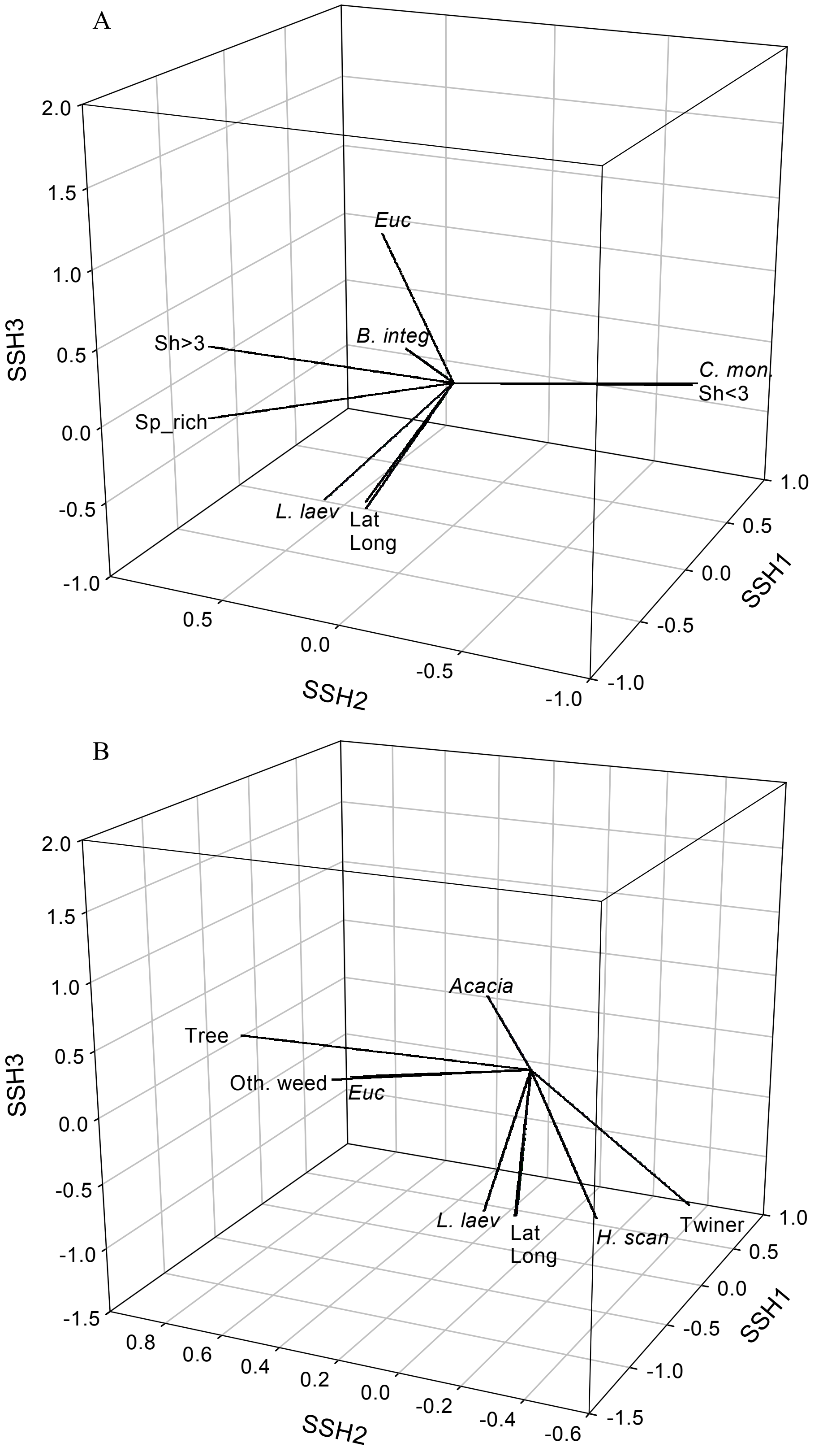


Figure 7.5 - Vectors of environmental attributes imposed on the ordination of survey sites by bird composition: using bird abundance (A) and presence/absence (B) data

Vectors plotted within the ordinations of Figure 7.3 (A and B respectively), for the environmental attributes with the ten highest correlation coefficients (r>0.85 for A, 0.76 for B). Key: Lat = latitude; Long = longitude; Sp_rich = Species richness of plants; Vegetation cover of: *C. mon = Chrysanthemoides monilifera*; Sh<3 = shrubs <3m; *Euc = Eucalyptus* spp.; *B. integ = Banksia integrifolia*; Sh>3 = shrubs and trees >3m; *L. laev = Leptospermum laevigatum*; *Acacia = Acacia* spp.; Tree = trees >5m; Oth. weed = weeds other than *C. monilifera*; *H. scan = Hibbertia scandens*; Twiner = ground twiners.

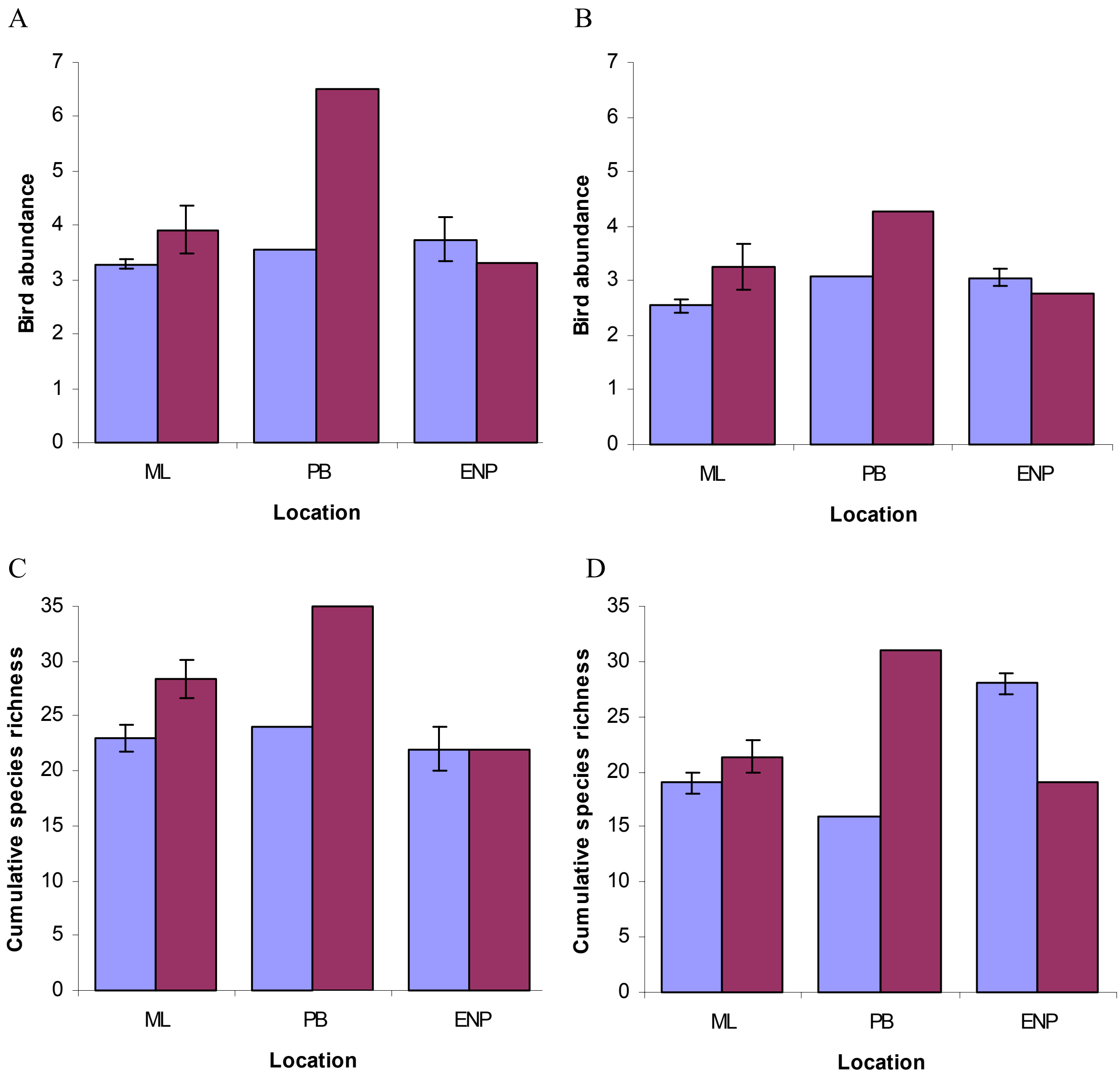


Figure 7.6 – Abundance and cumulative species richness of birds over the two analysis periods: autumn-winter and spring-summer

Mean abundance of birds per point count: (A) autumn-winter and (B) spring-summer. Cumulative species richness of birds: (C) autumn-winter and (D) spring-summer.

■ Habitat dominated by *C. monilifera*
■ Habitat of native vegetation

The three locations (ML, PB, ENP) are plotted separately, with the mean and standard error taken for multiple habitat samples within a location. All survey data was broken up into two periods: autumn-winter and spring-summer.

7.4 Discussion

7.4.1 Effects of habitat

Habitat (native or dominated by invasive *C. monilifera*) significantly affected a number of measures of the bird community. A number of previous studies have similarly detected changes in bird community composition and/or abundance attributed to the effects of invasive plants (Braithwaite *et al.* 1989; Giffin *et al.* 1989; Wilson and Belcher 1989; French and Zubovic 1997). In each of these studies, including mine, the impact of the invasive plant(s) in changing the bird community is inferred from comparisons of nearby sites that are assumed to have had similar bird communities prior to changes induced by the invasive plant(s). This assumption is, of course, impossible to test after invasion has taken place. However, given the number of similarly conducted studies that have mostly demonstrated differences between habitats (with vegetation dominated by the invasive plant providing less suitable habitat), it is extremely unlikely that all of these would be due to chance sampling effects.

Total abundance and relative abundance within the bird community

Habitat type affected the overall abundance of birds and the relative abundance of species, across the three locations sampled. More birds overall were found in native vegetation than in *C. monilifera* stands. In terms of community composition, the two habitats were significantly different, and substantially more different than were the locations. In contrast, French and Zubovic (1997) found no difference in the overall abundance of birds between native and *C. monilifera* vegetation. Suggestions to explain why these studies differ are discussed below.

Abundance of species groups and individual species

Once an impact of plant invasion has been established, it is important to investigate what components of the bird community are contributing most to this overall difference. Additionally, if the components of the bird community most affected by *C. monilifera* could be explained readily through measured differences in the habitats associated with *C. monilifera* invasion, this lends support to the conclusion that these differences between habitats are real and not chance sampling effects. Changes in the bird community were investigated through the abundance of species groups, individual species and by multivariate techniques.

Canopy and understorey insectivores comprised the group most substantially affected by *C. monilifera* invasion, with markedly lower abundance in *C. monilifera* than native vegetation. Species in this group that explain much of the dissimilarity between habitats are the Brown Thornbill, White-browed Scrubwren (*Sericornis frontalis*) and Variegated Fairy-wren. The abundance of Brown Thornbills and Variegated Fairy-wrens at native vegetation sites was more than double that at invaded sites. White-browed Scrubwrens, in contrast, were slightly more abundant in *C. monilifera* vegetation. This contrasts the assertion (apparently not based on any field data) made by Dodkin and Gilmore (1984) that White-browed Scrubwrens (and other ground-feeding insectivores) rarely occur in stands of *C. monilifera.* These three species exhibit one of the problems in analysing species in groups: that changes in one direction in some species may be cancelled out by changes in the opposite direction by others. Overall, however, canopy and understorey insectivores were substantially less abundant in *C. monilifera* vegetation, even if some species were more abundant.

It is unknown whether it is changes in floristics, with possible consequences for the invertebrate foods of these birds, or changes in vegetation structure, that are most important in rendering the habitat less suitable for insectivores. Both possibilities are plausible and may both be important. In regard to the food resources of insectivores, it would be unlikely that these would increase following *C. monilifera* invasion. Some food resources may be lost, however, following the reduction in floristic diversity in invaded habitats (Weiss 1983; NSW NPWS 2001), or the composition of the invertebrate community may change. The substantial nectar resources provided by winter-flowering plants (*e.g. Banksia integrifolia*) attract invertebrates and several of the insectivores, including Brown Thornbills and Variegated Fairy-wrens, feed on these invertebrates around the inflorescences (Gosper 1999a), as well as the nectar itself (Chapter 3).

French and Eardley (1997) found no change in the abundance or species richness of litter invertebrates in *C. monilifera* vegetation compared to native vegetation. However, community composition, specifically the relative abundance of some invertebrate orders, differed between habitats. The subsequent impact of these changes on their avian (and other) predators has not been directly determined, but such species would make up a component of the canopy and understorey insectivores group here. There are, however, many species in this group that would be affected more by changes in the invertebrate community at the foliage level. Impacts of *C. monilifera* invasion on these invertebrates have not been published.

French and Zubovic (1997) considered that insect availability in *C. monilifera* vegetation could be high, as a result of the sometimes-greater abundance of a group of bird species in *C. monilifera* vegetation. I suggest that these contrasting results have arisen from the different methods of classifying species into groups between the two studies. Specifically, the inclusion of some regularly frugivorous species (including of *C. monilifera* fruits) into the group of ‘understorey insectivores’ (such as Silvereyes, Lewin’s Honeyeaters and Yellow-faced Honeyeaters) by French and Zubovic (1997), but not in this study, may have produced this result. In this study, frugivores of *C. monilifera* (of which these species form a significant proportion), were unaffected by habitat differences

Alternatively, changes in vegetation structure could equally explain some of the differences in insectivore habitat preference. Vegetation dominated by *C. monilifera* is typically very dense in the lower shrub layer (0.5-3m off the ground), usually denser than native vegetation (Appendix A2). Native vegetation tends to be more variable in structure, including taller shrub species with a canopy height of 3-5m. Vegetation dominated by *C. monilifera* produces a litter layer of lower cover, lower average temperature, more moist and less light than native vegetation (K. French and E. Lindsay, University of Wollongong, pers. comm.). These changes could explain the divergent responses of these bird species, as White-browed Scrubwrens are associated with more mesic habitats than Brown Thornbills or Variegated Fairy-wrens (Higgins *et al.* 2001; Higgins and Peter 2002). The invertebrate order (Collembola) that increased in abundance in *C. monilifera* vegetation in the study of French and Eardley (1997) also favoured moist litter, further suggesting that *C. monilifera* invasion may render the low shrub and ground layer increasingly mesic.

Other groups of bird species, specifically frugivores (other than those that consume *C. monilifera* fruits) and nectarivores of *B. integrifolia*, may be more abundant in native vegetation than that dominated by *C. monilifera*, although this effect was not conclusive. In these cases, different responses to habitat were observed at different locations. This result is readily explicable for the nectarivores. In *C. monilifera*-invaded sites, the availability of nectar resources for birds depends largely on the cover of large trees and shrubs, such as *B. integrifolia* and *Eucalyptus* spp., that are the most widely used nectar sources for birds (Chapter 3). French and Eardley (1997) suggested that *C. monilifera* flowers provide an important, alternate food source for many animals over winter. Invertebrate use of *C. monilifera* flowers was not assessed in this study, however, few birds fed at flowers, and no visits could be attributed to taking nectar. *Chrysanthemoides monilifera* does not appear to provide significant floral resources used directly by birds. Consequently, the nectar resources for birds may decline following *C. monilifera* invasion, if there is an associated loss of native nectar-producing plants (which is likely; Dodkin and Gilmore 1984; Weiss and Noble 1984a).

Some *C. monilifera* sites in this study had a substantial cover of emergent *B. integrifolia* and *Eucalyptus* spp., which attracted many nectarivorous birds. The *C. monilifera* sites at ENP, for example, had similar cover of *B. integrifolia* to most sites in native vegetation (Appendix A2). This was the only location in which nectarivores of *B. integrifolia* were not more abundant in native vegetation. Similarly, at the two locations where the cover of *Eucalyptus* spp. differed greatly between habitats (PB and ENP), more nectarivores of *Eucalyptus* spp. were recorded in the habitat with greater *Eucalyptus* spp. cover (native at PB, *C. monilifera* at ENP). This indicates that cover of remaining native plant species can substantially influence the bird community, probably to a greater extent than *C. monilifera* cover *per se*. Over time, if existing nectar sources were displaced, *C. monilifera* vegetation would increasingly provide fewer resources for nectarivores and their abundance would be expected to decline, as has probably already happened at some *C. monilifera* sites in this study. Time since invasion, which may be linked to the extent of vegetation change, was a factor suggested by French and Eardley (1997) to explain some of the variability in litter invertebrate communities among *C. monilifera* sites.

The abundance of several of the commonest individual species in the nectarivorous groups contributed much of the dissimilarity between habitats, suggesting that they are showing habitat preference. Eastern Spinebills were substantially more abundant in native vegetation, and New Holland Honeyeaters, Little Wattlebirds and White-cheeked Honeyeaters slightly more abundant. In contrast, Silvereyes were more abundant in *C. monilifera* vegetation.

No habitat (or location) effects on the abundance of species that consume *C. monilifera* fruits were detected. This was interesting, because these species were presumed to be benefiting from *C. monilifera* invasion, specifically in the provision of a substantial new food source. The only conclusion that can be drawn from this result is that *C. monilifera* vegetation is only providing a replacement resource from that in native vegetation, and native vegetation is equally suitable habitat for this group of species overall. Frugivores of *C. monilifera* contain some of the species with the broadest (Gosper 1999a) and probably most flexible (in terms of adopting new food sources) diets in coastal vegetation. For example, Silvereyes and Lewin's Honeyeaters both consume a wide range of fruit, nectar and invertebrate resources, including those from a number of invasive plants (Barker and Vestjens 1990; Loyn and French 1991; Gosper 1999a). However, it should be noted that while several bird groups appear to have lower overall abundance in *C. monilifera* vegetation than native vegetation, this is not the case for this group, indicating that they are still doing better than other groups in invaded vegetation.

Individual species within the frugivores of *C. monilifera* group that contributed much of the dissimilarity between habitats again exhibited differing responses to *C. monilifera* dominance. For example, the abundance of Silvereyes and Red Wattlebirds was greater in invaded vegetation, while Little Wattlebirds were less abundant. Red Wattlebirds use a wide diversity of nectar sources (Higgins *et al.* 2001; Chapter 3), yet it was the extensive consumption of fruits, of both native plants and C. *monilifera*, which probably most strongly influenced their habitat preference. Red Wattlebirds were particularly abundant at ENP in spring-summer when feeding on the fruits of *Leucopogon parviflorus, Acacia sophorae* and *C. monilifera* (Gosper 1999a). Interestingly, Red Wattlebirds appear to have expanded their range northwards in coastal NSW (Higgins *et al.* 2001). While the influence of *C. monilifera* invasion on this expansion is unknown, it may be significant, in conjunction with other changes in near-coastal habitats. Silvereyes are well-known consumers of *C. monilifera* fruits (Dodkin and Gilmore 1984; Chapter 4) and those of other invasive plants (Gosper 1994; Williams and Karl 1996; Stansbury 2001), and have a particularly diverse diet in coastal vegetation (Chapter 3). It is of no surprise, therefore, to see this generalist species doing comparatively well in the invaded vegetation.

Additional individual bird species are likely to be affected by dominance of the vegetation by *C. monilifera*, but at the current time, the empirical data available are insufficient to examine their patterns of occurrence more robustly. These species occurred in low abundance, and hence have contributed little to the dissimilarity between habitats in analysis (which was driven by the most abundant species). However, it is likely that other species that are strongly insectivorous and/or nectarivorous (Appendix A4), such as Spangled Drongos (*Dicrurus bracteatus*), White-naped Honeyeaters (*Melithreptus lunatus*), Brown-headed Honeyeaters (*M. brevirostris*), Scarlet Honeyeaters (*Myzomela sanguinolenta*) and the endangered Swift Parrot, may be adversely affect by *C. monilifera* invasion, as the most abundant members of these resource-use groups were. These birds may be particularly reliant on nectar, and invertebrates attracted to the nectar, over the winter period. For example, Spangled Drongos and Swift Parrots are winter visitors to near-coastal vegetation in central and southern NSW (Pizzey and Knight 1997; Garnett and Crowley 2000), particularly to nectar-rich sites where they feed on nectar (Chapter 3), and in the case of Spangled Drongos, invertebrates (including invertebrates such as honeybees (*Apis mellifera*) attracted to flowers, pers. obs.). The key resource for Swift Parrots in coastal vegetation is the flowers of Swamp Mahogany (*Eucalyptus robusta*) (Kennedy 2001). PB, because of this study, has been listed as an important area for Swift Parrots on the Australian mainland (Kennedy 2001).

Alternatively, some additional, less common frugivores of *C. monilifera* may be benefiting from its dominance of coastal vegetation, such as Crimson Rosellas (*Platycercus elegans*), Eastern Rosellas (*P. eximius*), Pied Currawongs (*Strepera graculina*) and Australian Ravens (*Corvus coronoides*). Each of these species share a number of characteristics that facilitate use of habitat dominated by *C. monilifera*, but separate them from other species that use *C. monilifera* fruits (with which they, as a group, did not differ in abundance between habitats). All these species are quite mobile and have regularly been observed flying into coastal vegetation infested with *C. monilifera* from other habitats further inland. Therefore, they are able to exploit the rich fruit food source provided by *C. monilifera*, while obtaining other resource requirements elsewhere.

These species have broad diets (Chapter 3), being either generalist omnivores or herbivores, including using fruits of other invasive plants (Loyn and French 1991) and they have adapted well to human-induced habitat modification elsewhere and have increased in abundance or range (Smith *et al.* 1989; Higgins and Davies 1996; Higgins 1999; Reid 1999; Fulton and Ford 2001; Higgins *et al.* 2001).

French and Zubovic (1997) assessed the abundance of groups of bird species in native and *C. monilifera* vegetation. However, they classified bird species on different criteria from those I used, and consequently had some markedly different groupings of species. They found consistently more birds of two groups in native vegetation than that dominated by *C. monilifera*: carnivorous birds and birds "that are rarely insectivorous and instead feed on nectar, fruit, flowers and seeds in the understorey". The carnivorous groupings were roughly similar in this study to French and Zubovic (1997), yet they showed different responses to habitat. For both studies, however, assessment of this group was hampered by their consistently low abundance. The other group more abundant in native vegetation the French and Zubovic (1997) study was not similar to any of the groups in this study. The species were spread between four of the groups I defined, making comparison of the studies for this group impossible.

Invasive birds

The development of new mutualisms between invasive birds and plants has been widespread (Richardson *et al.* 1992; 2000b). Invasion of *C. monilifera* has also been suggested to favour non-indigenous fauna (Dickman 1999). Most invasive birds present in coastal vegetation consume *C. monilifera* fruits (Chapter 4), however, there is no evidence from this study to suggest that they uniformly preferred invaded vegetation to native. Common Blackbirds (*Turdus merula*) were recorded infrequently from both habitats but also occur in native coastal vegetation in regions completely free of *C. monilifera* (and largely free of other invasive plants) (Gosper and Baker 1997). Spotted Turtle-Doves (*Strepopelia chinensis*) and Red-whiskered Bulbuls (*Pycnonotus jocosus*) were both slightly more abundant in native vegetation, although in this study they were only regularly recorded at one location which is a habitat fragment isolated among urban, industrial and weed-invaded areas. House Sparrows (*Passer domesticus*) and Common Starlings (*Sternus vulgaris*), however, were only recorded in *C. monilifera* vegetation, and as both widely consume *C. monilifera* fruits, may have benefited from invasion.

Species Richness and presence/absence within the bird community

In contrast to results for bird abundance, no consistent effects of different habitats could be detected on measures of species richness or presence/absence species composition of the bird community. Likewise, French and Zubovic (1997) found no consistent effect of *C. monilifera* on bird species richness. There was a significant interaction of location and habitat for cumulative species richness in this study, however. The total number of species inhabiting coastal vegetation was greater in native vegetation at ML and PB, but less in native vegetation at ENP. This provides weak evidence that *C. monilifera* vegetation overall may not provide habitat for as broad a diversity of species as native vegetation, even if the species richness at a particular point in time and space (*i.e.* mean species richness per point count) did not differ. It is inferred from the results of ANOSIM that it would be the less common species in coastal vegetation (that may not occur at all native vegetation sites) that are not occurring in *C. monilifera* vegetation, as no strong habitat effects were detected in the analysis of presence/absence data. ENP was the exception to this pattern, and probably again emphasises the significance of remnant native vegetation among *C. monilifera* in determining community composition, in this case the total number of species inhabiting an area.

These results indicate that *C. monilifera* is primarily impacting on the bird community through changes in the abundance of birds overall and of particular species, as instantaneous species richness and the species composition of the bird community seem to be largely unaffected by habitat type. To this time, the changes to vegetation induced by *C. monilifera* seem not to be having the effect of displacing significant numbers of species from, or attracting significant numbers of species into, the coastal bird community. Rather, the abundance of species already present is altered. This may be occurring because many coastal bird species are flexible in diet and forage opportunistically (Chapter 3), and consequently some individuals being able to find sufficient resources in *C. monilifera* vegetation to persist. This may be coupled with the continued persistence of some of the key resource-providing native plant species in invaded vegetation (discussed above) and the capacity of many species to move throughout coastal vegetation and adjoining vegetation types in search of resources.

7.4.2 Environmental attributes

The environmental attributes with the greatest principal axis correlation coefficients give an indication of what variables may be associated with differences in the bird community. Causal relationships are, however, not able to be determined from this analysis. As with the ANOSIM, the vectors of the environmental variables analysed here suggest important influences of both location and habitat on the composition of the bird community, although only when bird abundance data are used. Of particular interest is how habitat variables other than the one used directly to select sites (cover of *C. monilifera*) are associated with the division of the bird community. Interpretations are clouded by the confounded nature of many of these variables, as *C. monilifera* cover is probably negatively associated with the cover of many native plant species, as it is for *Acacia sophorae* (Weiss and Noble 1984a). However, that some species, such as *B. integrifolia*, have such high correlation coefficients (nearly as great as that for *C. monilifera* cover) suggests that they also have an important role in determining bird community composition. The vector for the cover of *B. integrifolia* is orientated slightly towards the groups of native vegetation sites in the ordination (Figures 7.3 and 7.5). Several vegetation structural variables are also highly correlated with the composition of the bird community, but again are highly confounded with the abundance of *C. monilifera*. Sites dominated by *C. monilifera* had much more vegetation cover less than 3m in height (much of which was *C. monilifera*), while sites where it was absent had more vegetation greater than 3m (Appendix A2). Consequently, this result does not provide a strong indication of the relative importance of vegetation structural versus floristic changes to the bird community. Similarly, although not supported by analyses, French and Zubovic (1997) suggested that availability of native resources might be important in determining bird community composition.

In the ordination of sites by the presence or absence of bird species, most of the highly correlated environmental variables were location effects, even if they were a vegetation measure. For example, several plant species or groups (such as *Eucalyptus* spp., *Hibbertia scandens* and twiners) were substantially more abundant at one location than the others, rendering the effect of the plants inseparable from location. Cover of *C. monilifera* was not strongly correlated with the ordination. This result supports the ANOSIM in indicating that the presence or absence of bird species is substantially impacted by location, but not habitat.

7.4.3 Effects of time of year

Bird abundance was greater in native vegetation than in that dominated by *C. monilifera.* Most of this difference, however, is from surveys conducted over the autumn-winter period. No difference in bird abundance was found between the habitats from analysis of spring-summer data. This is particularly interesting, as French and Zubovic (1997) found no overall effect of *C. monilifera* invasion on bird abundance, in a study that was conducted over spring-summer. Both these studies indicate that dominance of the vegetation by *C. monilifera* has a minor impact on the overall abundance of birds over this period. This study, however, suggests that there is an overall impact, and that it occurs in autumn-winter. Autumn-winter was also the only period that a significant effect of habitat was detected on any measure of species richness, with higher cumulative species richness being found in native vegetation.

Major changes in plant food resources in native and *C. monilifera* vegetation occur between these periods. Native coastal vegetation provides substantial nectar resources for nectarivorous species over autumn-winter. This is the main period of inflorescence production of the two most widely used nectar sources by birds: *B. integrifolia* and *E. robusta* (Chapter 3). Bird communities in eastern Australia also vary substantially seasonally, with many bird species, or portions of their population, moving into near-coastal vegetation in winter (Nix 1976; Dodkin and Gilmore 1984). Some of these directly use winter nectar flows, such as Swift Parrots and Spangled Drongos (Chapter 3). Other species may not, although the possibility of indirect reliance cannot be ruled out. Some insectivores, for example, regularly feed on invertebrates attracted to nectar sources (Gosper 1999a). Overall, invertebrate activity is likely to be lower over this period due to the colder temperatures, perhaps making these invertebrate-rich foraging sites particularly significant. *Eucalyptus robusta* is regarded by Law *et al*. (2000) as a key resource for nectarivorous vertebrates because of its flowering season, wide distribution, prolific flowering on an almost annual basis and production of enough nectar and pollen to be regularly visited by large vertebrates. Law *et al.* (2000) regard the late winter-early spring period (August-September) as one of general blossom shortage in coastal myrtaceous forests.

Autumn-winter is also the period of peak production of *C. monilifera* fruits, although some are produced throughout the year (Chapter 2). Few native plant species, however, produce fruits over this period. Consequently, coastal vegetation with *C. monilifera* is now fruit-rich over the autumn to winter period. The greatest differences in the bird community between *C. monilifera* and native vegetation occurred over the period of probably the greatest plant resource availability; of nectar combined with the fruits of *C. monilifera*.

The cumulative species richness of birds was greater in native than in *C. monilifera* vegetation over autumn-winter. This may be quite significant, as it suggests that native vegetation supports a higher diversity of species. The nectar-rich vegetation attracted some of these species, such as the Swift Parrot and Scaly-breasted Lorikeet (*Trichoglossus chlorolepidotus*), two nectar-feeders only recorded from native vegetation and in winter. Over spring-summer, a significant interaction of habitat and location affects cumulative species richness, which is the same effect when all survey data are combined.

Any reduction in winter nectar resources in coastal vegetation due to *C. monilifera* (Dodkin and Gilmore 1984) is of concern, and possibly the greatest conservation threat to birds posed by *C. monilifera* in the medium to long-term. However, other changes also need to be considered. For example, *E. robusta* habitat has become increasing fragmented by development along the NSW coastline (Brereton 1996) and one ecological community of which *B. integrifolia* is characteristic (Eastern Suburbs (of Sydney) Banksia Scrub), is endangered by massive habitat clearance and fragmentation (NSW NPWS 2003). The landscape-scale effects of depletion of winter, coastal nectar sources cannot be accurately predicted at this stage, but may be substantial. The proportions of species' populations that use these nectar sources are not known in most cases. However, in some years, most of the Regent Honeyeater (*Xanthomyza phrygia*; an endangered species) population occurs in coastal *E. robusta* forests (D. Geering, NSW NPWS, pers. comm. 2002). It is likely, given the mobile nature of many nectarivorous species that use these winter nectar sources, that the effects of any reduction in their quantity or quality will be felt well beyond coastal habitats. Such impacts on pollinator assemblages can have significant ramifications for other ecosystems and plant-animal interactions (Kearns *et al.* 1988; Cox and Elmqvist 2000).

7.4.4 Effects of location

A number of significant location effects were detected. I consider these only briefly here, because examining differences between locations was not an aim of this study. No effect of location on the overall abundance or species richness of birds was detected using all survey data or in either of the two periods tested separately. This indicates that on these broad measures of the bird community, any location effects were substantially less than between habitat effects. Locations were significantly different in the ANOSIM of the bird community using abundance data and presence/absence data. Habitat did not have a significant effect in the later case, which indicates that when bird community data are reduced to presence/absence, location is the more significant influence on the bird community. That the difference in the bird community between locations appears similarly great using presence/absence and abundance data indicates that it is change in species, rather than their abundance, between locations that are particularly important. That locations differed in this range of measures of the bird community is not surprising or particularly worthy of comment. Distances between locations were over several hundred kilometres, and hence it would be expected that species composition and abundance would change substantially.

In this study, the impact of *C. monilifera* invasion on all variables appeared weakest at ENP. This location is quite close to several areas used by French and Zubovic (1997), so this may indicate lower impacts of *C. monilifera* on the bird community towards the southern limits of *C. monilifera* invasion. However, an equally likely explanation relates to the vegetation composition at this location. Both *C. monilifera* sites at ENP had substantial cover of *B. integrifolia*, an important tree species for nectarivorous birds (Gosper 1999a). The cover of *B. integrifolia* at these sites was roughly equivalent to native vegetation sites, and greater than all other *C. monilifera* sites (Appendix A2). *Banksia integrifolia* cover was also highly correlated with the division of the bird community with bird abundance data. It is unknown if the substantially greater cover of *B. integrifolia* at sites in ENP with *C. monilifera* is characteristic of the region (and could therefore possibly explain some of the differences between French and Zubovic (1997) and this study) or a chance result.

7.4.5 Conclusion

This study has revealed that the bird community differs, in a number of ways, between vegetation dominated by *C. monilifera* and native species, across the range of locations surveyed. Total bird abundance and the abundance of several groups of bird species and individual species differed, with in most cases, more birds in native vegetation. In vegetation dominated by *C. monilifera*, there is a shift in the composition in the bird community away from extensively insectivorous and nectarivorous species to mobile, generalist species that include substantial quantities of fruits in their diet. Most of the difference between the habitats can be attributed to differences over the autumn-winter period, with no differences detected over spring-summer. These differences are readily explicable in terms of the floristic changes induced by invasion and consequent changes to the temporal availability of food resources for birds. Species richness of birds did not appear to be affected by habitat, nor did the bird communities of the two habitats differ when assessed on the presence or absence of bird species. This further emphasises that *C. monilifera* induces changes to the abundance of bird species already present, without (at this stage) rendering the habitat completely unsuitable for many species or attracting many new species in.

The existing impact of *C. monilifera* invasion on bird communities of coastal vegetation is likely to have been substantial, as *C. monilifera* now infests over 80% of the NSW coastline (NSW NPWS 2001). This impact will increase in the absence of continued efforts at control. The area currently invaded by *C. monilifera* is substantial, and in the absence of active management, the native vegetation component of these areas will probably continue to degrade. Additionally, *C. monilifera* continues to infill areas currently uninfested. Of high importance to bird communities are the remaining native plants, particularly those that provide winter nectar resources, such as *B. integrifolia* and *E. robusta*. If declines of these plants occur on a regional scale (due to *C. monilifera* and/or other processes), the consequences for many bird species (and other fauna) reliant directly or indirectly on coastal winter nectar flows, may be substantial.

The demonstration of a temporal difference in the impact of an invasive plant is significant, and needs to be considered in similar studies. A short-term assessment may conclude that there is no substantial impact (in all probability correctly, such as French and Zubovic 1997). Yet, when the same system was examined at a different time (both in a different year and over different seasons), impacts that are more substantial were detected, although largely outside of the period of the French and Zubovic (1997) study. Of course, knowledge of the ecology of the invasive plant and the invaded ecosystem would suggest periods likely to have higher impacts than others, and would obviously provide a starting point for such impact assessment.

CHAPTER 8. GENERAL DISCUSSION

Overlooking the beach at Perkins Beach Recreation Reserve, Wollongong (patches of dead *Chrysanthemoides monilifera* post-herbicide application can be seen on the dune summit). How severely has invasion and dominance of coastal vegetation along much of the NSW coastline by *C. monilifera* affected plant-animal interactions and the bird community?

8.1 Overview

In this research project, I have investigated several aspects of the effect of a plant invasion on native biota, specifically focusing on the interaction between birds and vegetation. The invasive plant was Bitou Bush (*Chrysanthemoides monilifera* spp. *rotundata*), a vertebrate-dispersed species native to South Africa, and the invaded system coastal vegetation in NSW, Australia. Specifically, I have investigated how the fruits of *C. monilifera* compare with native alternatives, and how its invasion and control affect bird communities and the plant-bird interaction of fruit-removal. Combined with previous research, this study clarifies the impacts of this invasive plant on native biota, and some of the priorities for conservation management. Of equal importance, I have investigated some consequences for native biota of one of the main methods currently used to control *C. monilifera*, aerial herbicide application. This assessment is particularly important, as control strategies in degraded areas of indigenous vegetation are usually an attempt to enhance habitat value for native species, but the control methods themselves may have unforseen adverse impacts that need to be considered.

In this general discussion, I summarise my research and bring together the separate components, focussing on an assessment of the degree to which *C. monilifera* and its control have disrupted plant-animal interactions and the composition of the bird community. Additionally, in the final section, I identify management actions and further research that will be necessary for conservation of native biota in ecosystems invaded, or at risk of invasion, by *C. monilifera*. Chapters 2 to 4 will only be summarised briefly here, as the observations and measurements within them assist in the interpretation of the experiments that are discussed in more detail.

Chrysanthemoides monilifera, and a range of native plant species with which it co-occurs, have fruits dispersed by vertebrates, particularly birds. The range of *C. monilifera* fruit characteristics suggest that the species would be attractive to frugivorous vertebrates, as is the case in its indigenous range (Knight 1986). Specifically, compared to native vertebrate-dispersed plant species, *C. monilifera* fruits are intermediate in fruit and endocarp size (length, width and weight) and in nutrient content. *Chrysanthemoides monilifera* produced fruits throughout the year, but with a peak of production over the period April to July. By far the majority of native plant species produce fruits at other times of the year (often over September to January), and very few throughout the year. Of those native plant species that do fruit over April to July, most either are large-seeded or differ substantially in other aspects of fruit morphology. Consequently, the fruits of few native species overlap with those of *C. monilifera* in both phenology and morphology

In predicting the possible impacts of *C. monilifera* invasion and control on avifauna, it is important to know how native bird species use the resources in the habitat, both prior to and after *C. monilifera* invasion. Many of the common bird species in coastal habitats have quite broad diets: more diverse than previously recognised. Many species used a variety of fruit, seed, nectar and invertebrate foods. For several species, the dominant plant-derived component of the diet changed over periods of the year, for example Yellow-faced Honeyeaters (*Lichenostomus chrysops*), Red Wattlebirds (*Anthochaera carunculata*) and Noisy Friarbirds (*Philemon corniculatus*) mainly fed on fruits over one period, and nectar over another. Highly used fruit sources in coastal vegetation were *C. monilifera*, Coastal Beard-heath (*Leucopogon parviflorus*) and *Monotoca elliptica.* Highly used nectar sources were Coastal Banksia (*Banksia integrifolia*) and Swamp Mahogany (*Eucalyptus robusta*). The temporal pattern of change in plant-derived food resources for birds in native vegetation was higher nectar availability over autumn and winter and higher fruit availability over spring and early summer. Invasion by *C. monilifera* skewed this temporal pattern, such that fruits were available in high quantities throughout most of the year. Winter nectar resources were likely to decrease, however, as *C. monilifera* probably displaces some native plant species (Dodkin and Gilmore 1984; Weiss and Noble 1984a).

8.2 Some predictions

Based upon these preliminary observations, I predicted that *C. monilifera* invasion and control might affect plant-bird interactions and bird communities. The effects on the removal of fruits of vertebrate-dispersed native plant species was difficult to predict, as logical cases could be made for the rates of fruit-removal to increase, decrease or not change. Birds differ in their patterns of resource use, including of foods. I expected groups of bird species with different resource requirements to respond differently to *C. monilifera* invasion and removal, related to associated changes in their food resources:

1) Frugivorous and granivorous birds may exhibit inconsistent responses to *C. monilifera* invasion. Some species may adapt well to the changed resource patterns and may increase in abundance. These birds may, however, be vulnerable to *C. monilifera* removal. Species with more specialised fruit requirements, species that are highly facultatively frugivorous or use little fruit, may be unaffected or adversely affected by invasion, but this would depend on other vegetation and resource changes;
2) Nectarivores are likely to be, on the whole, negatively affected by *C. monilifera* invasion, as *C. monilifera* itself provides no floral resources used by birds, but it probably displaces nectar-producing native plants. The degree to which individual species are affected may vary according to their specialisation of resource use: those that can supplement their diet

with fruits are less likely to be adversely affected. The short-term impacts of *C. monilifera* removal of nectarivores are likely to be negligible, as plant resources used heavily by these birds are not likely to change in abundance.

3) I made no predictions on the response of mainly insectivorous species to *C. monilifera* invasion or removal, as the relevant data on which to base such predictions was not collected. However, the work of French and Eardley (1997) indicated shifts in species composition, but not abundance or species composition in litter invertebrates with *C. monilifera* invasion. This may have subsequent impacts on insectivorous birds.

I then tested these predictions experimentally, by measuring fruit-removal and bird community composition in three habitats: native vegetation, vegetation dominated by *C. monilifera*, and vegetation from where *C. monilifera* had been removed. These results are discussed in the following section.

8.3 Key Results

In the first experiment, I tested whether invasion or control of *C. monilifera* affected the rate of removal of fruits of *C. monilifera* and of several native plant species. An effect was thought possible, as changes in fruit availability may induce unpredictable changes in frugivore assemblages, with consequent effects on frugivory of other plant species. The key result of this experiment was that there was no difference in the rate of removal of any native fruit type between any of the habitats (native vegetation, vegetation dominated by *C. monilifera* and vegetation from which *C. monilifera* had been removed through application of herbicide). In addition, I found that the rate of removal of *C. monilifera* fruits was greater in vegetation dominated by *C. monilifera* than from where *C. monilifera* had been removed.

The use of aerial herbicide application to control *C. monilifera* could potentially affect fauna species that now use this modified habitat. The bird community of these areas was compared to untreated *C. monilifera* vegetation. Aerial herbicide application affected the bird community over the locations sampled in this study, with the main findings being:

1) There was no evidence for changes in any aspect of the bird community from before herbicide application to immediately afterwards, before *C. monilifera* died. Changes occurred after *C. monilifera* died, however, indicating that the impact was due to the death of *C. monilifera* rather than any disturbance during the herbicide application procedure;
2) Bird abundance declined after *C. monilifera* death relative to unsprayed sites. This decline was attributed to the group of species that consume *C. monilifera* fruits. Other groups of species with different resource requirements were unaffected;

3) The species richness and species composition of the bird community were unaffected by *C. monilifera* control.

Vegetation dominated by *C. monilifera* supported a different bird community than native vegetation, over the locations sampled in this study. The main differences detected were:

1) More individual birds (all species combined) occurred in native vegetation, although this was only the case over surveys in April-September (autumn-winter). No difference in the abundance of birds was detected over October-March (spring-summer);
2) Among groups of bird species, more canopy and understorey insectivorous birds occurred in native vegetation, and at some locations, this was also the case for nectarivorous birds. For nectarivores, abundance at *C. monilifera* sites was related to the cover of important native nectar-producing plants (*Banksia integrifolia* and *Eucalyptus* spp.). There was no difference in the abundance of consumers of *C. monilifera* fruits, carnivores/scavengers or aerial insectivores between the habitats;
3) Changes in the bird community were in the relative abundances of species. There was no evidence of changes in species richness or composition;
4) The foliage cover of some native plant species, such as *B. integrifolia*, was probably a stronger determinant of the bird community than cover of *C. monilifera per se*. Remnant native vegetation appears to exert a strong influence on the coastal bird community.

8.3.1 Plant-frugivore interactions

Several of the findings of this study suggested that *C. monilifera* could disrupt plant-frugivore interactions. Fruits of *C. monilifera* were of suitable size and nutrient content for use by frugivores, the fruits were used by many bird species, and *C. monilifera* invasion altered the temporal availability of fruits in coastal vegetation. Invasion and control of *C. monilifera* also affected the bird community. Dominance of the vegetation by *C. monilifera* did not, however, alter the rate of removal of fruits of the native plant species tested. Why was the plant-frugivore interaction of fruit-removal largely unaffected by the other bird and plant community changes? That no impact was detected can be attributed to four, non-exclusive factors.

(i) Facultative plant-frugivore relationships

The relationships between bird-dispersed plants and frugivores may be strongly facultative and contain a large amount of redundancy. Many of the most common frugivorous bird species in coastal vegetation have generalist diets, forage opportunistically (Chapter 3 and 4) and are relatively mobile. Consequently, changes in the temporal availability of the type of plant food may not have flow-on effects to the dispersal of fruits of other plants. For example, many of the bird species that consume *C. monilifera* fruits also feed on not only native fruits, but nectar as well. Nearly half of all the species that take *C. monilifera* fruits (Chapter 4) also feed at *B. integrifolia* inflorescences (Chapter 3). These species may be taking whatever plant food is available and not be limited by only the availability of, for example, fleshy fruits. Additionally, characters of the fruits of plants in coastal vegetation may indicate a highly generalised dispersal system. For example, the size of most fruits is less than the average size of fruits in other vegetation types in south-eastern Australia (Chapter 2, *c.f.* French 1991). This indicates that few plants require frugivores able to ingest large fruits. Similarly, short fruiting seasons and periods of fruit abundance and shortage may also facilitate the development of a generalised dispersal syndrome and/or mobile, generalist frugivores, rather than sedentary, specialist frugivores.

The occurrence of highly facultative relationships between plants and frugivores is supported by the comparison of the bird communities of native and *C. monilifera* habitats (Chapter 7). No consistent differences in the overall abundance of frugivores (either those that eat *C. monilifera* fruits or those that do not) occurred between habitats, indicating that dominance by *C. monilifera* has not substantially altered the frugivore community. Some of the constituent species may have preferred one or other of the habitats, for example Silvereyes (*Zosterops lateralis*), but not fruit-eating birds as a whole. This suggests several important points. First, *C. monilifera* habitat does not support more frugivores than native vegetation, hence the rate of removal of native plant fruits is unlikely to increase in *C. monilifera* habitat as suggested may occur in the introduction to Chapter 5. Second, that no difference in the abundance of frugivores overall was found indicates that these birds are largely using fruit resources as available, and are, as a whole, not responding solely to changes in fruit abundance. The frugivore community appears well buffered against changes in fruit abundance, with these birds able to use supplementary non-fruit resources in native vegetation and remain roughly equally abundant.

Consequently, the lack of a difference in the rate of removal of native fruits between habitats could be explained by the lack of changes in the abundance of frugivores (although changes in fruit-removal would still be possible with changes in bird community composition). Removal of fruits in coastal habitats, for small-fruited plants, may be largely opportunistic with birds taking fruits in proportion to fruit abundance. Fruit-removal may not be affected by changes in overall fruit abundance. If this is the case, competition between plant species and individuals for dispersal agents may be low, and the impact of a new species in this system on this bird-plant interaction may also be low.

(ii) Temporal differences in fruit production

Most native plant species produce fruits in spring and early summer, over periods of the year when lower quantities of *C. monilifera* fruits are available than in peak periods (Chapter 2).

These plants would avoid maximum competition with *C. monilifera* for dispersal agents, and possibly to changes to the frugivorous bird community induced by *C. monilifera* invasion if these were seasonal and associated with *C. monilifera* fruit availability. This last suggestion is not supported by the comparison of the bird community of vegetation dominated by native plants and *C. monilifera* (Chapter 7). Although differences between habitats were apparent over autumn-winter and not spring-summer, these changes were largely due to impacts on insectivorous and nectarivorous species (although many of these do use a limited range of fruits). In autumn-winter, more birds were found in native vegetation (most likely associated with nectar abundance) and less in *C. monilifera* vegetation, despite the fruit glut.

(iii) Fruit morphology differences

Those native plant species that do fruit over autumn-winter nearly all have fruits that differ substantially in size (being larger) from those of *C. monilifera* or differ in other aspects of morphology (Chapter 2). Fruit size influences choice by birds (Herrera 1984), as dispersers of fruits are limited by their gape width (although see Williams and Karl 1996; Gosper 1999a). Subsequently, a more limited range of bird species usually disperses large-fruited species than smaller-fruited species (Herrera 1984). Hence, removal of the large-fruited native plant species was likely to be by a substantially different disperser assemblage than that of *C. monilifera*.

(iv) Similar fruit nutrition

Chrysanthemoides monilifera fruits are moderately attractive in nutrient composition to birds. It is unlikely, however, from the fruit quality information available, that birds would consistently take *C. monilifera* fruits in preference to native fruits, hence not placing *C. monilifera* at a competitive advantage if dispersers were limiting (although there is no evidence that dispersers were limiting in this system).

In contrast, the rate of removal of *C. monilifera* fruits did differ between habitats; with more rapid removal in *C. monilifera* stands than from where *C. monilifera* had been eliminated by herbicide treatment. Why did this occur, while removal of native fruits was unaffected?

First, the lower rate of removal of *C. monilifera* fruits is supported by the changes in the bird community that occurred after *C. monilifera* death (Chapter 6). Birds that consumed *C. monilifera* fruits were less abundant, and probably contributed the bulk of the overall differences in bird abundance between the treatments. This readily explains why fruits of *C. monilifera* were removed at a lower rate in sprayed vegetation. *In-situ*, native fruits were taken by many of the same species as took *C. monilifera* fruits (Chapter 3 and 4), yet no differences in fruit-removal rates of these species were detected. Insufficient records of frugivores/granivores (species other than those that feed on *C. monilifera*) were available for the analysis in Chapter 6 to be particularly robust. Therefore, I cannot say with confidence what the effect of *C. monilifera* removal was on these birds. However, if they weren't as adversely affected as species that consume *C. monilifera* fruits, this may partly explain why removal of native fruits was not affected by elimination of *C. monilifera*.

Second, differences in periods of fruit production, combined with differences in morphology, may also have contributed to the lack of difference in fruit-removal rates of native species between habitats. The experiments using *Elaeocarpus reticulatus* and *C. monilifera* fruits were conducted concurrently. However, the disperser assemblage of *E. reticulatus* is likely to differ from that of *C. monilifera* because of the size of the fruits (Chapter 2), which may explain the different effects of *C. monilifera* elimination on fruit-removal between these species. Although *Monotoca elliptica* and *C. monilifera* have probably more similar dispersal assemblages, the *M. elliptica* trial was conducted in a different season (when *M. elliptica* fruits were available). This may be important in explaining why the rate of fruit-removal of *M. elliptica* was not affected by *C. monilifera* removal. At this time, native fruits are abundant (Chapter 2), and birds may be responding more to the abundance of native than *C. monilifera* fruits.

8.3.2 Control of *Chrysanthemoides monilifera* and bird communities

Control of *C. monilifera* through aerial herbicide application had an impact on birds. This was a decline in the abundance of frugivores of *C. monilifera* and occurred after *C. monilifera* died (rather than after herbicide application only). This impact is thus generalisable to other methods of *C. monilifera* control that involve rapid and broad-scale vegetation change. The impact on the bird community is readily explicable in terms of the vegetation and resource changes that occur with *C. monilifera* death. Aerial herbicide application typically kills over 95% of *C. monilifera* plants (Toth *et al.* 1996). As the control is conducted in winter, towards the end of peak *C. monilifera* fruit production (Chapter 2), there would be a substantial and rapid decline in the *C. monilifera* fruit resource at control sites. Consequently, it is not surprising that the group of bird species that use this resource directly for food are impacted by *C. monilifera* removal. No other resource-use group of birds was affected.

There was no indication of any groups of birds increasing in abundance at sprayed sites after *C. monilifera* death. This was expected, as the *C. monilifera* control program effectively removed one portion of the vegetation community, without any replacement. It contrasts several other studies that have assessed bird response after invasive plant removal (Linz *et al.* 1996a; 1997; Linz and Blixt 1997), although each of these studies was conducted over a much longer term than mine. No new resources for birds become available immediately after *C. monilifera* death. In the medium to longer term, however, newly germinated plants would replace the *C. monilifera*. Consequently, many bird species may benefit if replacement vegetation contains more native plant species (Chapter 7).

From the perspective of avifauna conservation, the changes induced by the species-specific removal of *C. monilifera* appear relatively minor and may not require any particular ameliorative measures. None of the individual species in the frugivores of *C. monilifera* resource-use group are of conservation concern, and for many individual species and as a group, are likely beneficiaries of *C. monilifera* invasion (or at least not disadvantaged), and probably also invasions by other bird-dispersed plants (Buchanan 1989; Spennemann and Allen 2000; Fulton and Ford 2001). Some of these mobile, generalist omnivores were more abundant in vegetation dominated by *C. monilifera* than that of native species (Chapter 7). Additionally, the changes induced to bird communities are only likely to be transient, although this has not been empirically confirmed. Over time, new vegetation will replace the killed *C. monilifera*, and the ultimate impact of the control program will depend on the composition of this vegetation. In the absence of further management (or inadequate management), newly germinated *C. monilifera* could regrow and the vegetation community could return to similar to the pre-control state. In other cases, native plants, other invasive plants, or a combination of both may replace the killed *C. monilifera*, which would also affect the bird community. It appears unjustified, given the changes observed, to devote substantial resources to reducing the transient impact on the bird community of *C. monilifera* removal. Likewise, given the changes to bird communities linked to dominance of the vegetation by *C. monilifera*, abandoning or limiting the most cost-effective current method of *C. monilifera* control (NSW NPWS 2001) would not be justified due to the minor impacts on avifauna detected in this study.

8.3.3 Invasion of *Chrysanthemoides monilifera* and bird communities

The bird community of areas dominated by *C. monilifera* differed substantially from areas of native vegetation. Birds were less abundant overall, specifically insectivores in the canopy and understorey, and at some locations, nectarivores. Bird abundance only differed between habitats over the autumn-winter period.

The lower abundance of insectivores in *C. monilifera* habitat was not predicted from resource and habitat use assessments of birds (Chapter 3), although invertebrate resources were not explicitly considered. However, it suggests that research on how *C. monilifera* affects canopy and understorey invertebrate communities may be valuable. The variable impacts on nectarivores are readily explicable by vegetation composition and subsequent resource availability. This resource-use group was predicted to decline in *C. monilifera* vegetation, as nectar-producing plants may be displaced by *C. monilifera*. The only location at which nectarivores were not less abundant in *C. monilifera* vegetation had similar foliage cover of *B. integrifolia* (an important nectar source for birds; Chapter 3) as native vegetation sites (and much more than other *C. monilifera* sites; Appendix A2). This suggests that cover of this key native plant species may be more important in determining nectarivore abundance than cover of *C. monilifera per se*. Frugivorous birds were, on the whole, unaffected by dominance of the vegetation by *C. monilifera*. These species may be able to use alternative resources in *C. monilifera* and native vegetation and stay at similar abundances. Most bird species that fed on *C. monilifera* fruits, for example, also used fruits and nectar from a variety of native plants (Chapter 3 and 4). Individually, a few species were more abundant in *C. monilifera* vegetation. Several of these were, as predicted, omnivorous, dietary generalists that consume *C. monilifera* fruits. Many of these have also responded positively to human-induced landscape changes elsewhere (Chapter 7).

The seasonal differences in the effect of dominance of the vegetation by *C. monilifera* on bird communities ties in well with changes in plant resource availability. With *C. monilifera* invasion, there is probably less nectar available over the autumn-winter months, but more fruits (Chapter 2 and 3). Nectar availability may also decline over spring-summer, although this may be less significant as bird use of nectar resources in coastal vegetation at this time appears less. The decline in autumn-winter nectar resources appears to be the most significant plant resource change for birds with *C. monilifera* invasion. These nectar losses may be important regionally, as the two most important coastal nectar sources (*B. integrifolia* and *E. robusta*; Chapter 3) flower during a period of regional blossom shortage (Law *et al.* 2000) and when many birds move into coastal vegetation (Nix 1976; Dodkin and Gilmore 1984).

8.3.4 Differences across locations

An interesting question arising from various analyses in this study is whether the impact of *C. monilifera* on plant-bird interactions and bird communities varies with location, and if it does so, is a regular pattern apparent? Although my study was not set up specifically to investigate this, there is some suggestion that this may indeed be occurring. The rate of removal of *C. monilifera* fruits declined consistently across locations from north to south (Chapter 5). This may be related to the observed slower pace of invasion of *C. monilifera* in the south of its invasive range (Lane 1984). The ENP location (the most southerly in this study) was the only location at which the abundance of nectarivores of *B. integrifolia* and cumulative species richness was not less in invaded than native vegetation (Chapter 7).

A range of factors, independent or correlated with *C. monilifera* invasion, may have caused these observed patterns. However, the possibility that some of the biodiversity impacts of *C. monilifera* may be reduced towards the southern limits of its invasive range may be worth further investigation. Nearly all research prior to this study on the biodiversity impacts of *C. monilifera* (such as Weiss 1984; Weiss and Noble 1984a; 1984b; French and Eardley 1997; French and Zubovic 1997; Vranjic *et al.* 2000) has been conducted near the southern limits of *C. monilifera* invasion in NSW. This could be significant if impacts of *C. monilifera* do differ along the north-south climatic gradient on the NSW coastline.

8.4 Implications for management and research

8.4.1 Management actions

From the findings of my research, what steps can be taken to avoid or ameliorate the impacts of *C. monilifera* invasion and control on bird communities and plant-bird interactions? Some suggestions are outlined below.

1) **Strategy for conservation of coastal communities**

I believe management of *C. monilifera* should be considered as part of an overall strategy for the conservation of coastal vegetation and fauna, with invasion by *C. monilifera* treated as one of the major threats to desired conservation outcomes. This is instead of control of *C. monilifera* being the main focus of management efforts while other processes are possibly neglected. Many of the following issues should be included in such a strategy.

2) **Consider biodiversity impacts in prioritising invasive plant control**

Biodiversity considerations should rank highly in weed control strategies; in deciding which weeds to control, where to undertake control, and what measures are needed to minimise impacts on biodiversity. To date for *C. monilifera*, there has been an emphasis on research on how to control it, and undertaking control actions (see Chapter 1). In my opinion, this has been while there is insufficient understanding of how *C. monilifera* invasion and control (through use of control methods with unknown effects) is affecting biodiversity (and probably other aspects of ecosystem function). Such approaches to invasive plant management may not be atypical (Williams and West 2000). With an understanding of the species and communities most at risk from *C. monilifera* invasion, and the impacts of some control techniques, the limited resources available for invasive plant control can be directed where they will have the most benefit in preserving ecosystem structure and function.

3) **Continue efforts at *C. monilifera* control**

Continued efforts directed to limit the spread and density of *C. monilifera* will be important to minimise the adverse impacts of invasion on birds identified here. Much progress in this regard has recently been made. Broad strategies to combat *C. monilifera* invasion and limit its impact on a range of levels (*e.g.* biotic, abiotic, economic) have already been developed on a national and state scale (ARMCANZ, ANZECC and Forestry Ministers 2000; NSW NPWS 2001), arising from the listing of *C. monilifera* as a weed of national significance (Thorp and Lynch 2000). Additionally, a threat abatement plan is in preparation (by the NSW NPWS) to minimise the threat posed by invasion of *C. monilifera* to native communities.

4) **Target control strategies to important communities**

A key resource for many bird species in coastal communities, and probably other biota, is winter-flowering plants. The most important of these are *B. integrifolia*, *E. robusta*, and in some years, *E. pilularis*. Many nectarivorous birds and other vertebrates (Chapter 3; Dodkin and Gilmore 1984; Law *et al.* 2000) use these species over a period of regional blossom shortage (Law *et al*. 2000) and when many bird species move to near-coastal habitats (Nix 1976; Dodkin and Gilmore 1984). Targeting control efforts at a local scale to protect the best stands of these plant species from *C. monilifera* invasion may be an effective mechanism to minimise local and probably regional effects of the degradation of these nectar sources. For example, the native vegetation site at PB in this study has stands of each of these plants. Yet, since this study had been completed, the native vegetation has been substantially invaded by *C. monilifera*, even though broad-scale control efforts have been mounted on adjoining extensive *C. monilifera* stands. From the perspective of bird conservation, it would have been more beneficial to manage this small patch of good-quality, mostly intact native vegetation, than direct resources at the extensive infestations that have proved unable to be controlled with the resources available.

Similarly, control strategies can be targeted to conserve other highly significant communities and sites important for specific biota. These may include areas of littoral rainforest, threatened species, endangered ecological communities and seabird and shorebird breeding colonies. In this last case, several significant nest predators are able to extensively use *C. monilifera* stands, for example Pied Currawongs (*Strepera graculina*) and Australian Ravens (*Corvus coronoides*) (Chapter 4). Near important breeding sites continued *C. monilifera* control may reduce the local density of these predators. Of course, the potential impacts of any control methods on the community or species that are to be protected need to be considered. For example, Matarczyk *et al.* (2002) describe the off-target effects of aerial herbicide application to control *C. monilifera* on the threatened *Pimelea spicata*.

5) **Consider other potential weed problems**

After successful control of an invasive plant, there exists a temporary gap in the vegetation that could be exploited by another invasive plant. If this invasive plant proved to be either more difficult to control that the one just removed (Westman 1990), or have greater impacts on biota, than the control efforts would have been in vain. This certainly needs to be considered in areas currently infested with *C. monilifera*, as a range of other invasive plants have become problems where *C. monilifera* control has been undertaken (NSW NPWS 2001) or elsewhere in coastal vegetation (Batianoff and Franks 1998). At the current time, it is not possible to judge if these potential replacement invasive plants would have greater, lesser or just different impacts on biodiversity than *C. monilifera* if they were to replace it.

6) **Manage coastal vegetation after major disturbances to ensure plant recruitment**

Any major disturbance has the potential to exacerbate the problem of invasive *C. monilifera*. The disturbance could facilitate initial invasion by *C. monilifera*, or increase the pace of vegetation change after initial invasion. For example, intense fire in coastal vegetation has the dual effects of stimulating the soil seed bank of *C. monilifera* (Vranjic 2000) and also affecting the native plants co-occurring with it, particularly obligate-seeding species. The period post-disturbance is likely to be critical for plant recruitment, as seedlings may be faced with extensive competition with seedling *C. monilifera* (Weiss and Noble 1984b). This management consideration is also critical in areas where *C. monilifera* control is carried out with the use of a fire treatment (Vranjic 2000). Unless indigenous plant species are conserved and/or recruitment of new individuals occurs, then there may be no bird conservation benefit from the control program. Management of this type of problem may be an ideal case to use adaptive management, whereby existing control strategies may be implemented, as long as changes in vegetation composition are recorded and management actions modified accordingly.

7) **Encourage widespread planting of native, winter-flowering plants**

As discussed at length in several sections, winter-flowering plants appear to be of great importance to many coastal bird species, and possibly to fauna on a regional scale. One method by which reduction in the quantity and quality of this resource, due to *C. monilifera* and other factors, can be reduced, is by the more widespread use of these species elsewhere in coastal situations. This may include in public and private landscaping, street trees and revegetation, including after invasive plant control. Local species should be used, and for much of the NSW coastline, these would include the key nectar-producers *B. integrifolia* and *E. robusta*.

Many of the nectarivorous birds that use winter nectar sources are quite mobile, on either a regional or a local scale (*e.g.* Swift Parrot (*Lathamus discolor*), Spangled Drongo (*Dicrurus* *bracteatus*) and Scarlet Honeyeater (*Myzomela sanguinolenta*); Pizzey and Knight 1997; Clarke *et al.* 1999; Garnett and Crowley 2000). Each of these species has used isolated remnants of these vegetation types, for example at PB, a small remnant highly isolated from other native habitat by urban, industrial and *C. monilifera*-dominated areas. Consequently, I believe these birds would be able to utilise appropriate urban or otherwise isolated patches of remnant or regenerated habitat of the winter nectar producing plants.

A potential problem is dominance of these nectar sources by aggressive species that can exclude small birds, affecting avian diversity and abundance, such as the Noisy Miner (*Manorina melanocephala*) (Clarke *et al.* 1995). However, I have not observed small patches of either *B. integrifolia* or *E. robusta* dominated by colonies of Noisy Miners as small remnants of other vegetation types can be. They do use both these plants for nectar (Higgins *et al.* 2001), however, the seasonal nature of major nectar flows in coastal vegetation may limit the success of the mainly sedentary, colonial Noisy Miner (Higgins *et al.* 2001). In contrast, most native nectarivores in this system move locally or regionally following nectar sources and/or switch diets to other foods.

8) **Experiments with rapid revegetation of sprayed areas**

Control of *C. monilifera* by aerial herbicide application only had relatively minor impacts on birds. To date, regeneration at sites that *C. monilifera* has been controlled by this method has largely been left to natural processes. This may be reasonably effective at recently invaded sites that retain a substantial seed bank of native plants. However, in more degraded sites and those colonised by *C. monilifera* for longer periods, other measures may be necessary to encourage regeneration and enhance the habitat after *C. monilifera* has been removed. Spreading seed of native plants at the same time as herbicide application could be considered. Some trials and costs for this have been completed (John Toth pers. comm.), however with only a limited range of species. Species that could be used include those killed by the herbicide treatment (listed in Toth *et al.* 1996) and dominant and key species in the community. For birds, these key plant species include those that structure the seasonal availability of plant-derived foods *i.e.* fleshy fruits over spring and summer (*e.g. Acacia sophorae, Monotoca elliptica* and *Leucopogon parviflorus*) and nectar over autumn and winter (*e.g. B. integrifolia, E. robusta* and *E. pilularis*).

8.4.2 Research priorities

In addition to management actions that can be initiated on the basis of existing knowledge, further research is required in aspects of the ecology of coastal birds and vegetation, and the impacts of *C. monilifera* invasion and control on native biota, to have full confidence that coastal habitats can be managed for conservation in the long-term. This research is just as important as management of *C. monilifera* infestations, and much of this work can be incorporated into existing management programs by adding a data collection objective. In the next section I will outline, in a rough order of priority, what I consider important further research in this area to be. I will briefly identify the gaps in knowledge and their importance.

1) **Effects of *C. monilifera* invasion on pollination and plant-nectarivore interactions**

Floral visits and pollination were not assessed as part of this study. However, several of my findings suggest that *C. monilifera* may impact on plant-nectarivore interactions more substantially than on plant-frugivore interactions. It would be useful to investigate this. For example, it could be hypothesised that pollination success, say of *B. integrifolia*, may be reduced when plants are isolated and less dense in stands of *C. monilifera*. This may occur because there are less pollinating birds in such habitats. Other hypotheses regarding pollen movements and outcrossing could also be developed.

2) **Impact of *C. monilifera* on canopy and understorey invertebrates**

In this study, birds that feed mainly on invertebrates in the understorey and canopy were much less abundant in *C. monilifera* than native vegetation. I have put forward alternative hypotheses attempting to explain this occurrence, namely that changing vegetation floristics and structure may affect invertebrate abundance or the ability of bird species to forage effectively. Investigating the composition of the invertebrate community of the understorey and canopy of *C. monilifera* stands compared to native vegetation may allow refinement of this hypothesis. Results could also be compared to the effects of *C. monilifera* invasion on litter invertebrates (French and Eardley 1997), and further expand our knowledge on effects of *C. monilifera* on biota. Although experimentally and logistically more difficult, determining any change in the invertebrate community at sites being invaded (and compared to control, uninvaded sites) over the period of invasion would be more powerful that the comparison of invaded and uninvaded sites used in this and previous studies. It may allow the allocation of causality, rather than having the underlying assumption of similarity of composition prior to invasion.

3) **Impact of *C. monilifera* invasion on plants**

Since the work by Weiss and Noble (1984a; 1984b), there has been little additional work on the effects of *C. monilifera* invasion on plants (Vranjic *et al.* 2000 being an exception). Consequently, for coastal plant species not covered by this research, such as *B. integrifolia, Eucalyptus* spp. and epacrids, I could find no empirical assessments of the impacts of *C. monilifera* invasion. This research may not have been done because the outcome seems obvious, but some quantitative assessment of the pace, scale and consequences would be very useful.

4) **Impact of aerial herbicide application on biota**

Aerial herbicide application, usually with glyphosate, or rarely, metsulfuron methyl (Toth *et al.* 1996), is one of the most cost-effective methods of *C. monilifera* control (NSW NPWS 2001). Consequently, it has been widely used throughout coastal NSW. Some of the impacts on biota of this control method have been investigated, such as on birds (this study), litter invertebrates (Liz Lindsay, University in Wollongong, in progress) and some plants (Kohler *et al.* 1995; Toth *et al.* 1996). Impacts on amphibians, reptiles and mammals are yet to be investigated, although the extent to which these groups use *C. monilifera* vegetation is also largely unknown. Some native plant species are adversely affected by the procedure (Toth *et al.* 1996; Martarczyk *et al.* 2002), but this has, to date, not prevented its use or stimulated investigation into how adverse impacts may be mitigated. Additional work on plants is undoubtedly justified. The continued use of this technique appears to be an example of prioritising weed control objectives over the ecological impacts of the control methods (Williams and West 2000).

5) **Bird movements**

The movement patterns of Australian birds are poorly known (Griffioen and Clarke 2002). This lack of knowledge hinders the effective conservation of these species, and through their interactions with other species, whole communities of biota. For example, the movements of birds that use the winter nectar resources of *B. integrifolia* and *E. robusta* in coastal vegetation are either only known very generally (*e.g.* Swift Parrots move between Tasmania and mainland Australia) or not at all. Any research that would increase knowledge on the movements of nectarivores, in particular, would assist in conservation measures. For example, it could contribute to assessing the importance of coastal vegetation for those species, and for plant and bird communities elsewhere where these species spend other periods of the year. Relating movements to resource availability (*e.g.* Woinarski *et al.* 2000) would be especially beneficial.

6) **Dispersal of native fruits**

In Chapter 5, the presence of *C. monilifera* was not found to have any impact on the rate native fruits were removed from experimental stations. In this chapter it was suggested that the rate of removal of fruits of additional plant species should be investigated. However, few native plant species have fruiting strategies likely to be affected by *C. monilifera*, and the facultative relationships between frugivorous birds and plants also suggest that disperser limitation is unlikely. Consequently, this research would probably be of much lower priority that the other topics identified above.

8.5 Conclusion

In this study, I have investigated a number of aspects of the effect invasion of native vegetation by *C. monilifera* is having on birds and bird-plant interactions in coastal NSW, Australia. The fruits of *C. monilifera* are bird-dispersed, and *C. monilifera* fruit size and nutrient content are within the range of co-occurring indigenous fruits, although the time of year of peak fruit production occurs when few indigenous fruits are available (autumn-winter). *Chrysanthemoides monilifera* invasion dramatically alters the pattern of fruit availability in coastal vegetation, from being naturally fruit-rich over spring-summer to being fruit-rich over much of the year. Native coastal vegetation is nectar-rich over autumn and winter, which plays an important role in structuring the native bird community, and which may be being affected by *C. monilifera* invasion and other processes. Despite these changes to the availability of resources used by birds, *C. monilifera* invasion does not appear to have substantially disrupted the interactions between native plants and frugivores. This is largely due to facultative relationship between plants and frugivores, especially the flexibility of diet for many of the more abundant coastal birds. *Chrysanthemoides monilifera* also differs sufficiently in fruit characteristics (particularly phenology) from many native plant species to be likely competitors for dispersers.

Dominance of the vegetation by *C. monilifera* is having a substantial effect on coastal bird communities. The abundance of birds overall, of individual species and of groups of birds with similar resource-use patterns have been affected. Bird species that are more reliant on invertebrate and nectar resources are most affected, suggesting that these species and their interactions may be more substantially affected by *C. monilifera* invasion than generalist frugivores and seed dispersal.

I have also investigated the impacts on birds of one of the main methods used to control *C. monilifera* - aerial application of herbicide. It is important to examine these potential impacts, as it is possible that further loss of biodiversity may be instigated by a management measure designed for conservation benefit. Some birds decreased in abundance immediately after *C. monilifera* death, although these were only generalist frugivores that have in many cases benefited from broad-scale landscape modification.

Finally, I have identified several management and research actions that would either improve current management of coastal communities for biodiversity conservation or collect useful information for this task. The most important of these, is that management of *C. monilifera* should be considered as part of an overall strategy for the conservation of coastal vegetation and fauna, rather than control of *C. monilifera* being the main focus of management efforts.

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Appendices

A1. Published paper

A published paper arising from this work is included:

Gosper, C.R. (1999) Plant food resources of birds in coastal dune communities in New South Wales. *Corella* **23**, 53-62.

[NB - Not included in digital version]

Appendix A2 Survey site location and vegetation composition

A2.1 Maps of survey site location

Base Maps for Figures A2-A4 sourced from topographic maps produced by Land and Property Information, Dept. of Information, Technology and Management, Bathurst, NSW. Latitude and longitude values for each site are given in Table A1.

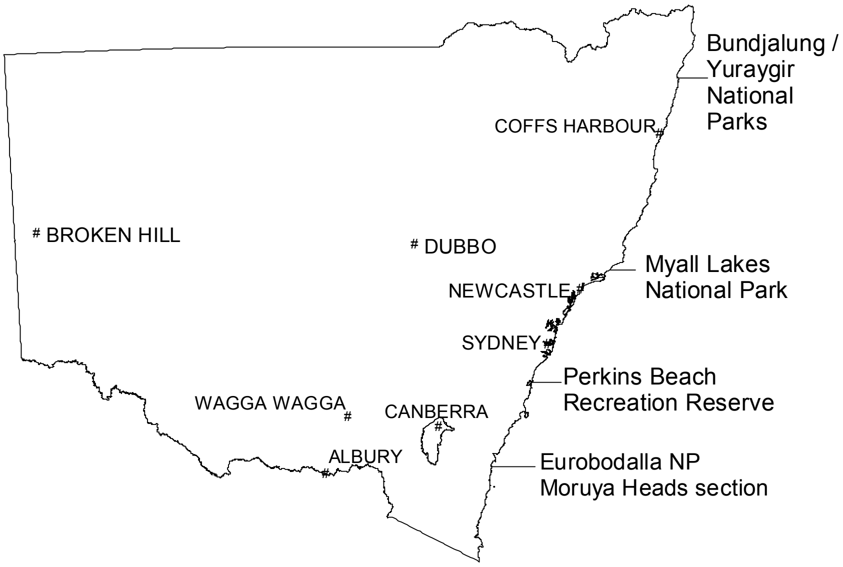

FIGURE A1.Distribution of the main study locations along the coast of the state of New South Wales, Australia

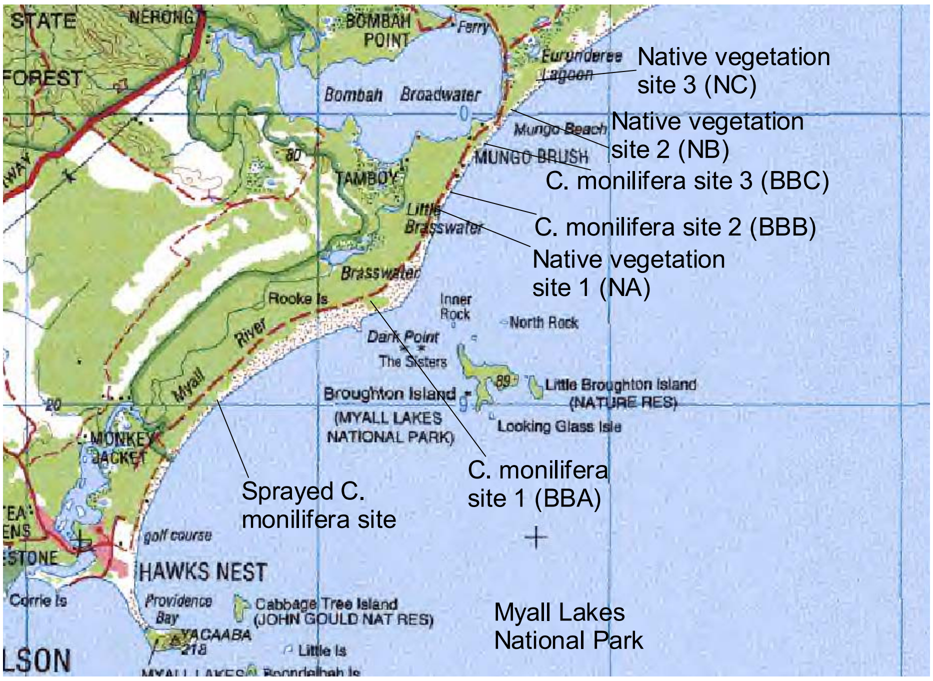

FIGURE A2.Topographic map of the Myall Lakes National Park area

Sites of various treatments/habitats used for the work in Chapter 6 and 7 are marked. Scale indicated by 10km grid over map.

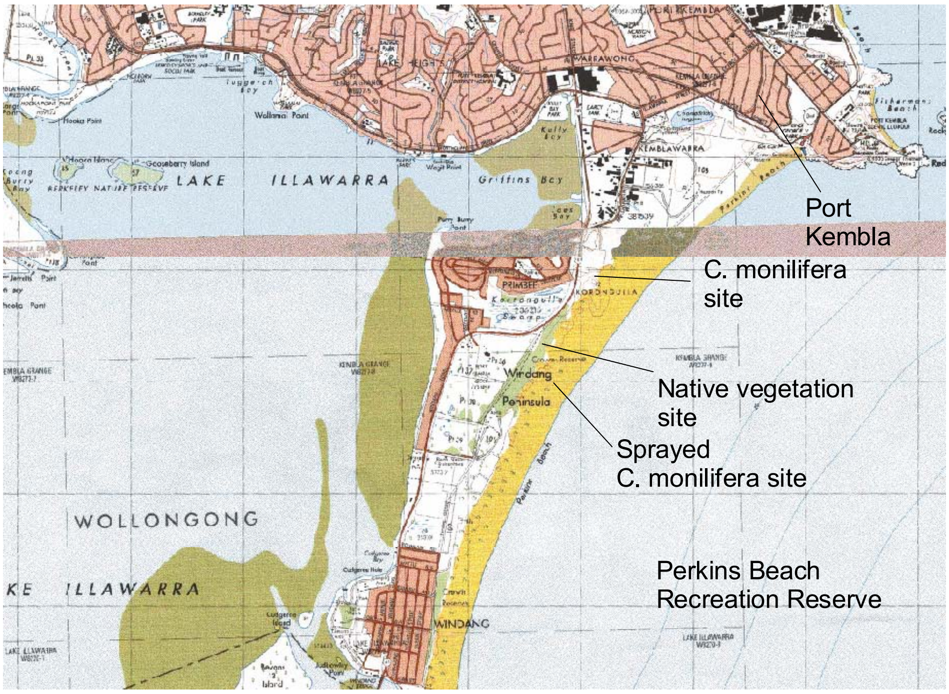

FIGURE A3.Topographic map of the Perkins Beach Recreation Reserve area

Sites of various treatments/habitats used for the work in Chapter 6 and 7 are marked. Scale indicated by 1km grid over map

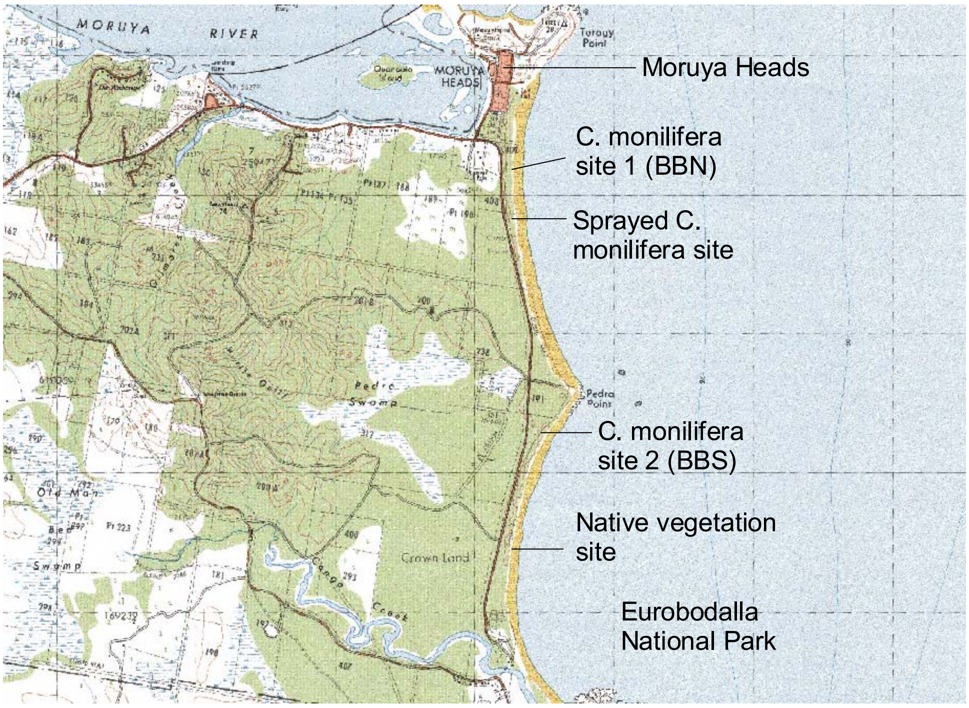

FIGURE A4.Topographic map of the Eurobodalla National Park area

Sites of various treatments/habitats used for the work in Chapter 6 and 7 marked. Scale indicated by 1km grid over map

A2.2 Habitat assessment

At all survey sites used for the assessment of the effect of *C. monilifera* removal and invasion on bird communities (Chapters 6 and 7), a range of habitat variables were measured. These were collected with the aim of providing additional confirmatory attributes for multivariate analyses, and thus contributing to explaining patterns in the survey data. Additional information of the distribution of survey points and site location are provided in the Methods sections of the relevant chapters.

Ten habitat assessment samples in each site were completed over a ten metre radius circle around a central point. This work was completed after the end of the bird surveys comparing native and *C. monilifera* vegetation. Assessments were not conducted on sites aerially treated with herbicide prior to this event. Consequently, the vegetation composition of these sites prior to herbicide application has to be pieced together from information collected in the assessments (for species not affected by the herbicide, *e.g. Banksia integrifolia*), with the presumption that these areas had similar levels of *C. monilifera* infestation to nearby untreated areas. Additionally, the habitat assessments at herbicide-treated sites were conducted some time after treatment (approximately 12 months at ML and ENP, two years at PB). Consequently, the cover of *C. monilifera* may have recovered somewhat from newly germinated plants by this time and so the cover of *C. monilifera* at the time of post-*C. monilifera* death surveys (Chapter 6) may be overestimated using these figures. Likewise, the native vegetation site at PB was not assessed until 12 months after bird surveys at that site had been completed (Chapter 7), and significant new infestations of *C. monilifera* were noted at this time.

The vegetation attributes measured were the percentage cover of the plot area of a number of plant species either frequent in coastal vegetation and/or thought to be highly significant for birds, and the total number of plant species within the plot. Cover of plants measured separately were *C. monilifera, B. integrifolia, B. serrata, Eucalyptus* spp., *Monotoca elliptica, Leucopogon parviflorus, Elaeocarpus* spp., assorted littoral rainforest species additional to *Elaeocarpus* spp., *Scaevola calendulacea, Hibbertia scandens, Acacia* spp., *Leptospermum laevigatum* and invasive species other than *C. monilifera*. Structural vegetation attributes measured were the projective foliage cover of vegetation layers (later classified into trees > 5m in height; tall shrubs 3-5m, shrubs 0.5-3m; groundcover <0.5m); cover of the ground surface by sand/soil, leaf litter and logs; topographic position (fore, mid or hind-dune or swale); and disturbance (time since fire, sand mining, herbicide treatment where relevant and if known). The structural type of the vegetation and adjoining (inland) vegetation structural type were also recorded.

In Table A1, the data collected has been re-arranged or combined in some cases for analysis. Attributes that were not considered relevant for analysis were excluded. At this stage, no additional analysis has been conducted on the habitat assessment data other than for use in explaining the pattern of occurrence of birds (Chapter 6 and 7).

Table A1 - Vegetation structural and floristic characteristics and location of bird survey sites

All measures are the average per 10m radius circle. Species included in the composite vegetation groups were: *Eucalyptus* spp. = *E. botryoides, E. pilularis, E. robusta, Corymbia gummifera.* Littoral rainforest species = different species, but commonly *Cupaniopsis anacardioides, Elaeocarpus reticulatus, Acmena smithii, Pittosporum undulatum, Polyscias elegans, Notelaea longifolia, Alphitonia excelsa, Breynia oblongifolia, Rapanea variabilis, Marsdenia rostrata, Kennedia rubicunda*, *Pandorea pandorana. Acacia* spp. = *A. sophorae, A. terminalis, A. saligna* (naturalised), *A. suaveolens.* Other invasive species = *Lantana camara, Solanum nigrum*

Habitat / Treatment	**Aerially herbicide-treated**			**Dominated by *Chrysanthemoides monilifera***						**Native coastal vegetation**				
Location	ML	PB	ENP		ML		PB	ENP			ML		PB	ENP
Site	MLS	PBS	ENPS	MLBA	MLBB	MLBC	PBB	EBBN	EBBS	MLNA	MLNB	MLNC	PBN	EBPN
Decimal latitude	-32.632	-34.511	35.920	-32.596	-32.559	-32.559	-34.506	-35.922	-35.937	-32.561	-32.54	-32.533	-34.510	-35.945
Decimal longitude	152.21	150.88	150.16	152.28	152.30	152.31	150.89	150.16	150.16	152.30	152.32	152.33	150.88	150.15
Floristics - % cover														
C. monilifera	10.00	9.30	12.25	38.00	33.25	31.50	69.60	60.00	57.50	1.50	2.50	1.75	2.10	6.00
Banksia integrifolia	0	4.60	15.00	3.75	2.00	2.25	0	17.25	17.25	8.25	13.75	12.50	5.00	15.00
B. serrata	6.75	0	0.25	0	8.00	4.00	0	0	2.00	0.50	9.75	10.50	4.60	1.75
Total *Banksia*	*6.75*	*4.60*	*15.25*	*3.75*	*10.00*	*6.25*	*0*	*17.25*	*19.25*	*8.75*	*23.50*	*23.00*	*9.60*	*16.75*
Eucalyptus spp.	0	0	1.50	0	0	0	5.7	0	3.80	0	0	0	28.60	0
Monotoca elliptica	3.25	0	1.25	0.50	2.00	2.75	0	1.50	4.25	22.75	4.75	8.75	1.10	29.5
Leucopogon parviflorus	8.75	0	8.50	8.50	8.25	7.50	0	19.00	9.50	0.50	6.75	0	0	2.25
Total epacrids	*12.00*	*0*	*9.75*	*9.00*	*10.25*	*10.25*	*0*	*20.50*	*13.75*	*23.25*	*11.50*	*8.75*	*1.10*	*31.75*
Littoral rainforest species	0	0	0.5	0	2.25	3.00	1.77	0	0	8.50	31.60	25.00	12.53	5.00
Acacia spp.	15.00	26.10	12.25	6.25	9.75	8.25	26.10	19.50	8.75	10.75	9.75	3.00	24.80	12.50
Leptospermum laevigatum	21.75	10.00	0	21.75	19.50	19.35	9.60	0	0	31.80	18.00	30.20	5.70	0
Scaevola calendulacea	2.25	0	0	3.50	15.50	20.25	0	0	0	2.75	15.00	0.50	0	0
Hibbertia scandens	11.25	0	0	1.00	1.75	3.25	0	0	0	2.50	3.50	1.00	0	0
Total groundcover twiners	*13.50*	*0*	*0*	*4.50*	*17.25*	*23.25*	*0*	*0*	*0*	*5.25*	*18.50*	*1.50*	*0*	*0*
Other invasive species	0	1.40	0	0	0	0	2.50	0	0	0	0	0	13.20	0
Av. Species richness	11.1	5.1	8.2	7.8	11.8	12.4	5.4	6.4	8.7	17.7	20.7	23.2	15.6	11.3
Structure – % cover														
Tree (height 5m+)	0	0	6.3	3.0	0	0	8.6	9.0	8.3	2.3	19.0	33.3	33.4	13.4
Tall Shrub (3-5m)	14.0	14.6	2.6	15.3	19.3	0	8.1	0	9.1	29.2	29.5	13.5	0.6	29.1
Total veg. >3m	*14.0*	*14.6*	*8.9*	*18.3*	*19.3*	*0*	*16.7*	*9.0*	*17.4*	*31.5*	*48.5*	*46.8*	*34.0*	*42.5*
Low Shrub (0.5-3m)	36.5	25.0	37.2	54.0	48.0	68.0	75.0	85.6	74.5	43.7	19.1	25.8	40.0	13.2
Ground cover veg.	17.2	7.0	23.0	13.5	25.5	33.5	5.1	9.5	24.0	14.0	29.0	31.0	37.9	42.5
Bare sand	30.5	38.7	37.0	31.0	28.5	25.0	33.6	50.0	26.0	27.5	22.0	14.0	19.3	17.0
Leaf litter and logs	52.2	54.3	40.0	55.5	45.5	41.5	61.3	39.5	50.0	58.5	49.0	55.0	41.4	41.5

Appendix A3 Classification of bird species into resource-use groups

(A. Species recorded during bird surveys for Chapter 6)

Birds were divided into five groups for analyses in Chapter 6: frugivores of *C. monilifera* fruits; frugivores/granivores not recorded consuming *C. monilifera* fruits; primarily nectarivorous species; primarily insectivorous species (based on data collected for Chapter 3); and carnivores. More details on classification in section 6.2.4. # = introduced bird species

Frugivores of *Chrysanthemoides monilifera*	
Crimson Rosella	*Platycercus elegans*
Eastern Rosella	*P. eximius*
Red Wattlebird	*Anthochaera carunculata*
Little Wattlebird	*A. chrysoptera*
Lewin's Honeyeater	*Meliphaga lewinii*
Yellow-faced Honeyeater	*Lichenostomus chrysops*
Satin Bowerbird	*Ptilonorhynchus violaceus*
Pied Currawong	*Strepera graculina*
Forest Raven	*Corvus tasmanicus*
# Red-whiskered Bulbul	*Pycnonotus jocosus*
Mistletoebird	*Dicaeum hirundinaceum*
# House Sparrow	*Passer domesticus*
Silvereye	*Zosterops lateralis*
# Common Starling	*Sturnus vulgaris*
Primarily insectivores	
Shining Bronze-Cuckoo	*Chrysococcyx lucidus*
White-browed Scrubwren	*Sericornis frontalis*
Superb Fairy-wren	*Malurus cyaneus*
Variegated Fairy-wren	*M. lamberti*
Brown Thornbill	*Acanthiza pusilla*
Yellow-rumped Thornbill	*A. chrysorrhoa*
Yellow Thornbill	*A. nana*
Striated Thornbill	*A. lineata*
Eastern Yellow Robin	*Eopsaltria australis*
Eastern Whipbird	*Psophodes olivaceus*
Golden Whistler	*Pachycephala pectoralis*
Grey Shrike-thrush	*Colluricincla harmonica*
Grey Fantail	*Rhipidura fuliginosa*
Welcome Swallow	*Hirundo neoxena*
Tree Martin	*H. nigricans*
Primarily nectarivores	
Rainbow Lorikeet	*Trichoglossus haematodus*
Scaly-breasted Lorikeet	*T. chlorolepidotus*
White-naped Honeyeater	*Melithreptus lunatus*
New Holland Honeyeater	*Phylidonyris novaehollandiae*
White-cheeked Honeyeater	*P. nigra*
Eastern Spinebill	*Acanthorhynchus tenuirostris*
Carnivores	
Osprey	*Pandion haliaetus*
Whistling Kite	*Haliastur sphenurus*
White-bellied Sea-Eagle	*Haliaeetus leucogaster*
Swamp Harrier	*Circus approximans*
Brown Goshawk	*Accipiter fasciatus*
Collared Sparrowhawk	*A. cirrhocephalus*
Grey Butcherbird	*Cracticus torquatus*
Frugivores/granivores (but not of *C. monilifera*)	
Brown Quail	*Coturnix ypsilophora*
Bar-shouldered Dove	*Geopelia humeralis*
Galah	*Cacatua roseicapilla*
# Common Blackbird	*Turdus merula*

Appendix A4 Classification of bird species into resource-use groups

(B. Species recorded during bird surveys for Chapter 7)

Birds were allocated to the following groups for analyses in Chapter 7. Frugivores of *C. monilifera* (Chapter 4); frugivores/granivores not recorded consuming *C. monilifera* fruits; nectarivores of *Banksia integrifolia*; nectarivores of *Eucalyptus* spp. (Chapter 3); primarily insectivorous species of the canopy and understorey; primarily insectivorous species that forage above the canopy; and carnivores and/or scavengers. Bird species could be classified into more than one resource-use group. Further explanation of classification given in section 7.2.4. # = introduced bird species. [1] = no resource use recorded for Tawny-crowned Honeyeater, hence classified with congeneric species.

Frugivores of *Chrysanthemoides monilifera*

Crimson Rosella	*Platycercus elegans*
Eastern Rosella	*P. eximius*
Common Koel	*Eudynamys scolopacea*
Superb Fairy-wren	*Malurus cyaneus*
Red Wattlebird	*Anthochaera carunculata*
Little Wattlebird	*A. chrysoptera*
Lewin's Honeyeater	*Meliphaga lewinii*
Yellow-faced Honeyeater	*Lichenostomus chrysops*
White-cheeked Honeyeater	*Phylidonyris nigra*
Black-faced Cuckoo-shrike	*Coracina novaehollandiae*
Olive-backed Oriole	*Oriolus sagittatus*
Figbird	*Specotheres viridis*
Regent Bowerbird	*Sericulus chrysocrphalus*
Satin Bowerbird	*Ptilonorhynchus violaceus*
Pied Currawong	*Strepera graculina*
Australian Raven	*Corvus coronoides*
Forest Raven	*C. tasmanicus*
# Red-whiskered Bulbul	*Pycnonotus jocosus*
Mistletoebird	*Dicaeum hirundinaceum*
# House Sparrow	*Passer domesticus*
Silvereye	*Zosterops lateralis*
# Common Starling	*Sturnus vulgaris*
# Common Blackbird	*Turdus merula*

Other frugivores/granivores

Brown Quail	*Coturnix ypsilophora*
# Spotted Turtle-dove	*Strepopelia chinensis*
Bar-shouldered Dove	*Geopelia humeralis*
Emerald Dove	*Chalcophaps indica*
Common Bronzewing	*Phaps chalcoptera*
Brush Bronzewing	*P. elegans*
Wonga Pigeon	*Leucosarcia melanoleuca*
Yellow-tailed Black Cockatoo	*Calyptorhynchus funereus*
Galah	*Cacatua roseicapilla*
Red-browed Finch	*Neochmia temporalis*

Nectarivores of *B. integrifolia*

Scaly-breasted Lorikeet	*Trichoglossus chlorolepidotus*
Rainbow Lorikeet	*T. haematodus*
Crimson Rosella	*Platycercus elegans*
Eastern Rosella	*P. eximius*
Variegated Fairy-wren	*Malurus lamberti*
Brown Thornbill	*Acanthiza pusilla*
Red Wattlebird	*Anthochaera carunculata*
Little Wattlebird	*A. chrysoptera*
Lewin's Honeyeater	*Meliphaga lewinii*
Yellow-faced Honeyeater	*Lichenostomus chrysops*
Brown-headed Honeyeater	*Melithreptus brevirostris*
White-naped Honeyeater	*M. lunatus*
New Holland Honeyeater	*Phylidonyris novaehollandiae*
White-cheeked Honeyeater	*P. nigra*
[1]Tawny-crowned Honeyeater	*P. melanops*
Eastern Spinebill	*Acanthorhynchus tenuirostris*
Scarlet Honeyeater	*Myzomela sanguinolenta*
Brown Honeyeater	*Lichmera indistincta*
Regent Bowerbird	*Sericulus chrysocrphalus*
Satin Bowerbird	*Ptilonorhynchus violaceus*
Pied Currawong	*Strepera graculina*
Forest Raven	*Corvus tasmanicus*
Silvereye	*Zosterops lateralis*

Nectarivores of *Eucalyptus* spp.

Scaly-breasted Lorikeet	*Trichoglossus. chlorolepidotus*
Rainbow Lorikeet	*T. haematodus*
Swift Parrot	*Lathamus discolor*
White-throated Treecreeper	*Cormobates leucophaeus*
Red Wattlebird	*Anthochaera carunculata*
Little Wattlebird	*A. chrysoptera*
Noisy Friarbird	*Philemon corniculaus*
Lewin's Honeyeater	*Meliphaga lewinii*
Yellow-faced Honeyeater	*Lichenostomus chrysops*
Brown-headed Honeyeater	*Melithreptus brevirostris*
White-naped Honeyeater	*M. lunatus*
New Holland Honeyeater	*Phylidonyris novaehollandiae*
White-cheeked Honeyeater	*P. nigra*
Eastern Spinebill	*Acanthorhynchus tenuirostris*
Scarlet Honeyeater	*Myzomela sanguinolenta*
# Red-whiskered Bulbul	*Pycnonotus jocosus*
Spangled Drongo	*Dicrurus bracteatus*
Silvereye	*Zosterops lateralis*

Carnivores/scavengers

Whistling Kite	*Haliastur sphenurus*
White-bellied Sea-Eagle	*Haliaeetus leucogaster*
Brown Goshawk	*Accipiter fasciatus*
Grey Goshawk	*A. novaehollandiae*
Collared Sparrowhawk	*A. cirrhocephalus*
Swamp Harrier	*Circus approximans*
Brown Falcon	*Falco berigora*
Nankeen Kestrel	*F. cenchroides*
Southern Boobook	*Ninox novaehollandiae*
Laughing Kookaburra	*Dacelo novaeguinese*
Sacred Kingfisher	*Todiramphus sanctus*
Grey Butcherbird	*Cracticus torquatus*
Australian Magpie	*Gymnorhina tibicen*
Pied Currawong	*Strepera graculina*
Australian Raven	*Corvus coronoides*
Forest Raven	*C. tasmanicus*

Canopy and understorey insectivores

Fan-tailed Cuckoo	*Cacomantis flabellifomis*
Shining Bronze-Cuckoo	*Chrysococcyx lucidus*
White-throated Treecreeper	*Cormobates leucophaeus*
Superb Fairy-wren	*Malurus cyaneus*
Variegated Fairy-wren	*M. lamberti*
Southern Emu-wren	*Stipiturus malachurus*
Spotted Pardalote	*Pardalotus punctatus*
White-browed Scrubwren	*Sericornis frontalis*
Brown Gerygone	*Gerygone mouki*
Brown Thornbill	*Acanthiza pusilla*
Yellow Thornbill	*A. nana*
Striated Thornbill	*A. lineata*
Eastern Yellow Robin	*Eopsaltria australis*
Eastern Whipbird	*Psophodes olivaceus*
Golden Whistler	*Pachycephala pectoralis*
Rufous Whistler	*P. rufiventris*
Grey Shrike-thrush	*Colluricincla harmonica*
Black-faced Monarch	*Monarcha melanopsis*
Leaden Flycatcher	*Myiagra rubicula*
Satin Flycatcher	*M. cyanoleuca*
Willie Wagtail	*Rhipidura leucophrys*
Rufous Fantail	*R. rufifrons*
Grey Fantail	*R. fuliginosa*
Black-faced Cuckoo-shrike	*Coracina novaehollandiae*
White-bellied Cuckoo-shrike	*C. papuensis*
Australian Magpie-lark	*Grallina cyanoleuca*
Spangled Drongo	*Dicrurus bracteatus*

Aerial insectivores

White-throated Needletail	*Hirundapus caudacutus*
Dollarbird	*Eurystomus orientalis*
Dusky Woodswallow	*Artamus cyanopterus*
Welcome Swallow	*Hirundo neoxena*
Tree Martin	*H. nigricans*
Fairy Martin	*H. ariel*