

**QUANTIFYING AND PREDICTING
MAMMALIAN HERBIVORE DAMAGE IN
TASMANIAN EUCALYPT PLANTATIONS**

by

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ABSTRACT

In Tasmania, the Bennett's wallaby (*Macropus rufogriseus rufogriseus*), Tasmanian pademelon (*Thylogale billardierii*), brushtail possum (*Trichosurus vulpecula*) and European rabbit (*Oryctolagus cuniculus*) browse eucalypt seedlings growing in forestry plantations. This damage is thought to reduce plantation productivity by reducing seedling growth, reducing seedling survival and changing seedling form. Forestry companies attempt to reduce damage by reducing herbivore populations with 1080 poison. This control strategy, as it is currently employed, is not without problems. No real data exist relating the severity of browsing damage to losses in productivity, herbivore abundance or plantation characteristics. Thus, control tends to be carried out at most plantations without regard to recouped benefits. Furthermore, the use of 1080 is strongly criticised by some community groups. Unfortunately, existing alternatives are either ineffective or too costly to use on a large scale.

This study aimed to quantify the spatial distribution, effect and cost of browsing damage; determine relationships between herbivore abundance and damage; and determine environmental variables important in influencing herbivore abundance and damage. From 1994 to 1997 data were collected at 35 forestry plantations, 32 planted with *Eucalyptus nitens* and three with *Eucalyptus globulus*. Forestry companies carried out poisoning at 28 of the plantations. Growth and damage data were collected at planting and then at six and 12 months after planting. Herbivore abundance was assessed at each plantation over 320 days using scat plots established at planting. Plantations and adjoining cover habitats were characterised by measuring environmental variables. At seven of the plantations, fenced control plots were established at planting (1996). Growth rate, survival

and number of leading shoots for these seedlings were compared to unfenced seedlings which were also monitored for damage.

Damage severity varied between plantations, ranging from low (17% of seedlings damaged at 12 months after planting with a mean browse score of 0.5) to severe (100% of seedlings damaged at 12 months after planting with a mean browse score of 4.7). Damage reduced seedling growth but did not influence seedling survival. The amount by which growth was reduced was linearly related to damage severity, but was only reduced significantly when damage exceeded a critical level. Unfenced seedlings developed more leading shoots than fenced seedlings at plantations with intermediate levels of damage. Severity of browsing damage at each plantation was linearly related to cumulative density of possum scats. For most herbivore species cumulative scat density at each plantation was negatively related to frequency of poisoning. A regression model incorporating possum scat density, grass cover and fern cover explained 71% of between-site variation in damage. A second regression model, incorporating variables that could be measured prior to planting, explained 52% of between-site variation. Important variables in this model were the area to perimeter ratio of the plantation, the proportion of the perimeter that was adjacent to cover habitat, canopy closure in adjacent cover habitat and the amount of vegetation at ground level in adjacent cover habitats.

These results can be used to assign meaningful measures of loss to observed damage and to identify 'acceptable' damage levels. Additionally, relationships between herbivore abundance and damage, and plantation characteristics and damage have potential for use in predicting damage severity at plantations.

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CHAPTER 1

INTRODUCTION

Since the early history of large scale commercial tree farming in Tasmania, Australia, seedlings have been observed to be damaged by native and introduced herbivorous mammals (Anon 1958). This browsing damage is most evident during the early stages of seedling establishment (less than one year since planting) and typically takes the form of damaged or removed foliage, broken stems and uprooted plants (Statham 1983; Coleman *et al.* 1997). This damage is thought to reduce seedling growth rates, reduce seedling survival and change the form of seedlings so that tree quality at harvesting is reduced. The net effect is thought to be a reduction in plantation productivity and losses in subsequent economic return. In 1990, Forestry Tasmania estimated the costs of browsing damage at \$2.8 million per annum (as reported in Coleman *et al.* 1997).

Ever since studies of browsing damage were first reported in the 1960s (Mollison 1960; Gilbert 1961; Cremer 1969), problematic browsing damage has generally been attributed to four species (Mollison 1960; Gilbert 1961; Cremer 1969; Statham 1983; Coleman *et al.* 1997). These are the Bennett's wallaby (*Macropus rufogriseus rufogriseus*), the Tasmanian pademelon (*Thylogale billardierii*), the common brushtail possum (*Trichosurus vulpecula*) and the European rabbit (*Oryctolagus cuniculus*). The rabbit is the only species which is not native to Tasmania.

In the following sections a brief overview is given of the techniques that are used to control browsing damage, some recent criticisms of vertebrate pest control programs are discussed and the aims of this study are outlined.

1.1 CONTROLLING BROWSING DAMAGE IN TASMANIAN FORESTRY PLANTATIONS

For over 50 years managers of Tasmanian forestry plantations have actively attempted to reduce the potential impact of 'pest' species. At present, managers are reported to spend approximately 10% of their annual maintenance budget on mammalian herbivore control (Coleman *et al.* 1997). Control is most frequently attempted using lethal means. Historically, snaring and commercial shooting operations have been favoured methods (Driessen 1992). However, snaring is now illegal and collapse of the skin trade has severely reduced the viability of large scale shooting operations (Driessen 1992). Today, the most widely utilised method of control is the use of the poison, sodium monofluoroacetate (FCH_2COONa), to reduce the localised abundance of deleterious target species.

Sodium monofluoroacetate, more commonly referred to as 1080, is an organic fluoride which occurs in two genera of Australian plants (Oelrichs and McEwan 1961; Aplin 1971; Twigg 1994) as well as African and Brazilian species (Statham 1983; Coleman *et al.* 1997). If ingested, it is toxic because it disrupts the Krebs cycle, blocking the metabolism of glucose for energy (Atzert 1971). Due to its low cost, ease of application, relatively rapid action and apparent effectiveness, it has become a popular poison for pest control throughout the world (eg. Atzert 1971; Kinnear *et al.* 1988; Lazarus 1989; Thomas 1994; Staples and McPhee 1995; Thomas and Hickling 1995).

In Tasmanian forestry plantations, poisoning operations are typically carried out shortly before seedlings are planted. Baits are distributed from roads around the perimeter of plantation coupes. Poisoning usually follows a period (about 2 weeks) of 'free feeding' with baits which have not been poisoned. Where seedling damage is monitored and judged to be excessive, further post-planting poisoning may be carried out. Most plantation coupes are poisoned at least once. Coleman *et al.* (1997) report that approximately 5-6 kg of 1080 concentrate is used by Tasmania forestry companies during any given year. In the 1994/1995 period, 614 separate poisoning operations were carried out (Coleman *et al.* 1997).

1.1.1 Concerns about 1080

In recent years community groups have raised concerns regarding the use of 1080, both in Tasmania and elsewhere in the world (Eason 1995; Coleman *et al.* 1997). Animal welfare groups, conservationists and environmental activists have actively lobbied for a complete ban on the use of 1080. In Tasmania, the opposition to poisoning has been particularly strong because several of the traditionally targeted ‘pest’ species are native animals. In response to the concerns raised by these groups, the Tasmanian Government established the Browsing Animal Research Council (BARC) in 1993. As part of its brief, the Council was to :

- i) review the current use of 1080
- ii) organise and direct the undertaking of a research programme with the twin aims of defining the effects of 1080 on populations of native animal species and developing alternative control methods (Coleman *et al.* 1997).

1.1.2 Non-lethal alternatives to 1080

The efficacy of a range of non-lethal control techniques for the mitigation of pest damage have been examined in studies worldwide. Those that pertain to the protection of commercially farmed tree species are described below.

Physical barriers

The erection of barriers to physically prevent animals from gaining access to plants has been a long used method of crop protection. In forestry areas, these barriers typically take the form of fences or tree guards. In Tasmania, considerable effort has been invested into designing an electric fence suitable for large scale use. Gregory (1988) was able to substantially reduce the abundance of herbivores in an agricultural area using an electric fence which had been specifically designed for the purpose. Using a modified fence, Statham (1994) was able to improve on Gregory’s (1988) results. In one trial area, the signs of Bennett’s wallaby and

pademelon were reduced by 99 and 80% respectively. More traditional wire netting fences have also been effective at protecting some plantations. However, the high cost associated with these fences has meant that their use has generally been restricted to plantations of high value specialty timber such as blackwood (*Acacia melanoxylon*).

Tree guards are typically made from a flexible plastic and are designed to enclose individual seedlings. There are a variety of commercially produced tree guards available (Jenkin 1991, as cited in Montague 1993). Montague (1993) assessed the efficacy of various tree guards to prevent browsing of *Eucalyptus regnans* seedlings by a group of captive swamp wallabies (*Wallabia bicolor*). While the majority of guards he tested proved inadequate, he found that rigid plastic tubing provided good protection for seedlings. In the United States, Engeman *et al.* (1995a) have found that plastic mesh tubes provide excellent protection from gophers (*Thomomys spp.*).

Reduced palatability of seedlings

It has been observed that, for a given herbivore species, seedlings are not all of equal palatability or 'attractiveness'. Some seedlings appear to be preferred whilst others are not. It has often been suggested, for example, that trees which have been fertilised are more susceptible to browsing than trees which have not (Horne 1975; also see review by Gill 1992). The obvious implication of this for management is that herbivore damage might be reduced by planting seedlings which have an inherently low palatability or by 'artificially' reducing seedling palatability. Studies which have explored this potential have had encouraging results. Hood and Libby (1980) found that mule deer (*Odocoileus hemionus*) browsed seedlings from three provenances of *Pinus radiata* 8-19% less than seedlings from other provenances. Montague (1995) attempted to reduce the palatability of seedlings by the application of systemic additives including fertiliser, bitrex and selenium. He observed an encouraging, though non-significant, reduction in damage to the seedlings treated with selenium. In a recent Tasmanian

field study, Marsh (1998) compared the damage levels for five different *E. nitens* seed-lots. She observed that one of the seed-lots was consistently damaged less than the others.

Repellents

The principle behind repellent use for crop protection is that the palatability of a crop can be lowered by applying substances which are distasteful to the pest species. In the United States and Europe, the efficacy of several repellent formulations have been tested for the protection of commercial tree species. The repellents which have been tested include a range of commercially available formulations (Harris *et al.* 1983; Engeman *et al.* 1995a), the diluted urine of predators (Melchior and Leslie 1985; Epple *et al.* 1993; Engeman *et al.* 1995a) and traditional protein-based recipes (Harris *et al.* 1983; Engeman *et al.* 1995a). Repellent trials have also taken place within Australian plantation areas. Statham (1983) observed that browsing damage to eucalypt seedlings in Tasmanian plantations was reduced by between 18 and 26% if seedlings were treated with an egg powder formulation. In a captive trial Marks *et al.* (1995) used repellents, with some success, to reduce the amount of swamp wallaby and rabbit browsing damage on *Eucalyptus regnans* seedlings. However, in one of the few trials of systemic repellents, Montague (1995) was unable to report a significant reduction in damage to eucalypt seedlings by swamp wallabies.

1.1.3 Criticism of alternatives to 1080

In Tasmania, the general criticisms which are made of existing alternatives to 1080 are that they are either ineffective at preventing damage, or they are too expensive to use on a large scale (Coleman *et al.* 1997). Repellents do not always work and, when they do, their effects can be short lived, either because animals become habituated to the repellent, or it is washed off by rain (Morgan and Woolhouse 1993). Tree guards and fencing, though effective, are expensive and it is argued that they are too costly to use on a large scale. Suitable fences range in cost from around A\$2500 to A\$4500 per kilometre (Anon 1991; Coleman *et al.* 1997). Tree

guards can cost between A\$250 and A\$4500 per hectare (Anon 1991; Coleman *et al.* 1997). Even where economics and terrain allow their use, animal activity and theft mean that fences and guards require regular attention. Planting seedlings of low palatability holds promise, but further trials and more specific isolation of causal factors may be required.

By comparison, poisoning with 1080 generally costs less than \$100/ha and it is widely regarded by forestry managers as an effective means of controlling browsing damage.

1.2 RECENT CRITICISMS OF VERTEBRATE PEST CONTROL

Vertebrate pest control programs have been the subject of several, relatively recent reviews (Putman 1989; Braysher 1993; Hone 1994, 1995a; Norton G. 1995). The tone of these reviews has largely been critical and, despite having been written from quite different perspectives, the general criticisms made of vertebrate pest control programs are remarkably uniform. The following is a synthesis of the concerns raised by these authors.

The damage caused by a particular species, perceived as being a pest, is often poorly defined and/or poorly quantified. In some cases, quantifying the amount and/or the effect of damage that a so-called pest species does, reveals that the species' impact is actually negligible. In other words, the 'pest' is found not to be a pest at all. Harrison and Symes (1989), for example, reviewed the economic damage caused by mink (*Mustela vison*), a species regarded as a damaging pest in England and Wales, and found that economic damage was negligible. Putman (1986) found that highly seasonal and apparently severe damage caused by roe deer (*Capreolus capreolus*) did not significantly reduce the economic return from English cereal crops. In contrast, there are also many examples where quantification of damage has served to confirm that a particular species does have a significant negative effect on resources (e.g. Short 1985; Crawley 1989; see also reviews by Myers *et al.* 1994 and Thompson 1994).

The basic premise that vertebrate pest control should bring about some net gain (e.g. economic) is rarely qualified. Control programs are frequently implemented without weighing the potential benefits of the control against the costs of its implementation. It is possible that, even where a species is found to have a negative impact on a resource, the cost of trying to control the damage will actually outweigh the recouped benefit. In a comprehensive review, Hone (1994) observed that case studies did not provide uniform support for pest control. For example, Dolbeer (1981) applied a cost-benefit analysis to the damage caused in Ohio corn crops by black birds (*Agelaius phoeniceus*). He found that control was economically justifiable at only two out of the 21 fields surveyed.

There is rarely any consideration paid to the distribution of damage within the environment. many studies have shown that damage varies on both a spatial and temporal scale (Buckle *et al.* 1984; Hothem *et al.* 1988; Hannan and Whelan 1989; Hone 1995b). Consequently, expending control effort uniformly across sites and overtime is likely to be wasteful and/or ineffective (e.g. Dolbeer 1981). Control effort is wasted on those sites with a low potential for damage, while extra effort may be necessary at those sites with a high potential for damage. Control options need to be carefully evaluated for each specific situation (Braysher 1993; Choquenot and Lukins 1995).

Control programs tend to rely too heavily on lethal control measures. Lethal control often proves successful in the short term, but a continued and sometimes increasing effort is required in the longer term. The very nature of pest species means that any reduction in population levels, short of actual extinction, will be compensated for by increased fecundity, reduced mortality and immigration (Begon and Mortimer 1986; Begon *et al.* 1990; Driessen 1992). Thus, lethal control programs frequently require an ongoing investment of effort. Where poisons are regularly used as the means of reducing population levels, control efforts may actually have to increase over time. Individuals can acquire phenotypic resistance to the toxin if they are repeatedly exposed to sub-lethal

doses, or they may simply modify their behaviour so that they avoid the toxins (Hickling 1994; Morgan *et al.* 1995; Thomas and Hickling 1995; Hickling *et al.* 1998). Furthermore, poisons can act as powerful selecting agents so that whole populations evolve a genetic resistance. Many European populations of rodents are now resistant to warfarin, a poison widely used for their control in the 1950s (see reviews by Lazarus 1989; and Richards 1989).

Despite the heavy reliance on lethal control measures, the relationship between pest abundance and damage is rarely quantified. In many cases, managers assume that any reduction in pest abundance will bring about a corresponding decrease in damage. This assumption may hold true for those systems where the relationship between pest abundance and damage is linear. However, if the relationship is non-linear (e.g. Hone 1995b), a control effort which brings about a reduction in pest abundance may not actually reduce damage by a significant amount (Hone 1994, 1995a,b).

The effectiveness of control efforts is rarely quantified or monitored following implementation (Braysher 1993; Hone 1995a). When post control evaluation is carried out, the success or failure of the effort is frequently assessed in terms of pest abundance. However, as has already been discussed, this is only valid where the relationship between the abundance of the pest and damage is known. The end result may be that costly but ineffective and/or inappropriate control efforts are repeatedly carried out. The dangers of measuring the success of a control operation, in terms of reduced pest numbers is well illustrated by Foran *et al.* (1985), who observed that pasture biomass did not increase following rabbit control, despite significant reduction in rabbit numbers.

1.2.1 Criticisms of vertebrate pest control in relation to Tasmanian forestry

Many of the above criticisms could also be levelled at pest control programs as they are presently carried out in Tasmanian forestry plantations. Though it has

long been recognised that several vertebrate herbivores damage eucalypt seedlings in plantations, the effects of this damage have been poorly quantified, particularly in regards to financial costs. At present, no formalised framework exists for assigning cost estimates to observed levels of browsing damage. On those occasions where damage is so severe that replanting is required, the costs of herbivore damage become obvious (e.g. price of seedlings, wages, delayed time to harvest). However, at present it is not possible to quantify the long-term costs of any lesser degree of damage. This is a problem which is not likely to be overcome easily. The long rotational lengths of forestry crops (>10 years) means that very long term studies may be required to determine the losses in productivity caused by early browsing damage.

Though damage is observed to vary in both space and time, there is presently little knowledge regarding the factors which make some plantations more prone to damage than others. Though foresters may often have a 'feel' for which plantations are likely to be severely damaged, there are essentially no hard data relating plantation characteristics to damage levels. Indeed, even the fundamental link between herbivore abundance and damage has not yet been demonstrated. With such gaps in knowledge, foresters are unable to predict, with any degree of certainty, the degree to which a particular plantation will be damaged. Coupled with the poorly understood relationship between damage and cost, this means that control programs are invariably carried out without any real idea of how much benefit will be derived from their implementation. Control tends to be carried out at all plantations 'just in case', even though there may not always be a net benefit.

The control programs implemented in Tasmanian forestry plantations rely almost exclusively on reducing herbivore populations through the use of 1080 poison. A major criticism of such a control strategy is that any derived benefit tends to be only temporary (Putman 1989). However, for the Tasmanian situation this is actually a benefit rather than a problem. Of the species which have been implicated in causing damage, only the arboreal brushtail possum is able to cause

significant, deleterious damage to seedlings once they are beyond 1.5 m in height. Thus, seedlings only need to be protected from herbivores for a relatively short period of time (usually less than two years). Since the herbivores in question are valued native species a temporary, rather than a long term, reduction in numbers is the preferred option. Furthermore, the problem of populations developing a genetic resistance to 1080 may also be avoided because, for any given site, operations tend to be carried out over a relatively short period of time. In effect, the selection pressure for 1080 tolerance may not be very strong.

Most forestry companies carry out some form of post-control damage monitoring. At some plantations, further control operations may be carried out if damage levels are deemed to become excessive. Thus, the effectiveness of each control operation is monitored to some extent. However, this monitoring tends to be qualitative rather than quantitative. This makes it impossible to assign costs to damage levels and it severely reduces the value of between-site comparisons. From a scientific point of view this is also a great pity because large and valuable data sets could be generated in relatively short periods of time.

The goals of control operations tend to be poorly defined. The general aim is to 'reduce damage', but no concrete guidelines exist as to what levels of damage should be aimed for. Furthermore, short of the complete decimation of seedlings, there are no real criteria by which a control program can be deemed to have failed. Even if a plantation is heavily damaged despite control operations, the result could be justified by reasoning that 'things would have been much worse if nothing had been done'. Such hypotheses are not made from a position of strength, however, because there is a lack of empirical damage data for plantations where control operations have not been carried out. Any ideas about 'what could have been' are consequently rather nebulous.

1.3 THE AIMS OF THIS STUDY

Broadly, this project aimed to garner a range of data that could be used to improve current pest control techniques in Tasmania and improve the viability of alternatives to 1080. Specifically the aims of the project were to :

- i) quantify the scope of the browsing problem in terms of spatial distribution, effect and cost;
- ii) quantify relationships between herbivore abundance and damage;
- iii) determine which environmental characteristics are important in influencing severity of damage; and
- iv) explore the potential for predicting damage.

CHAPTER 2

STUDY ANIMALS, STUDY SITES AND FREQUENTLY USED TECHNIQUES

2.1 STUDY ANIMALS

In the following sections a brief summary is given of the ecology of the four herbivore species that are considered to damage eucalypt seedlings in Tasmanian forestry plantations.

2.1.1 The Bennett's wallaby (*Macropus rufogriseus rufogriseus*)

The Bennett's wallaby is a medium sized, native marsupial belonging to the family Macropodidae. On average, adult males weigh around 20 kg (Calaby 1983). As is typical of the sexually dimorphic macropod species, females tend to be considerably lighter than males, having an average adult body weight of 14 kg (Calaby 1983). Bennett's wallabies occur throughout Tasmania and many of its offshore islands. The range of its closely related subspecies, the red-necked wallaby (*Macropus rufogriseus banksianus*), extends along the coast from the south-east of South Australia through eastern New South Wales to the south-east of Queensland (Calaby 1983). The species is abundant throughout its range and it is believed to be one of the few marsupial species to have benefited from land-clearing (Calaby and Grigg 1989; Johnson *et al.* 1989). It occurs in many habitat types but typically prefers eucalypt forests with a recognisable shrub stratum or tall coastal heath communities (Calaby 1983).

As with other macropod species, *M. rufogriseus* gives birth to only one young at a time. It can achieve a continuous breeding cycle through post partum ovulation and embryonic diapause (Tyndale-Biscoe 1989). However, in the Tasmanian subspecies, breeding tends to be seasonal, with a peak in births occurring in February and March (Merchant and Calaby 1981; Curlewis 1989). Like most other marsupial species, the young spend the majority of their early development in a pouch, leaving at approximately 270 days of age and continuing to suckle to



Plate 1 : The Bennett's wallaby (*Macropus rufogriseus*).

around 360 days (Lee and Ward 1989). Females reach sexual maturity between 11 and 21 months of age and males at around 12 months (Lee and Ward 1989).

Macropus rufogriseus is essentially nocturnal and spends the majority of the day sheltering in bushland areas. In those areas where bushland abuts cleared areas, the Bennett's wallaby will typically move from shelter into the open where it will feed throughout the night (Mooney and Johnson 1979). It is essentially a grazer, preferring to feed on monocotyledons such as grasses and soft leaved dicotyledons (Calaby 1983; Statham 1983; Jarman and Phillips 1989; Sprent 1997). In Tasmania, home ranges have been found to average around 100 ha (Mooney and Johnson 1979); while on the mainland, median home ranges of 16.3 ha have been recorded (Johnson 1987).

2.1.2 The Tasmanian pademelon (*Thylogale billardieri*)

The Tasmanian pademelon is another native marsupial belonging to the Macropodidae family. Adult male pademelons on average weigh 7 kg (Johnson and Rose 1983). Females have an average weight of 4 kg. Pademelons occur throughout Tasmania and the larger islands of Bass Strait (Johnson and Rose 1983). Its distribution once extended into the south of Victoria and South Australia (Wood-Jones 1925) but it is now thought to be extinct on the mainland (Calaby 1971). In Tasmania, it is considered abundant and land clearing is thought to have lead to a general increase in numbers (Frith 1973; Driessen and Hocking 1992). Throughout its present range the pademelon occurs in a variety of habitats, though it generally prefers densely vegetated areas such as wet sclerophyll forest and rainforest, which afford it protection during daylight hours (Green 1973; Johnson and Rose 1983; Tasmanian Conservation Trust 1987).

Breeding occurs throughout the year but peaks from April to June. Gestation takes around 30 days, after which the young spend approximately 200 days in the pouch (Rose and McCartney 1982; Johnson and Rose 1983). Sexual maturity is reached at around 14 months (Rose and McCartney 1982).

The pademelon is essentially nocturnal and forages predominantly at night (Johnson and Rose 1983). There is limited information regarding the home range of pademelons in Tasmania. One radio-tracking study carried out by Johnson (1978) found that the home ranges of five male pademelons varied from 148 ha to



Plate 2 : The Tasmanian pademelon (*Thylogale billardierii*).

169 ha. Where forest adjoins cleared areas, pademelons may move into the open to feed (Johnson 1978), though they are thought to rarely venture far from shelter (Johnson and Rose 1983). Pademelons feed on a wide range of plant species, preferring soft leaved dicotyledons and grasses (Statham 1983; Sprent 1997).

2.1.3 The common brushtail possum (*Trichosurus vulpecula*)

The brushtail possum is an abundant, medium sized, arboreal marsupial. Adult males weigh between 2 and 4.5 kg (How 1983). Adult females weigh between 1.5 and 3.5 kg (How 1983). Three subspecies are described (How 1983): *T. v. johnstoni*, extending from northern Queensland through to southern Victoria; *T. v. vulpecula*, extending from southern Victoria through central Australia to south-western West Australia; and *T. v. fuliginosus* which is distributed throughout Tasmania. The combined range of these three subspecies is one of the largest for any Australian marsupial (Kerle 1984). It lives in a wide variety of treed habitats, ranging from the arid zone in central Australia through to the rainforests of Tasmania. The brushtail possum is also well adapted for survival in the urban environment (Fitzgerald 1975).

Longevity is typically less than 11 years (Flannery 1994). It is a continuous breeder but there is a peak in births during autumn (Green 1984). Sexual maturity is reached after around one year in females (How 1983). Females give birth to only one young at a time, with one to two young being produced each year. Following a gestation period of around 18 days, the young spends 4-5 months in the pouch (Flannery 1994). Another 1-2 months are spent suckling and riding on the mother's back (Flannery 1994).

The brushtail possum is essentially a nocturnal animal. During the day it nests in tree hollows and logs which it defends from other possums (Flannery 1994). At night it forages for food both in the tree tops and on the ground (MacLennan 1984). It is typically a browser, feeding predominantly on soft and hard leaved dicotyledons (Statham 1983; Fitzgerald 1984; Green 1984; Kerle 1984; Evans 1992; Owen and Norton 1995). However, a wide range of other foodstuffs are more occasionally eaten including fruit, buds, bark, insects and meat (Flannery 1994). Home range size for the brushtail possum varies between habitat types. Ranges determined from Australian studies have varied in size from 3.0 ha to 7.4 ha for males and 1.0 ha to 4.7 ha for females (see review by Green 1984).



Plate 3 : The brushtail possum (*Trichosurus vulpecula*, photograph courtesy of Dr. Clare McArthur).

2.1.4 The introduced European rabbit (*Oryctolagus cuniculus*)

The European rabbit, a placental mammal, was introduced into south-eastern Australia in the 1800s (Rolls 1969). Since then, the rabbit has spread over much of mainland Australia, Tasmania and several offshore islands. Throughout Australia, the rabbit occurs in a wide range of habitat types and is apparently limited only by the availability of water and soil suitable for burrows in which to shelter (Myers 1983; Parer and Libke 1985). For both sexes, average adult body weight is around 1.6 kg (Myers 1983).

Longevity is typically short (approximately 3 years) and mortality of juveniles high (Cooke 1982, 1983). The rabbit is a prolific breeder. Sexual maturity is reached after 4 months in some areas (Myers *et al.* 1994). On average, around four or five young are born in each litter (Myers *et al.* 1994). In favourable areas up to five litters can be born in a year (Myers 1983).

Rabbits have a diet typical of grazers, though they are known to be remarkably adaptable where preferred vegetation is scarce (Cooke 1982; Myers *et al.* 1994). In Tasmania the rabbit has been observed to feed primarily on soft leaved dicotyledons and grasses (Statham 1983). Home ranges vary markedly between environments (Myers *et al.* 1994), measuring up to 4.7 ha in dry sparsely vegetated areas (Fullagar 1981, as cited in Myers *et al.* 1994).

2.2 STUDY SITES

Thirty-five forestry plantations (Figure 2.1) were used during the studies outlined in this thesis. These plantations were provided for study by four forestry companies. Australian Newsprint Mills provided nine plantations, predominantly in the Florentine and Styx valley areas in central Tasmania. Forestry Tasmania provided four plantations situated in the Dover and Geeveston districts in Southern Tasmania. Boral Timber Resources provided 12 plantations, scattered from Deloraine in the north through to Scottsdale and Ringarooma in the north-east. North Forest Products provided nine plantations, two of these near Ridgley in the

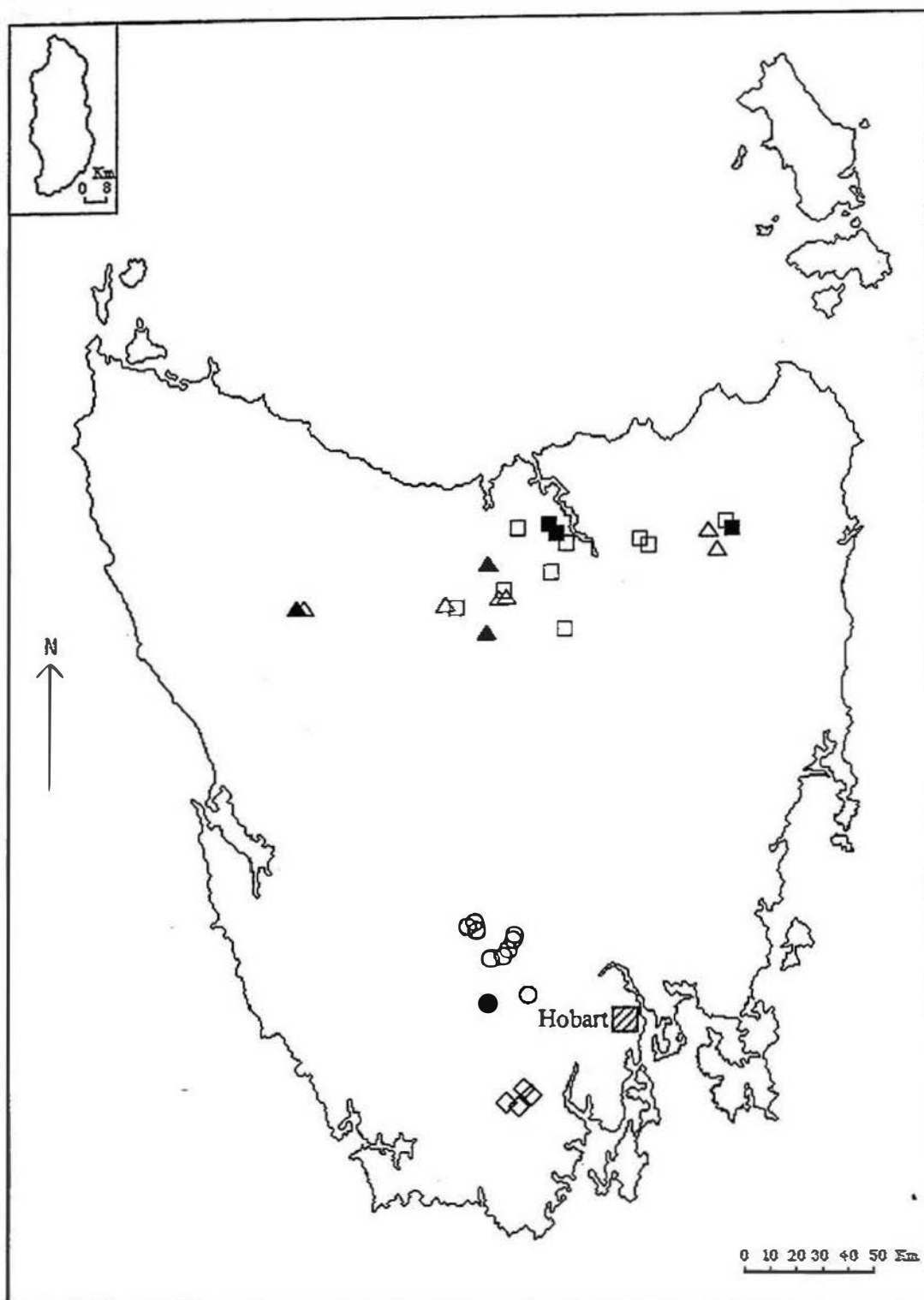


Figure 2.1 : The location of the 35 forestry plantations used during this study. Plantations from four forestry companies were used; Australian Newsprint Mills (O), Boral Timber (□), Forestry Tasmania (◇) and North Forest Products (△). Plantations where poison was used are marked with black symbols (e.g. ◆). Plantations where no poison was used are marked with open symbols (e.g. ◇).

north-west of the state, and the remainder scattered from Deloraine through to Ringarooma in the north-east.

Prior to studies commencing at a plantation, forestry company employees harvested the majority of standing timber in the plantation and then prepared it for planting seedlings. Pre-planting preparation included stacking waste timber into windrows, ripping along planting lines with a bulldozer, and reducing ground vegetation cover with herbicide. At 28 of the plantations, 1080 poisoning programs were carried out in an attempt to reduce the local abundance of browsing species and thereby reduce the potential for browsing damage. At each of these plantations carrot baits, stained with blue dye and poisoned with 1080, were distributed around the plantation perimeter. This was usually preceded by a period of 'free-feeding' with non-poisoned carrot. In general these were carried out from around two weeks prior to planting to one week after planting. At the remaining seven plantations no attempt was made to control browsing damage. These plantations were not poisoned because they were to be used in a growth experiment being conducted jointly by three forestry companies. Thus, these plantations were selected at random from the large group of plantings that were proposed by these companies during 1994 and it is important to stress that the decision not to poison was made irrespective of preconceptions concerning the degree to which seedlings would be damaged.

Eucalyptus nitens seedlings were planted at 32 of the plantations and *Eucalyptus globulus* were planted at the remaining three. Seedlings were taken from various nursery stocks and hand planted. The timing of planting varied between plantations, but generally occurred between August and November. The majority of data collection was carried out at each plantation from the time when seedlings were first planted until one year later. Sampling was spread out over three years, with 13 plantations being monitored from 1994 to 1995, 15 different plantations from 1995 to 1996 and the remaining seven plantations from 1996 to 1997 (refer to planting dates in Table 2.1).

A summary of the descriptive characteristics of each plantation is presented in Tables 2.1, 2.2 and 2.3. Altitude, slope and aspect were determined from official Tasmanian Government, 1:25000 topographic maps (Hays *et al.* 1981). The aspect of each plantation was also ground truthed. Plantation areas were obtained from forestry company surveying records. Climatic data were obtained from the Bureau of Meteorology. Further information describing each plantation is presented in Chapter 8.

2.3 THE TREE SPECIES

Of the plantations used during the studies outlined in this text, 32 were planted with *Eucalyptus nitens* and the remaining three were planted with *Eucalyptus globulus*. Both are favoured species for forestry plantations. A brief description of their biology follows.

2.3.1 Shining gum (*Eucalyptus nitens*)

Aside from the forestry areas where it has been introduced, *Eucalyptus nitens* occurs in scattered populations from the Great Dividing Range in New South Wales to the alpine regions of Victoria (Boland *et al.* 1984). It typically occurs between 600 m and 1600 m. It occurs on slopes and mountain tops on a variety of soil types including basalt, granite schist, shale and sandstone. It tolerates severe frosts and snow falls which can occur throughout its distribution. Mean annual rainfall for most localities where it is found range from 750-1750 mm. *E. nitens* is a tall tree which can reach heights of up to 90 m. Its form and rapid growth rate make it a favoured species for forestry plantations. Its pulp is used for paper and sawn timber for building and construction (Norton P. 1995).

2.3.2 Tasmanian blue gum (*Eucalyptus globulus*)

Eucalyptus globulus is distributed throughout southern Victoria and along the east coast of Tasmania. It is also found on Flinders island and King island in Bass Strait. It is typically a lowland species, occurring from near sea level up to around

Table 2.1 : General information for each plantation, including the owner company (ANM : Australian Newsprint Mills; Boral : Boral Timber Resources; Forestry Tas : Forestry Tasmania; NFP : North Forest Products), easting (metres), northing (m), the date on which seedlings were planted, species of eucalypt planted, and the type of mammal control used at each plantation.

Plantation	Owner company	Easting (m)	Northing (m)	Planting date (d/m/y)	Species planted	Mammal control
Adu29	ANM	475500	5278600	26/08/94	<i>E. nitens</i>	Poisoning
Alo48	ANM	455000	5286725	30/10/96	<i>E. nitens</i>	Poisoning
Alo49	ANM	456575	5286000	25/11/96	<i>E. nitens</i>	Poisoning
Amark	ANM	474750	5278000	11/11/96	<i>E. nitens</i>	Poisoning
Aseol	ANM	456000	5287875	15/11/96	<i>E. nitens</i>	Poisoning
Asx13	ANM	473400	5259500	25/08/94	<i>E. nitens</i>	None
Asx51	ANM	490500	5257100	12/08/95	<i>E. nitens</i>	Poisoning
Atu18	ANM	471200	5269900	20/07/95	<i>E. nitens</i>	Poisoning
Atu20	ANM	473300	5272100	20/07/95	<i>E. nitens</i>	Poisoning
Awv55	ANM	453600	5315000	25/07/95	<i>E. nitens</i>	Poisoning
Beam	Boral	533300	5423500	15/11/95	<i>E. nitens</i>	Poisoning
Bfia	Boral	562800	5437000	27/10/94	<i>E. nitens</i>	Poisoning
Bkui	Boral	489600	5376900	15/11/95	<i>E. nitens</i>	Poisoning
Brfl	Boral	488500	5432000	11/10/94	<i>E. nitens</i>	None
Brf2	Boral	488000	5431700	11/10/94	<i>E. nitens</i>	None
Brin	Boral	563200	5435600	27/10/94	<i>E. nitens</i>	None
Bsch	Boral	528900	5426000	27/10/94	<i>E. nitens</i>	Poisoning
Bswan	Boral	482800	5415500	17/10/95	<i>E. nitens</i>	Poisoning
Bswan2	Boral	483825	5415000	19/11/96	<i>E. nitens</i>	Poisoning
Bwal	Boral	488300	5423300	12/10/94	<i>E. nitens</i>	Poisoning
Bwee	Boral	457000	5407400	14/11/95	<i>E. nitens</i>	Poisoning
Bwfr	Boral	475400	5424800	31/10/95	<i>E. nitens</i>	Poisoning
Fha21h	Forestry Tas	492550	5194300	22/11/96	<i>E. nitens</i>	Poisoning
Fhp23g	Forestry Tas	498300	5205200	18/07/95	<i>E. nitens</i>	Poisoning
Fkd13b	Forestry Tas	490300	5217200	19/06/95	<i>E. nitens</i>	Poisoning
Fsolb	Forestry Tas	495100	5198000	03/07/95	<i>E. nitens</i>	Poisoning
Nalans	NFP	482500	5407725	15/11/96	<i>E. nitens</i>	Poisoning
Nbless	NFP	546400	5409300	15/10/95	<i>E. nitens</i>	Poisoning
Ncw	NFP	386500	5409800	19/04/95	<i>E. nitens</i>	Poisoning
Ndig	NFP	476800	5425500	15/10/94	<i>E. globulus</i>	Poisoning
Npear	NFP	381600	5407600	01/05/95	<i>E. nitens</i>	Poisoning
Nped	NFP	556300	5433800	16/10/95	<i>E. nitens</i>	Poisoning
Npu	NFP	482000	5417900	15/11/94	<i>E. nitens</i>	None
Nrit	NFP	473900	5394300	15/10/94	<i>E. globulus</i>	None
Nspill	NFP	445300	5415800	15/10/94	<i>E. globulus</i>	Poisoning

Table 2.2 : Physical characteristics of each plantation, including altitude above sea level in metres (m), average slope (degrees), aspect and the area of the plantation (ha).

Plantation	Altitude (m)	Slope (degrees)	Aspect	Area (ha)
Adu29	490	12	NW	24.0
Alo48	365	1	E	59.6
Alo49	395	1	NE	12.7
Amark	500	4	E	10.4
Aseol	385	2	SW	17.8
Asx13	460	3	NE	16.0
Asx51	475	14	W	26.7
Atl18	375	14	S	14.7
Am20	405	6	E	30.4
Aww55	600	5	NE	80.0
Bcam	690	11	NE	12.9
Bfra	345	25	S	9.6
Bkui	450	10	SE	12.5
Brf1	560	11	NE	9.5
Brf2	165	11	NW	12.7
Brin	225	5	W	13.5
Bsch	505	14	S	40.5
Bswan	255	13	W	18.0
Bswan2	255	11	E	37.0
Bwal	130	5	NW	21.0
Bwee	160	16	SW	43.1
Bwfr	175	2	W	10.5
Fha21h	45	8	SW	27.2
Fbp23g	195	12	SW	38.0
Fkd13b	175	13	NE	52.0
Fsolb	135	9	W	64.0
Nalans	250	5	NE	108.8
Nbless	510	16	SW	41.0
Ncw	630	2	NW	42.9
Ndig	250	9	S	17.0
Npear	645	1	NE	34.9
Nped	365	9	NE	20.0
Npu	340	10	W	39.0
Nrit	330	12	NE	24.0
Nspill	275	13	SE	28.0

Table 2.3 : Climatic information for each plantation, including average annual rainfall measured in millimetres (mm), average maximum summer temperature measured in degrees Celsius (°C), average summer minimum temperature (°C), average maximum winter temperature (°C) and the average winter minimum temperature (°C). Averages are based on Tasmanian Bureau of Meteorology data collected from the weather station nearest to each plantation. Averages are taken over all years since recording began.

Plantation	Rainfall		Temperature		
	Annual rainfall (mm)	Summer max (°C)	Summer min (°C)	Winter max (°C)	Winter min (°C)
Adu29	520	20.9	10.0	10.9	1.5
Alo48	942	18.7	6.8	8.0	-0.1
Alo49	942	18.7	6.8	8.0	-0.1
Amark	520	20.9	10.0	10.9	1.5
Aseol	942	18.7	6.8	8.0	-0.1
Asx13	943	21.3	8.8	10.4	2.3
Asx51	783	18.6	7.9	8.0	2.4
Atm18	976	21.3	8.8	10.4	2.3
Atm20	836	21.3	8.8	10.4	2.3
Aww55	1145	18.7	6.8	8.0	-0.1
Bcam	1131	22.7	10.4	11.9	3.3
Bfra	1176	22.7	10.4	11.9	3.3
Bkui	903	23.0	10.2	10.9	1.8
Brf1	695	23.9	11.9	12.3	2.0
Brf2	695	23.9	11.9	12.3	2.0
Brin	1176	22.7	10.4	11.9	3.3
Bsch	1131	22.7	10.4	11.9	3.3
Bswan	819	22.7	10.4	11.9	3.3
Bswan2	819	22.7	10.4	11.9	3.3
Bwal	929	20.5	12.0	12.4	4.1
Bwee	950	20.9	10.0	10.6	2.5
Bwfr	1059	20.5	12.0	12.4	4.1
Fha21h	1348	20.3	9.9	11.9	3.3
Fhp23g	838	21.2	9.5	11.9	1.7
Fld13b	838	21.2	9.5	11.9	1.7
Fsolb	1348	20.3	9.9	11.9	4.1
Nalans	819	20.9	10.0	10.2	2.0
Nbless	883	23.1	10.1	10.8	2.3
Ncw	1772	17.6	6.3	7.2	0.8
Ndig	1059	20.5	12.0	12.4	4.1
Npear	2154	17.6	6.3	7.2	0.8
Nped	1176	22.7	10.4	11.9	3.3
Npu	819	22.7	10.4	11.9	3.3
Nrit	928	23.0	10.2	10.9	1.8
Nspill	1160	20.9	10.0	10.6	2.5

400 m, though it is occasionally found as high as 820 m (Williams and Potts 1996). It occurs in a wide range of habitats ranging from wet, fertile gullies to infertile, dry dune areas (Williams and Potts 1996). It occurs on dolerite and on shallow humus soils over mudstone. It prefers a warm to cool, humid to sub-humid, climate. Mean annual rainfall in the localities where it is found range from around 600-1400 mm. *E. globulus* is highly variable in form, growing up to 70 m, but it is sometimes reduced to a mallee-like shrub where conditions are poor (Boland *et al.* 1984). It is a favoured forestry species because it is fast growing and has a long straight bole to around two thirds of its total height (Norton P. 1995). It has a very high pulp yield and is one of the most important species for pulpwood production in temperate regions of the world (Eldridge *et al.* 1993). Its sawn timber is also popular for use in both heavy and light construction (Norton P. 1995).

2.3.3 Potential confounding effects

In some of the chapters that follow, some analyses appear to ignore the potential between-species difference in the palatability of the two eucalypt species. Ideally only one species would have been used throughout. However, for the three year duration of field studies, forestry companies only made a total of seven unpoisoned plantations available for study. One of these was planted with *E. globulus*. Not including this plantation would have reduced sample size for unpoisoned sites to only six. Furthermore, I felt the inclusion of the *E. globulus* plantation was justified on the grounds that existing evidence suggests that the palatability of *E. nitens* and *E. globulus* is similar. Scott (1997) recorded levels of brushtail possum damage to saplings of various eucalypt species in Tasmania and found no significant difference in severity of damage recorded for *E. nitens* and *E. globulus* at any plantation. In a similar study, Dungey (1996) observed little difference between the two species in the extent to which they were damaged by possums. In a series of preference trials McArthur and Turner (1997a) found that captive brushtail possums exhibited no preference for the sapling foliage of one species over that of the other.

Data from the three *E. globulus* plantations have been excluded from any analyses where a between-species difference in palatability might have confounded results. In all analyses where these data were included, residuals plots have been examined to

ensure that results have not been biased by the introduction of outlying or highly influential data points.

2.4 FREQUENTLY USED TECHNIQUES

During the studies outlined in the following chapters, several techniques were repeatedly used. Rather than repeat the descriptions of these techniques in each section, a general discussion of them is included below. The reader will be referred back to this chapter whenever the following techniques are used.

2.4.1 Measuring seedling height and diameter

Data were collected on seedling height and diameter so that daily height and diameter growth could be calculated. Seedlings were measured at the time of planting and then again at approximately six and 12 months after planting. Seedling height was measured to the nearest centimetre. Seedlings which were less than one metre tall were measured with a one metre ruler ('Sands' brand). The ruler was placed at the base of the seedling and the seedling was then held by its topmost leaves against the ruler. Care was taken not to damage the seedling. Height measurements were taken at the tallest point where the seedling touched the ruler. Seedlings which were taller than one metre were measured in the same way except that a five metre collapsible surveyors staff ('Surmac' brand) was used. The diameter of seedlings was measured to the nearest 0.1 mm using a pair of Vernier callipers. The callipers were placed at the base of the seedling and closed until the jaws were just touching the plant. Care was taken not to bruise the seedling itself.

2.4.2 Assessing browsing damage to seedlings

Distinguishing mammal damage from other forms of damage

Seedlings can be damaged by a number of agents including wind, frost, insects and browsing mammals. A description of each is given below :

- i) wind damage - seedlings are typically incompletely broken off part way up the main stem. Foliage may be snapped off and deposited near the base of the seedling.
- ii) frost damage - some or all of the seedling's leaves may be brown and dead. Dead leaves can accumulate at the base of the seedling. Stems and branches do not appear damaged.
- iii) insect damage - usually confined to the periphery of leaves, where relatively small circular or curved cuttings are made. Stems and branches do not appear damaged. Usually foliage has not accumulated at the base of the seedling.
- iv) mammalian damage - typically foliage is partially or wholly sheared off, leaving straight, rather than curved edges. Damage is not confined to leaves only and may include leaf stems, shoots and branches. Usually foliage does not accumulate at the base of the seedling.

Although mammalian damage is generally quite easy to distinguish from frost and wind damage it is possible that, on occasion, insect damage was mistakenly identified as mammal damage. This is particularly true where leaves were damaged right back to the leaf stem, leaving little foliage by which to make an identification of the responsible agent. On these occasions it was generally assumed that mammals were responsible. Conversely, it is felt that there would have been few instances where mammal browsing was mistaken as insect damage.

Thus, the damage estimates reported in this thesis may represent a slight overestimate of mammal damage.

Assessing damage severity

Seedlings were examined for signs of vertebrate browsing at approximately six and 12 months after seedlings were planted. An assessment of the severity of damage to seedlings was carried out following Marsh (1998). Browsing damage

severity was scored using a six point scale (Table 2.4). The scale, which ranges from 0-5, is based on a visual estimate of the percentage of a seedling's biomass which has been removed by mammal browsing.

Table 2.4 : The six point scale used to assess severity of browsing damage to seedlings.

Browse score	Estimate of the biomass removed (% of total)
0	0
1	1-29
2	30-49
3	50-69
4	70-89
5	90-100

As a guide to how the scale is used, several photographs of seedlings damaged to varying severities have been included along with their damage scores (Plates 4 to 9). To ensure that this relative measure of damage severity remained consistent throughout, I was the only person who ever scored browsing damage.

2.4.3 Estimating vegetation cover

During some of the following studies measures were made of the vegetation cover on each plantation and in areas of adjoining bushland. The method used to assess components of ground vegetation cover was consistent throughout. Quadrats with sides 1 m × 1 m were the basic sampling unit used. Estimation of coverage within each quadrat was achieved using the ocular estimation of cover method outlined by Hays *et. al* (1981; also see Kent and Coker 1992). Briefly, this method involves visually subdividing a quadrat into 25 units with the aid of markings made on the quadrat (0.2 m apart). Given that each of these units represents 4% of the total area of the quadrat, the total coverage of a particular ground cover component can be determined by estimating the number of units it occupies. Coverage estimates were made to the nearest 10% and the same observer was used for all measurements.



Plate 4 : A eucalypt seedling which has not been damaged by vertebrate herbivores.
This seedling would be assigned a '0' for browse score.



Plate 5 : A eucalypt seedling which is estimated to have had between 1 and 29% of its biomass removed by vertebrate herbivores. This seedling would be assigned a browse score of 1.



Plate 6 : A eucalypt seedling which is estimated to have had between 30 and 49% of its biomass removed by vertebrate herbivores. This seedling would be assigned a browse score of 2.



Plate 7 : A eucalypt seedling which is estimated to have had between 50 and 69% of its biomass removed by vertebrate herbivores. This seedling would be assigned a browse score of 3.



Plate 8 : A eucalypt seedling which is estimated to have had between 70 and 89% of its biomass removed by vertebrate herbivores. This seedling would be assigned a browse score of 4.



Plate 9 : A eucalypt seedling which is estimated to have had over 90% of its biomass removed by vertebrate herbivores. This seedling would be assigned a browse score of 5.

2.5 Statistics and notation

For each of the statistical analyses employed during this text, a brief description is given in the methods section of the chapter where it is first used. All analyses were performed using either the SAS or SYSTAT statistics packages. In all cases the level of statistical significance has been set at $\alpha < 0.05$ (Zar 1984). Table 2.5 outlines the notation and abbreviation associated with the techniques used.

Table 2.5 : Explanation of the statistical notations and abbreviations commonly employed throughout this text.

Notation	Description
χ^2	the chi-squared test statistic
df	degrees of freedom
F	Fisher statistic for analysis of variance
n	the sample size, number of observations
no.	number of
ns	denotes a result that is not significant ($p > 0.05$)
p	probability
r	Pearson's correlation coefficient
r^2	coefficient of determination
adjusted r^2	r^2 adjusted for sample size
rho	the Spearman rank test statistic
se	standard error
sd	standard deviation
Sr^2	semi-partial correlation coefficient
t	the t-test statistic
T	the Wilcoxin signed-rank test statistic
U	the Mann-Whitney test statistic
*	denotes a result that is significant ($p < 0.05$)
**	denotes a result that is highly significant ($p < 0.01$)

CHAPTER 3

SEEDLING GROWTH AND BROWSING DAMAGE

3.1 INTRODUCTION

Braysher (1993) considers that the first step to take when considering a pest control program must be to determine whether or not the threats to the crop are real or perceived. In Tasmania, the browsing damage caused by mammalian herbivores is thought to reduce plantation productivity by reducing seedling growth rates, reducing seedling survival and changing the form of seedlings so that tree quality is reduced. Thus, Braysher's (1993) original question might be rephrased as : How and by how much does vertebrate browsing reduce the overall volume and value of timber harvested from plantations ? Answering this question, in quantifiable terms, can provide a suitable basis for making management decisions regarding control. For example, if it is found that the effect of browsing damage on productivity is negligible, then control is unnecessary. If, on the other hand, browsing damage does reduce productivity it is important to know by how much, and under what circumstances, so that the potential benefits of control can be weighed against the costs.

The link between damage caused by herbivores and losses in plant productivity or growth is not clear. It is inherently appealing to assume that any biomass reduction caused by herbivory will negatively affect a seedling's growth. However, some studies suggest that herbivore damage has no negative effect on overall plant growth (e.g. Strauss 1991; Stock *et al.* 1993) and that it may even be beneficial (e.g. Owen and Wiegert 1976; McNaughton 1979; review by Belsky 1986). In most cases it appears that the observed effect of browsing on plant growth is dependent on the severity of damage.

Cremer (1969, 1972, 1973), Nielsen and Pataczek (1991) and Wilkinson and Nielsen (1995) carried out studies examining how the removal of varying

quantities of foliage can affect eucalypt seedling growth. In summary, their results showed that seedlings are not greatly affected by removal of small amounts of foliage. However, if the extent of foliage removal is extreme (e.g. complete crown removal) then both seedling growth and survival can be reduced.

Wilkinson and Nielsen (1995) demonstrated that these effects can extend for up to seven years, although these results may have partly reflected the effects of weed competition.

The degree to which the results from studies of simulated browsing damage can be extrapolated to field conditions may be somewhat limited. Although there are some exceptions (e.g. Bergstrom and Danell 1989; Nielsen and Pataczek 1991), simulated browsing studies generally involve a one-off removal of foliage.

However, in a natural system browsing attacks of varying severity may occur periodically over time. Furthermore, researchers may not be able to simulate the feeding patterns of animals during feeding bouts. Animals may, for example, feed randomly on a plant or concentrate their attacks on certain preferred parts.

Finally, it is difficult to know how the 'damage' imposed by the researcher equates to damage observed in the field.

Considering the importance of establishing the links between damage severity and seedling growth and survival, the extent of published data from Tasmanian field trials is disappointing. The study reported by Gilbert (1961) indicated that *E. regnans* seedling growth over a 2-3 year period was adversely affected by exposure to browsing. Cremer (1969) also found that seedlings of *E. regnans* were adversely affected by browsing damage, with browsed seedlings suffering high mortality and retarded growth. Both of these studies were restricted to a single study site of regenerating eucalypts. To date no one has examined how growth rates of plantation seedlings are affected by the wide range of browsing severities observed in the field.

This chapter outlines a large scale field study which aimed to quantify the relationships between observed levels of browsing damage, and eucalypt seedling growth and survival across a wide range of environmental conditions.

3.2 METHODS

3.2.1 Study sites

Data were collected from the 35 plantations which were described in detail in Section 2.2. Data collection was carried out for approximately one year after seedlings were planted. Thirteen plantations were monitored from 1994 to 1995, 15 plantations from 1995 to 1996 and the remaining seven plantations from 1996 to 1997 (see planting dates in Table 2.1).

3.2.2 Techniques

Immediately after each coupe was planted, 40 plots were established, each containing five seedlings (total of 200 seedlings). Wire 'pig-tail' markers, brightly coloured surveyors tape and 'Permotags' (Hortico brand) were used to mark and identify plots. Plots were favoured over the marking of single seedlings for logistic reasons. At many plantations seedlings and other vegetation were quite high after only a few months of growth, making it difficult to locate plots. Forty large plots took far less time to locate than 200 smaller plots. Plots were placed out on a plantation according to randomly selected points of intersection on a one hectare grid overlay. At smaller plantations all of the intersection points could end up being selected.

Seedling height and basal diameter were measured at the time of planting and then again approximately six and 12 months later (Section 2.4). These measurements were only taken for seedlings still living at the time of sampling. Due to the timing of planting, data on the initial size of seedlings could not be collected from two of the study sites (Ncw and Npear). Each seedling was also assessed for signs

of damage at approximately six and 12 months after planting (Section 2.4). The severity of damage to individual seedlings was estimated using a six point assessment scale (Section 2.4). Damage data were collected from all plantations. While all efforts were made to keep sampling intervals equal (i.e. six months) it was rare that seedlings were assessed at exactly six or 12 months after planting. Sampling intervals ranged from 5.1 months (153 days) to 6.8 months (203 days).

3.2.3 Statistics

In the majority of the following analyses the use of raw height and diameter data for plantations was thought to be inappropriate for two reasons. Firstly, when seedlings were planted they were not all of equal height and diameter. As a consequence, observed differences in raw height or diameter data may have reflected initial seedling size rather than any difference in seedling growth rates between individuals or sites. Secondly, it was impossible to keep sampling intervals exactly equal for all sites. This meant that at some plantations seedlings were allowed to grow for slightly longer periods than at other plantations.

I considered average daily 'growth' in height or diameter to be a superior variable for use. Daily growth is defined here as the average amount by which a seedling changed in height or diameter per day. Values for height growth were calculated by first subtracting a seedling's starting height from the measurements taken at six or 12 months after planting to give the total amount by which height had changed. Total height change was then divided by the number of days over which the seedling had been growing to give average daily height growth (mm/day).

$$\text{height growth} = \frac{\text{final height} - \text{starting height}}{\text{number of days growing}}$$

Values for average daily diameter growth were calculated in the same way except that diameter values were substituted for height values. It should be noted that, as

it is used here, the term 'growth' is applied irrespective of whether there was an overall positive or negative change in seedling proportions.

Two values were used as measures of the overall damage severity at a plantation. These were the 'percentage of seedlings damaged' and 'mean browse score'. Percentage of seedlings damaged was calculated by dividing the number of seedlings which were damaged by the total number of living seedlings sampled and then multiplying the result by 100. Mean browse score was calculated by averaging the browse scores (Section 2.4) for all living sampled seedlings.

The data collected at six months was compared to that collected at 12 months using a two-tailed paired t-test (Zar 1984). Mean values for two sets of 'unpaired' variables were compared using the two-tailed unpaired t-test. An example is the comparison between the growth rates of undamaged and damaged seedlings. The t-test statistic, degrees of freedom and associated probability values have been given for all analyses.

Linear regression was used to analyse linear trends between two variables (Draper and Smith 1966). Non-linear regression models, such as the logarithmic, were used in those instances where variables were suspected to observe a non-linear relationship. For all regressions, the Pearson's correlation coefficient has been used as a measure of the 'goodness of fit' of regression lines. The associated probability value and the sample size have also been given. On occasion a modified t-test was used to compare two regression lines to see if they were significantly different from each other (Zar 1984). An example of this is the comparison made between the regression line relating seedling diameter to height measures at six months and the analogous regression generated for the 12 month data.

It should be noted that, because measurements of the height and diameter of seedlings were not collected at two sites, growth data could only be calculated for 33 sites. Damage data on the other hand was collected from all 35 sites. As a

consequence, degrees of freedom and sample sizes vary during the following analyses.

3.3 RESULTS

3.3.1 Seedling growth

Tables 3.1 and 3.2 (respectively) show the mean daily growth (mm/day) in height and diameter values for seedlings at 33 plantations. Growth values have been calculated for the first six months of seedling growth (0-6 months), for the second six months of growth (6-12 months) and for the whole year (0-12 months). Mean daily growth values varied between sampling periods. A paired t-test shows that during their first year of growth seedlings increased their height significantly faster during the first six months than during the second (height : $t=39.911$, $df=5746$, $p<0.01$). The trend is consistent for mean daily diameter growth where seedlings increased their diameter more rapidly during the first six months ($t=22.31$, $df=5746$, $p<0.01$).

Daily growth values varied markedly between plantations. The mean daily height growth calculated over one year for seedlings ranged from -0.01 to 3.24 mm/day. The mean daily diameter growth calculated over one year ranged from 0.002 to 0.087 mm/day. For both measures the lowest growth rate was less than 5% of the highest rate.

3.3.2 Damage to seedlings

Browsing damage was recorded at all 35 plantations sampled. Table 3.3 shows the results of the seedling damage assessments carried out at six and 12 months after planting. For the six month data the percentage of seedlings damaged at a plantation ranged from 14% to 97%. For the data collected at 12 months the range was from 17% to 100%. Mean browsing scores ranged from 0.17 to 3.84 for the six month data and from 0.24 to 4.69 for the 12 month data, where

Table 3.1 : The mean daily height growth (mm/day) for seedlings calculated for the 0-6, 6-12 and 0-12 month periods. Standard deviations are given in parenthesis.

Plantation	0-6 months	6-12 months	0-12 months
Adu29	2.4 (1.57)	0.6 (0.10)	1.6 (0.87)
Alo48	0.9 (0.80)	0.1 (1.70)	0.7 (0.68)
Alo49	0.8 (0.90)	-1.0 (1.10)	0.4 (0.63)
Amark	0.5 (0.70)	0.0 (0.09)	0.4 (0.56)
Aseol	0.9 (0.63)	-0.5 (0.11)	-0.0 (0.05)
Asx13	0.7 (0.84)	0.5 (0.32)	0.6 (0.59)
Asx51	2.3 (1.29)	0.9 (0.87)	1.8 (0.96)
Atm18	2.7 (1.29)	1.0 (2.08)	2.3 (1.14)
Atm20	0.7 (0.50)	0.0 (0.14)	0.6 (0.43)
Aww55	2.1 (1.00)	0.1 (3.00)	1.7 (0.95)
Bcam	1.9 (1.13)	1.7 (1.00)	1.9 (0.76)
Bfr	3.5 (2.33)	0.7 (0.58)	3.1 (1.33)
Bkui	0.9 (0.99)	1.3 (0.90)	1.3 (0.85)
Brf1	0.3 (0.19)	0.2 (0.12)	0.5 (0.34)
Brf2	1.9 (1.07)	0.6 (0.54)	1.4 (1.21)
Brin	0.5 (0.57)	0.2 (0.32)	0.2 (0.46)
Bsch	2.7 (1.10)	0.8 (1.03)	1.7 (0.97)
Bswan	3.7 (1.40)	0.6 (0.93)	2.4 (0.92)
Bswan2	2.7 (1.68)	0.5 (1.44)	2.0 (1.21)
Bwal	4.3 (2.63)	0.4 (0.25)	3.2 (1.72)
Bwee	1.3 (1.20)	0.5 (1.29)	0.8 (1.01)
Bwfr	2.9 (2.00)	0.7 (0.45)	1.9 (1.13)
Fha21h	1.0 (1.04)	0.2 (0.99)	0.8 (0.84)
Fhp23g	1.9 (1.20)	1.1 (1.77)	1.8 (1.07)
Fkd13b	2.8 (1.30)	2.1 (2.45)	2.7 (1.23)
Fsolb	1.6 (1.10)	1.4 (1.96)	1.6 (0.98)
Nalans	3.8 (1.87)	0.8 (1.17)	2.7 (1.32)
Nbless	3.3 (1.70)	0.7 (0.70)	2.2 (1.05)
Ndig	2.9 (1.70)	0.7 (0.54)	2.1 (1.40)
Nped	2.5 (0.19)	1.8 (1.28)	2.2 (1.42)
Npu	0.3 (0.39)	0.3 (0.35)	0.3 (0.38)
Nrit	1.7 (1.11)	0.8 (0.65)	1.3 (1.23)
Nspill	3.3 (2.85)	0.5 (0.35)	2.9 (1.40)

Table 3.2 : The mean daily diameter growth (mm/day) for seedlings calculated for the 0-6, 6-12 and 0-12 month periods. Standard deviations are given in parenthesis.

Plantation	0-6 months	6-12 months	0-12 months
Adu29	0.07 (0.048)	0.03 (0.020)	0.05 (0.030)
Alo48	0.02 (0.018)	0.00 (0.074)	0.02 (0.021)
Alo49	0.01 (0.013)	0.01 (0.013)	0.01 (0.009)
Amark	0.01 (0.010)	0.00 (0.014)	0.01 (0.007)
Aseol	0.01 (0.010)	0.00 (0.013)	0.01 (0.007)
Asx13	0.02 (0.017)	0.01 (0.010)	0.01 (0.015)
Asx51	0.05 (0.027)	0.02 (0.014)	0.04 (0.023)
Atm18	0.05 (0.032)	0.01 (0.012)	0.05 (0.028)
Atm20	0.01 (0.010)	0.01 (0.009)	0.01 (0.010)
Awv55	0.04 (0.023)	0.02 (0.030)	0.04 (0.020)
Bcam	0.04 (0.029)	0.04 (0.030)	0.05 (0.025)
Bfr	0.08 (0.056)	0.06 (0.054)	0.08 (0.033)
Bkui	0.03 (0.031)	0.03 (0.037)	0.04 (0.027)
Brfl	0.00 (0.008)	0.00 (0.005)	0.00 (0.007)
Br2	0.07 (0.067)	0.02 (0.056)	0.04 (0.033)
Brin	0.02 (0.013)	0.01 (0.012)	0.01 (0.013)
Bsch	0.06 (0.027)	0.05 (0.028)	0.05 (0.026)
Bswan	0.07 (0.037)	0.04 (0.023)	0.06 (0.027)
Bswan2	0.07 (0.039)	0.04 (0.048)	0.06 (0.032)
Bwal	0.09 (0.056)	0.06 (0.052)	0.08 (0.039)
Bwee	0.04 (0.031)	0.02 (0.023)	0.03 (0.022)
Bwfr	0.06 (0.048)	0.01 (0.028)	0.04 (0.023)
Fha21h	0.03 (0.029)	0.02 (0.043)	0.03 (0.062)
Fhp23g	0.03 (0.024)	0.03 (0.042)	0.03 (0.025)
Fkd13b	0.06 (0.034)	0.08 (0.058)	0.06 (0.032)
Fsolb	0.03 (0.029)	0.04 (0.049)	0.03 (0.027)
Nalans	0.10 (0.045)	0.06 (0.058)	0.09 (0.038)
Nbless	0.08 (0.046)	0.06 (0.035)	0.07 (0.035)
Ndig	0.07 (0.054)	0.03 (0.045)	0.05 (0.037)
Nped	0.06 (0.047)	0.07 (0.042)	0.06 (0.041)
Npu	0.02 (0.013)	0.01 (0.016)	0.01 (0.013)
Nrit	0.04 (0.028)	0.03 (0.029)	0.03 (0.025)
Nspill	0.08 (0.079)	0.05 (0.052)	0.07 (0.034)

Table 3.3 : The percentage of marked seedlings with signs of browsing damage and the mean browse score calculated for all surviving seedlings at each of 35 surveyed forestry plantations. The data from both the six and 12 month sampling periods are presented. Standard deviations are shown in parenthesis. Sample size (n) is given.

Plantation	Six months			12 months		
	Seedlings damaged (%)	Mean Browse Score	n	Seedlings damaged (%)	Mean Browse Score	n
Adu29	31	0.31 (0.486)	198	45	0.60 (0.795)	191
Alo48	49	0.86 (1.226)	176	82	1.84 (1.554)	172
Alo49	72	1.60 (1.574)	180	98	3.89 (1.358)	176
Amark	20	0.38 (1.519)	168	60	0.98 (1.694)	168
Aseol	78	3.84 (1.599)	170	100	4.69 (0.812)	142
Asx13	58	1.02 (1.136)	188	85	2.13 (1.433)	168
Asx51	29	0.38 (0.704)	192	54	0.96 (1.261)	192
Atn18	16	0.23 (0.609)	188	42	0.66 (1.000)	186
Atn20	74	1.51 (1.327)	200	90	2.49 (1.359)	200
Aww55	17	0.17 (0.373)	196	26	0.25 (0.435)	196
Bcam	27	0.48 (0.938)	184	65	0.97 (1.008)	184
Bfr	24	0.28 (0.532)	148	22	0.24 (0.488)	148
Blui	88	2.12 (1.211)	192	31	1.25 (0.838)	184
Brf1	67	1.22 (1.064)	98	89	2.36 (1.163)	92
Brf2	33	0.42 (0.662)	160	47	0.71 (0.942)	160
Brin	97	2.56 (1.114)	136	94	2.58 (1.146)	132
Bsch	25	0.29 (0.552)	186	31	0.36 (0.585)	184
Bswan	15	0.17 (0.430)	196	26	0.34 (0.687)	196
Bswan2	49	0.90 (1.229)	185	56	1.33 (1.524)	183
Bwal	47	0.83 (1.132)	178	32	0.35 (0.585)	170
Bwee	94	2.21 (1.202)	188	88	2.33 (1.495)	166
Bwfr	43	1.11 (1.574)	180	47	1.40 (1.801)	176
Fha21h	46	1.05 (1.330)	192	60	1.54 (1.667)	176
Fhp23g	17	0.19 (0.462)	188	25	0.28 (0.582)	178
Fhd13b	16	0.22 (0.577)	196	24	0.29 (0.658)	196
Fsolb	27	0.41 (0.870)	194	24	0.43 (0.950)	194
Nalans	26	0.58 (1.270)	184	17	0.47 (1.211)	184
Nbless	34	0.48 (0.776)	192	40	0.66 (1.028)	186
Ncw	91	2.47 (1.631)	188	91	1.98 (1.381)	176
Ndig	48	0.55 (0.691)	166	29	0.32 (0.522)	166
Npear	49	0.93 (1.220)	198	86	2.01 (1.389)	196
Nped	33	0.44 (0.709)	159	21	0.29 (0.722)	149
Npu	94	3.34 (1.488)	172	97	2.53 (1.116)	172
Nrit	37	0.64 (0.971)	182	75	1.21 (1.042)	175
Nspill	14	0.18 (0.459)	194	30	0.32 (0.527)	170

browsing scores of 1 and 5 correspond to estimates that 1-29% and 90-100% (respectively) of a seedlings biomass has been removed (see Table 2.4).

Relationships between damage measures

Figures 3.1 and 3.2 show that for both the six and 12 month data there was a very strong relationship between the percentage of seedlings damaged at a plantation and the mean browsing damage score for that plantation (six months : $r^2=0.901$, $df=34$, $p<0.01$; 12 months : $r^2=0.878$, $df=34$, $p<0.01$).

The relationships are best described using exponential curve functions :

$$\text{Six months : } y = 0.150 \times e^{0.032x}$$

Equation 3.1

$$\text{12 months : } y = 0.176 \times e^{0.030x}$$

Equation 3.2

where 'y' is mean browse score and 'x' is the percentage of seedlings damaged.

To test whether the two regression lines were significantly different from each other a natural log transformation of the mean browse score values was performed such that two straight line regressions were obtained (six months : $\log_e(y) = 0.033x - 1.898$; 12 months : $\log_e(y) = 0.030x - 1.735$). The two regressions were then compared using a modified t-test (Zar 1984). The results indicate that the two lines do not differ significantly in either slope or elevation (slope : $t=0.0002$, $df=66$, $p>0.05$; elevation : $t=0.416$, $df=67$, $p>0.05$).

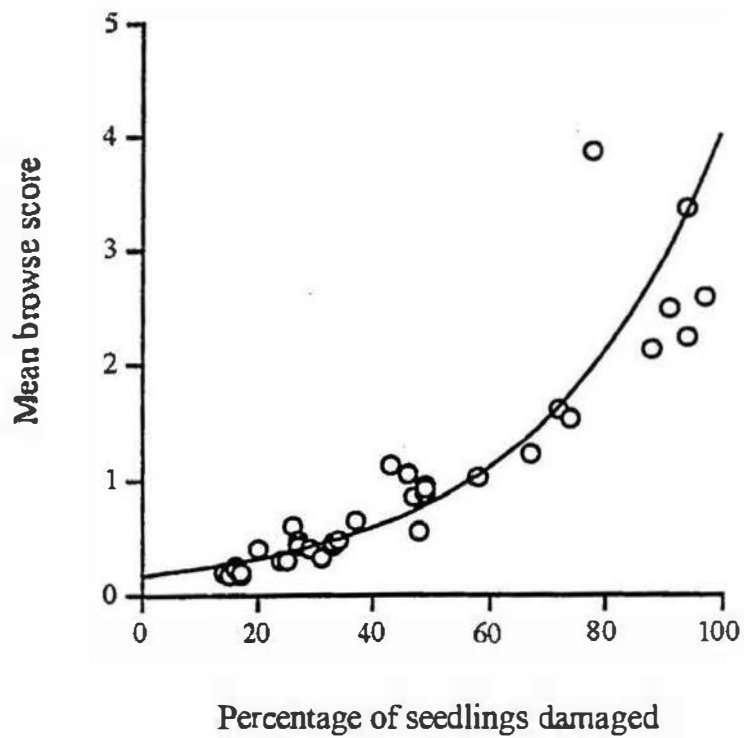


Figure 3.1 : Exponential regression relating the percentage of living, sampled seedlings showing signs of damage at six months after planting to mean browse score calculated at six months.

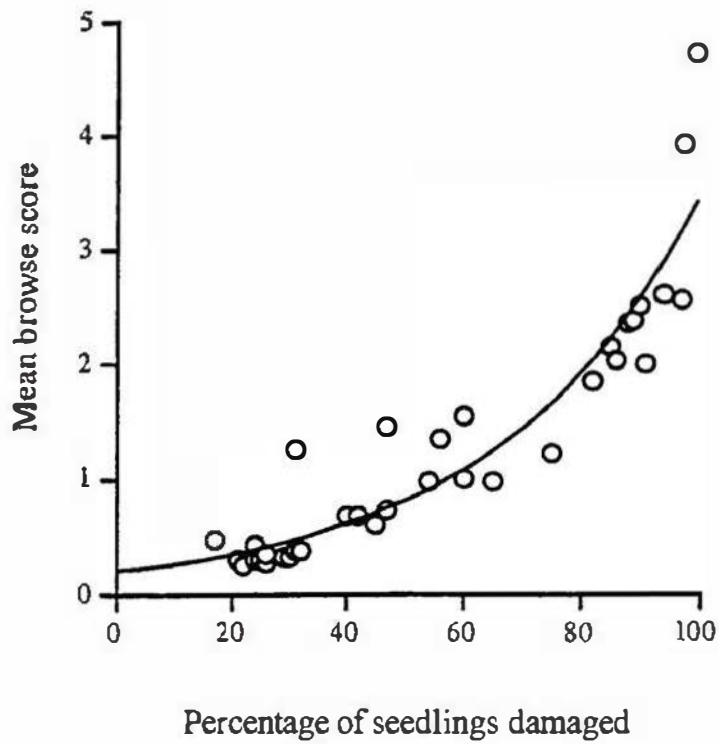


Figure 3.2 : Exponential regression relating the percentage of living, sampled seedlings showing signs of damage at 12 months after planting to mean browse score calculated at 12 months.

In Figures 3.3 and 3.4 two further regressions are presented. Both relate the percentage of seedlings that were damaged at a plantation to the mean of browse scores calculated for seedlings which exhibited signs of browsing. Hence, they differ from the regressions presented in Figures 3.1 and 3.2 in that seedlings with a browse score of zero have been excluded from the calculation of mean browse score. Figure 3.3 shows the relationship for the six month data and Figure 3.4 the relationship for the 12 month data. Both are highly significant and best described by exponential functions (six months : $y = 1.075 \times e^{0.009x}$, $r^2=0.585$, $df=34$, $p<0.01$; 12 months : $y = 0.938 \times e^{0.012x} + 18.297$, $r^2=0.742$, $df=34$, $p<0.01$).

Perusal of the data in Table 3.3 suggests that mean browse score values tended to be greater at 12 months than at six months. A paired t-test shows that the trend is a significant one for both the percentage of seedlings damaged data ($t=9.829$, $df=34$, $p<0.01$) and the mean browse score data ($t=2.988$, $df=34$, $p<0.01$).

3.3.3 Damage and seedling growth

Effects on the relationships between seedling height and diameter

In Figure 3.5 a scatterplot of seedling diameter and height measures taken six months after planting from *E. nitens* plantations is presented. Plots are presented for both unbrowsed seedlings and seedlings recorded with browsing damage. Linear regression indicates that, for both groups of seedlings, there is a highly significant relationship between seedling height and diameter (unbrowsed: $r^2=0.763$, $df=2820$, $p<0.01$; browsed: $r^2=0.725$, $df=2445$, $p<0.01$). These relationships are described by the following equations :

for unbrowsed seedlings	$y = 3.47x + 19.99$	Equation 3.3
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for browsed seedlings	$y = 3.54x + 9.09$	Equation 3.4
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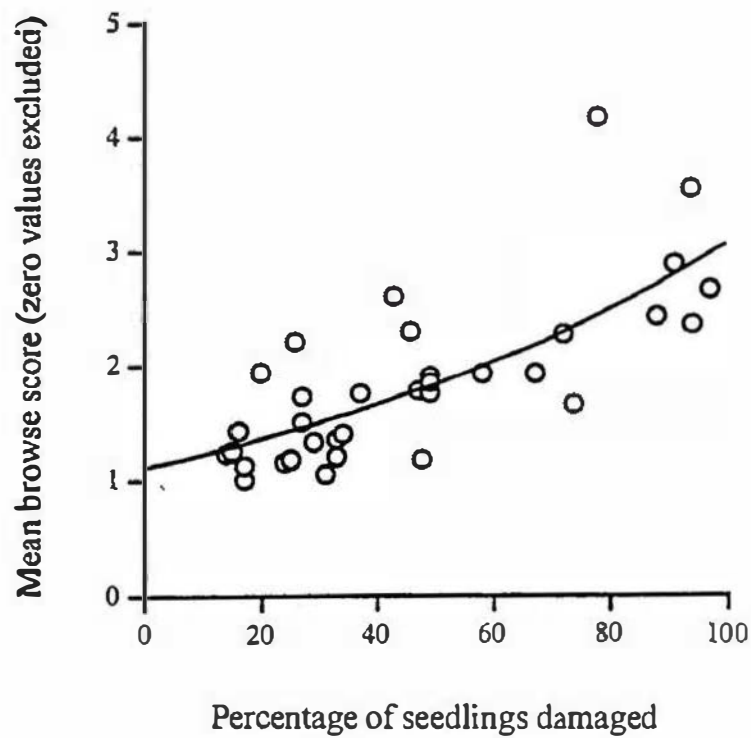


Figure 3.3 : Exponential regression relating the percentage of living, sampled seedlings showing signs of damage at six months after planting to the mean browse score calculated only for those seedlings exhibiting signs of browsing damage at six months.

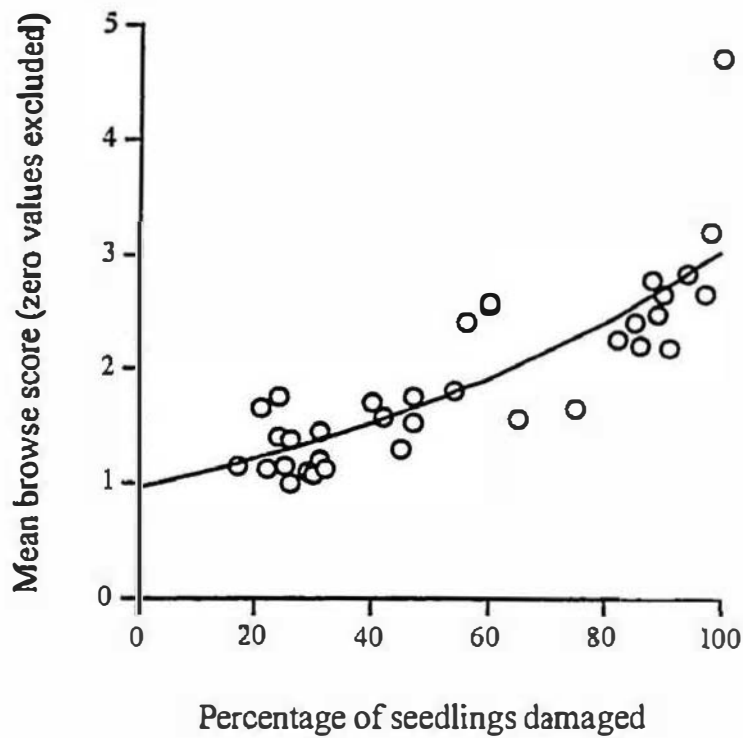
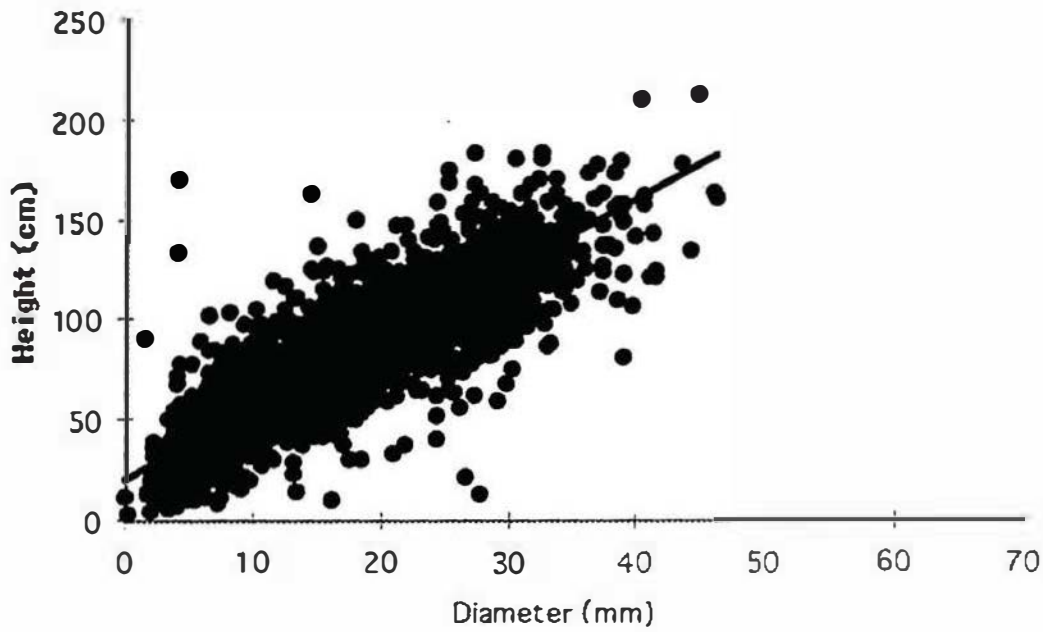


Figure 3.4 : Exponential regression relating the percentage of living, sampled seedlings showing signs of damage at 12 months after planting to the mean browse score calculated only for those seedlings exhibiting signs of browsing damage at 12 months.

A



B

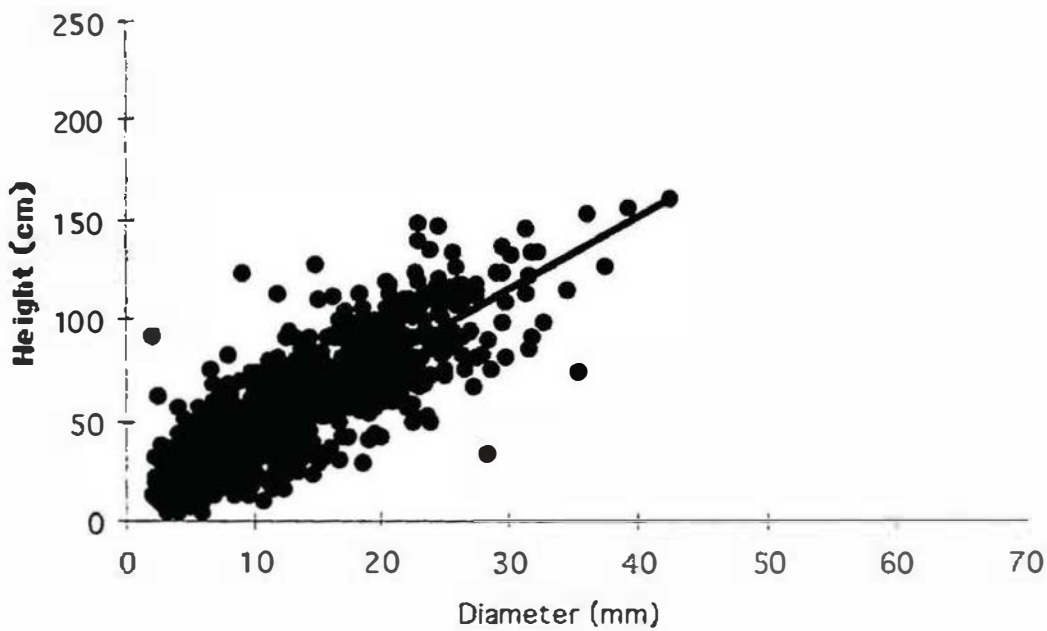


Figure 3.5 : Scatterplot comparing the diameter (mm) of seedlings at six months after planting to the height (cm) at six months after planting for (A) unbrowsed and (B) browsed seedlings.

where 'y' is the height (cm) of a seedling at six months and 'x' is the diameter (mm) at six months.

The same analysis was applied to the 12 month diameter and height data (Figure 3.6), again yielding two highly significant relationships (unbrowsed: $r^2=0.724$, $df=2937$, $p<0.01$; browsed: $r^2=0.722$, $df=2299$, $p<0.01$) described by the following equations :

$$\text{for unbrowsed seedlings} \quad y = 2.67x + 34.08 \quad \text{Equation 3.5}$$

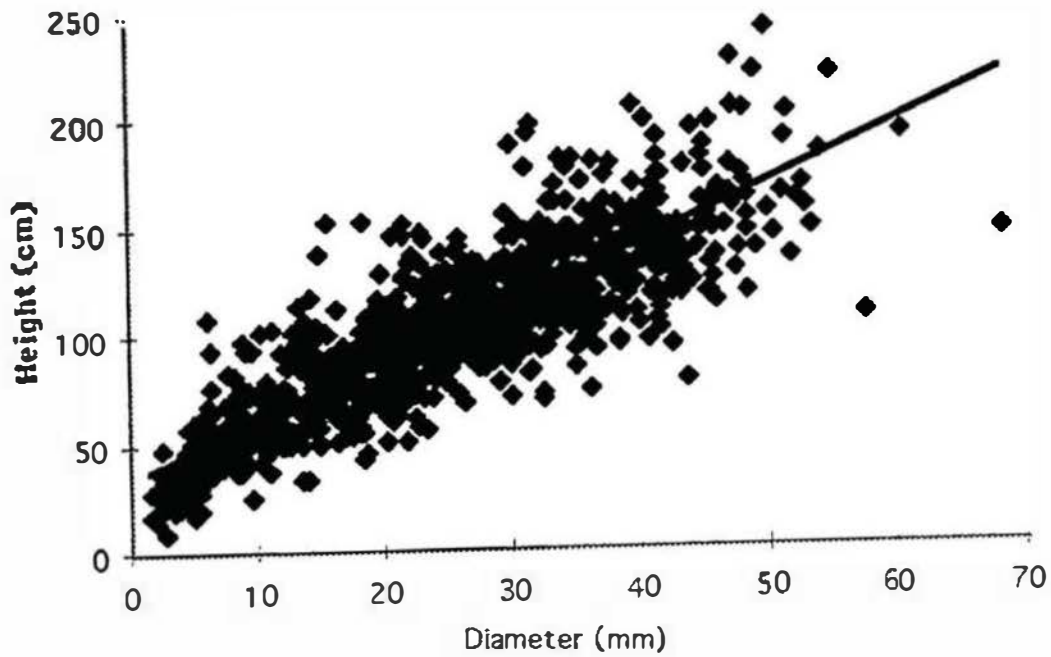
$$\text{for browsed seedlings} \quad y = 3.26x + 8.18 \quad \text{Equation 3.6}$$

Effects on rate of growth

To determine whether browsing had any effect on the average daily growth of seedlings, t-tests were carried out for each plantation comparing the differences between the mean growth of seedlings which were never seen to be browsed ('unbrowsed') and the mean growth for those seedlings which were recorded as having been damaged. T-tests were carried out for both the mean daily height growth and mean daily diameter growth values calculated for the 0-12 month period. A data summary and the results of this analysis are presented in Tables 3.4 and 3.5. Seven plantations have sample sizes less than 10 for unbrowsed seedlings and the t-test results for these plantations have not been considered.

At 18 out of the 35 plantations there was a significant difference between the mean daily height growth in height of seedlings observed to be damaged and those which were never recorded as having been damaged. At 17 of the plantations, seedlings which were not observed to be damaged, increased their height significantly faster than seedlings recorded with browsing damage (as indicated by the positive t-statistic values). However, at one of the plantations (Aww55), seedlings recorded with browsing damage actually increased in height significantly faster than those which were not observed to be damaged.

A



B

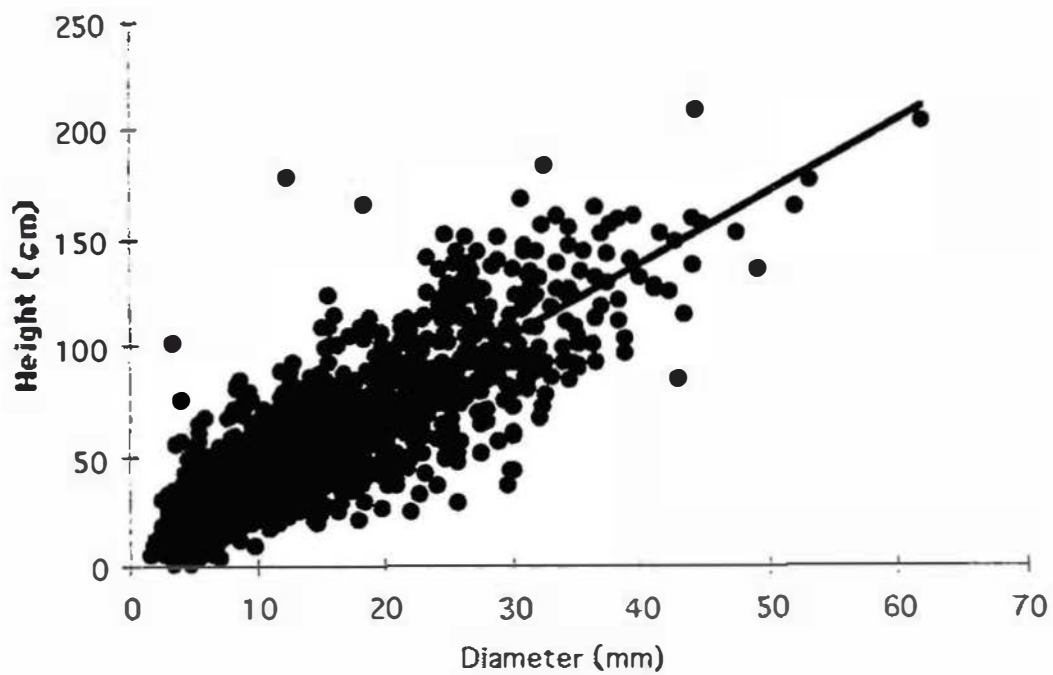


Figure 3.6 : Scatterplot comparing the diameter (mm) of seedlings at 12 months after planting to the height (cm) at 12 months after planting for (A) unbrowsed and (B) browsed seedlings.

Table 3.4 : Results of t-tests comparing mean daily height growth values (calculated for 0-12 months) for browsed and unbrowsed seedlings. Standard deviations are given in parenthesis. Sample size (n), t-statistic values (t) and associated levels of significance (p) have been included. Calculated levels of significance have not been included for those comparisons where the sample size for 'unbrowsed' seedlings is < 10.

Plantation	Browsing status	Mean growth (mm/day)	n	t	p
Adu29	Unbrowsed	1.86 (1.05)	102	3.689	p<0.01
	Browsed	1.32 (0.96)	89		
Alo48	Unbrowsed	0.42 (0.79)	4	-0.787	-
	Browsed	0.68 (0.65)	168		
Alo49	Unbrowsed	0.20 (0.56)	4	-0.503	-
	Browsed	0.36 (0.63)	172		
Amark	Unbrowsed	0.28 (0.52)	63	1.382	p>0.05
	Browsed	0.15 (0.47)	105		
Aseol	Unbrowsed	-	0	-	-
	Browsed	-0.11 (0.51)	142		
Asx13	Unbrowsed	0.83 (0.61)	22	2.909	p<0.01
	Browsed	0.43 (0.60)	146		
Asx51	Unbrowsed	2.36 (0.75)	91	7.312	p<0.01
	Browsed	1.45 (0.95)	101		
Atm18	Unbrowsed	2.57 (1.10)	100	3.222	p<0.01
	Browsed	2.04 (1.14)	86		
Atm20	Unbrowsed	0.53 (0.44)	15	-0.740	p>0.05
	Browsed	0.61 (0.40)	185		
Aww55	Unbrowsed	1.57 (0.95)	128	-3.687	p<0.01
	Browsed	2.09 (0.92)	68		
Bcam	Unbrowsed	1.95 (0.74)	54	0.482	p>0.05
	Browsed	1.89 (0.78)	130		
Bfr	Unbrowsed	3.45 (1.25)	98	5.219	p<0.01
	Browsed	2.34 (1.17)	50		
Bkui	Unbrowsed	1.39 (0.53)	19	0.952	p>0.05
	Browsed	1.20 (0.85)	165		
Brf1	Unbrowsed	0.62 (0.59)	4	0.882	-
	Browsed	0.47 (0.32)	88		
Brf2	Unbrowsed	1.91 (1.25)	74	5.304	p<0.01
	Browsed	0.97 (0.99)	86		

Table 3.4 (continued) : Results of t-tests comparing mean daily height growth values (calculated for 0-12 months) for browsed and unbrowsed seedlings.

Standard deviations are given in parenthesis. Sample size (n), t-statistic values (t) and calculated levels of significance (p) have been included. Calculated levels of significance have not been included for those comparisons where the sample size for 'unbrowsed' seedlings is < 10.

Plantation	Browsing status	Mean growth (mm/day)	n	t	p
Brin	Unbrowsed	0.20 (0.40)	7	-0.056	-
	Browsed	0.21 (0.46)	125		
Bsch	Unbrowsed	2.02 (0.85)	111	1.762	p>0.05
	Browsed	1.78 (0.98)	73		
Bswan	Unbrowsed	2.53 (0.91)	141	2.930	p<0.01
	Browsed	2.11 (0.88)	55		
Bswan2	Unbrowsed	2.76 (0.95)	65	13.927	p<0.01
	Browsed	1.53 (0.11)	118		
Bwal	Unbrowsed	4.12 (1.41)	89	8.288	p<0.01
	Browsed	2.27 (1.50)	81		
Bwee	Unbrowsed	3.27 (1.06)	7	7.530	-
	Browsed	0.69 (0.88)	159		
Bwfr	Unbrowsed	2.55 (0.77)	89	11.715	p<0.01
	Browsed	1.19 (0.77)	87		
Fha21h	Unbrowsed	0.84 (0.74)	59	0.658	p>0.05
	Browsed	0.75 (0.91)	117		
Fhp23g	Unbrowsed	1.93 (1.13)	116	2.167	p<0.05
	Browsed	1.57 (0.90)	62		
Fkd13b	Unbrowsed	2.83 (1.20)	144	3.128	p<0.01
	Browsed	2.22 (1.22)	52		
Fsolb	Unbrowsed	1.76 (0.94)	119	3.564	p<0.01
	Browsed	1.26 (0.97)	75		
Nalans	Unbrowsed	3.19 (0.94)	125	8.273	p<0.01
	Browsed	1.70 (1.48)	59		
Nbless	Unbrowsed	2.70 (0.79)	95	8.960	p<0.01
	Browsed	1.54 (0.97)	91		
Ndig	Unbrowsed	2.19 (1.55)	100	0.393	p>0.05
	Browsed	2.10 (1.27)	66		
Nped	Unbrowsed	2.48 (1.34)	91	3.192	p<0.01
	Browsed	1.74 (1.44)	58		
Npu	Unbrowsed	0.42	1	-	-
	Browsed	0.27 (0.38)	171		
Nrit	Unbrowsed	1.50 (0.96)	42	1.051	p>0.05
	Browsed	1.27 (1.31)	133		
Nspill	Unbrowsed	3.21 (1.32)	120	5.256	p<0.01
	Browsed	2.06 (1.25)	50		

Table 3.5 : Results of t-tests comparing mean daily diameter growth values (calculated for 0-12 months) for browsed and unbrowsed seedlings. Standard deviations are given in parenthesis. Sample size (n), t-statistic values (t) and calculated levels of significance (p) have been included. Calculated levels of significance have not been included for those comparisons where the sample size for 'unbrowsed' seedlings is < 10.

Plantation	Browsing status	Mean growth (mm/day)	n	t	p
Adu29	Unbrowsed	0.053 (0.033)	102	0.700	p>0.05
	Browsed	0.050 (0.025)	89		
Alo48	Unbrowsed	0.018 (0.036)	4	0.000	-
	Browsed	0.018 (0.018)	168		
Alo49	Unbrowsed	0.002 (0.009)	4	-2.197	-
	Browsed	0.012 (0.009)	172		
Amark	Unbrowsed	0.006 (0.006)	63	-1.715	p>0.05
	Browsed	0.008 (0.008)	105		
Aseol	Unbrowsed	-	0	-	-
	Browsed	-0.007 (0.007)	142		
Asx13	Unbrowsed	0.021 (0.012)	22	2.685	p<0.01
	Browsed	0.012 (0.015)	146		
Asx51	Unbrowsed	0.052 (0.021)	91	4.586	p<0.01
	Browsed	0.037 (0.024)	101		
Atn18	Unbrowsed	0.052 (0.029)	100	1.721	p>0.05
	Browsed	0.045 (0.026)	86		
Atn20	Unbrowsed	0.006 (0.010)	15	-0.913	p>0.05
	Browsed	0.008 (0.008)	185		
Aww55	Unbrowsed	0.034 (0.019)	128	-1.659	p>0.05
	Browsed	0.039 (0.022)	68		
Bcam	Unbrowsed	0.045 (0.021)	54	-0.236	p>0.05
	Browsed	0.046 (0.028)	130		
Bfr	Unbrowsed	0.084 (0.032)	98	4.039	p<0.01
	Browsed	0.062 (0.030)	50		
Bkui	Unbrowsed	0.037 (0.022)	19	-0.311	p>0.05
	Browsed	0.039 (0.027)	165		
Brf1	Unbrowsed	0.003 (0.012)	4	0.271	-
	Browsed	0.002 (0.007)	88		
Brf2	Unbrowsed	0.051 (0.036)	74	4.207	p<0.01
	Browsed	0.030 (0.027)	86		

Table 3.5 (continued) : Results of t-tests comparing mean daily diameter growth values (calculated for 0-12 months) for browsed and unbrowsed seedlings. Standard deviations are given in parenthesis. Sample size (n), t-statistic values (t) and calculated levels of significance (p) have been included. Calculated levels of significance have not been included for those comparisons where the sample size for 'unbrowsed' seedlings is < 10.

Plantation	Browsing status	Mean growth (mm/day)	n	t	p
Brin	Unbrowsed	0.030 (0.032)	7	1.827	-
	Browsed	0.012 (0.025)	125		
Bschr	Unbrowsed	0.055 (0.024)	111	1.904	p>0.05
	Browsed	0.048 (0.025)	73		
Bswan	Unbrowsed	0.057 (0.028)	141	0.000	p>0.05
	Browsed	0.057 (0.024)	55		
Bswan2	Unbrowsed	0.079 (0.030)	65	6.536	p<0.01
	Browsed	0.050 (0.028)	118		
Bwal	Unbrowsed	0.097 (0.033)	89	7.780	p<0.01
	Browsed	0.057 (0.034)	81		
Bwee	Unbrowsed	0.077 (0.011)	7	6.558	-
	Browsed	0.027 (0.020)	159		
Bwfr	Unbrowsed	0.054 (0.016)	89	9.608	p<0.01
	Browsed	0.027 (0.021)	87		
Fha21h	Unbrowsed	0.035 (0.027)	59	3.423	p<0.01
	Browsed	0.022 (0.022)	117		
Fhp23g	Unbrowsed	0.031 (0.026)	116	0.261	p>0.05
	Browsed	0.030 (0.021)	62		
Fkd13b	Unbrowsed	0.068 (0.032)	144	3.643	p<0.01
	Browsed	0.050 (0.026)	52		
Fsolb	Unbrowsed	0.037 (0.028)	119	2.812	p<0.01
	Browsed	0.026 (0.024)	75		
Nalans	Unbrowsed	0.099 (0.032)	125	7.629	p<0.01
	Browsed	0.058 (0.038)	59		
Nbless	Unbrowsed	0.088 (0.031)	95	7.820	p<0.01
	Browsed	0.053 (0.030)	91		
Ndig	Unbrowsed	0.050 (0.039)	100	0.340	p>0.05
	Browsed	0.048 (0.034)	66		
Nped	Unbrowsed	0.072 (0.039)	91	3.291	p<0.01
	Browsed	0.050 (0.041)	58		
Npu	Unbrowsed	0.016	1	-	-
	Browsed	0.013 (0.013)	171		
Nrit	Unbrowsed	0.039 (0.042)	42	1.665	p>0.05
	Browsed	0.029 (0.031)	133		
Nspill	Unbrowsed	0.075 (0.034)	120	4.553	p<0.01
	Browsed	0.050 (0.029)	50		

3.3.4 Quantifying the relationship between damage and growth

In order to quantify the relationship between browsing damage severity and the daily growth in height and diameter for *E. nitens* seedlings, the daily growth data for each *E. nitens* plantation was sorted by dividing seedlings into six categories, based on the damage score (0-5) recorded from seedlings at 12 months. Thus, for each *E. nitens* plantation, the daily growth values for each seedling with a browsing score of zero were grouped together, the daily growth values for each seedling with a browse score of 1 were grouped together and so on. The mean daily growth values for seedlings from each plantation in each browsing damage category were then calculated and plotted in Figures 3.7 (mean daily height growth) and 3.8 (mean daily diameter growth). The trend is for mean daily growth values to decrease with increases in browsing damage score. However, for both height and diameter there is a considerable amount of scatter of data points, particularly at low levels of damage.

The results of the analyses carried out with the daily diameter growth data largely reflect the results obtained from the daily height growth data. In total there were significant differences in mean daily diameter growth values at 14 plantations. All of the significant t-statistic values obtained indicated that the mean daily diameter growth of the unbrowsed seedlings was significantly larger than that for the browsed seedlings.

Because of variation in environmental conditions, genetic stock and planting practices, seedling growth is likely to be inherently variable between plantations. This inherent variation in growth confounds any analysis relating daily growth values to browsing damage and may explain the scatter of points observed in Figures 3.7 and 3.8. A further treatment of the data was performed to take these differential growth rates into account. The daily growth and damage data for each *E. nitens* plantation was sorted as above. The 'base' daily growth was then set as the mean daily growth value obtained for seedlings which never showed signs of browsing damage (i.e. seedlings in the zero browsed category at both 6 and 12

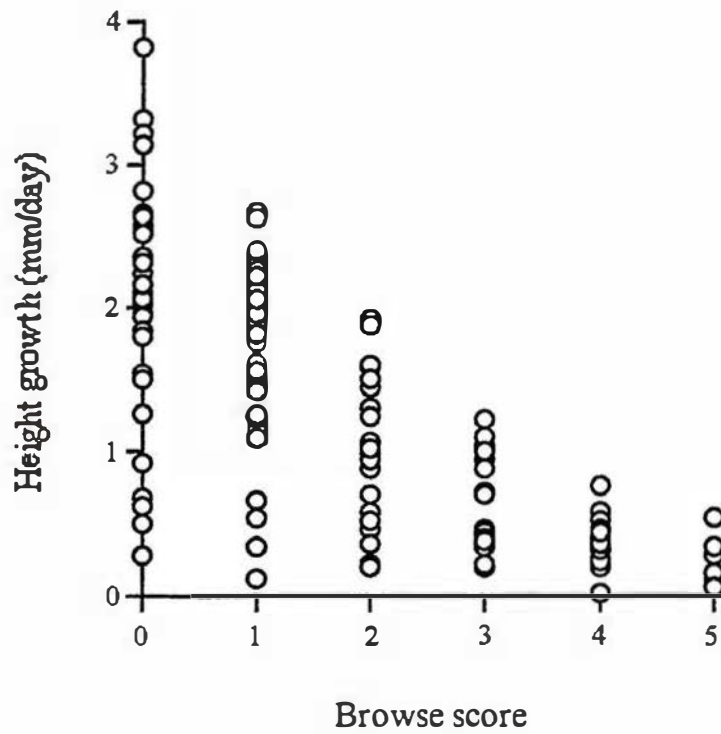


Figure 3.7 : The mean daily height growth (mm/day) calculated for seedlings from each *E. nitens* plantation, in each browsing score. The data presented are those collected 12 months after seedlings were planted.

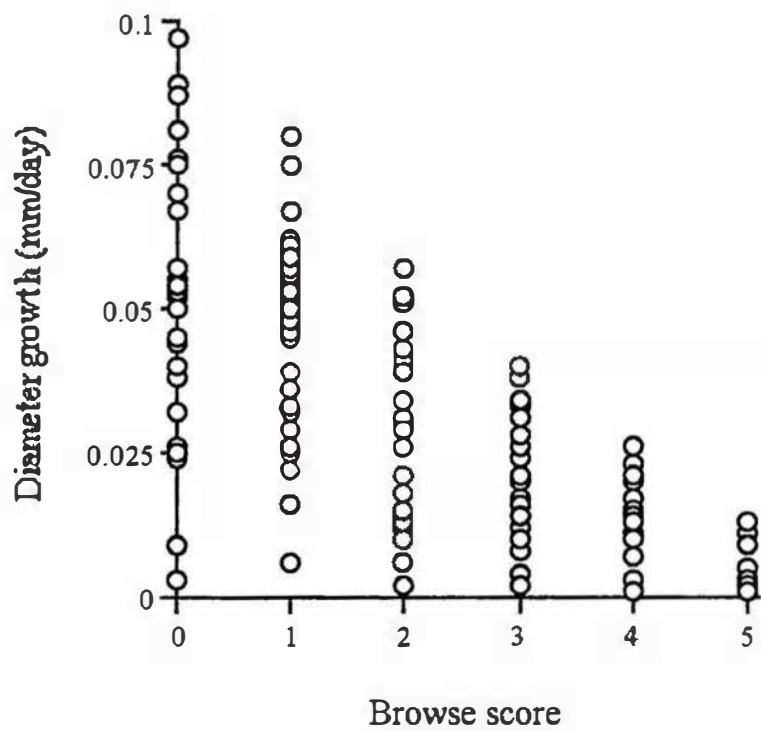


Figure 3.8 : The mean daily diameter growth (mm/day) calculated for seedlings from each *E. nitens* plantation, in each browsing score. The data presented are those collected 12 months after seedlings were planted.

months). The assumption made was that the zero browsed seedlings did not differ physiologically from the browsed seedlings and, thus, the daily growth values for the unbrowsed seedlings are representative of the average rate for all seedlings in the absence of browsing¹. The difference between the daily growth values of unbrowsed seedlings and seedlings which were browsed was expressed as a percentage using the following formula :

$$\% \text{ difference} = \frac{(\text{growth of unbrowsed} - \text{growth of browsed})}{\text{growth of unbrowsed}} \times 100$$

For the purposes of further discussion this value will be referred to as the 'percentage reduction' in daily growth caused by browsing damage. In Figures 3.9 and 3.10 percentage reduction in mean daily height and diameter growth (0-12 months) values have been regressed against browsing damage levels. Both regressions are highly significant (height : $r^2=0.798$, $df=105$, $p<0.01$; diameter $r^2=0.812$, $df=105$, $p<0.01$) and can be described by the following equations :

$$\text{Height : } y = 19.055x + 1.458 \quad \text{Equation 3.7}$$

$$\text{Diameter : } y = 17.054x + 0.712 \quad \text{Equation 3.8}$$

where 'y' is the percentage reduction in daily growth and 'x' is the browse score.

The speed at which an *E. nitens* seedling increases its height or diameter is strongly related to browsing damage severity. Browsed seedlings tend to increase their height and diameter more slowly than those seedlings which were never observed to be browsed. The amount by which mean daily height and diameter growth of browsed seedlings is reduced, when compared to unbrowsed seedlings, approaches 100% (i.e. no growth) as mean browse score (12 months) approaches the maximum level of 5.

¹ A discussion of the validity of this assumption is provided in section 3.4.6.

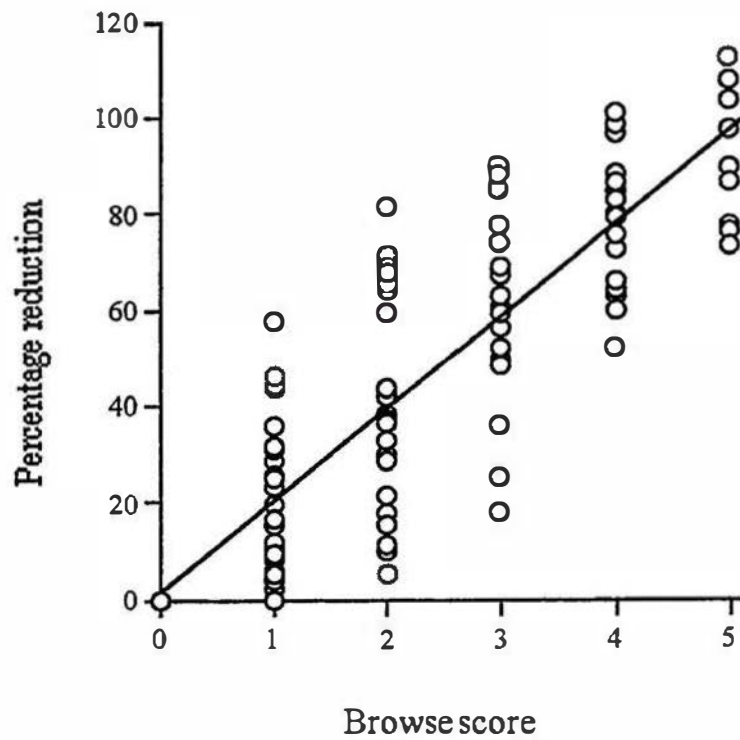


Figure 3.9 : Linear regression comparing calculated percentage reduction in mean daily height growth (mm/day) to browse score. The results presented are based on calculations made using the data collected from *E. nitens* plantations 12 months after seedlings were planted.

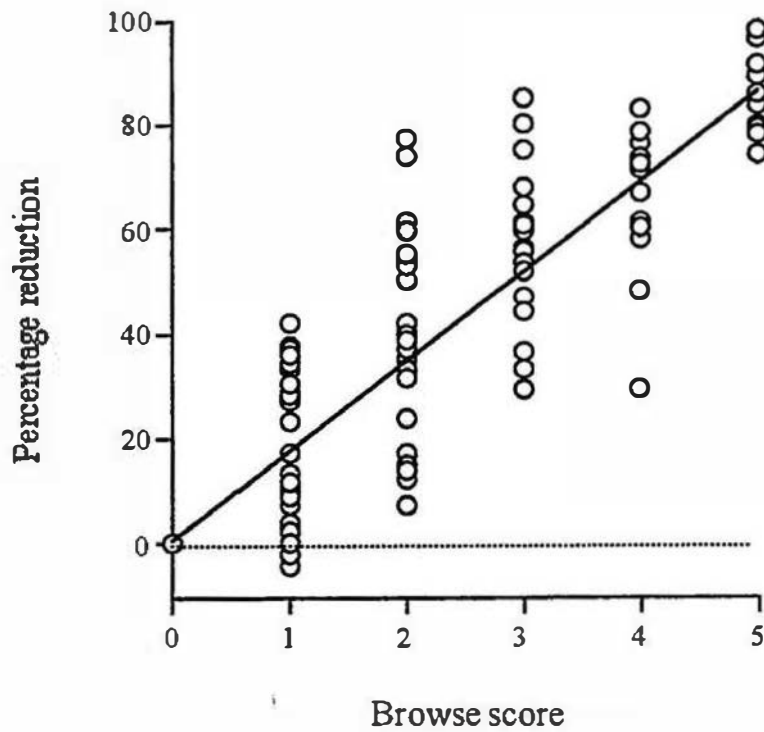


Figure 3.10 : Linear regression comparing calculated percentage reduction in mean daily diameter growth (mm/day) to browse score. The results presented are based on calculations made using the data collected from *E. nitens* plantations 12 months after seedlings were planted.

3.3.5 Seedling survival

The percentage of sampled seedlings which were still alive at six and 12 months is shown in Table 3.6. Seedling survival to six months ranged from 49 to 100%. Seedling survival to 12 months ranged from 46 to 100%. The average survival for all plantations to six months was 90.9% (sd=9.9). The average survival to 12 months was 88.3% (sd=10.7). This represents a drop in mean survival of only 2.6%, indicating that the majority of seedlings surviving to six months would continue to survive through to 12 months.

Table 3.6 : The percentage of marked seedlings surviving to six and 12 months at each of the 35 surveyed forestry plantations.

Plantation	% survival (6 months)	% survival (12 months)	Plantation	% survival (6 months)	% survival (12 months)
Adu29	99	96	Bswan2	96	95
Alo48	88	86	Bwal	89	85
Alo49	90	88	Bwee	94	83
Amark	84	84	Bwfr	90	88
Aseol	85	71	Fha21h	96	95
Asx13	94	84	Fhp23g	94	89
Asx51	96	96	Fkd13b	98	98
Atm18	94	93	Fsolb	97	97
Atm20	100	100	Nalans	94	94
Aww55	98	98	Nbless	96	93
Bcam	92	92	Ndig	83	83
Bfr	99	99	Nped	86	83
Blui	96	92	Npu	86	86
Brf1	49	46	Nrit	91	90
Brf2	80	80	Nspill	97	85
Brin	68	66	Ncw	94	88
Bsch	93	92	Npear	99	98
Bswan	98	98			

Survival and browsing damage

In Figures 3.11 and 3.12 the percentage of seedlings which were still surviving at six and 12 months (respectively) at *E. nitens* plantations, have been regressed against the mean browse score for each plantation. There was no significant relationship between the percentage of seedlings still surviving at six months and

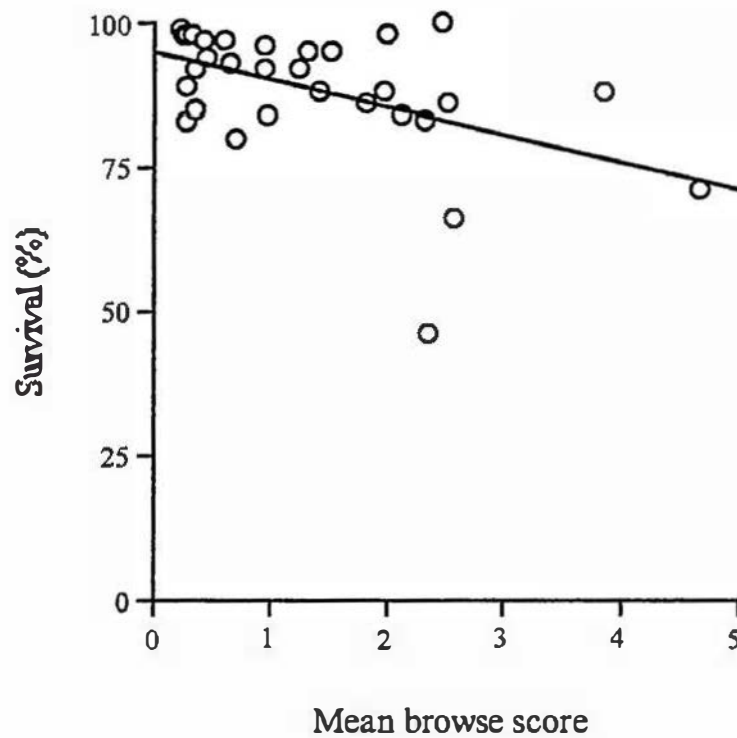


Figure 3.11 : Linear regression relating the percentage of seedlings which were surviving at each *E. nitens* plantation six months after planting, to mean browse score recorded at six months.

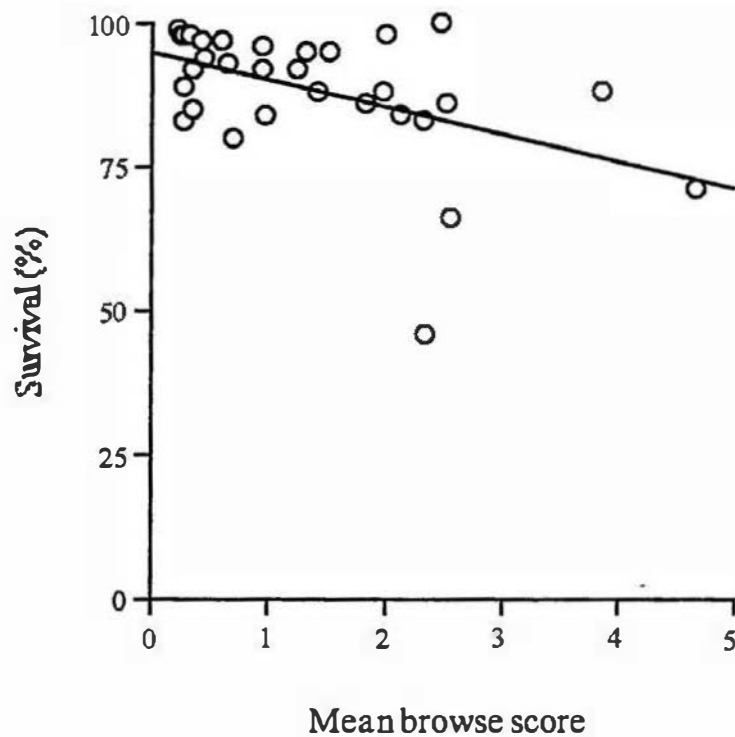


Figure 3.12 : Linear regression relating the percentage of seedlings which were surviving at each *E. nitens* plantation at 12 months after planting, to mean browse score recorded at 12 months.

the mean browse score at six months ($r^2=0.083$, $df=31$, $p>0.05$). There is a significant, negatively sloped relationship between the survival of seedlings to 12 months and the mean browse score measured at 12 months ($r^2=0.227$, $df=31$, $p<0.01$). However, the trend is not convincing given the large amount of variation of data around the fitted line.

Survival and average daily growth

In Figures 3.13 and 3.14 the percentage of seedlings at each *E. nitens* plantation surviving to six and 12 months, respectively, has been regressed against mean daily height growth values for the 0-6 month and 0-12 month periods. Both regressions are significant (six months : $r^2=0.145$, $df=29$, $p<0.05$; 12 months : $r^2=0.260$, $df=29$, $p<0.01$) but both models have low associated correlation coefficients. The trend from both data sets is for seedling survival to be slightly greater at plantations with high values for mean daily height growth. However, the small values for the slope of the regressions (six months : 3.29; 12 months : 6.30) indicate that the effect is not strong. The value of the trend is further reduced because the two very low survival rates to the left of the plot have a strong influence on slope.

Regressions were also carried out between seedling survival measures and the mean daily diameter growth data at six and 12 months. Figures have not been presented because the trends are very similar to those obtained using the mean daily height growth data. Both regressions are significant (six months : $r^2=0.161$, $df=29$, $p<0.05$; 12 months $r^2=0.268$, $df=29$, $p<0.05$) but have low correlation coefficients. The small positive slope associated with the regression lines is, once again, strongly influenced by the two low, outlying, survival values.

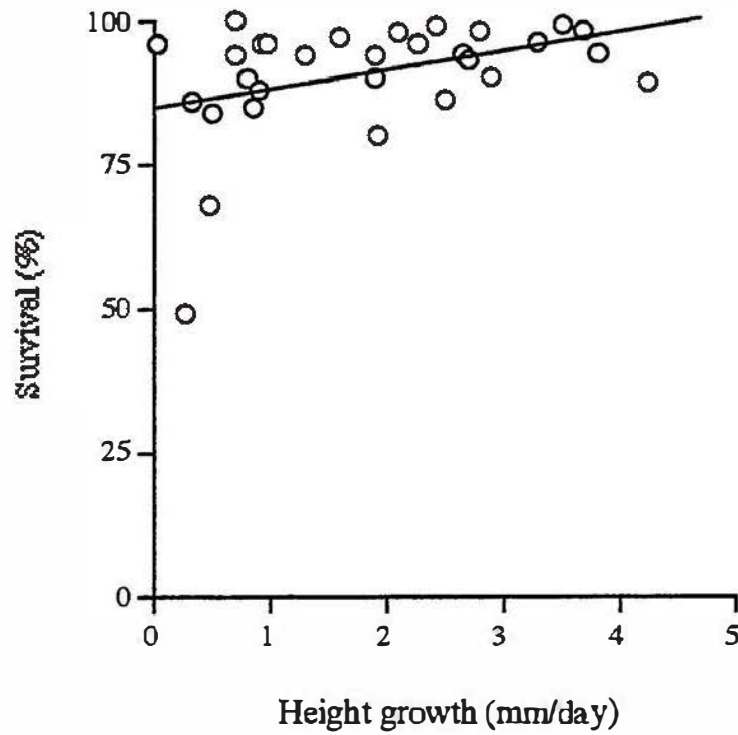


Figure 3.13 : Linear regression relating the percentage of marked seedlings at each *E. nitens* plantation still surviving at six months after planting, to mean daily height growth values calculated for the 0-6 month period.

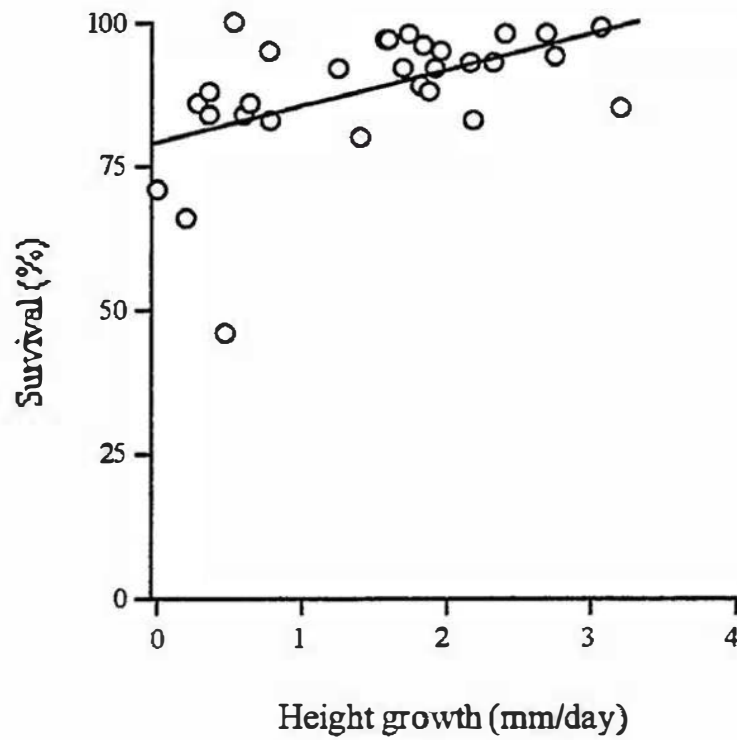


Figure 3.14 : Linear regression relating the percentage of marked seedlings at each *E. nitens* plantation still surviving at 12 months after planting, to mean daily height growth values for each plantation calculated for the 0-12 month period.

3.4 DISCUSSION

3.4.1 Seedling growth

Curtis (1967) showed that the relationship between Douglas fir (*Pseudotsuga menziesii*) height and diameter measures can be generally described using linear functions. Following this, West (1979) demonstrated that linear functions work well for describing the same relationship for established eucalypt trees of 20-100 years of age. In this study strong, positive, linear relationships were observed between the diameter and height of *E. nitens* seedlings up to one year of age. Work by Donald (1976) indicated that such relationships may remain constant regardless of site variation. However, a more recent study by Snowdon (1981), which showed that the application of fertiliser can affect the relationship between height and diameter for *Pinus radiata*, has cast serious doubt on Donald's (1976) conclusions.

Generally, the mean daily height growth values recorded at plantations were similar to the range of growth rates which Turnbull *et al.* (1993) reported for apparently unbrowsed first year growth of *E. nitens* and *E. globulus* in Tasmania. However, in my study there was a large variation in the growth rates of seedlings from site to site, and at some plantations mean daily height growth values were considerably lower than those reported by Turnbull *et al.* (1993). The variation observed both within and between studies is likely to reflect differences in genetic stock, silvicultural practices and environmental conditions such as soil type, aspect, altitude, rainfall, nutrient levels and the level of attack from insects and mammals.

3.4.2 Browsing damage

The occurrence of browsing damage is widespread. Of the 35 plantations surveyed there was not one at which no browsing damage was recorded. However,

browsing damage severities were not uniform across sites, ranging from a very low value of 0.24 to a severe damage score of 4.69 (mean browse scores recorded at 12 months). Such spatial variation in damage is consistent with anecdotal evidence for Tasmanian forestry plantations (Wilkinson and Nielsen 1995; Coleman *et al.* 1997) and results reported from studies of a wide range of crops and their pest species. Montague (1996), for example, observed browsing damage to vary amongst eucalypt plantations in Victoria. Hone (1995b) observed great spatial variation in the amount of damage caused by feral pigs (*Sus scrofa*), Buckle *et al.* (1984) found variation in rat damage to rice crops, and Hothem *et al.* (1988) found that sunflowers were highly damaged at some sites but not damaged at many others.

The factors which are thought to influence such spatial variation in damage are varied. For example, Conover (1989) observed wide variation between nursery sites in the percentage of yew shoots (0-94%) which were damaged by white-tailed deer (*Odocoileus virginianus*). He successfully related this variation to a measure of deer abundance and the size of woodlots adjacent to nurseries. Kay (1993) observed spatial variation in levels of deer damage to coppiced woodlands. The variation observed in her study was best explained by the presence/absence of fences, the length of perimeter of the woodlot and the abundance of deer species.

Anecdotal evidence would suggest that the damage caused to eucalypt seedlings in Tasmania may also vary in time (Cremer 1969; Wilkinson and Nielsen 1995). Coleman *et al.* (1997) suggest that damage may be greatest in the June to August period when alternative foods are in shortest supply. Montague (1996) observed browsing damage to occur to eucalypts in Victorian plantations throughout the year, but damage peaked between September and November. In my study the damage levels recorded at 12 months were significantly higher than the damage levels recorded at six months. Unfortunately, little can be inferred from this result in relation to the timing of damage. It was impossible to distinguish between the damage caused to seedlings in the 0-6 month period and the damage caused in the

6-12 month period. It is possible that the higher values observed at 12 months reflect a cumulative effect rather than any real temporal difference. In other words, the higher damage levels recorded at 12 months may be a result of recording the 'old' damage from the six months assessment plus any 'new' damage which occurred since then.

A very strong relationship was observed between the two measures of browsing damage used in this study : the percentage of seedlings observed with damage at a plantation and the mean browsing score for that plantation. This held true for both the mean values calculated for browse scores for all seedlings (includes zero browsed seedlings) and the mean values calculated for just those seedlings with signs of browsing damage. For any given plantation then, the average severity of damage to individual seedlings appears to be related to how widespread the damage is at the plantation. The more widespread the damage, the more severe the damage to individual seedlings is likely to be. One possible explanation for this trend is that, whilst feeding in plantations, Tasmanian herbivores do not expend a large proportion of total feeding effort on actively seeking out seedlings to consume. Instead they may consume small portions of seedlings as they are encountered during foraging bouts. However, this data is not conclusive and further experimentation will be required to confirm or invalidate this suggestion.

3.4.3 Browsing damage and seedling growth

Much of the literature concerning the effects of browsing on plant growth comes from the northern hemisphere. The results of these studies are conflicting and the reported effects of browsing or simulated browsing range from deleterious (Grant *et al.* 1982; Elliot *et al.* 1993) through neutral (Strauss 1991) to beneficial (see review by Belsky 1986). For woody tree species, researchers have generally found that if defoliation is extensive enough then growth will be affected (e.g. Bergstrom and Danell 1987, 1995), although this may also depend on the timing and persistence of the browsing attack.

of this study indicate that browsing damage has a strong negative effect on seedling growth. At 18 of the surveyed plantations, seedlings recorded with browsing damage grew significantly slower than seedlings which were not recorded with damage. This result is consistent with other Tasmanian and mainland Australian studies. Both Gilbert (1961) and Cremer (1969) found that fenced *E. regnans* seedlings grew better than unfenced seedlings exposed to browsing damage. Various simulated browsing experiments (Cremer 1969, 1972, 1973; Nielsen and Pataczek 1991; Wilkinson and Nielsen 1995,) have also shown that extreme levels of defoliation reduce the growth rates of eucalypt seedlings, although it should be noted that the magnitude of these effects might have been inflated by the effects of competing weed species.

Nielsen and Pataczek (1991) and Wilkinson and Nielsen (1995) observed that the effect of simulated browsing damage on seedling growth was dependent on the amount of foliage removed. Low levels of foliage removal did not result in significantly reduced growth rates. However, when foliage removal was severe, growth was significantly reduced, an effect which lasted for several years. The results of this study also show that the effect of browsing damage on growth is highly dependent on damage severity. The amount by which a browsed *E. nitens* seedling's growth was reduced, as compared to unbrowsed seedlings, was found to increase linearly with increasing browsing damage score. As browse score approached the maximum value of five, reductions in seedling growth of up to 100% could be expected. Indeed where browsing damage was extreme, some seedlings actually decreased in height.

3.4.4 Seedling survival

Although there was some variation between plantations, there was generally a high rate of seedling survival. The average survival of seedlings to 12 months was 88%. This result is remarkably similar to that reported by Montague (1996) for survival of *E. regnans* seedlings to nine months in Victorian plantation areas (84% still present and alive). Unlike Montague (1996) however, it cannot be

concluded from the weak evidence presented here that browsing damage had any effect on seedling survival. It appears that, while browsing damage may exercise a subtle influence on seedling survival, other factors are more likely to be responsible for early losses. In the short term this may include the stresses associated with transferring seedlings from a nursery environment to the forestry coupe. At one plantation (Brf1), for example, an exceptionally low survival rate to six months (46%) was attributed to seedlings being allowed to dry out before planting (P. Naughton pers. comm. 1994). In the longer term, contributing factors to mortality may include water stress, disease, insect attack, competition from weed species, stochastic weather events like heavy frosts, as well as the damage caused by browsers.

Other authors have also found plant mortality to be unaffected by removal of biomass (e.g. Bergstrom and Danell 1987). However, it generally appears that the mortality of seedlings increases when browsing is severe. In a North American study, Myster and McCarthy (1989) found that only 10% of *Carya* seedlings which were unprotected from herbivores survived to an age of 2 years. Cremer (1969, 1972, 1973) and Wilkinson and Nielsen (1995) observed that heavy levels of simulated browsing (e.g. total crown removal) significantly reduced the survival of eucalypt seedlings. This effect could extend for up to seven years (Wilkinson and Nielsen 1995). In separate field studies, Gilbert (1961) and Cremer (1969) demonstrated that *E. regnans* seedlings exposed to browsing suffered greater mortality than those protected by fencing. It should be noted that mortality may have been more the result of a reduced ability to out-compete weed species than a direct effect of biomass removal.

It is possible that, if the seedlings in this study continued to suffer browsing pressure, long term seedling survival may be adversely affected. Persistent browsing may, for example, reduce a plant's ability to compete with its neighbours. The literature is replete with studies showing that plant species in a community can attain dominance by virtue of low susceptibility to browsing (eg.

Neave and Tanton 1989; Anderson and Katz 1993). Wilkinson and Nielsen (1995) observed that some heavily defoliated eucalypt seedlings in their study were totally suppressed by competing species.

3.4.5 Management implications

The results presented in this chapter confirm that the browsing problem is a real one. Damage to seedlings was recorded at all plantations and in some instances it was severe. The main effect of this damage was to reduce the growth rates of seedlings. By quantifying the relationship between damage and seedling growth a framework now exists whereby managers are able to determine what constitutes an acceptable level of damage and what levels of damage should be aimed for if eucalypt plantations are to be successfully established. The implications of these findings for management will be explored in detail in Chapter 5.

3.4.6 Limitations and confounding effects

This study has been largely one of observation, rather than experimentation. There are a number of limitations associated with such an approach. Several comparisons were made between the growth of browsed and 'unbrowsed' seedlings. Effectively those seedlings which were not scored with damage were treated as a control group. One assumption was that seedlings which were scored as having no damage were not attacked by browsers and their growth was consequently not affected by browsing. However, this assumption does not necessarily hold. It is possible that seedlings which were not scored as being browsed could have actually been damaged at some stage. Damage was recorded at six monthly intervals. Conceivably, if a seedling was damaged sufficiently early, it could recover from browsing damage and still be scored as a zero browsed seedling. This is despite the fact that its growth rate may have been reduced by the browsing attack. As a result, the data presented in this chapter may actually underestimate the negative effects of browsing damage on seedling growth.

In order to use seedlings which were not observed to be browsed as a control group, it must also be assumed that they are not, in some way, inherently different from those seedlings which were observed to be browsed. If this assumption is incorrect the conclusions drawn in this chapter would be compromised. Consider, for example, the comparisons made between the mean daily growth in height of browsed seedlings to that of unbrowsed seedlings (Table 3.4). The results strongly suggest that browsing damage exerts a strong, negative influence on daily height growth. However, this cannot be stated unequivocally because the same pattern in results would be observed if herbivores deliberately chose to feed on seedlings with a slower growth rate.

These limitations will be addressed in Chapter 4.

CHAPTER 4

AN EXPERIMENTAL STUDY OF THE EFFECTS OF BROWSING DAMAGE ON SEEDLINGS

4.1 INTRODUCTION

In Chapter 3, a study was outlined which aimed to quantify the relationships between severity of browsing damage and the growth and survival of eucalypt seedlings during their first year. The study was carried out at a large number of sites, encompassing a wide range of environmental conditions. The results suggested that the primary effect of browsing damage was to exert a strong negative influence on the rate at which seedlings increased their height and diameter. The degree to which daily height/diameter growth was reduced was linearly related to the severity of browsing damage, as measured by mean browse score, which varied greatly between plantations.

The study was largely observational rather than experimental. As such, the findings are weakened by the lack of control seedlings (free from browsing) for use as a comparison. Instead, comparisons of daily height growth, diameter growth and survival were made between seedlings observed to be damaged by browsers, and seedlings which showed no signs of browsing at either six or 12 months after planting. These comparisons strongly implicated browsing in reducing the rate at which seedlings increased their height and diameter, however, the causal effects could not be totally isolated. For example, an alternative explanation for the results may be that, for some reason, browsers chose to feed on slower growing seedlings.

It is also uncertain whether the seedlings which were never scored with damage were, in fact, totally free from damage themselves. The intervals between damage sampling were large (six months). It is possible that any given seedling could

have been severely browsed before being sampled and, provided browsing events did not persist, may still have regrown sufficiently that they were assessed as having zero damage. Gilbert (1961) noted that *E. regnans* seedlings regenerated rapidly following release from browsing attack. In effect, the results presented in Chapter 3 may underestimate the effects of browsing damage on the growth of seedlings.

In this chapter, an experimental field study is described which aimed to determine more accurately the effect of browsing damage on seedling growth. The data were used to test some of the assumptions made in Chapter 3. Additionally, the effects of browsing damage on seedling form were studied.

4.2 METHODOLOGY

4.2.1 Study sites

This study was carried out at seven geographically distinct forestry plantations scattered around Tasmania (Figure 4.1). These plantations are the Alo48, Alo49, Amark, Aseol, Bswan2, Fha21h and Nalans plantations described in detail in Section 2.2. Coupes were prepared for planting as in Section 2.2. Before planting, coupes were sprayed with herbicide to control weed growth. Carrot baits treated with 1080 poison were laid following a period of free feeding. All coupes were planted with *E. nitens* seedlings between September and October 1996.

4.2.2 Techniques

The study was begun in September/October 1996, following the planting of each site. Ten, randomly located, control plots were established at each plantation immediately after seedlings were planted. A control plot consisted of five seedlings, which were fenced to exclude all vertebrate browsers. Fences were constructed using wire poultry netting (1800×50×1 mm). The top of each plot

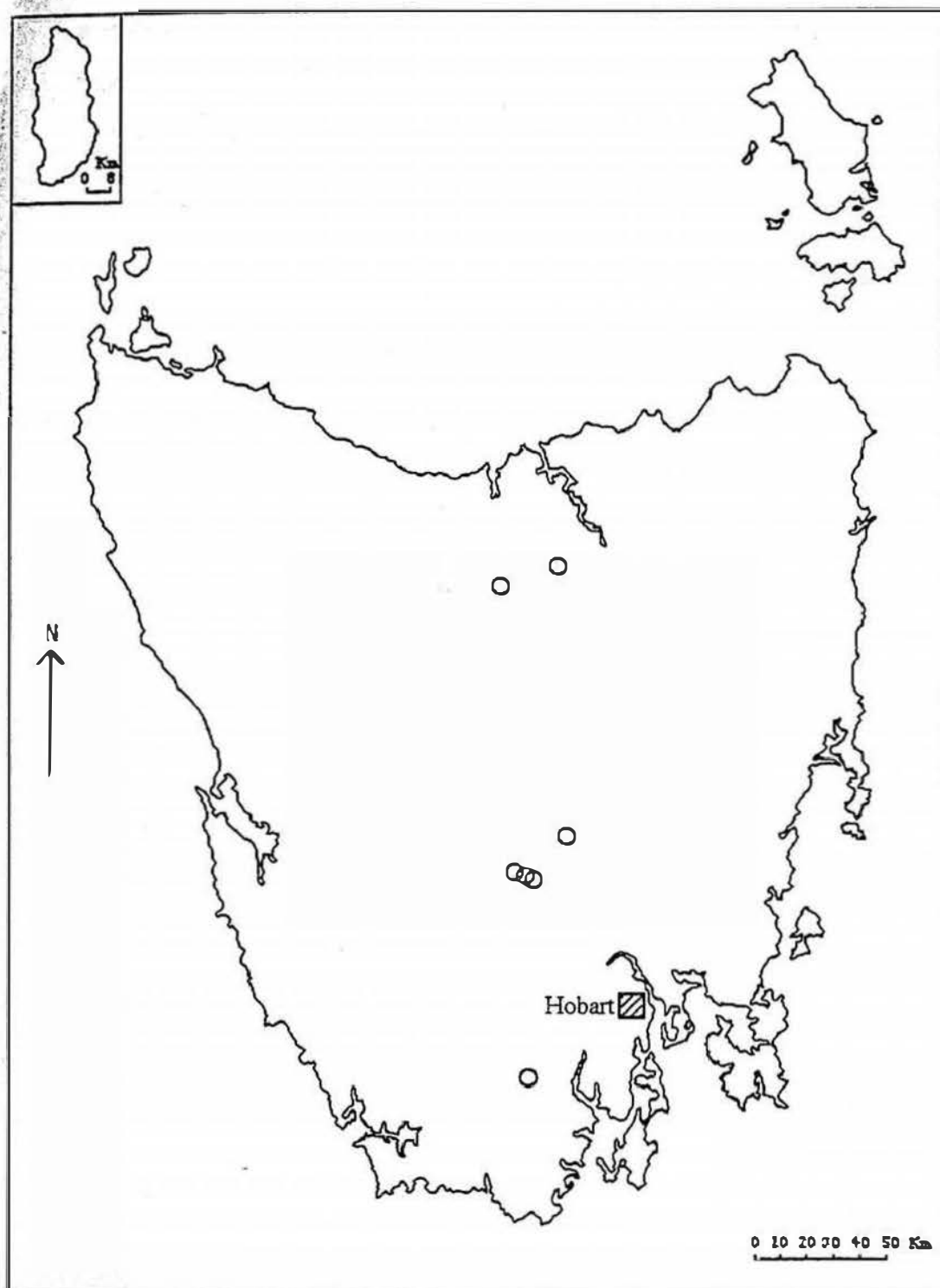


Figure 4.1 : Location of the seven forestry plantations used during this experimental study of the effects of browsing damage on *E. nitens* seedlings.

was covered over with netting (Figure 4.2) and the netting at the bottom of the fence was buried. The condition of enclosure fences was assessed at two monthly intervals. In the 12 months that the study was run, there was never any evidence to suggest that browsers had gained access to fenced seedlings.

Additionally, at each plantation, ten 'treatment' plots were established adjacent to control plots. The distance between control and treatment plots was generally around 10 m, though this could vary slightly depending on the placement of tree rows. Treatment plots were identical to control plots except that they were not fenced so that browsers had free access to seedlings. Treatment plots were marked using wire 'pig-tails' and coloured surveyors tape. Aluminium 'Permotags' were used to identify plots and individual seedlings.

Basal diameter and height measurements (Section 2.4) were taken for both the fenced and unfenced seedlings at the time of planting and then again at six and 12 months after planting. Number of leading shoots (Plate 10) was recorded for all seedlings. Browsing damage was assessed for all seedlings at six and 12 months using the six point scale outlined in Section 2.4.

4.2.3 Problems

During this experiment fencing materials were stolen from four of the control plots. Two of these were at the Fha21h plantation and one each from Amark and Aseo1. Materials from three of the plots, one at each plantation, were taken shortly before the six month sampling period. The materials from the remaining plot (Fha21h) were taken between the six and 12 month sampling period. As a result the seedlings in these plots were exposed to browsing damage and they have been excluded from much of the analysis.

At one plantation, Bswan, seedling growth was so rapid that the tops of plants were in danger of coming through the roof of plots before the experiment was completed. At this plantation the wire netting covering the tops of plots was

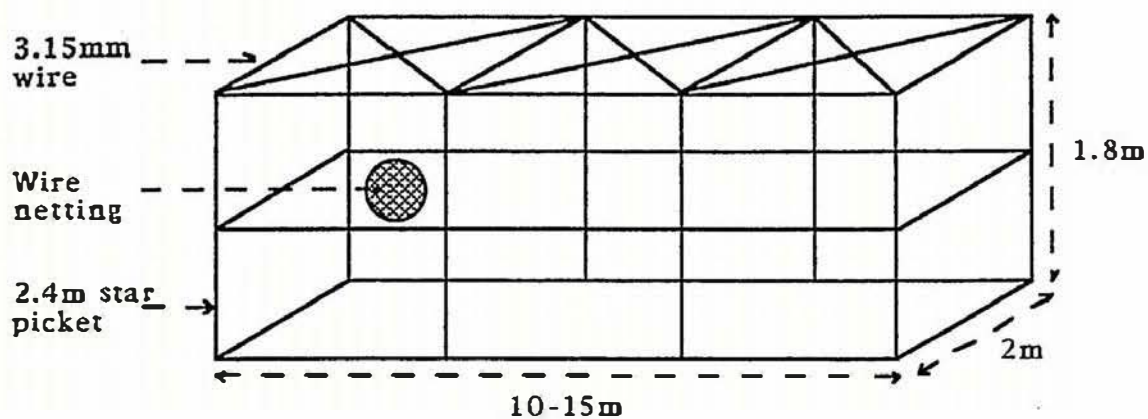


Figure 4.2 : Design of the fenced plots. Plots were 2 metres wide and between 10 m and 15 m long depending on the spacing of seedlings. Star pickets, 2.4 m long, were driven 0.6 m into the ground using a pile driver. These formed the support for the fencing. Wire (3.15 mm diameter) was run through the star pickets to form a frame-work. Wire poultry netting (180×5×1 mm) was then stapled to the sides and top of this frame-work. The bottom edge of the netting was buried under soil.

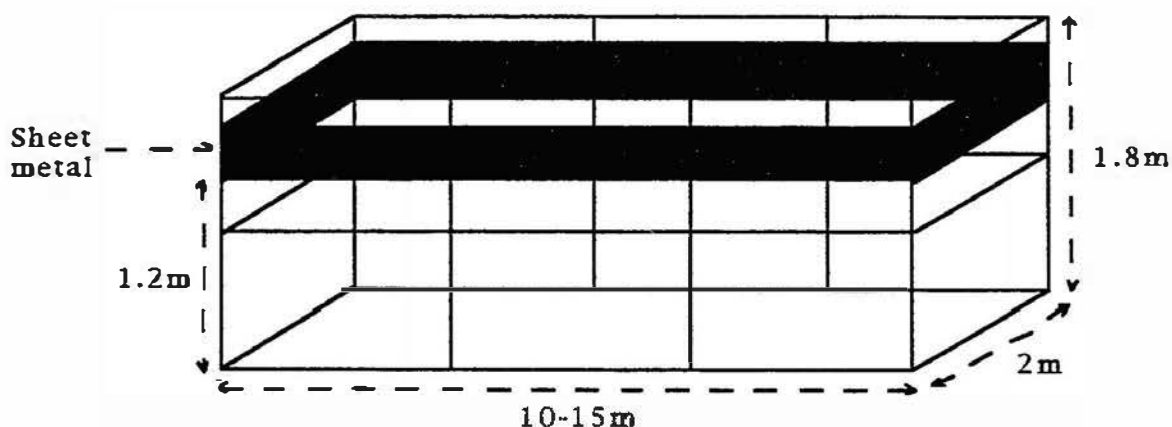


Figure 4.3 : Design of modified plots. The wire netting ceilings in these plots were removed. Galvanised sheeting (2400×40×0.375 mm) was then wired to the sides of the plots so that possums could not climb over.

NB : Drawings not to scale.



Plate 10 : A seedling with two leading shoots.

removed. To ensure possums would not climb over into the plots, galvanised sheeting (thickness : 0.4 mm; dimensions : 2.4×0.4 m) was wired to the sides. Details of the modified plot design are shown in Figure 4.3.

4.2.4 Statistics

The t-test was used to compare differences between control (fenced seedlings) and treatment (unfenced seedlings) groups. Where applicable, the paired t-test was also used to test hypotheses about differences between groups. Two techniques were employed to test for relationships between variables. Linear regression was used where assumptions of normality were not violated. The alternative non-parametric test used was Spearman rank correlation. Because this test is influenced by tied data values, a correction has been applied in all cases where tied data exist (Zar 1984).

In Section 4.3.3 a relationship is described between percentage reduction in daily growth and mean browse score. These ‘percentage reduction’ values were calculated in a similar manner to those discussed in Section 3.3.4. Firstly, it was assumed that the daily height or diameter growth¹ of fenced seedlings (ΔF) was the maximum that can be expected without browsing. It was then assumed that the difference between the height or diameter growth of these and the unfenced seedlings (ΔUF), represents the reduction in growth caused by browsing. This reduction can be expressed as a percentage using the following formula :

$$\text{Percentage reduction} = \frac{(\Delta F - \Delta UF)}{\Delta F} \times 100$$

Because materials were stolen from some of the control plots, these plots had to be excluded from some of the analyses. As a consequence sample sizes vary throughout.

¹ Once again, the term ‘growth’ refers here to the net daily change in a seedling’s proportions and may take positive or negative values (refer Section 3.2.3)

4.3 RESULTS

4.3.1 Damage to unfenced seedlings

In Figure 4.4 the mean browse scores, calculated from the browse scores assigned to each seedling in the ten unfenced plots, is shown for all seven plantations. Data are given for the assessments carried out at both six and 12 months after planting. Though browsing damage to seedlings was evident at all seven plantations, the severity of damage varied greatly. For the six month data mean browsing scores ranged from 0.38 (sd=0.98) to 3.82 (sd=1.62). At 12 months the range was from 0.02 (sd=0.14) to 4.69 (sd=0.53).

4.3.2 Comparing average daily growth for seedlings

Comparison using t-tests of the differences between mean height of fenced seedlings at planting and mean heights for unfenced seedlings indicates that there was no significant difference for any plantation (Table 4.1). Nor was there any significant difference in mean starting diameters.

In Figure 4.5, mean daily height growth for seedlings is shown for the 0-6 months after planting interval. Daily height growth for fenced seedlings varied greatly between plantations, ranging from 0.5 to 5.4 mm per day. The height growth for unfenced seedlings also varied from site to site, ranging from 0.0 to 4.4 mm per day. The trend across all sites was for the fenced seedlings to increase in height faster than the unfenced seedlings. However, the t-test analyses shown in Table 4.2 indicate that the difference is only significant at three plantations (Alo49, Aseol and Bswan2). All three of these plantations have relatively high levels of browsing damage (Figure 4.4).

A similar pattern was observed for daily diameter growth during the 0-6 month period (Figure 4.6). Fenced seedlings tended to increase in diameter at a faster rate than unfenced. The unfenced seedlings on average increased their diameter by 0.01 to 0.12 mm per day. Fenced seedlings on average increased their

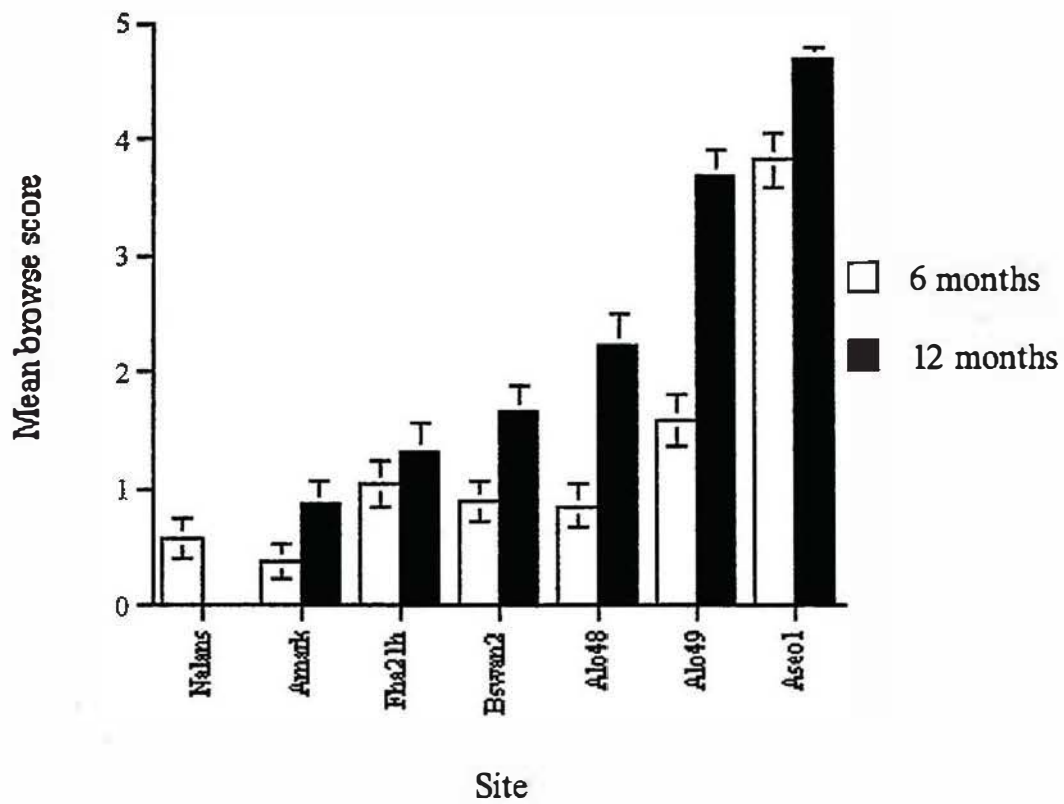


Figure 4.4 : The mean (\pm se) browse scores for all plantations calculated from unfenced seedling data collected at both six and 12 months after planting. The results are ranked from left to right according to increasing mean browse score recorded at 12 months. Browsing damage was recorded at the Nalans plantation at 12 months, but at such a low level that it does not show clearly on this graph.

Table 4.1 : Results of t-tests comparing mean seedling height (cm) and diameter (mm) for fenced and unfenced seedlings at the time of planting. Standard deviations are given in parenthesis. For all tests $df=98$, $p>0.05$.

Seedling height at planting (cm)

Plantation	Fenced seedlings	Unfenced seedlings	t
Nalans	15.0 (3.4)	14.8 (2.7)	0.399
Amark	27.2 (9.2)	29.7 (6.4)	-1.577
Fha21h	28.1 (7.0)	28.6 (8.9)	-0.338
Bswan2	22.3 (5.5)	21.7 (5.4)	0.550
Alo48	22.5 (5.4)	20.1 (8.1)	1.743
Alo49	19.8 (8.4)	22.7 (9.8)	-1.589
Aseol	20.1 (6.9)	21.5 (10.3)	-0.810

Seedling diameter at planting (mm)

Plantation	Fenced seedlings	Unfenced seedlings	t
Nalans	2.78 (0.82)	2.55 (0.70)	1.508
Amark	4.49 (1.32)	5.09 (1.08)	-0.829
Fha21h	2.93 (0.94)	2.82 (0.92)	0.614
Bswan2	2.93 (0.79)	2.93 (0.83)	0.025
Alo48	3.98 (1.41)	3.53 (1.21)	1.713
Alo49	3.06 (1.60)	3.36 (1.46)	-0.979
Aseol	3.42 (1.26)	3.75 (1.46)	-1.093

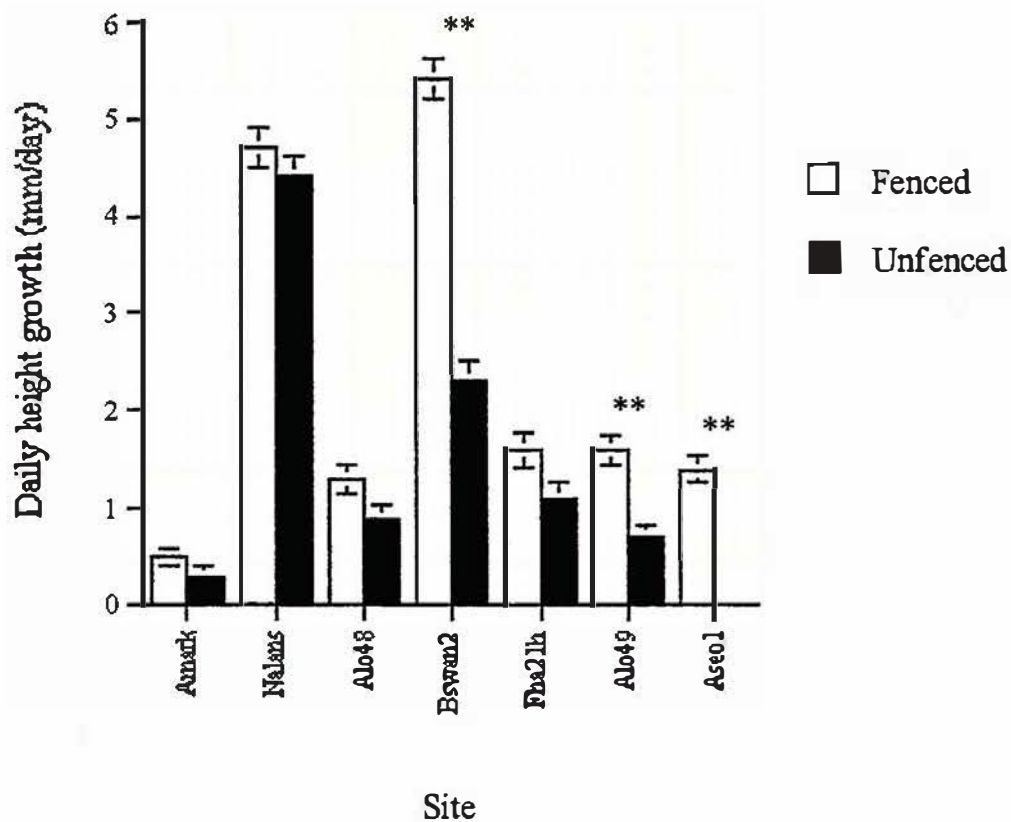


Figure 4.5 : Mean (\pm se) daily height growth (mm/day) for fenced and unfenced seedlings at each plantation, calculated for the 0-6 month period. The data have been ranked from left to right in order of increasing mean browse score measured from unfenced seedlings at six months after planting. Highly significant differences ($p < 0.01$) have been marked with a double asterisk, otherwise differences are not significant ($p > 0.05$).

Table 4.2 : The results of t-tests comparing mean daily height/diameter growth (mm/day) values, calculated for the 0-6 months after planting period, for fenced and unfenced seedlings. Standard deviations (sd) are given in parenthesis. The sample size (n) and values for the t-test statistic (t) are also given. Non-significant differences are marked with 'ns', significant differences ($p < 0.05$) with a single asterisk and highly significant differences ($p < 0.01$) with a double asterisk.

Daily height growth (mm/day)								
Plantation	Mean	Fenced sd	n	Mean	Unfenced sd	n	t	
Nalans	4.7	(1.5)	49	4.4	(1.4)	50	1.029	ns
Amark	0.5	(0.6)	38	0.4	(0.8)	43	0.630	ns
Fha21h	1.6	(1.2)	43	1.1	(1.2)	43	1.932	ns
Bswan2	5.4	(1.5)	50	2.3	(1.4)	48	10.566	**
Alo48	1.3	(0.9)	40	0.9	(0.8)	38	2.071	ns
Alo49	1.6	(0.9)	42	0.7	(1.0)	48	2.974	**
Aseol	1.4		45	0.0		39	8.625	**

Daily diameter growth (mm/day)								
Plantation	Mean	Fenced sd	n	Mean	Unfenced sd	n	t	
Nalans	0.12	(0.04)	49	0.12	(0.04)	50	0.381	ns
Amark	0.01	(0.01)	38	0.01	(0.01)	43	0.000	ns
Fha21h	0.05	(0.03)	43	0.04	(0.03)	43	1.546	ns
Bswan2	0.11	(0.04)	50	0.08	(0.04)	48	4.483	**
Alo48	0.02	(0.02)	40	0.02	(0.02)	38	0.172	ns
Alo49	0.02	(0.01)	42	0.01	(0.01)	48	4.732	**
Aseol	0.02	(0.02)	45	0.01	(0.01)	39	2.829	**

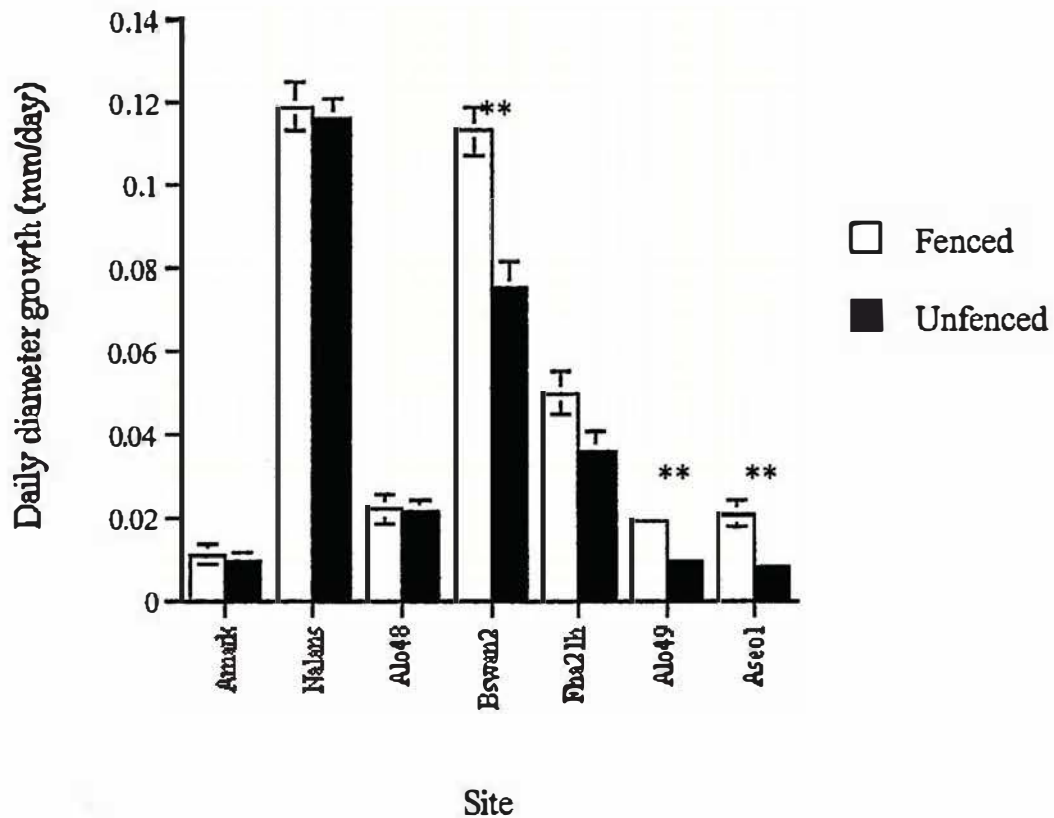


Figure 4.6 : Mean (\pm se) daily diameter growth (mm/day) for fenced and unfenced seedlings at each plantation, calculated for the 0-6 month period. The data have been ranked from left to right in order of increasing mean browse score recorded at six months after planting. Highly significant differences ($p < 0.01$) have been marked with a double asterisk, otherwise differences are not significant ($p > 0.05$).

diameter by between 0.01 and 0.12 mm per day. Once again, the difference was significant (Table 4.2) at three plantations.

Data for the mean daily growth in seedling height and diameter, calculated for the 0-12 month period, are shown in Figures 4.7 and 4.8 respectively. Mean daily height growth varied between plantations. For the fenced seedlings the range was from 0.3 to 3.8 mm/day. Height growth of unfenced seedlings ranged from -0.1 to 3.2 mm/day. The trend was for fenced seedlings to increase in height at a faster rate than unfenced seedlings. This difference was significant (Table 4.3) at five plantations. There was no significant difference at the two plantations where the lowest recorded mean browse values (at 12 months) were recorded.

Similar trends are evident in the daily diameter growth data (Figure 4.8) for the 0-12 month period. Once again, there was wide variation in growth between plantations. The range for fenced seedlings was from 0.01 to 0.10 mm/day. The range for the unfenced seedlings was from 0.00 to 0.10 mm/day. Fenced seedlings always increased their diameter as fast, or faster, than unfenced seedlings. At four plantations this difference was statistically significant (Table 4.3).

4.3.3 Describing the relationship between damage and growth

In Figure 4.9 the reduction in daily height growth, calculated from the 0-12 month data for fenced and unfenced seedlings (see Section 4.2.4), is regressed against the mean browse score measured at 12 months after planting. The regression is highly significant ($r^2=0.907$, $df=6$, $p<0.01$) and can be described by the following equation :

$$y = 20.73x + 7.76$$

Equation 4.1

where 'y' is the percentage reduction in daily height growth and 'x' is the mean browse score. The regression indicates that as browse score increases seedling height is reduced.

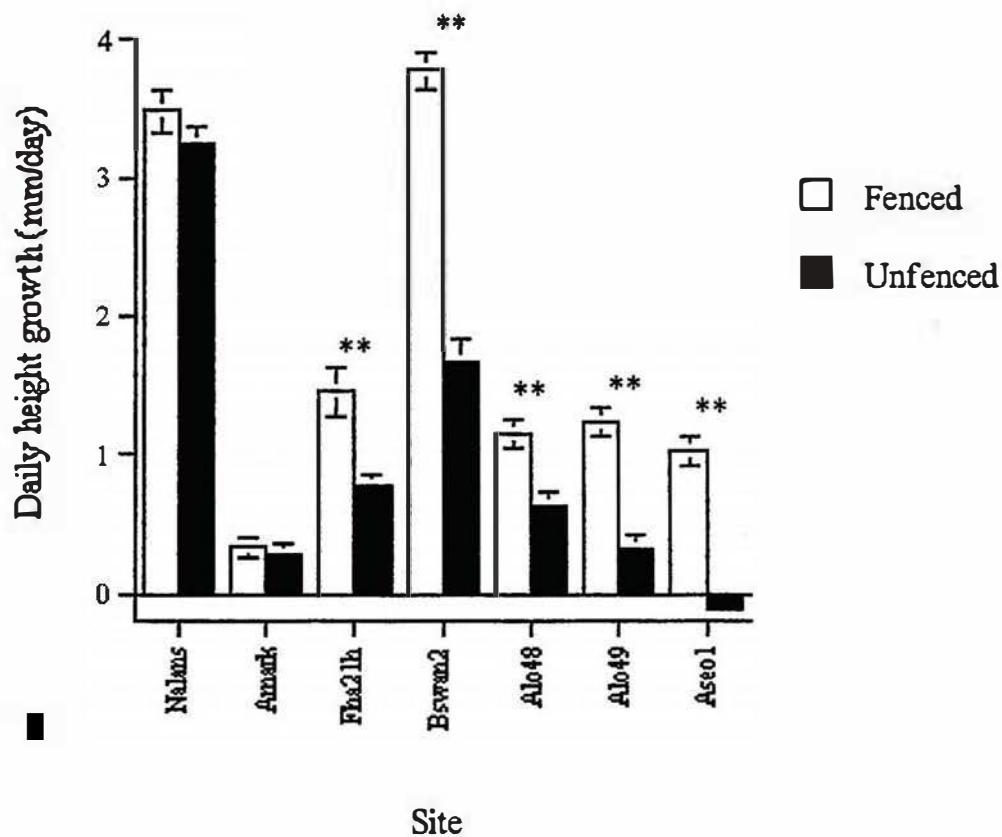


Figure 4.7 : Mean (\pm se) daily height growth (mm/day) for fenced and unfenced seedlings at each plantation, calculated for the 0-12 month period. The data have been ranked from left to right in order of increasing mean browse score recorded at 12 months after planting. Highly significant differences ($p < 0.01$) have been marked with a double asterisk, otherwise differences are not significant ($p > 0.05$).

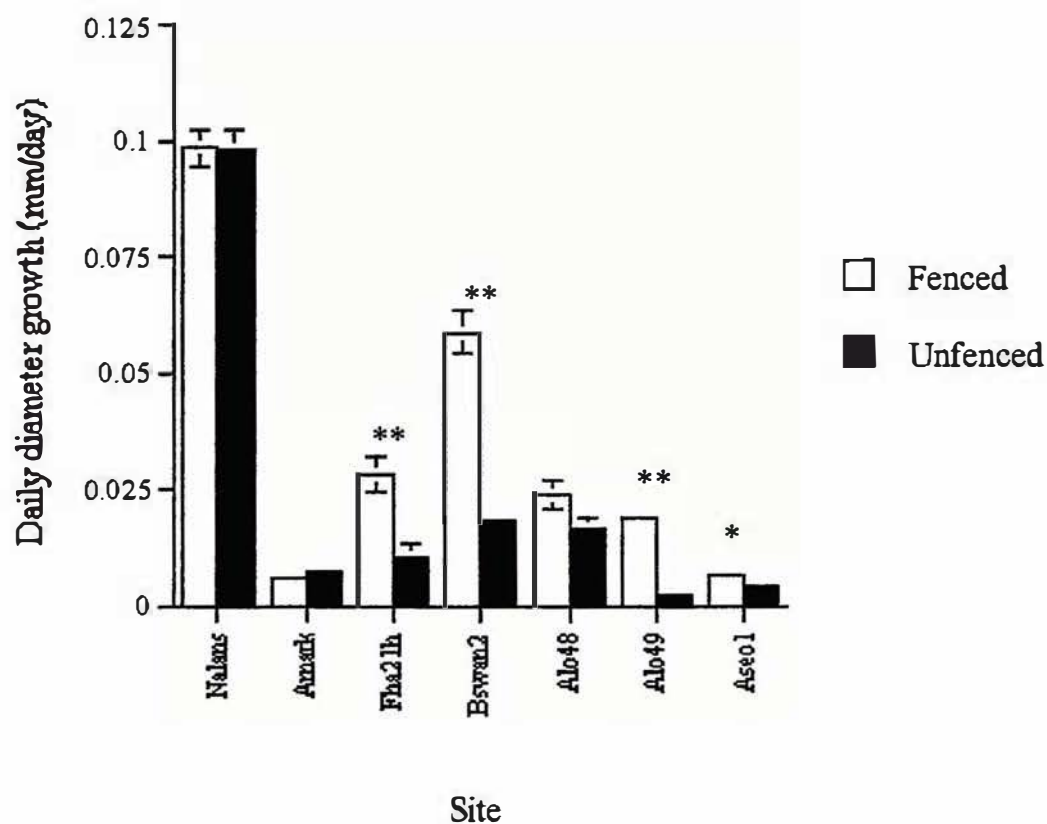


Figure 4.8 : Mean (\pm se) daily diameter growth (mm/day) for fenced and unfenced seedlings at each plantation, calculated for the 0-12 month period. The data were ranked from left to right in order of increasing mean browse score recorded at 12 months after planting. Highly significant differences ($p < 0.01$) have been marked with a double asterisk, and significant differences ($p < 0.05$) with a single asterisk. Non-significant differences have not been marked.

Table 4.3 : The results of t-tests comparing mean daily height and diameter growth (mm/day), calculated for the 0-12 month period, for fenced and unfenced seedlings. Standard deviations (sd) are given in parenthesis. The sample size (n) and values for the t-test statistic (t) are also given. Non-significant differences ($p>0.05$) are marked with 'ns', significant differences ($p<0.05$) with a single asterisk and highly significant ($p<0.01$) differences with a double asterisk.

Daily height growth (mm/day)

Plantation	Fenced			Unfenced			t
	Mean	sd	n	Mean	sd	n	
Nalans	3.5	(1.1)	49	3.2	(1.0)	50	1.212 ns
Amark	0.3	(0.4)	38	0.3	(0.5)	42	0.000 ns
Fha21h	1.5	(1.0)	36	0.8	(0.7)	46	3.621 **
Bswan2	3.8	(1.0)	50	1.7	(1.2)	48	9.573 **
Alo48	1.2	(0.7)	40	0.6	(0.6)	38	3.582 **
Alo49	1.2	(0.7)	43	0.3	(0.7)	47	6.278 **
Aseol	1.0	(0.7)	45	-0.1	(0.4)	36	8.976 **

Daily diameter growth (mm/day)

Plantation	Fenced			Unfenced			t
	Mean	sd	n	Mean	sd	n	
Nalans	0.10	(0.03)	49	0.10	(0.03)	50	0.094 ns
Amark	0.01	(0.01)	38	0.01	(0.01)	42	-0.534 ns
Fha21h	0.05	(0.02)	36	0.03	(0.02)	46	4.494 **
Bswan2	0.09	(0.03)	50	0.06	(0.03)	48	4.949 **
Alo48	0.02	(0.02)	40	0.02	(0.02)	38	1.947 ns
Alo49	0.02	(0.01)	43	0.00	(0.00)	47	9.026 **
Aseol	0.01		45	0.00		36	2.305 *

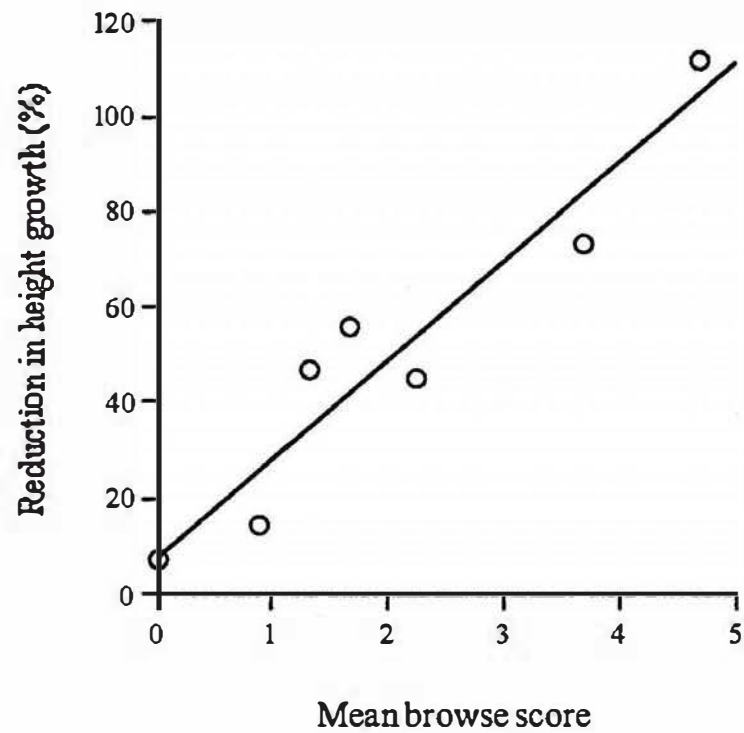


Figure 4.9 : Linear regression relating the percentage reduction in daily height growth (0-12 months) for unfenced seedlings to the mean browse score recorded at 12 months after planting for each plantation.

A similar regression carried out with the daily diameter growth data is shown in Figure 4.10. The general trend is for seedling daily diameter growth to be reduced as browse score increases ($r^2=0.758$, $df=6$, $p>0.05$).

4.3.4 Leading shoots

The mean number of leading shoots per seedling at 12 months is shown for both fenced and unfenced seedlings in Table 4.4. The unfenced seedlings tended to have more leading shoots than the fenced seedlings. Exceptions to this were the Nalans plantation, where shoot numbers were approximately equal, and the Alo48 plantation where the fenced seedlings tended to have more shoots. T-tests between the mean values for fenced and unfenced seedlings indicated that unfenced seedlings had significantly more leading shoots at Amark, Fha21h, Bswan2, and Alo49.

Table 4.4 : The results of one tailed t-tests comparing the mean number of leading shoots at 12 months after planting for fenced and unfenced seedlings. Standard deviations (sd) are given in parenthesis. Sample sizes (n) and values for the t-test statistic (t) are shown. Non-significant differences ($p>0.05$) are marked with 'ns', significant differences ($p<0.05$) with a single asterisk and highly significant differences ($p<0.01$) with a double asterisk. The data is ranked from top to bottom according to increasing mean browse score recorded at 12 months after planting.

Plantation	Fenced			Unfenced			t	
	Mean	sd	n	Mean	sd	n		
Nalans	1.1	(0.4)	49	1.1	(0.4)	50	0.000	ns
Amark	1.2	(0.4)	38	1.4	(0.5)	42	-2.185	*
Fha21h	1.1	(0.4)	36	1.3	(0.5)	43	-1.999	*
Bswan2	1.2	(0.5)	50	1.7	(1.0)	48	-3.150	**
Alo48	1.4	(0.7)	40	1.2	(0.5)	38	1.445	ns
Alo49	1.1	(0.5)	43	1.4	(0.6)	47	-2.542	**
Aseol	1.3		45	1.4		36	-0.643	ns

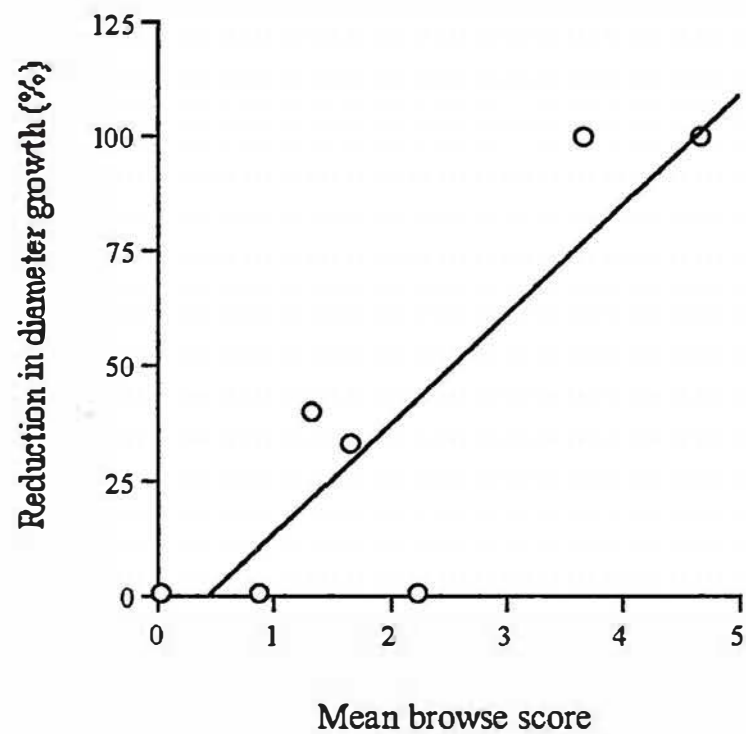


Figure 4.10 : Linear regression relating the percentage reduction in daily diameter growth (0-12 months) for unfenced seedlings to the mean browse score recorded at 12 months after planting for each plantation.

4.3.5 Seedling survival

The percentage of sampled seedlings at each plantation which were still surviving to six months is shown in Figure 4.11. Survival of fenced seedlings to six months ranged from 84 to 100%. The percentage of unfenced seedlings surviving to six months ranged from 76 to 100%. Perusal of Figure 4.11 suggests that there is no real difference in the survival of fenced and unfenced seedlings. This observation is confirmed by the results of a paired t-test for differences between the two groups to six months ($t=0.372$, $df=6$, $p>0.05$).

Data for the percentage of seedlings surviving to 12 months are shown in Figure 4.12. Survival to 12 months ranged from 78 to 100% for the fenced seedlings and from 72 to 100% for the unfenced seedlings. Once again, there was no significant difference between the survival of the fenced and unfenced seedlings ($t=0.644$, $df=6$, $p>0.05$). Nor did there appear to be any overall difference in survival values obtained from the 6 month sample and the 12 month sample (fenced : $t=1.542$, $df=6$, $p>0.05$; unfenced : $t=1.256$, $df=6$, $p>0.05$). The one exception was the severely damaged Aseol plantation, where the survival of the unfenced seedlings dropped by 18% from the six month sampling period to the 12 month sampling period.

Spearman rank correlation indicates that there was no relationship between seedling survival to six months and mean browse score at six months (unfenced seedlings : $\rho=0.116$, $n=7$, $p>0.05$). Nor could a relationship be demonstrated for survival to 12 months and browse score measured at 12 months ($\rho=0.571$, $n=7$, $p=0.162$). However, a significant correlation is observed between fenced seedling daily height growth and fenced seedling survival to 12 months ($\rho=0.811$, $n=7$, $p<0.05$). There is also a significant correlation between daily height growth and survival to 12 months for the unfenced seedlings ($\rho=0.821$, $n=7$, $p<0.05$). Both correlations indicate that the faster seedlings grew at a plantation the greater their survival tended to be, irrespective of whether they were exposed to browsing damage or not.

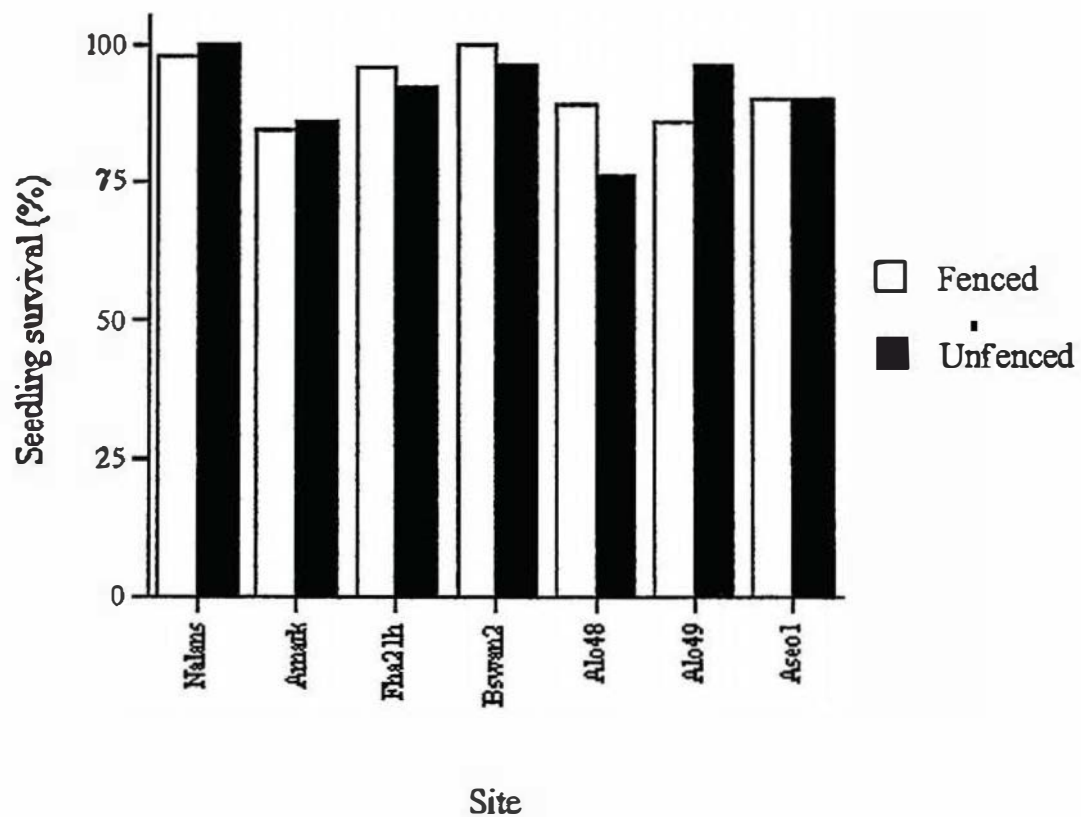


Figure 4.11 : Percentage of seedlings surviving to six months for both fenced and unfenced plots. The data are ranked from left to right according to increasing mean browse score recorded for unfenced seedlings at 12 months after planting.

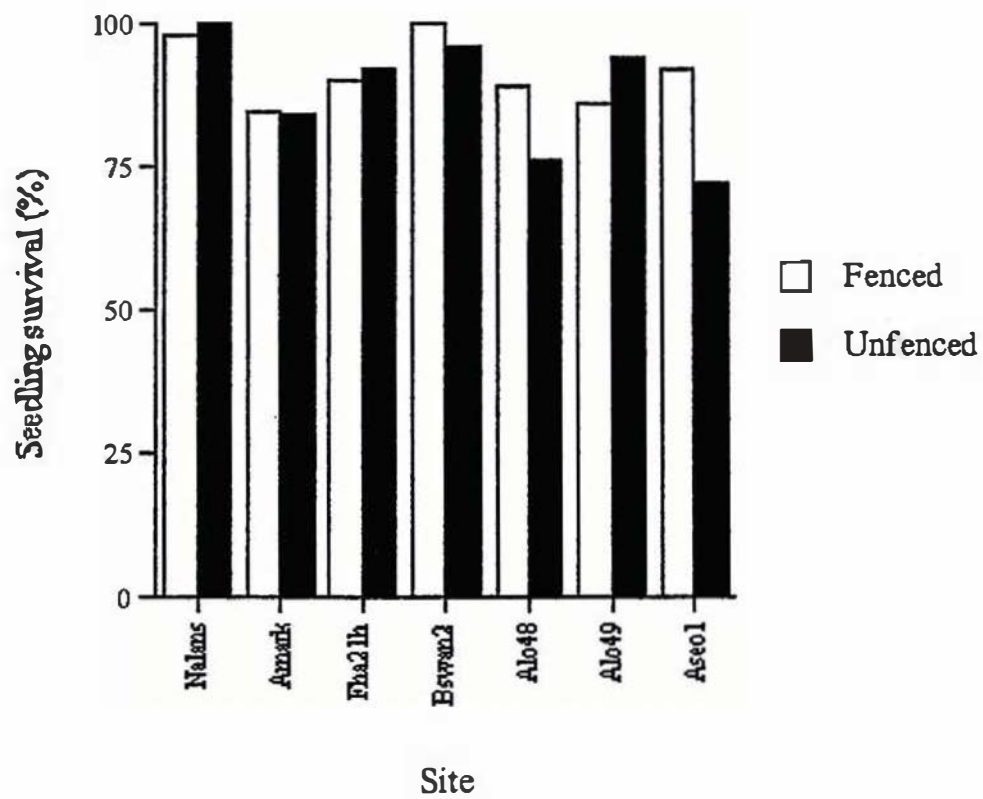


Figure 4.12 : Percentage of seedlings surviving to 12 months for both fenced and unfenced plots. The data are ranked from left to right according to increasing mean browse score recorded for unfenced seedlings at 12 months after planting.

4.3.6 Testing the assumptions of Chapter 3

A series of t-tests were carried out to test the assumption made in Chapter 3, that seedlings which were recorded with no damage (browsing score of zero) had daily growth rates equivalent to seedlings known to have never been browsed. The results are shown in Table 4.5. For daily height growth there was a significant difference between the fenced and unfenced, 'undamaged' seedlings at two plantations (Alo49 and Aseo1). However, due to low sample size ($n=5$), the t-test results for Aseo1 are considered unreliable. There was a significant difference in daily diameter growth at only one plantation. It can be seen that significant differences were prevalent for plantations with high overall levels of browsing damage (high mean browse score).

In Figure 4.13 the regression equation relating mean browse score to reductions in seedling daily height growth derived in Section 3.3.4 (Equation 3.7) is compared to that derived in this chapter (Equation 4.1). It appears that Equation 3.7 slightly underestimates the effect of browsing damage. However, comparison of the slope of the two lines indicates that they are not significantly different ($t=0.393$, $df=133$, $p<0.05$).

Table 4.5 : Results of t-tests comparing the mean daily height and diameter growth (mm/day) of fenced seedlings (calculated for the 0-12 month data) to growth of unfenced seedlings which were not observed to be damaged. Standard deviations (sd) are shown in parenthesis. Non-significant differences ($p>0.05$) are marked with 'ns' and highly significant differences ($p<0.01$) with a double asterisk. The data are ranked from top to bottom, according to increasing mean browse score (unfenced seedlings) recorded at 12 months after planting.

Daily height growth (mm/day)

Plantation	Fenced			Unfenced			t	
	Mean	sd	n	Mean	sd	n		
Nalans	3.48	(1.09)	49	3.68	(1.39)	35	-0.739	ns
Amark	0.34	(0.43)	35	0.25	(0.67)	33	0.663	ns
Fha21h	1.45	(0.99)	36	1.11	(1.33)	22	1.112	ns
Bswan2	3.77	(1.02)	50	3.38	(1.02)	20	1.445	ns
Alo48	1.15	(0.66)	40	1.12	(0.78)	17	0.149	ns
Alo49	1.23	(0.70)	43	0.64	(0.06)	13	3.016	**
Aseol	1.02	(0.69)	45	-0.28	(0.67)	5	4.006	**

Daily diameter growth (mm/day)

Plantation	Fenced			Unfenced			t	
	Mean	sd	n	Mean	sd	n		
Nalans	0.098	(0.030)	49	0.102	(0.037)	35	-0.509	ns
Amark	0.006	(0.010)	35	0.008	(0.010)	33	-0.649	ns
Fha21h	0.028	(0.022)	36	0.028	(0.033)	22	0.055	ns
Bswan2	0.059	(0.032)	50	0.053	(0.045)	20	0.613	ns
Alo48	0.024	(0.018)	40	0.020	(0.020)	17	0.713	ns
Alo49	0.019	(0.016)	43	0.006	(0.010)	13	2.797	**
Aseol	0.007	(0.008)	45	0.003	(0.004)	5	1.138	ns

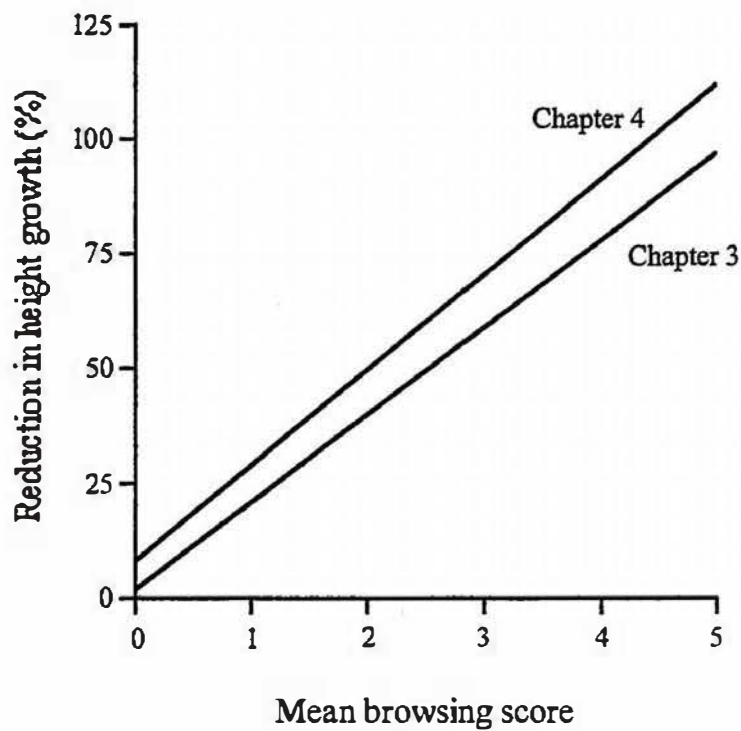


Figure 4.13 : Comparison of two regression lines, one derived in Chapter 3 (Equation 3.7) and the other in Chapter 4 (Equation 4.1), relating mean browse score measured at 12 months to the percentage reduction in mean daily height growth (0-12 months).

4.4 DISCUSSION

4.4.1 Effect of browsing damage on seedling growth

Browsing damage reduced the rate at which unfenced seedlings increased their height. At six months, significant differences between the daily height growth of fenced and unfenced seedlings were evident at three of seven plantations. At 12 months there were significant differences at five plantations. The trends observed for daily diameter growth were similar. This result is consistent with that reported in Chapter 3, other studies of eucalypt/browser interactions (e.g. Gilbert 1961; Cremer 1969; Horne 1975; Montague 1996), and a wide range of studies of the effects of defoliation on plant growth (e.g. Bergstrom and Danell 1987, 1995; Nielsen and Pataczek 1991; Strauss 1991; Elliott *et al.* 1993; Wilkinson and Nielsen 1995).

The magnitude of the differences in growth between fenced and unfenced seedlings depended on the severity of browsing damage. While any level of measured damage appeared to reduce the daily growth values of unfenced seedlings, significant differences were restricted to those plantations where browsing damage was most severe. A highly significant positive linear relationship was demonstrated between the percentage difference in height growth (fenced and unfenced) and a measure of browsing severity (mean browse score). The greater the browsing severity the more height growth was reduced for the unfenced seedlings, with height growth reduction exceeding 100% as mean browse score approached 5 (at 12 months).

No other Tasmanian study has so explicitly and quantitatively linked natural browsing damage levels to losses in growth of *E. nitens* seedlings. However, links between *E. nitens* growth and severity of simulated browsing have been recently observed. Nielsen and Pataczek (1991) found that the height growth of *E. nitens* and *E. regnans* decreased with increasing levels of biomass removal. Height growth of seedlings which had only their tips removed were not greatly

reduced, while seedlings which had been clipped to within 25 mm of the ground achieved heights of less than 31% of controls. A similar study carried out by Wilkinson and Nielsen (1995) revealed similar trends.

4.4.2 Inherent differences in seedling growth

The rate at which the fenced, unbrowsed seedlings increased in height varied widely among plantations. It appears then, that seedling growth rates are inherently quite variable between sites and that browsing damage is not the sole agent responsible for regulating seedling growth. This result is not surprising. Differential growth rates of eucalypt seedlings between sites have been demonstrated in other studies (e.g. Turnbull *et al.* 1993). Such variation between sites may be a result of differing genetic stock (Pederick 1979; Griffin *et al.* 1982), silvicultural practices (Hopmans *et al.* 1993) and variation in environmental conditions such as rainfall (Bachelard 1986; Turnbull, *et al.* 1993, Honeysett *et al.* 1996), temperature (Sa-ardavut 1984; Turnbull *et al.* 1993), soil type (Bachelard 1986), soil nutrient levels (Stone 1993), fungal pathogens (Burdon and Chilvers 1974) and levels of insect attack (Strauss 1991; Elliot *et al.* 1993).

4.4.3 The effect of browsing damage on seedling form

Anecdotal evidence (e.g. Coleman *et al.* 1997) would suggest that browsing damage to eucalypt seedlings promotes the development of multiple leading shoots. Ultimately this is thought to reduce the value of harvested logs by increasing the branching and knotting of tree trunks. There is certainly a wide range of studies which show that trees and shrubs undergo morphological changes after natural or simulated mammal browsing damage (e.g. Krefting *et al.* 1966; Jacobs 1969; Danell *et al.* 1985), however, evidence to support the hypothesis that browsing promotes multiple shoot development in plantation species is scarce. In one of the few studies to demonstrate a positive link, Montague (1996) found that severe browsing and/or repeated browsing attack produced double leaders in 20%

of *E. regnans* seedlings. In contrast, Strauss (1988) observed that the numbers of shoots produced by smooth sumac (*Rhus glabra*) was apparently unaffected by deer browsing. Furthermore, in two simulated browsing studies on birch trees (*Betula spp.*), Bergstrom and Danell (1987, 1995) found that biomass removal either did not promote shoot development or actually reduced the number of shoots developed.

In this study browsing damage appeared to promote the development of multiple leaders in *E. nitens* at four plantations. All four plantations had intermediate levels of browsing damage. An explanation for why seedlings should only produce more shoots at plantations with intermediate damage is not immediately apparent. Perhaps seedlings were not recorded with multiple shoots at plantations with low browsing damage either because a critical level of severity is required before shoot development is promoted, or because apical buds were not removed. This would be consistent with the results of Montague (1996) who found that double leader formation was promoted only when browsing damage was severe and/or repeated. Conversely, seedlings may not be recorded with multiple shoots at plantations where browsing damage is exceptionally high (e.g. Aseol) because so much biomass is removed that seedlings are typically reduced to a single stem cropped close to the ground.

4.4.4 The effect of browsing damage on seedling survival

Seedling survival to 12 months did not appear to be strongly related to browsing damage. No real difference was observed between the survival of the fenced and unfenced seedlings. For both the fenced and unfenced seedlings, most deaths occurred within the first six months after planting, suggesting that much of the observed mortality was associated with the stress caused by removal from a nursery environment.

The findings presented in this chapter are consistent with those presented in Chapter 3 and a number of other studies of the effects of damage on tree survival.

Montague (1996) observed that browsing damage had only a slight impact on the overall survival of *E. regnans* seedlings in Victorian plantations. Horne (1975) found that even severely browsed eucalypt seedlings did not suffer significantly greater mortality than unbrowsed seedlings. These findings are apparently contradictory to recent simulated browsing studies carried out with *E. nitens*. Nielsen and Pataczek (1991) found that seedling survival to age three years was significantly reduced when biomass removal was severe. In a similar experiment Wilkinson and Nielsen (1995) also observed that heavy levels of defoliation reduced the survival of seedlings to age two years.

Perhaps the divergence of results obtained from these studies highlights the artificial nature of the damage regimes implemented in simulated browsing trials. Nielsen and Pataczek (1991) and Wilkinson and Nielsen (1995) both observed that survival was significantly affected only when they removed all foliage down to 25 mm above ground level. This would be the equivalent of a mean browsing score of 5 in this study. Perusal of the range of browsing severities reported in this chapter and Chapter 3 indicate that, at the majority of plantations, damage levels observed in the field tend to be substantially less than this. This holds true even for those plantations where no effort is made to control browsing damage.

An alternative explanation for why the results of this study are different from the simulated browsing studies may be that the effect of damage to seedling survival is not immediate. Nielsen and Pataczek (1991) and Wilkinson and Nielsen (1995) carried their studies out over 3 and 7 years respectively. This study, on the other hand, was restricted to the first year after planting. It is possible that if this study were to continue, the survival of the fenced seedlings may eventually exceed that for unfenced seedlings. The survival data from the Aseol plantation partially support this hypothesis. This plantation had extremely high levels of browsing damage and there was a drop in survival from 90% at six months to 72% at 12 months suggesting that persistent, severe browsing was starting to reduce seedling survival. It is certainly difficult to imagine that the seedlings at this site, which

are actually decreasing in height, would not eventually suffer from increased mortality.

4.4.5 Possible confounding effects

It is possible that the differences observed between fenced and unfenced seedlings, and the lack of difference in survival, may actually underestimate the effects of browsing damage because of confounding 'cage' effects (Hulberg and Oliver 1980; Walde and Davies 1984). The netting used in this study is unlikely to have affected seedlings directly, since the thin gauge and wide spacing did not appear to alter lighting levels, water levels or microclimatic conditions. However, in plantations browsing animals feed not only on seedlings but on surrounding vegetation as well (Statham 1983; Sprent 1997). Fencing may therefore have indirectly improved the competitiveness of weed species growing within the enclosure and/or altered weed species composition. In several plots there appeared to be distinct differences in the amount of weed growth within the plot compared to that outside it. Competition from weeds has been shown to reduce the growth and survival of woody species (Tesch and Hobbs 1989; Bush and van Auken 1990; Ross *et al.* 1990). If similar mechanisms were in operation during this study, weeds may have reduced both growth and survival of fenced seedlings.

4.4.6 Testing the assumptions made in Chapter 3

It appears that the assumption made in Chapter 3, that seedlings which were not recorded with browsing damage would have daily height growth rates equivalent to seedlings which were never damaged, was valid for five out of the seven plantations. At the two remaining plantations (Alo49 and Aseo1) there were significant differences between the daily height growth for fenced seedlings and unfenced, apparently unbrowsed, seedlings. It is considered, however, that the data from the Aseo1 plantation should be disregarded as there was a very low sample size ($n=5$) for unfenced, 'unbrowsed' seedlings at this site. For diameter growth there was a significant difference at the Alo49 plantation only.

In effect, the reductions in daily height growth caused by browsing damage reported in Chapter 3 may be an underestimate of the 'true' effect at some plantations. The extent of this underestimation can be observed by comparing the regression line obtained in Chapter 3 (Equation 3.7) to that obtained in Chapter 4 (Equation 4.1). However, while a difference in elevation exists, no significant difference was observed for slope. Thus the nature of the relationship between browsing damage and height growth was essentially the same for both studies.

4.4.7 Management implications

The findings reported within this chapter have several important implications for forestry management. These will be addressed in detail in the following chapter.

CHAPTER 5

MANAGEMENT IMPLICATIONS OF CHAPTERS 3 AND 4

5.1 INTRODUCTION

The studies outlined in Chapters 3 and 4 have shown that the browsing problem is a very real one. Damage to seedlings was evident at all of the 35 plantations which were surveyed, varying in severity from very low to severe. The main measured effect of browsing damage was the reduction in the growth rates of seedlings. By quantifying the relationships between damage and growth, a framework now exists which can aid managers in determining what constitutes an acceptable level of damage, and what levels of damage should be aimed for if seedling growth is to be maximised. The aim of this chapter is to more explicitly develop this framework through a synthesis of the management implications of Chapters 3 and 4. The chapter is divided into discrete sections and, where appropriate, all methods, results and conclusions are discussed within each section.

5.2 ASSESSING BROWSING DAMAGE

In the previous chapters two measures were used to assess the severity of browsing damage at forestry plantations. These were the 'percentage of seedlings damaged' (PSD) and the 'mean browse score' (MBS). The methods employed in obtaining these measures were outlined in detail in Chapters 2 and 3. For the damage data at both six and 12 months these two measures of damage are curvilinearly related to one another. These relationships are described by Equations 3.1 and 3.2.

The high correlation coefficients associated with these regressions (six months : $r^2=0.901$; 12 months : $r^2=0.878$), combined with the apparent temporal consistency of the relationship (see Section 3.3.2 for a comparison between curves generated for the six and 12 month data) suggests that obtaining values for both damage severity measures is unnecessary because trends observed in one measure will be reflected in the other. Deciding which measure to use is dictated to some extent by the aim of the observer. Logistically, the most economic measure to use is PSD. Measuring PSD only requires an observer to examine a seedling for signs of damage and then decide if it has been browsed or not. On the other hand, obtaining a browse score for a seedling requires the observer to first examine the seedling for signs of damage, decide if the seedling has been damaged, and then assign the seedling to a browse score category, based on an estimate of the percentage of biomass which has been removed. As a result, obtaining a PSD measure for a plantation takes less than one third of the time required to obtain a MBS measure. Additionally, because assigning a browse score to a seedling requires the observer to actually estimate how much biomass is removed, the potential influence of observer bias and error is likely to be lower for PSD (Wint 1983; Landsberg 1989; Armstrong and MacDonald 1992).

The PSD measure does, however, suffer from reduced fidelity when browsing damage is severe. Consider the curve generated from the 12 month data (Figure 3.4). It can be seen that when 100% of seedlings exhibit signs of damage mean browse scores can range anywhere from 4.1 to 5.0. In effect, the PSD measure does not differentiate damage severities when they are extremely high (mean browse score > 4.1). Because of the coarse nature of PSD, the problem of reduced fidelity will also be evident whenever inference is to be drawn from data for individual seedlings (e.g. Section 3.3.4). Assume, for example, that two seedlings have been damaged by herbivory. One seedling has had only one leaf removed by a

herbivore and the other has had over 90% of its leaves removed. Clearly both seedlings have been damaged to very different severities, but using the PSD measure alone will only provide the information that they are both damaged.

5.3 ASSESSING SEEDLING GROWTH

In the past it has been common practice for foresters to estimate the volume of timber in even aged forest stands from tree height and diameter measurements (e.g. Curtis 1967). However, it is very difficult and time consuming to obtain measurements of height for all the trees within a particular stand. As a consequence tree height is often estimated from tree diameter using a mathematical function relating diameter to height (e.g. West 1979). Describing this function, or relationship, involves comparing a sample of tree diameter measures to tree height measures, usually using a linear regression technique (Curtis 1967; West 1979). Such relationships have been described for mature commercial pine (Curtis 1967; Donald 1976) and eucalypt trees (West 1979).

In Chapter 3, similar relationships were observed between *E. nitens* seedling diameter and height. These very strong, linear relationships were described by Equations 3.3, 3.4, 3.5, and 3.6.

The fact that there is such a strong relationship between *E. nitens* seedling diameter and height suggests that only one need be measured because any trends observed in one measure will be reflected in the other. However, unlike the techniques used by foresters for estimating stand volume I suggest that, excepting where sample size is small or specific questions regarding diameter are to be answered, measurements of seedling height be made preferentially to diameter because :

- i) assuming that vernier callipers are used for diameter measurements it will take at least twice as long to measure diameter as height.
- ii) where repeated measures (e.g. over time) are to be taken from seedlings, height measures may prove more reliable. The form of seedling stems is not always regular and diameter measures made on the same seedling can vary depending on the positioning of callipers. One way to reduce this problem is to average a number of diameter measures made on the same seedling, however, this greatly increases measuring time.
- iii) where specific questions are to be addressed relating to the costs of browsing damage, measured in growth terms, seedling height is probably the superior variable to measure because daily rate of change in seedling height appears to be the most responsive to browsing damage.

5.4 ASSESSING THE COSTS OF BROWSING DAMAGE AND ASSIGNING TARGET DAMAGE LEVELS

The most cost effective control of pest damage can only be achieved where managers are able to specifically identify the damage level at which the benefits of control outweigh the costs of its implementation (Braysher 1993). This is only possible where there is an explicit knowledge of the relationship between damage severity and the monetary costs of this damage. Montague (1996) assessed the costs of browsing damage caused by the swamp wallaby to Victorian *E. globulus*, *E. nitens*, *E. regnans* and *P. radiata* plantations. He based his assessment on estimates of the mortality of seedlings and subsequent loss of final timber yield at harvesting. However, the results of the studies outlined in Chapters 3 and 4 suggest that the mortality of eucalypt seedlings in Tasmanian plantations, at least during the first 12 months of growth, is largely unrelated to browsing damage. Hence performing a cost analysis similar to that of Montague (1996) is not particularly valid here.

Of greater value might be a consideration of costs in terms of losses in seedling growth. The results outlined in both Chapters 3 and 4 indicate that the rate at which *E. nitens* seedlings increase their height is strongly influenced by severity of browsing damage. The higher the observed damage score, the lower the rate of height increase. A mathematical function quantifying this relationship was described in both Chapter 3 (Equation 3.7) and Chapter 4 (Equation 4.1). The relationship described by Equation 4.1 (see also Figure 5.1) is considered the more accurate due to the stricter use of controls. Using this equation it is possible, for any given plantation, to estimate the mean percentage loss in *E. nitens* seedling height growth by measuring the MBS for that plantation. In Table 5.1 the MBS (12 months) values which were obtained for each of the 32 surveyed *E. nitens* plantations have been substituted into Equation 4.1 to estimate the percentage reduction in seedling height growth caused by browsing damage.

■

Although even small levels of measured damage can result in reduced seedling height growth, the comparison of fenced and unfenced seedling data outlined in Section 4.3.2 indicates that this reduction is statistically significant only when MBS (12 months) exceeds a certain critical level. For the 0-12 month data, the lowest MBS at which a significant difference was recorded between growth of fenced and unfenced seedlings was 1.33. The highest MBS at which no significant difference occurred was 0.88. Thus, the critical MBS at which overall seedling height growth at a plantation begins to be significantly affected by browsing damage appears to occur somewhere between 0.88 and 1.33 (Figure 5.1).

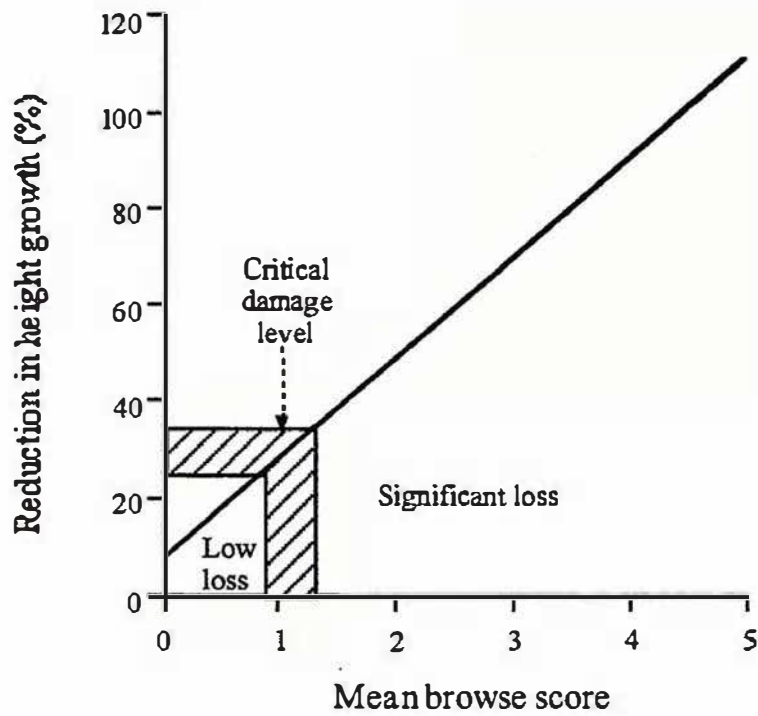


Figure 5.1 : The relationship between mean browse score and percentage reduction in daily height growth. The critical mean browse score, where seedling height growth begins to be significantly affected, appears to occur between 0.88 and 1.32 (cross hatched area).

Table 5.1 : Estimates of the reduction in daily rate of seedling height increase (0-12 months) caused by browsing damage for each of the 32 surveyed *E. nitens* plantations. Those plantations where damage exceeds only the lower, conservative estimate for mean browse score causing significant reduction (0.88) have been marked with a single asterisk. Plantations where damage exceeds both the lower and upper (1.33) estimate of mean browse score required to significantly reduce seedling height growth are marked with a double asterisk. Plantations where no poisoning or other damage control was carried out have been marked with 'np'. Damage data are also presented for the three surveyed *E. globulus* plantations (Ndig, Nrit, Nspill).

Plantation	MBS (12 months)	Reduction (%)	Plantation	MBS (12 months)	Reduction (%)
Adu29	0.597	20.1	Bswan2	1.332	35.4 **
Alo48	1.837	45.8 **	Bwal	0.354	15.1
Alo49	3.892	88.4 **	Bwee	2.331	56.1 **
Amark	0.979	28.1 *	Bwfr	1.403	36.8 **
Aseol	4.688	104.9 **	Fha21h	1.537	39.6 **
Asx13	2.131	51.9 ** np	Fhp23g	0.281	13.6
Asx51	0.957	27.6 *	Fkd13b	0.291	13.8
Atn18	0.658	21.4	Fsolb	0.425	16.6
Atn20	2.492	59.4 **	Nalans	0.471	17.5
Awv55	0.251	13.0	Nbless	0.661	21.5
Bcam	0.967	27.8 *	Ndig	0.319	-
Bfr	0.242	12.8	Nped	0.287	13.7
Bkui	1.250	33.7 *	Npu	2.534	60.3
Brfl	2.359	56.7 ** np	Nrit	1.212	- np
Brf2	0.706	22.4 np	Nsp	0.324	-
Brin	2.583	61.3 ** np	Ncw	1.977	48.7 ** np
Bsch	0.364	15.3	Npex	2.012	49.5 **
Bswan	0.342	14.9			

Assuming that the goal of management is to achieve maximal seedling growth then, control programs should aim to restrict mean browse score to between 0.88 and 1.33. These critical values have been applied to the data presented in Table 5.1 to determine at which of the 32 surveyed *E. nitens* plantations seedling height growth was likely to have been significantly reduced by browsing damage. Using the lower, conservative value for mean browse score (0.88), it is estimated that 18 plantations (56%) could expect to have significantly reduced seedling growth rates. Using the upper value it is estimated that 14 plantations (44%) would have significantly reduced growth.

Ideally, reductions in seedling growth could also be linked to economic losses so that the potential benefits of control could be weighed against the costs. Such analyses are beyond the scope of this work. The relationships between early browsing damage and the very long term growth of trees has not been greatly studied. Nielsen and Pataczek (1991) and Wilkinson and Nielsen (1995) have studied the effects of early simulated browsing on the growth of *E. nitens* to 3 and 7 years respectively. In both studies it was found that severe browsing treatments, applied to seedlings during their first year of growth, resulted in reduced growth. This effect persisted through to the end of each study. However, to date no study has been carried out through to harvesting and it is possible that, over an extended time span, even quite severely damaged seedlings might recover from browsing damage and achieve growth equivalent to unbrowsed seedlings. In effect, the most economic target damage level could occur above a mean browse score of 1.33.

5.5 SEEDLING ESCAPE TIMES

So far, much of this discussion has focused on the degree to which browsing damage reduces the rate at which seedlings increase in height and diameter. The

analysis upon which this discussion has been based was specifically tailored to avoid problems associated with making comparisons between sites where seedling growth is inherently variable. However, this inherent variation in growth rates between sites is an important factor in itself, and also carries a range of implications for management.

Although the arboreal brushtail possum has the potential to browse eucalypts of any age (Barnett *et al.* 1977), ground dwelling species such as the Bennett's wallaby, Tasmanian pademelon and rabbit, are only able to reach foliage whilst seedlings are still small. The tallest of these herbivores is the Bennett's wallaby, with males averaging 0.8 m in height (Calaby 1983). Though no quantitative data exist documenting how far up a seedling these animals can reach to feed, personal observation would suggest that foliage taller than 1.5 m (approximately 2 body lengths) is well clear of attack. This estimate is consistent with that reported by Coleman *et al.* (1997). ■

Essentially then, ground dwelling herbivores have a well defined window of opportunity within which they are able to feed on seedlings. As a consequence, browsing pressure is likely to be most heavy between the period when a seedling is planted and when it attains a height greater than 1.5 m. This is also likely to be the time when eucalypts are least able to cope with browsing because small seedlings have only a small nutrient reserve with which to compensate for the damage (Cremer 1969).

If it is assumed that, as the weight of evidence would suggest (Gilbert 1961; Cremer 1969, 1972, 1973; Nielsen and Pataczek 1991; Wilkinson and Nielsen 1995), browsing damage has a negative impact on the growth of eucalypts, it stands to reason that the longer a seedling is exposed to browsing attack the greater the potential effect the browsing will have. In other words, it is likely that

seedlings subject to browsing attack for only a short period of time are likely to be less affected than those seedlings subject to attack for more lengthy periods. How long it takes for a seedling to reach a height where it is relatively inaccessible to browsers could then, have serious implications for its long term survival and growth.

This adds a further complicating dimension to a discussion of the management implications of browsing damage. To demonstrate this, consider the results presented in Chapter 4, which indicate that growth rates of seedlings free from browsing can vary enormously between sites. At Amark, for instance, the daily height growth (0-12 months) for fenced seedlings was a low 0.3 mm/day, while at Bswan2 the rate was 3.8 mm/day. The browsing damage to the unfenced seedlings at these two plantations was also quite different, with seedlings at Bswan 2 sustaining much more severe damage than those at Amark. However, provided seedlings continue to grow at the same rates, seedlings at the Bswan2 plantation will still escape from ground-dwelling herbivores much faster than seedlings at Amark. The basal or inherent rate of growth at this plantation is so high that seedlings can suffer a 56% reduction in growth from browsing and still escape relatively quickly.

Thus, when considering the control options for a given plantation it may be beneficial to consider not only the potential for browsing damage but also the potential for seedling growth. In low growth areas even moderately severe browsing may act to reduce already low seedling growth to such an extent that escape time is very high. Conversely, in areas where growth is rapid, seedlings might escape from browsing in a short amount of time, even when damage is severe.

5.6 THE EFFECTIVENESS OF CURRENT POISONING PRACTICES FOR CONTROLLING DAMAGE

For over 37 years 1080 poison has been used as a means of controlling browsing damage in Tasmania (Mollison 1960). In recent years it has become the principle browsing reduction technique employed by foresters. Clearly the justification for using such control action, which carries with it a range of environmental, social and economic costs, must be that its implementation brings about a significant reduction in damage. However, to date there have been no published studies which quantify how effective 1080 is in this role. The data collected in Chapter 3 consequently provides a rare opportunity to compare damage data collected at a randomly selected group of poisoned plantations with those collected at a randomly selected group of unpoisoned plantations (see Chapter 2).

The data for mean browse score and percentage of seedlings damaged were presented in Table 3.3. In Figures 5.2 and 5.3 these data are summarised for poisoned and unpoisoned *E. nitens* plantations. Figure 5.2 summarises the mean browse score data which was collected at six and 12 months. The results of one-tailed unpaired t-tests indicate that mean browse score for the 26 plantations poisoned with 1080 was significantly lower than that for the six unpoisoned *E. nitens* plantations at both six ($t=2.473$, $df=30$, $p<0.01$) and 12 months after planting ($t=1.759$, $df=30$, $p<0.05$).

Similar trends are evident for the percentage of seedlings damaged data, which is summarised in Figure 5.3. Mann-Whitney tests (Zar 1984) indicate that the mean percentage of seedlings damaged for the 26 poisoned plantations was significantly lower than that for the six unpoisoned *E. nitens* plantations at both six ($U_{(6,26)}=132$, $p<0.01$) and 12 months after planting ($U_{(6,26)}=131$, $p<0.01$).

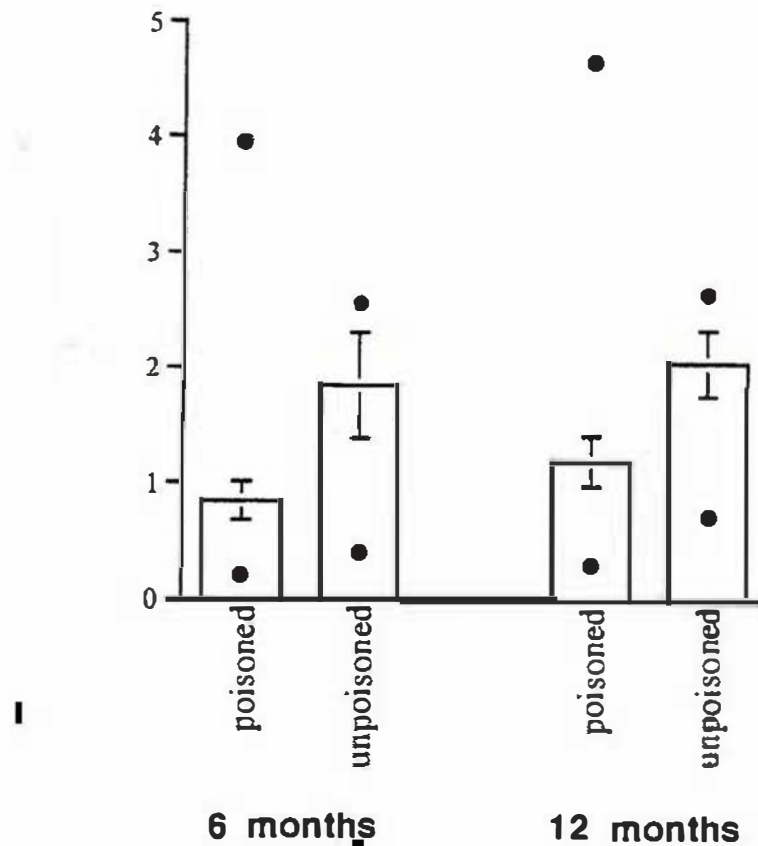


Figure 5.2 : Summary of the mean browse score data collected at *E. nitens* plantations six and 12 months after planting. Mean (\pm se) values for browse score are presented for plantations where poison was used (poisoned), and plantations which were not poisoned (unpoisoned). Solid points indicate the highest and lowest values recorded for each group in each period. Sample sizes (n) are also given

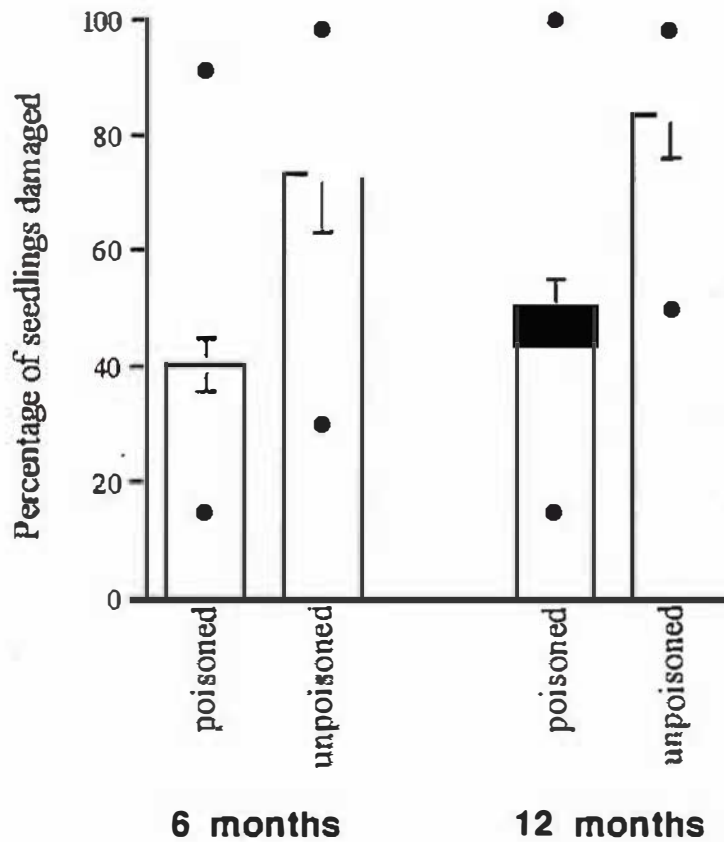


Figure 5.3 : Summary of the percentage of seedlings damaged data collected at *E. nitens* plantations six and 12 months after planting. Mean (\pm se) values are presented for plantations where poison was used (poisoned) and plantations where poison was not used (unpoisoned). Solid points indicate the highest and lowest values recorded for each group in each period. Sample sizes (n) are also given.

These results suggest that poisoning with 1080 does result in an overall significant reduction in browsing damage. This conclusion is, however, predicated upon the damage data collected at the unpoisoned *E. nitens* plantations being representative of the damage severities which could be expected at all plantations if poisoning were not carried out. Confidence in this being the case is somewhat eroded by the low sample size for this group of plantations (n=6).

Two further conclusions can be drawn from the data :

- i) poisoning with 1080 is not always a successful means of controlling browsing damage; and
- ii) browsing damage severity varies enormously between plantations, some appear to have an inherently low potential for damage and the benefits of using control at these plantations is likely to be negligible.

Following this, several prescriptions for management may be recommended :

1. Restrict control effort to problem plantations

The efficiency of current pest control programs could be immediately improved by restricting effort to those plantations where damage is going to be unacceptably high. The benefits of this are two-fold. Firstly, the wasteful expenditure of control effort on areas where the potential for damage is inherently low will be avoided. Secondly, specifically targeting problem areas improves the viability of alternative control measures, such as fencing, which are generally not cost effective for large scale use (Anon 1991; Coleman *et al.* 1997). Unfortunately, this can only be achieved by either assessing and responding to damage very rapidly or by predicting damage severity prior to it occurring. Currently no system exists which can deal effectively with either of these requirements.

2. Review efficacy of current poisoning techniques

Poisoning with 1080 can undoubtedly reduce, at least temporarily, the local abundance of pest species in forestry plantations (Statham 1983; Gregory 1988; also see Chapter 7). As such it can also reduce browsing damage and this is reflected in the generally lower damage levels which were observed during this study at poisoned plantations as compared to unpoisoned plantations. However, poisoning clearly failed to control browsing damage at some plantations, suggesting that the effectiveness of current poisoning techniques needs to be reviewed. It is possible that changes in methodology (e.g. delivery methods, timing, follow up monitoring, etc.) might more fully realise the potential of 1080 poisoning for use as a control tool.

3. Consider alternative strategies

At several plantations (e.g. Aseol[■], Alo49, Npu) the control effort expended on poisoning programs appears to have been wasted because damage was not controlled to an adequate level. Effectively, 1080 poisoning failed as a control technique at these plantations. For these sites a more effective approach might have been to use alternative control measures either in conjunction with, or in preference to, 1080 poisoning. Alternative control techniques may be relatively expensive to implement in the short term (Anon 1991). However, if they are more efficient than 1080 at reducing damage they may, in some instances, prove to be more cost effective in the long term (Engeman *et al.* 1995a). Physical barriers such as fencing have been shown to reduce pest abundance (Howard 1978; Wright 1978; Gregory 1988; Statham 1994) and damage (Engeman *et al.* 1995a). Gregory (1988) designed an electric fence which reduced the abundance of Bennett's wallabies and pademelons in pasture areas by up to 80%. An electric fence design used by Statham (1994) reduced the abundance (as measured by scat counts) of pademelon

and Bennett's wallaby in pasture areas by 80% and 99% respectively. These results are either equivalent or superior to the abundance reductions achieved by 1080 poisoning that were reported by Statham (1983) and Gregory (1988). It should be noted though that in some areas topography, accessibility and theft of materials will undoubtedly make the construction and maintenance of fences impractical.

Once again, acting on this management suggestion is reliant on being able to predict the damage severity which will occur at a plantation prior to it actually occurring. In the following chapters, studies are outlined which aim to isolate the factors that make some plantations more prone to browsing damage than others.

CHAPTER 6

OBSERVER BIAS AND ERROR IN SCAT COUNTS

Many techniques have been used to examine temporal and spatial trends in the abundance of animals. Several of these have been used with vertebrate herbivores in Tasmania, including harvest/catch-effort methods (Johnson 1977; Driessen 1992), examination of historical records (Johnson 1977; Barker and Caughley 1990), spotlight counts from transects (Johnson 1977; Gregory 1988), stake-out counts (Johnson 1977; Gregory 1988) and faecal pellet counts (Johnson 1977; Gregory 1988; Statham 1983). For the research outlined in Chapter 7 and Chapter 8, it was necessary to collect data on the relative abundance of animals at a large number of separate forestry plantations. The only logistically viable method to use for this was faecal pellet counts.

Faecal pellet count techniques have been used to study trends in the abundance of a wide range of vertebrate species including deer (Bennett *et al.* 1940; Rogers *et al.* 1958; Batchelor 1975), elk (Collins and Urness 1978), rabbits (Taylor and Williams 1956; Cochran and Staines 1961) and macropods (Floyd 1980; Arnold and Maller 1987; Johnson and Jarman 1987; Bulinski *et al.* 1997). The technique used in this thesis is referred to as the 'cleared plot' or 'recruitment' method. It is a relatively rapid and inexpensive technique that requires the interpreter to establish fixed plots of known dimension at the study site(s). When the plots are established they are cleared of pellets. Plots are then searched for pellets at regular time intervals. The number of scats deposited per metre per day can then be calculated by dividing the number of scats found by the area searched and the number of days over which pellets were deposited. While it is difficult to accurately convert these data to measures of absolute abundance (Southwell 1989), they do appear to have considerable value as measures of relative animal abundance (Johnson 1977; Johnson and Jarman 1987; Southwell 1989).

Several factors may reduce the value of scat count data. These include the inability of an observer to distinguish faecal pellets of sympatric species, the unequal propensity of observers to miscount scats at sites because of differences in site characteristics such as vegetation cover and, finally, the unequal decomposition rates of scats across sites. All of these have the potential to skew data so that it is related less to the abundance of animals than to observer bias and specific site characteristics. The following is a discussion of four experiments which examine various aspects of the scat count technique. Each experiment will be discussed separately and will be followed by a summary of conclusions.

6.1 EXPERIMENT 1 : SCAT IDENTIFICATION

6.1.1 Introduction

Where two or more species live sympatrically, the value of scat count data can be reduced if the faecal pellets of each species are not sufficiently distinctive that substantial errors are made when trying to distinguish between species. Consider a situation where it is desirable to obtain scat count data for species 'A' living in association with species 'B'. If the scats from species A are mistakenly identified as originating from species B then estimates for species A will be erroneously low. Conversely estimates for species B will be erroneously high.

In Tasmania the Bennett's wallaby, the Tasmanian pademelon and the brushtail possum live sympatrically across much of their range. Researchers who have studied populations of these animals tend to imply that scats from these three species are readily distinguished from each other (Johnson 1977; Statham 1983; Gregory 1988). From field observations and observations of captive animals, it would certainly seem that scats from the brushtail possum, in Tasmania at least, are easily distinguished from other sympatric herbivores. Their scats are relatively uniform in size and shape, much narrower than scats from either macropod species and composed of much finer vegetation particles. However, scats from the two macropod species are frequently observed to be very similar in

form to each other. Despite this, no study has addressed the possibility that macropod scats are misidentified in the field. This experiment aimed to determine the degree of error associated with distinguishing between the scats of these two species.

6.1.2 Methods

Two tests were employed to determine how easily people could distinguish between scats from the Bennett's wallaby and the Tasmanian pademelon.

Test 1

A total of 50 scats were placed in each of 6 containers. Each container held a different mix of scats (Table 6.1). For example, in container two, 40 pademelon scats and 10 Bennett's wallaby scats were present for a total of 50 scats.

Table 6.1 : The mix of scats which were placed in each container.

	Container					
	1	2	3	4	5	6
Number of pademelon scats	50	40	30	20	10	0
Number of Bennett's wallaby scats	0	10	20	30	40	50

Bennett's wallaby scats were obtained from captive animals kept at Bonorong Wildlife Park, located 5 kilometres to the east of Brighton, Tasmania. Pademelon scats were obtained from captive animals kept at the University of Tasmania, Hobart. All scats were collected as they were voided and then stored in a drying oven at sixty degrees until they were used. Scats were collected randomly with respect to the age, size and sex of individuals.

As a means of identification a small pin was inserted into each scat. The pins were identical except that small marks, made with a pair of pliers, were placed near the end of pins inserted into pademelon scats. By removing a pin and examining it I was able to identify the species from which the scat had originated.

Twelve people served as the subjects for this test. These people ranged in their experience of scat identification from completely inexperienced to highly experienced. Subjects were broadly grouped into two categories :

- i) inexperienced - six subjects with absolutely no exposure to scat identification; and
- ii) experienced - six subjects who had previously made identifications of scats from the two species.

Each person was told at the start of the test to treat each container as if it were the contents of a field plot which could contain any mix of scats from the two macropod species. Each person examined each of the 50 scats and identified them as Bennett's wallaby or pademelon. Subjects were forced to decide, even if they were uncertain. To assist people in making identifications a random selection of 20 reference scats from each species was provided in separate labelled containers. People were free to study both the test and reference scats for as long as they felt necessary.

Test 2

This test was carried out as for Test 1, except that participants were allowed to place scats into an 'unidentified' category. If they did not feel confident that they could identify a scat, they were not forced to choose between Bennett's wallaby and pademelon. Several months separated Test 2 from Test 1.

The percentage of scats that were misidentified by subjects was calculated for each container in both Test 1 and Test 2. For Test 1, this was done by dividing the number of scats from a particular species which were misidentified by the total number of scats from that species present and then multiplying by 100 :

$$\% \text{ of scats misidentified} = (\text{no. misidentified} \div \text{no. present}) \times 100$$

The calculation was the same for Test 2, except that those scats for which no attempt at identification was made were removed from the calculation such that :

$$\% \text{ of scats misidentified} = (\text{no. misidentified} \div \text{no. attempted}) \times 100$$

6.1.3 Results

Test 1

The mean estimated proportion of Bennett's wallaby scats in each container is shown in Figure 6.1. The results presented are the means (\pm se) calculated for all subjects (see Table A.1 in Appendix A). For comparison the actual proportions that were present have also been shown. The results for pademelon scats are shown in Figure 6.2. For both species the trends are similar. Subjects tended to underestimate the number of scats that were present from a species when they represented a high proportion of the total. Conversely, when scats from a particular species represented only a low proportion of total scats their numbers tended to be overestimated. A Mann-Whitney test indicated that there was no significant difference between the mean percentage of scats misidentified by inexperienced subjects and the mean percentage misidentified by experienced subjects for either species (Bennett's wallaby : $U_{(6,6)}=13.0$, $p>0.05$; pademelon : $U_{(6,6)}=12.0$, $p>0.05$).

Test 2

In Figure 6.3 the mean estimated proportion of Bennett's wallaby scats in each container is shown for Test 2. (see Table A.2 in Appendix A). Once again, those scats which could not be confidently identified have been excluded from the calculations. The results for pademelon scats are shown in Figure 6.4. The trends are similar to those observed in Test 1. For both species, subjects tended to make underestimates when scats from a particular species represented a high proportion of the total scats present. Conversely, when scats from a particular species

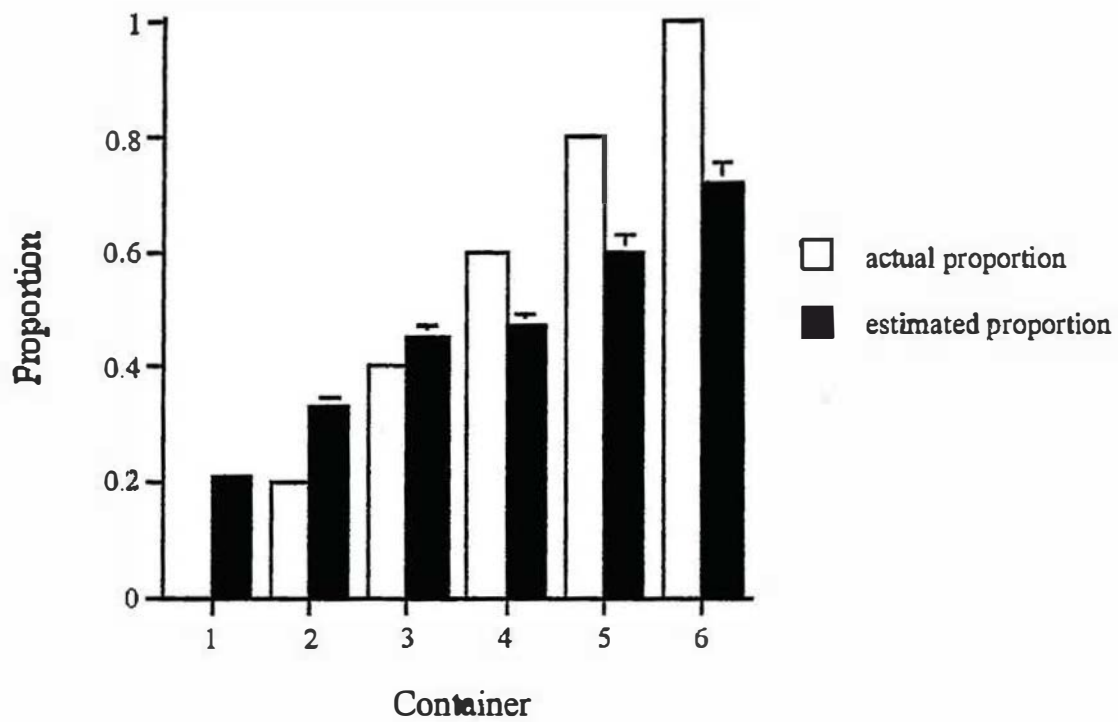


Figure 6.1 : Mean (\pm se) values for the estimates made during Test 1 of the proportion of Bennett's wallaby scats in each container. Actual proportions are given for comparison.

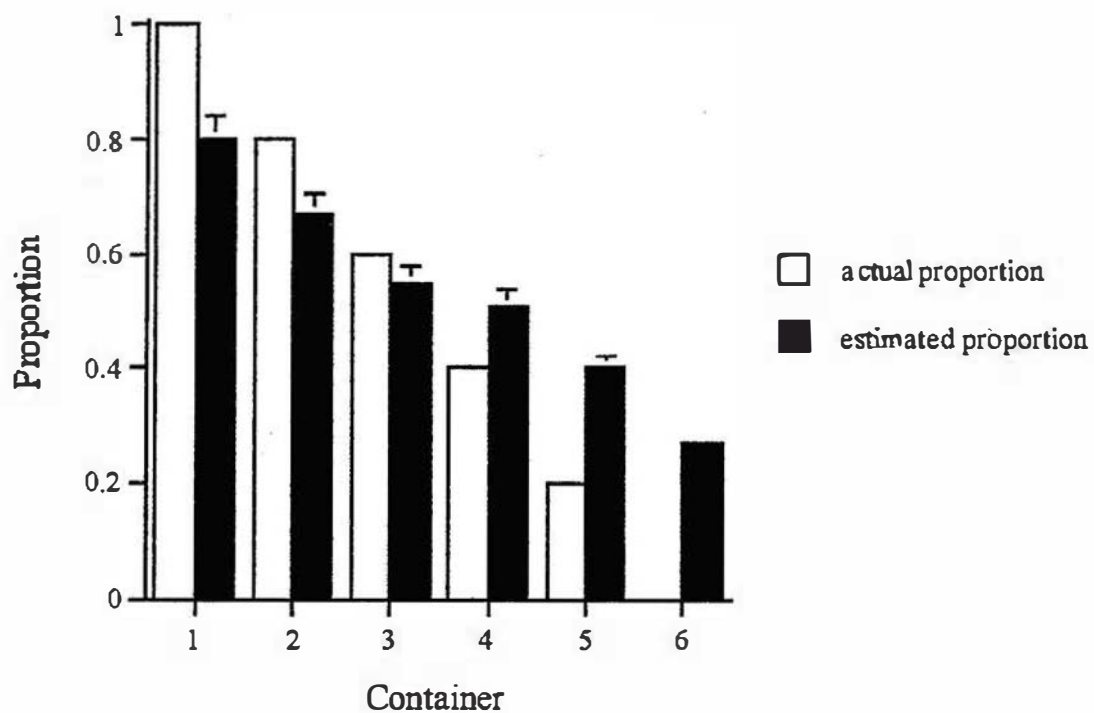


Figure 6.2 : Mean (\pm se) values for the estimates made during Test 1 of the proportion of pademelon scats in each container. Actual proportions are given for comparison.

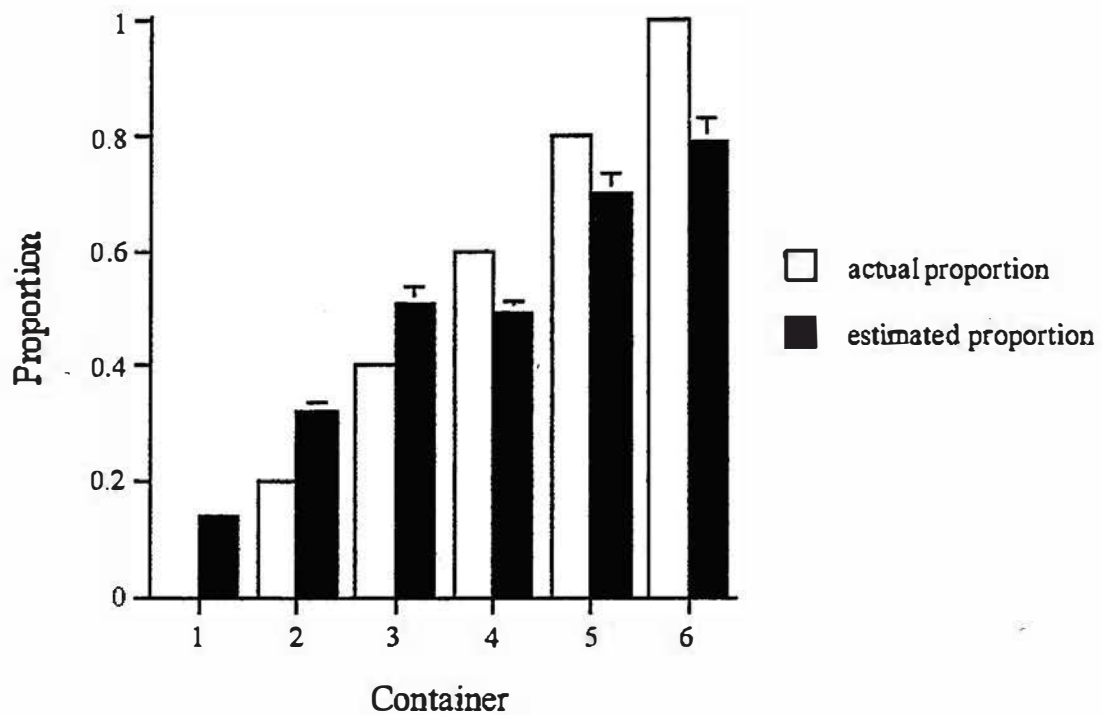


Figure 6.3 : Mean (\pm se) values for the estimates made during Test 2 of the proportion of Bennett's wallaby scats in each container. Actual proportions are given for comparison.

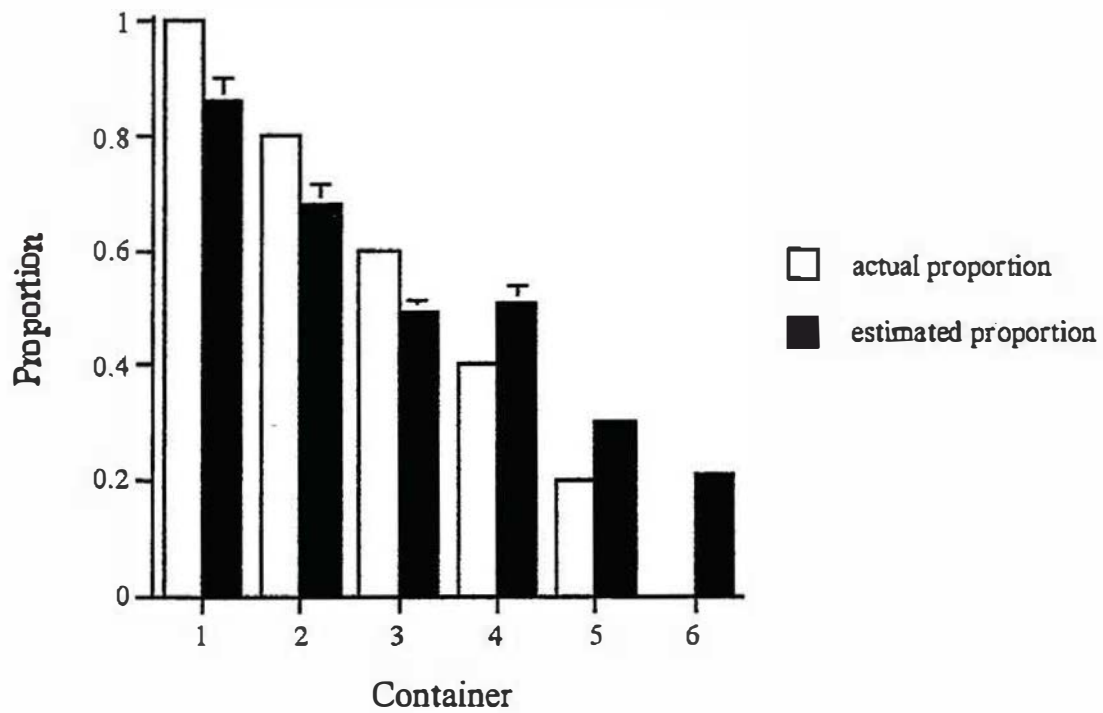


Figure 6.4 : Mean (\pm se) values for the estimates made during Test 2 of the proportion of pademelon scats in each container. Actual proportions are given for comparison.

represented only a small proportion of the total scats present their numbers tended to be overestimated. Results of a Mann-Whitney test indicate that there was no significant difference between the percentage of scats misidentified by inexperienced subjects and the percentage of scats misidentified by experienced subjects for either species (Bennett's wallaby : $U_{(6,6)}=15.0$, $p>0.05$; Pademelon : $U_{(6,6)}=18.0$, $p>0.05$).

A between-tests comparison indicated that there was no correlation between the mean percentage of scats a subject misidentified in Test 1 and the percentage misidentified in Test 2 (Bennett's wallaby : $\rho=0.587$, $n=12$, $p>0.05$; pademelons : $\rho=0.434$, $n=12$, $p>0.05$). A Wilcoxon Signed-rank test indicated that the overall difference in the mean percentage of scats misidentified was not significantly different between Test 1 and Test 2 (Bennett's wallaby : $T_+=17$, $T_-=61$, $n=12$, $p>0.05$; pademelon : $T_+=54$, $T_-=24$, $n=12$, $p>0.05$).

The mean percentage of scats from the Bennett's wallaby and pademelon which were categorised as 'unknown' is shown in Figure 6.5. This figure is based on the data presented in Table A.3 (Appendix A). The mean percentage unidentified for each container was relatively consistent, ranging from 11% ($sd=22$) to 18% ($sd=18$) for the Bennett's wallaby and from 11% ($sd=10$) to 19% ($sd=20$) for the pademelon. A Mann-Whitney test indicated that there was no significant difference in the percentage of scats categorised as 'unknown' by inexperienced and experienced subjects (Bennett's wallaby : $U_{(6,6)}=12.5$, $df=11$, $p>0.05$; pademelons : $U_{(6,6)}=9.0$, $df=11$, $p>0.05$).

6.1.4 Discussion

Estimates of animal abundance obtained from scat counts can be skewed if significant error is made in the identification of scats from species living sympatrically (Neff 1968). Johnson and Jarman (1987), for example, carried out a survey of eastern grey kangaroos and red-neck wallabies using scat counts. They found that kangaroo numbers tended to be underestimated while wallaby numbers

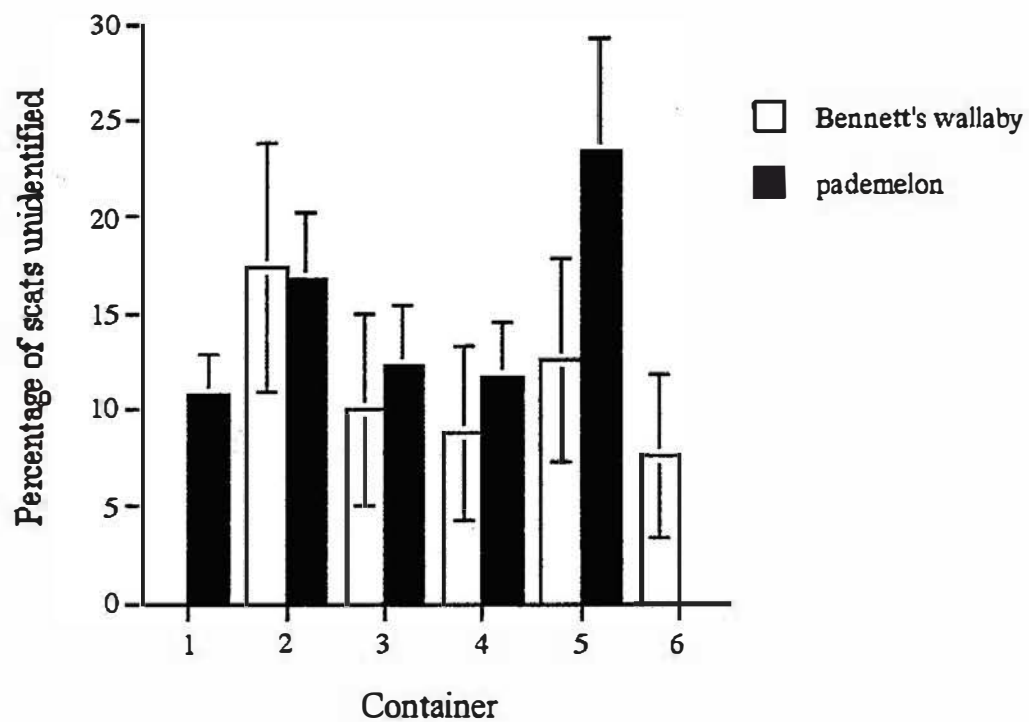


Figure 6.5 : The mean (\pm se) percentage of scats from each species which were unidentified in each container.

were overestimated. The authors attribute this result to mistakenly identifying wallaby pellets as kangaroo pellets.

Authors who have carried out population studies of the Bennett's wallaby, and Tasmanian pademelon have generally implied that the scats from these animals are readily distinguishable. Johnson (1977) states that : "Bennett's wallaby, pademelon and brush possum each deposit pellets of characteristic and identifiable form". He describes "typical" scats from the two macropod species in the following way :

"Bennett's wallaby - discrete, ovoid pellets which are dorso-ventrally flattened, and about 2.5 cm long.

Pademelon pellets - cylinders of about 2.5×1.1 cm. These are deposited in groups of two to about ten which are adhered end to end or partially overlapped."

Gregory (1988) does not describe the criteria by which he distinguished between the two species, stating simply "faeces were identified". Statham (1983) states that scats from pademelons are "distinguished from *M. rufogriseus* pellets by being cylindrical rather than flattened". None of the authors make reference to any difficulty in identifying scats although Johnson (1977) does concede that "forms vary between individuals, and within individuals over time".

The data collected during this study suggests that scats from the two macropod species are not as distinct from each other as these authors purport. The mean percentage of scats which were misidentified by each subject during Test 1 ranged from 9% to 63% for Bennett's wallabies and 13% to 34% for pademelons. In Test 2 the range was from 8% to 42% for Bennett's wallabies and 10% to 43% for pademelons.

For both tests this error affected the estimated proportions of scats present from each species. If scats from a species were low in number, or absent, the estimated

proportion of scats present tended to be overestimated. Conversely, if scats from a species were high in number, the estimated proportion of scats present tended to be underestimated. It is possible that this trend is an artefact of experimental design. For example, the results may reflect the subconscious expectation of subjects that scats from the two species were present in containers at a ratio of 1:1. This would certainly explain the tendency to homogenise data. If similar biases operate in the field observer prejudices about what animals 'should be there' might exert some influence on their ability to accurately identify scats.

The results of this experiment suggest that relative density estimates, based on scat count data, do have the potential to be biased because of misidentification. Indeed erroneous conclusions might be reached even where inference is to be drawn about the presence or absence of a species. However, even if these results are not an artefact of the experimental design, scat counts still appear to be a valuable means of measuring relative abundance. Changes in the actual proportions of scats present in a container were mirrored by changes in the estimated proportions. In other words, as the actual proportion of a species scats present in a container decreased so did the estimated proportion.

Unfortunately it appears that, without a large expenditure of effort, there is not much scope for reducing the errors associated with identifying scats. The 'experienced' subjects used in these tests, on average, performed no better than the inexperienced subjects. This suggests that only through very extensive and comprehensive training would there be any likelihood of improving observer identification skills.

Given that there appears to be an overlap in form for the scats of the two species it might be possible to reduce errors by not attempting to identify those scats which are not distinct. However, this reasoning is not supported by the results of Test 2, where people were allowed to opt out of identifying a scat by placing it in an 'unidentified' category. For both species there was no significant difference in the percentage of scats misidentified in Test 1, where subjects identified all scats, and

the percentage misidentified in Test 2, where subjects identified only those scats they felt confident about.

6.2 EXPERIMENT 2 : MISCOUNTING SCATS

6.2.1 Introduction

During scat counts it is unlikely that all the scats within a plot are counted. How many scats are missed may be linked to the nature of the substrate onto which they are deposited. Scats deposited in areas of extremely thick, high vegetation may be missed more often than those deposited on areas of bare ground. Where scat counts are used as a relative index of abundance across sites, or between trials, it is inherently assumed that the probability of finding scats is equal for all sites and trials. This assumption is rarely validated and it is possible that observed trends reflect site, or seasonal specific characteristics, rather than real trends in animal abundance. This experiment aimed to quantify the proportion of scats which were missed during scat counts in first year forestry plantations and determine if the magnitude of this error was related to the vegetation characteristics of the plantation.

6.2.2 Methods

Study sites

This study was conducted at ten forestry plantations. Their locations are shown in Figure 6.6. The study sites used were the Adu29, Bfra, Brf1, Brf2, Brin, Bwa, Ndig, Npug, Nrit, and Nsp plantations. A detailed description of each of these is given in Chapter 2.

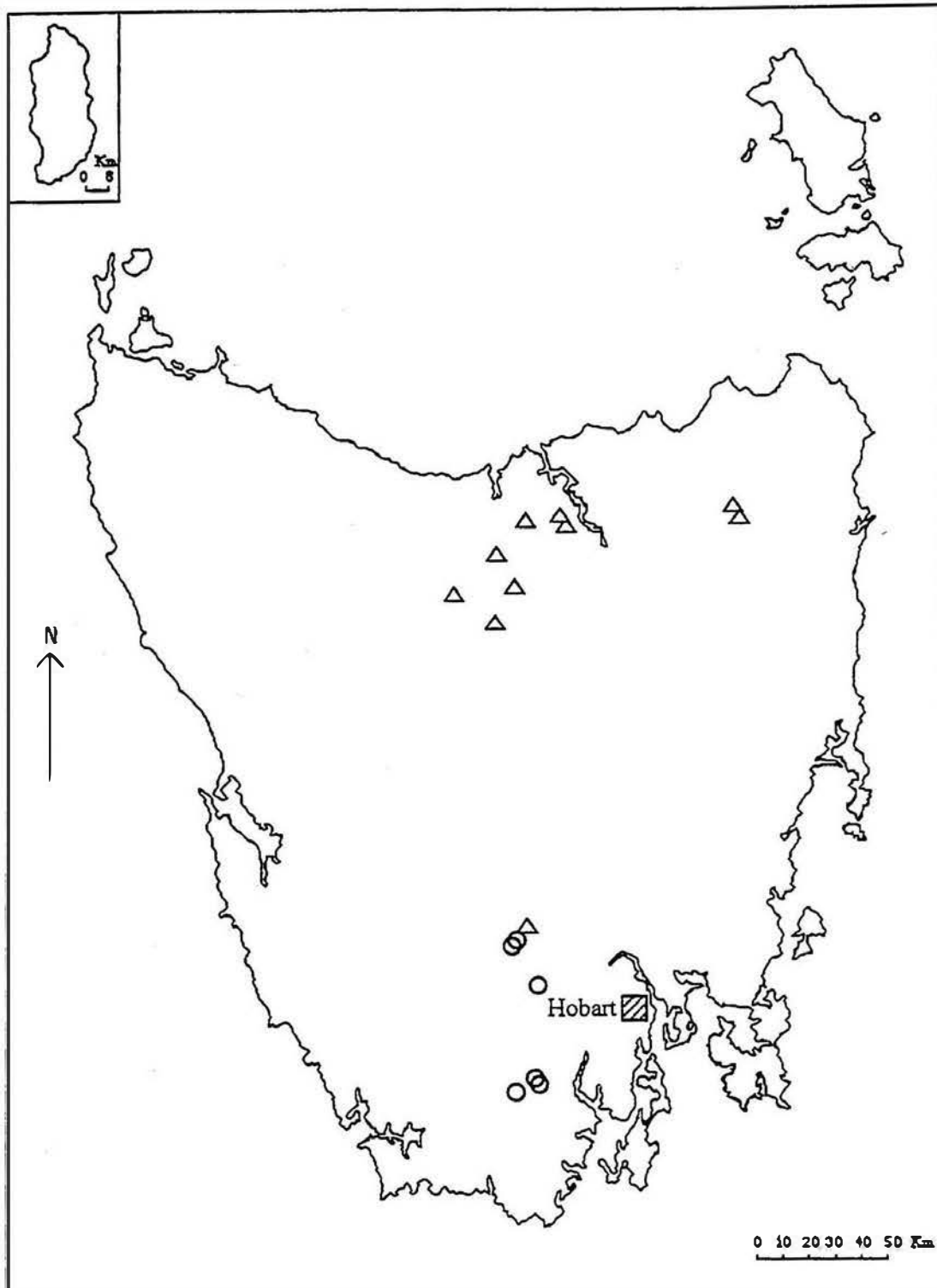


Figure 6.6 : Location of the study sites used during Experiment 2 (Δ), Experiment 3 (O) and Experiment 4 (Δ).

Techniques

At each of the study sites scat plots were established as part of a larger study (Chapter 7). Plots were 25 metres long and one metre wide. Ten randomly selected plots at each site were checked for scats and those found were counted and removed. The vegetation cover on each plot was assessed from six 1 m² quadrats placed at 5 metre intervals along the plot (includes 0 and 25 m points). In each quadrat a visual estimate of vegetation cover was made to the nearest 10% (see Chapter 2). The average of these values was used as an overall estimate of the vegetation cover on each plot. Vegetation height was measured to the nearest centimetre at 0.5, 2.5, 5, 7.5 ... 22.5 and 24.5 metres along each plot using a one metre ruler. The mean of these values was used as an overall estimate of the average vegetation height in each plot. Following this, vegetation along the entire length of the plot was slashed using a brush-cutter. Two observers then moved on hands and knees along the plot, sifting through the vegetation and recording any scats which were found.

Statistics

Stepwise logistic regression was used to model the probability of finding a scat in relation to vegetation data. The procedure is well suited to binary response data (Cox and Snell 1989), in this case whether a scat was found or not found. The model takes the form :

$$\text{logit}(p) = \log \left(\frac{p}{1-p} \right) = \alpha + \beta x$$

where 'p' is the response probability to be modelled, 'x' is a vector of explanatory variables, 'β' is the vector of slope parameters and 'α' is the intercept parameter. Thus an estimation of the parameters α and β allows calculation of logit (p) which in turn can be used to calculate the probability of an event occurring (p) from :

$$p = \frac{e^{(\ln(1/p))}}{(1 + e^{(\ln(1/p))})}$$

Data from individual scat plots were used for the analysis. Plots were excluded if no scats were found in them. The binary response variable was equal to the number of scats counted in each plot before slashing, divided by the total number of scats counted in the plot (includes scats found before and after slashing). The explanatory variables were vegetation height and vegetation cover.

The analysis was performed using the SAS statistics package (SAS Institute 1989a,b). SAS provides estimates for the joint significance of explanatory variables (Score statistic, -2 Log Likelihood), estimates of α and β , and measures of the predictive ability of the derived model (Concordance, Discordance). For a general discussion of the application of logistic regression to binary response data the reader is referred to Cox and Snell (1989). The SAS Users Guide (SAS Institute 1989a,b) can be consulted for discussion of the computing procedure and further definition of the terminology used in this chapter.

6.2.3 Results

In total there were 2831 scats found before slashing and 3077 after slashing. All the scats which were found were identified as being from either Bennett's wallaby or pademelon. In Table 6.2 the number of scats which were found at each plantation before and after slashing is shown along with mean vegetation height and coverage. The number of scats which were found at a plantation, prior to slashing, ranged from 0 to 526. The total number of scats found at a plantation, including those found before and after slashing, ranged from 0 to 574. The percentage of scats which were overlooked (i.e. those found after slashing) ranged from 2.5% to 12.3 %. A paired t-test for differences between the before and after slashing scat counts indicates that there is a significant difference between the two ($t=4.445$, $df=9$, $p<0.01$).

A chi-squared analysis comparing the observed number of scats missed to that which would have been expected if scats were missed randomly with respect to site, indicates that there is a highly significant difference between the two ($\chi^2 = 35.2$, $df=8$, $p<0.01$). This result suggests that the propensity for scats to be missed was not constant across sites. It should be noted that the 'Nsp' plantation data had to be excluded from the analysis because no scats were found.

Table 6.2 : The number of scats which were found before slashing (No. scats no slash), the total number of scats found (No. scats after slash) and calculated percentages for the number of scats missed (% missed) are shown. Mean vegetation height and cover are shown with standard deviations in parenthesis.

Plantation	No. scats no slash	No. scats after slash	% Missed	Vegetation height (cm)	Vegetation cover (%)
Adu29	255	270	5.6	6.3 (15.1)	16.0 (29.4)
Bfra	142	160	11.3	13.0 (24.1)	30.4 (36.2)
Brf1	526	574	8.4	10.2 (16.0)	25.5 (18.3)
Brf2	350	388	9.8	11.8 (18.3)	8.0 (8.5)
Brin	470	496	5.2	8.9 (17.3)	38.5 (34.0)
Bwa	309	317	2.5	6.8 (13.0)	59.2 (31.0)
Ndig	321	350	8.3	15.2 (19.2)	23.8 (23.7)
Npu	79	90	12.2	27.7 (24.5)	33.0 (27.9)
Nrit	379	432	12.3	10.8 (18.8)	13.0 (15.6)
Nsp	0	0	-	6.3 (10.4)	74.8 (34.0)

Differences in the percentage of scats missed may have resulted from differing vegetation characteristics at plantations. Mean vegetation height varied from 6.3 to 27.7cm while vegetation cover ranged from 8% to 75%. A stepwise logistic regression was carried out comparing the probability of finding a scat in a given plot to vegetation height, cover and interactions between the two (height \times cover). A -2 LOG L value of 28.7 ($df=3$, $p=0.0001$) was obtained, indicating that the combined effect of the three variables on the probability of finding a scat is significant. Using the parameter estimates the probability of counting a scat can be described by :

$$\text{logit}(p) = 2.4306 + [(-0.0276 \times \text{height}) + (0.0097 \times \text{cover})]$$

Equation 6.1

where 'p' is the probability of counting a scat. Thus a value for p can be then be derived from :

$$p = \frac{e^{(\text{logit}(p))}}{(1 + e^{(\text{logit}(p))})}$$

Equation 6.2

Unfortunately the confidence interval displacement value (c=0.593) associated with the model suggest that its predictive powers are poor.

6.2.4 Discussion

If scat counts are used as a relative measure of animal abundance then the degree of error involved with counts does not matter as long as it remains constant in space and time. Where this is not the case, relative abundance data may be biased. It has long been recognised that the error associated with direct counts of animals during aerial surveys, perhaps the census method most analogous to scat counting, can be biased by a range of factors (Caughley *et al.* 1976). However, there has been very little examination of the possible biases associated with scat counts, despite the wide range of species and habitat types for which the method has been used. Presumably this reflects the confidence of observers that either all the scats present are counted, or that the chances of finding scats is equal for all sites.

During this study it was found that not all scats were found during standard counts on Tasmanian forestry plantations. From a total pool of 3077 scats, 2831 (92%) were found prior to slashing vegetation. Overall then, the error associated with counts was quite low. However, this error was not constant for all sites. There was a greater propensity to miss scats at some sites than others with the percentage of missed scats ranging from 3% to 12%.

Where such site specific bias exists, the value of count data can be greatly improved by the application of correction factors. A range of correction factors have been formulated to account for biases in aerial survey counts (e.g. Bayliss

and Giles 1985). In this study, a logistic regression was used to model the relationship between the probability of finding a scat and vegetation measures. The results indicate that increases in vegetation height reduce the chances of finding a scat while increases in vegetation cover actually improve the chances. The latter result seems counter intuitive. One possible explanation may be that a low, verdant cover actually improves the visibility of scats by providing a starkly contrasting background. Macropod scats are typically black in colour when first deposited, fading to a dull brown over time. As a result, scats deposited on bare earth may be less visible than those deposited on a cover of green vegetation.

Unfortunately, the predictive power of the derived model was poor. It seems that further, as yet unmeasured factors play a part in influencing the probability of finding a scat. Before any correction factors can be formulated these variables will need to be isolated. There are a range of factors known to affect transect counts of animals which may also have the potential to affect scat counts. These include lighting (Low *et al.* 1981; Bayliss and Giles 1985; Short and Bayliss 1985) and observer ability (Caughley *et al.* 1976; Packard *et al.* 1985; Short and Bayliss 1985). Van Etten and Bennett (1965) have already demonstrated that observers do differ in their ability to find scats.

6.3 EXPERIMENT 3 : SCAT DECOMPOSITION

6.3.1 Introduction

Ideally, when using scat count methods, scats are counted and removed from plots before they begin to decompose. Where this is not practiced, data can become biased if scats decompose at different rates across sites or between trials. There are many environmental factors which could affect the degradation rate of faecal pellets including insect activity, temperature, exposure to wind and rainfall. Johnson and Jarman (1987) found that the rate of decomposition of eastern grey kangaroo (*Macropus giganteus giganteus*) and red-necked wallaby pellets was

related to weather, with more pellets disappearing during times of warm, moist weather than during periods of cold, dry weather.

Where sampling periods are separated by sufficient duration that decomposition of scats occurs, it is important to establish the extent of decomposition for each trial and site. In the study outlined in Chapter 7, scat counts were carried out at 80 day intervals. In this experiment I wanted to determine whether or not significant levels of scat decomposition could occur during this time, and if so, whether decomposition rates varied between sites. Additionally, decomposition data were collected beyond 80 days so that aspects of the long term degradation of scats could be examined.

6.3.2 Methods

Study sites

The Bennett's wallaby scats used during this study were collected from the Lime Bay area in south-eastern Tasmania. The site had a large population of Bennett's wallabies and significant grassland areas where scats could be easily located. The study proper was carried out at seven sites. One of these was located in bushland contained within the University of Tasmania (Uni) and the remaining six were the Atn18, Atn20, Asx51, Fso1b, Fhp23g and Fkd13b plantations described in detail in Chapter 2. The location of all sites is shown in Figure 6.6.

Techniques

Bennett's wallaby scats were collected shortly after dawn and refrigerated at 5°C until they were used 1-2 days later. The experiment consisted of two trials. The first began in January 1996 (summer) and the second in June 1996 (winter). For each trial five plots were established at each of the seven sites. A plot consisted of four wooden pegs driven into the ground to form a square with sides approximately 0.2 m long. Ten Bennett's wallaby scats were placed inside each plot, for a total of 50 scats at each site. The location of scats within a plot was

marked using small wire pegs. To prevent scats from being washed away and to mark the plot boundary, each plot had shade cloth wrapped around the wooden pegs at ground level to form a barrier approximately 0.1 m tall. For each trial, the condition of scats was checked approximately every 10 days over a 200 day period. Scats were considered to be decomposed when they could no longer be identified as macropod scats. Rainfall (mm) was measured at each site using a 100 mm capacity Nylex rain gauge.

6.3.3 Results

A plot of the mean number of scats still intact over time for Trial 1 (commenced in summer) and Trial 2 (commenced in winter) is shown in Figures 6.7 and 6.8 respectively. Average decomposition rates (scats lost per day) are shown in Table 6.3. The data which were obtained from the Uni site during Trial 1 have been omitted from these analyses and further discussion, because the plots involved in the trial were vandalised.

Table 6.3 : Results of the scat decomposition trials. The number of scats which were missing from each site at the end of each trial (200 days) is shown. The average rate of scat loss per day has been calculated.

Site	Trial 1		Trial 2	
	Scats lost over trial	Scats lost per day	Scats lost over trial	Scats lost per day
Fkd13b	8	0.040	20	0.100
Fhp23g	14	0.070	25	0.125
Fsolb	5	0.025	16	0.080
Atn20	7	0.035	13	0.065
Atn18	9	0.045	18	0.090
Asx51	13	0.065	22	0.110
Uni	na	na	8	0.040

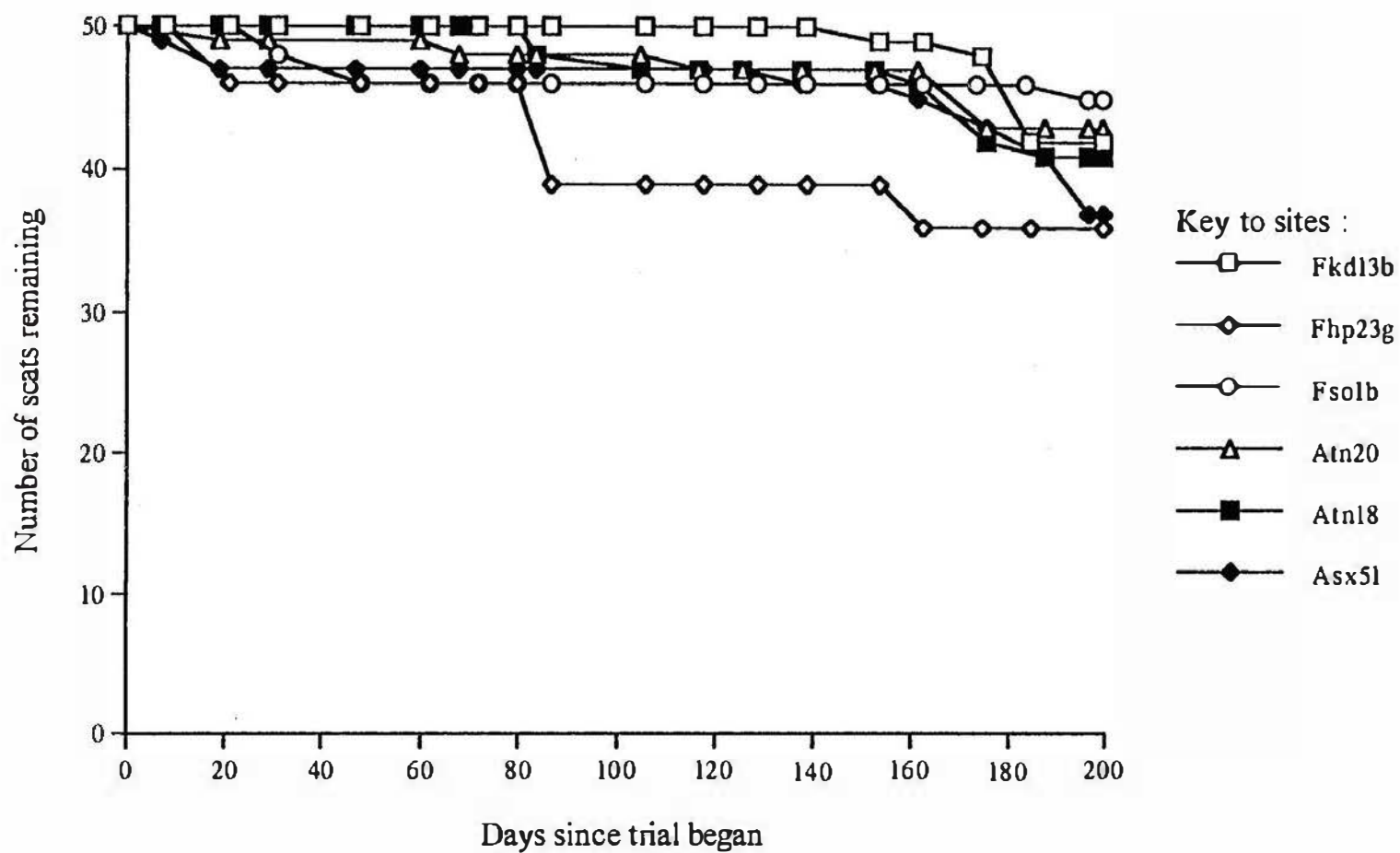


Figure 6.7 : The number of scats remaining at each sampling period during Trial 1. Results for the 'Uni' site have been excluded.

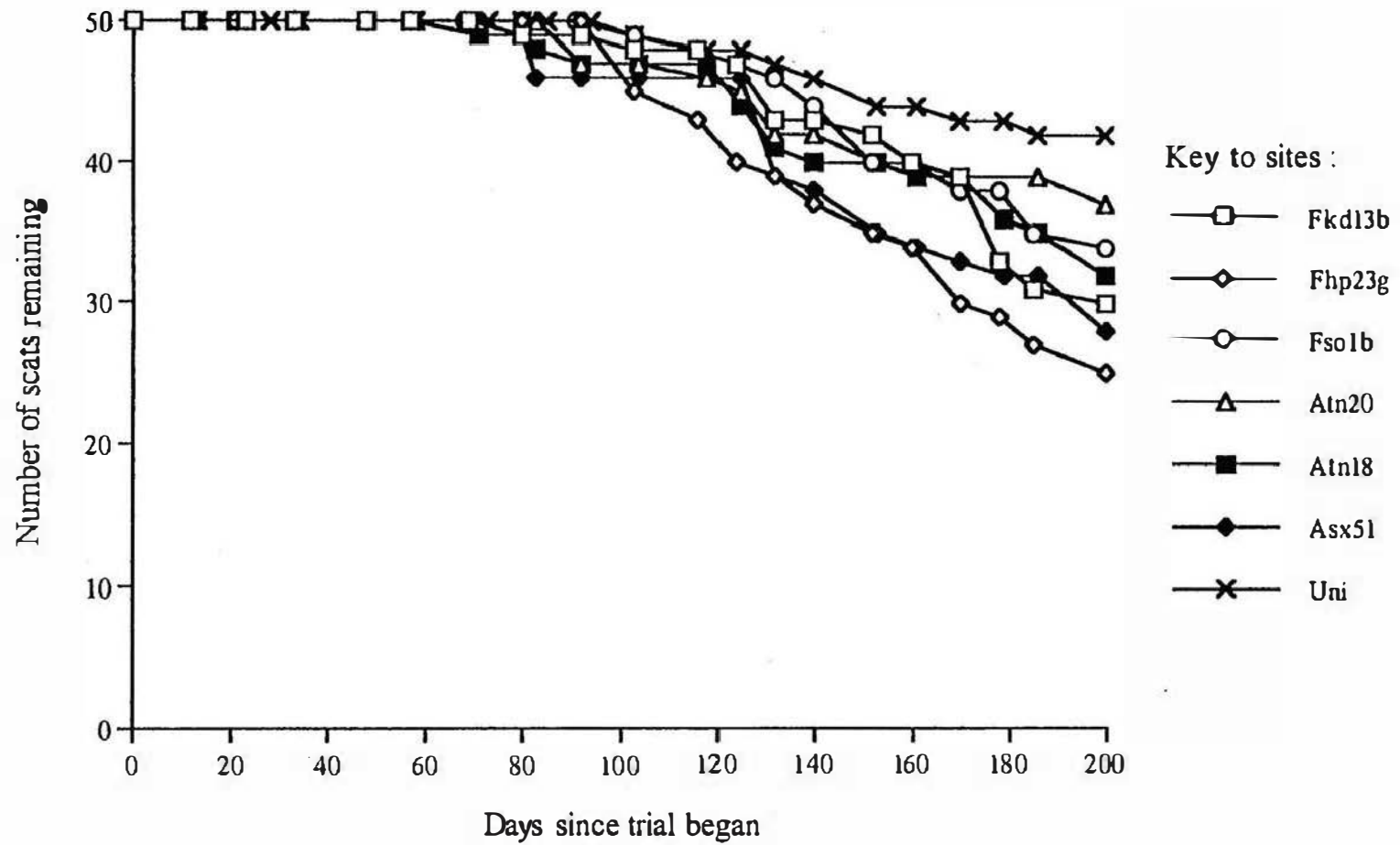


Figure 6.8 : The number of scats remaining at each sampling period during Trial 2.

In general, the number of scats which were missing from a site at the 80 day sampling period was low. The average number of scats lost was 2.2 (sd=1.8) for Trial 1 and 0.4 (sd=0.5) for Trial 2. A chi-squared analysis comparing the number of scats remaining at 80 days to the starting number (50) indicates that there is no significant difference between the two for either Trial 1 ($\chi^2=0.90$, df=5, $p>0.05$) or Trial 2 ($\chi^2=0.42$, df=6, $p>0.05$).

The number of scats which had decomposed at each site by the end of Trial 1 ranged from 5 to 14 with a mean of 11 (sd=6). For Trial 2, the range was from 8 to 25 with a mean of 17 (sd=6).

A plot of the cumulative rainfall data, which was collected for each site during Trial 1, is shown in Figure 6.9. The amount of rain to fall during the 200 day duration of Trial 1 ranged from 523.4 mm (Fhp23g) to 773.2 mm (Atn20). No correlation could be found between the amount of rainfall at a site during Trial 1 and the number of scats which decomposed ($\rho=0.657$, $n=6$, $p>0.05$).

There was a significant difference between the final number of scats which decomposed during Trial 1 and the final number which decomposed during Trial 2 ($t=10.961$, df=5, $p<0.01$).

6.3.4 Discussion

Up to day 80 of the decomposition trials there was very little decomposition of scats for any site or trial (maximum of 8%). At several sites no scats were lost at all during this period. This suggests that, over the range of environments sampled, scat density estimates are unlikely to be biased if sampling intervals are 80 days or less. In fact, the scats in this study were not readily broken down and, for any given site, over half of the scats still remained at the end of the trials (200 days). Conceivably then, the interval between counts could be even longer and still be reasonably unbiased.

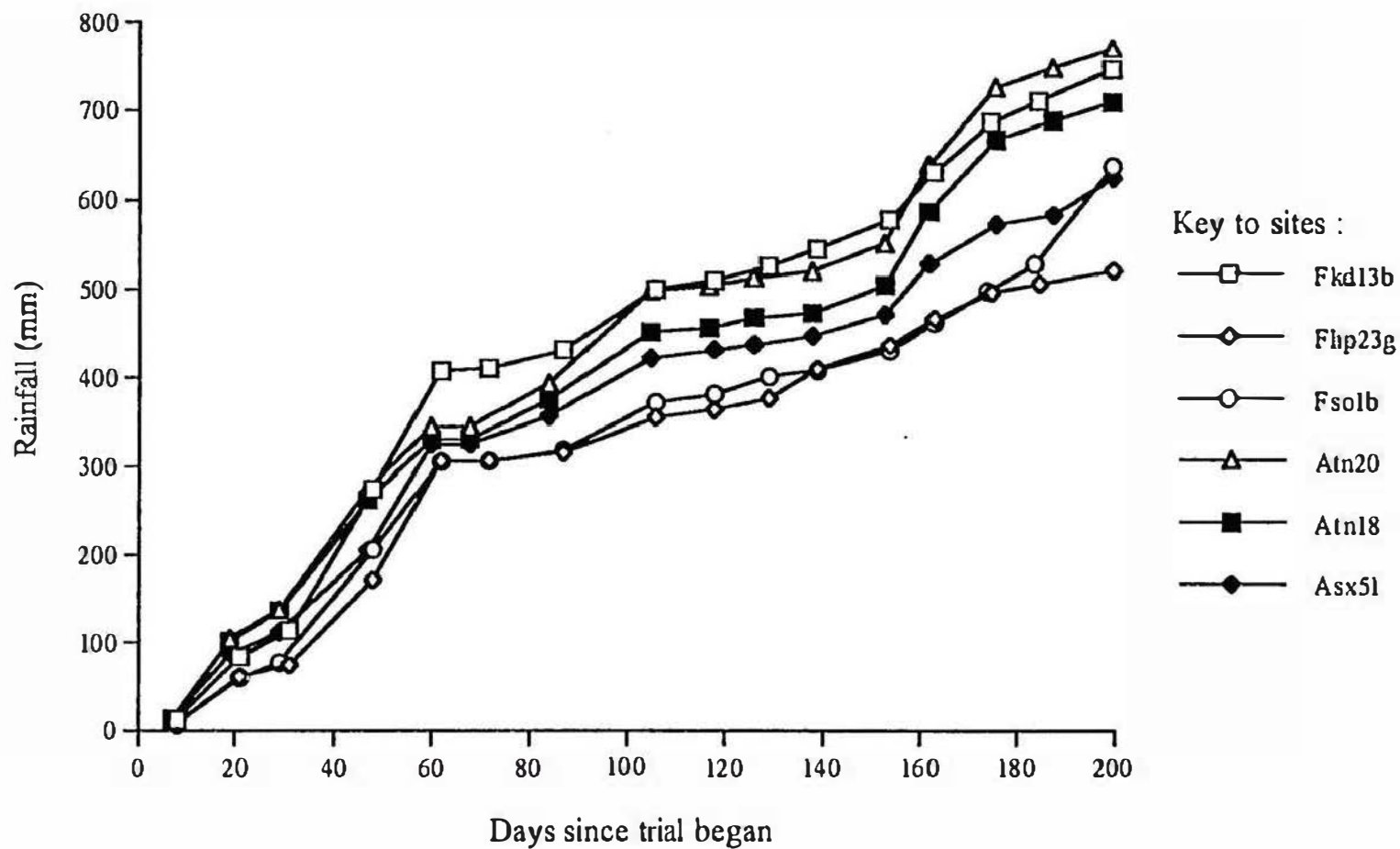


Figure 6.9 : The cumulative rainfall data collected at each site during Trial 1. Results for the 'Uni' site have been excluded.

In the majority of published studies for Australian species, decomposition trials have been run only long enough to establish the extent to which scats decompose between sampling periods (e.g. Johnson and Jarman 1987). There is little published data from longer term studies (Southwell 1989). Johnson (1977) examined the decomposition of scats from Bennett's wallabies, pademelons and possums in Tasmania. For all species he observed a linear decline in scats remaining over time. In contrast the results of this study indicate that, at the majority of sites, the decline of Bennett's wallaby scats is curvilinear at least until 200 days.

A further notable difference between the results of this study and that of Johnson's (1977) is the rate at which Bennett's wallaby scats decomposed. Johnson's (1977) linear regression equation for the rate of scat decomposition in pasture areas predicts that all scats would be decomposed after 138 days. This value is far lower than for the equivalent period in this study (Trial 1 : 78-100% of scats remaining; Trial 2 : 74-92% of scats remaining). These differences may primarily be attributable to differences in methodology. Johnson (1977) marked randomly selected groups of scats for observation. Apparently these were of mixed age so that some scats are likely to have already lain on the ground for an extended period of time. Conversely, during this study, only fresh scats were used so that they were all the same age. The results of this study may therefore provide a more accurate reflection of the decomposition rates of Bennett's wallaby scats.

The study of Perry and Braysher (1986) probably provides a better comparison than that of Johnson (1977). They studied the decay rate of pellets of eastern grey kangaroos at three sites in the Australian Capital Territory. Their study was of similar duration to this one (six months) and they used scats of even age. Their results show a similar trend in decomposition to the data presented here, with scat numbers declining curvilinearly with time. Decomposition was negligible to around 2 months and then declined fairly constantly to around six months.

One interesting difference in the results was that Perry and Braysher (1986) observed no real difference in decay rates between the three sites they used. Differences of up to 22% were observed in the number of scats remaining to 200 days during this study. It is possible that environmental conditions did not vary greatly between the three sites used by Perry and Braysher (1986). Two of the sites were close together and all

three were areas of native grassland. A number of other authors have found decomposition to vary between sites. Johnson (1977) found that decay rates for Bennett's wallaby and pademelon scats were higher in pasture areas than in areas of dry sclerophyll forest. Hill (1982) observed much faster decomposition of eastern grey kangaroo scats at one site compared to another, a result he attributed to termite attack. Johnson and Jarman (1987) found that the decay of pellets from the eastern grey kangaroo and red-necked wallaby was more rapid in areas covered in short grass than in areas covered in long grass. Finally, Harested and Bunnell (1987) found that deer pellets decomposed more rapidly on moist sites than dry ones.

There are many environmental factors which could affect decomposition rates. These include insect activity (Hill 1982), rainfall (Wallmo *et al.* 1962; Johnson and Jarman 1987), temperature (Cochran and Staines 1961) and vegetation cover (Harested and Bunnell 1987; Johnson and Jarman 1987). If these variables differ spatially or temporally then differential decomposition rates could be expected between sites and trials. Southwell (1989) performed a stepwise multiple regression on the data collected by a number of authors to determine what effect rainfall, average maximum temperature, grass density, shrub density, tree density and diet of study animals had on rate of scat decomposition. No clear results emerged from the analysis. Southwell (1989) attributed this to the coarse nature of the data, the exclusion of other relevant variables and the likelihood that the effects of variables are complex and interacting.

In this study, rainfall was considered to be the environmental factor most likely to vary between sites and consequently to be an important factor affecting decomposition rates. Other authors have found rainfall to be a very important influence on rate of scat decomposition (Wallmo *et al.* 1962; Van Etten and Bennett 1965; Harested and Bunnell 1987; Johnson and Jarman 1987). However, in this study rainfall alone could not be used to explain the difference in decomposition rates between sites. As Southwell (1989) has suggested, the relationship between rate of decomposition and environmental variables appears to be a complex one.

There appeared to be some difference in the rate of decomposition of scats between trials. Scats included in the trial that commenced in winter decomposed more rapidly than those used in the trial that commenced in summer ($t=10.96$, $df=5$, $p<0.01$). A number of other authors have noted seasonal trends in rate of scat decomposition, but these appear to largely contradict the results found in this study. Johnson and Jarman

(1987) found red-necked wallaby and eastern grey kangaroo pellets disappeared more rapidly during the moist warm weather of autumn than during the dry cold weather of winter. Hill (1982) also recorded low decay rates for eastern grey kangaroo scats compared to summer rates. The difference between the seasonal trends observed in this study and those observed by these authors probably reflects differences in the influence of decomposition agents. For example, a major factor influencing seasonal variation of decomposition rates observed by Johnson and Jarman (1987) and Hill (1982) appears to have been insect activity. It is possible that in Tasmania, insects play only a minor role in the decomposition of macropod scats. Of more importance might be periods of heavy rainfall and frost events, which are more prevalent in winter.

Limitations of the study

A major aim of this study was to determine if significant scat decomposition occurs within a sampling interval of 80 days. This was shown not to be the case for the Bennett's wallaby. However, no data were obtained for the three other important Tasmanian browsing species; the Tasmanian pademelon, the brushtail possum and the rabbit. Unfortunately, efforts to include these species were hampered by an inability to locate sites suitable for the collecting of large numbers of fresh scats over a short space of time.

If scats from these species do decompose significantly within 80 days, and if decomposition rates vary across sites, then it is possible that estimates of the relative abundance of these species may be biased. However, this is thought to be unlikely. The results of Johnson (1977) suggest that the decomposition rates of Bennett's wallaby and pademelon scats are similar, while scats from the brushtail possum are slower to decompose. Extension of Johnson's (1977) findings to this study would suggest that decomposition over an 80 day period is negligible for all three browsing species, particularly given that his study utilised scats of mixed ages.

6.4 EXPERIMENT 4 : EFFECTS OF PLOT SIZE

6.4.1 Introduction

When using scat count methods, the experimenter must decide on the design of plot to use. Factors to consider include the shape and dimension of each plot. In some cases, plot design may be determined by certain aspects of the study. In areas of thick vegetation, for example, smaller plots may be more efficient than large plots. However, in the majority of cases plot design seems to be almost arbitrary, particularly with reference to the size of the plot which is used. A review by Southwell (1989) indicates that the size of linear scat plots which have been used for monitoring macropod abundance has ranged from 10 m² (Johnson 1977) to 240 m² (Andrew and Lange 1986).

When using scat counts to determine the relative abundance of a species within a particular area, it obviously makes statistical sense that the greater the proportion of an area that can be sampled the more precise and accurate the overall estimate. From a logistical standpoint, however, sampling from large areas is rarely practical. At some stage a point must be reached where the per unit increase in effort for return on improvement of precision decreases towards zero. In logistical terms then, it makes sense to use the smallest plot that will still achieve adequate levels of precision. One way to determine what plot size to use is to compare the variation in scat counts associated with plots of varying sizes. This experiment aimed to determine how scat count data might be affected by plot size.

6.4.2 Methods

Study sites

This experiment was carried out at the Adu29, Bfr, Brf1, Brf2, Bri, Bwa, Ndi, Npu, Nrit and Nsp plantations. These are described in detail in Chapter 2 and shown in Figure 6.6.

Techniques

At each plantation 10 plots were established as described in Chapter 7. Plots were 25 metres long and 1 metre wide. For the purposes of this study they were also divided into 5 metre intervals so that data could be recorded for each consecutive five metre segment. In this way scat count data could be obtained for 5, 10, 15, 20 and 25x1 m plot sizes. When the plots were established, they were cleared of scats. Following this, a single count of scats was carried out eighty days after plot establishment (Chapter 7). For each five metre segment, the observer would move slowly from one end to the other with a one metre ruler abutting the tape measure. The ground immediately in front of the ruler would be searched for scats and any found would be identified, counted and then removed from the plot. Scats could be identified as being from a Bennett's wallaby, pademelon or other herbivore. If the species of origin of a macropod scat was uncertain, the scat was counted as an 'unknown macropod'. For some of the analyses these data were pooled with the data for the Bennett's wallaby and pademelon and called 'total macropod' scats.

Statistics

A measure of the precision of estimates of scat density obtained from plots of varying sizes was required. Southwell (1989) outlines a term for measuring the precision of scat counts called the coefficient of variation (CV) (see also Eberhardt 1978). This value is essentially the standard error (se) of counts expressed as a percentage of the mean :

$$CV = \frac{se}{mean} \times 100$$

6.4.3 Results

In Table A.4 (Appendix A) the mean (\pm sd) density of scats, calculated as the average of the number of scats found divided by the number of square metres of plot searched, is presented for the five different plot sizes at each of ten plantations. Data are presented for Bennett's wallaby, pademelon, total macropod

scats (incorporating Bennett's wallaby, pademelon, and macropod scats which could not be identified to species level) and 'other' herbivores.

Perusal of the data indicates that mean scat density estimates remain fairly constant regardless of plot size. The plantation with the widest range for pademelons was Brf1, where estimates ranged from 0.74 (20 m² plot) to 1.42 scats.m⁻² (5 m² plot). The narrowest range occurred at the Nsp plantation where estimates ranged from 0.00 (5,10,15 m² plots) to 0.01 scats.m⁻² (20 and 25 m² plots).

In Figure 6.10 the mean coefficient of variation (CV) values (Section 6.4.2) have been plotted for each plot size and species grouping. The mean values have been calculated from the data presented in Table A.5 (Appendix A). Very few scats were found at the 'Nsp' plantation and, because of this, it had extremely high CV values for the plot sizes where scats were found. I felt that this would disproportionately bias pooled results. Consequently, the 'Nsp' data was excluded from calculations of mean CV values.

For the Bennett's wallaby, pademelon, and total macropod scat groupings, there is little difference in mean CV values between plot sizes. The largest differences occur from the 5 to 10 to 15m² plot sizes where mean CV values progressively decrease with increasing plot size. Increases in plot size beyond 15 m² result in only minor improvements.

The mean CV values for the 'other' herbivore scat grouping were much higher than for the macropod groups. This probably reflects the low numbers of scats which were found from 'other' herbivores. The mean CV values for this grouping were very variable between sites and plot sizes. The lowest CV value was recorded for plots of 25 m². There were large decreases in CV values from the 5 to 10 m² plot sizes and also from the 15 to 20 m² sizes.

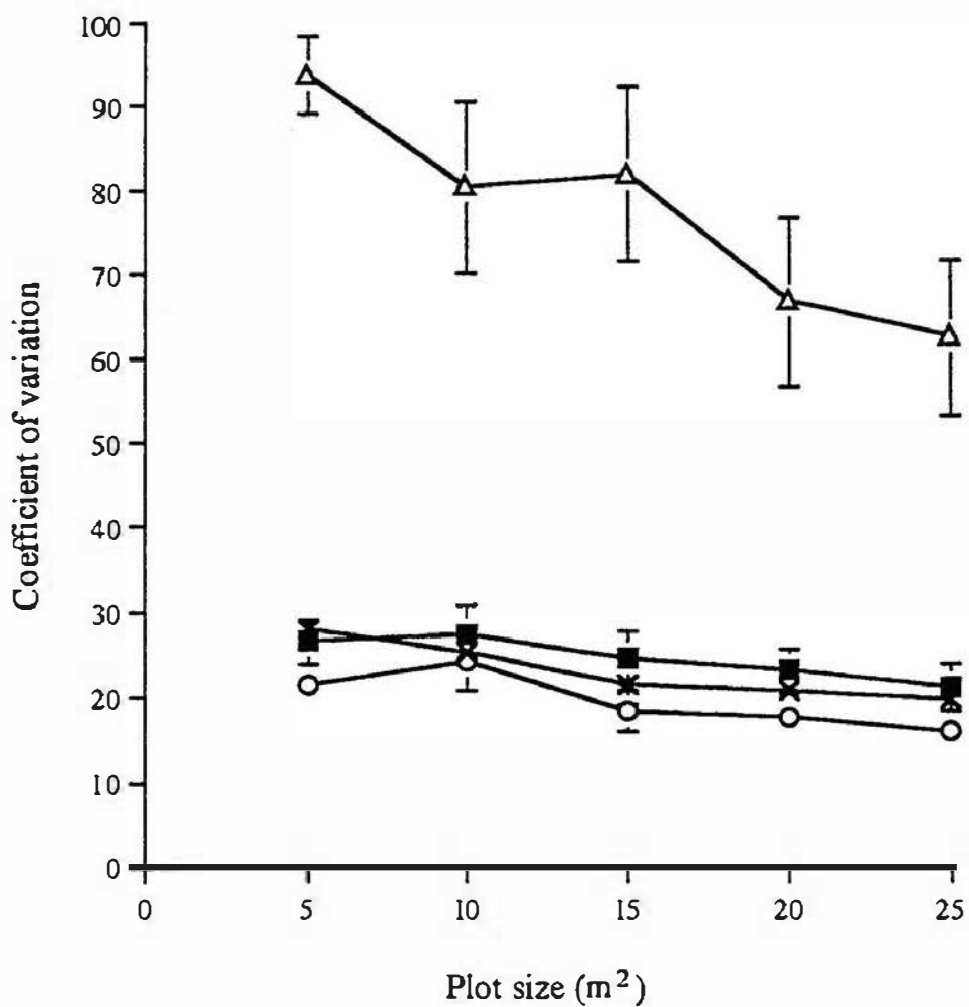


Figure 6.10 : The mean coefficient of variation values plotted for each plot size. Results are calculated from counts of pademelon (×), Bennett's wallaby (■), total macropod (○) and 'other' herbivore (Δ) scats. Standard error bars are included.

6.4.4 Discussion

There is remarkable variation in the design of the faecal pellet plots which have been used to study populations of terrestrial herbivores (Neff 1968; Southwell 1989). Two basic shapes are evident : circular plots and belt plots. However, the dimensions of these plots have varied greatly from study to study. A review by Southwell (1989) found that belt plots used in studies of macropod species have ranged in size from 10×1 m (Johnson 1977) to 120×2 m (Andrew and Lange 1986). In Tasmanian studies the following belt plot dimensions have been used : 10×1 m (Johnson 1977), 20×1 m (Gregory 1988) and 25 ×1 m (Statham 1983).

Despite this wide range in plot sizes, there has been very little study of the effect that plot size may have on scat density estimates and precision of count data for Australian herbivores. This is surprising given that a number of northern hemisphere studies have shown that faecal pellet density estimates and precision of counts can be dependent on plot size. For example Harris (1959), Neff (1960), Smith (1964) and Smith (1968) all found that density estimates for deer pellet groups were higher in smaller sized plots than large plots. They variously attribute this to observer boredom and the effects of plot edge to area ratio on the number of 'borderline' pellet groups counted as falling within the plot.

Density estimates

The results of this study suggest that density estimates are reasonably static with regard to plot size. The exception appears to be the density estimates for pademelons and Bennett's wallabies obtained from the smallest plot size, 5×1 m. These values were considerably different from those obtained for other plot sizes. The mean pademelon scat density estimate for the 5×1 m plot was 0.53 scats.m⁻² while the other plot sizes gave estimates of 0.45-0.46 scats.m⁻². For Bennett's wallabies the smaller sized plots actually gave a lower estimate (0.48 scats.m⁻²) than the larger sized plots (0.53-0.55 scats.m⁻²). Smith (1968) observed that smaller plots generally gave higher density estimates for deer pellets. He

attributes this result to not seeing some of the scats in larger plots. While this might also explain the result that was observed for pademelons, in this study it does not explain the results obtained for the Bennett's wallaby.

Precision

Precision is an important quality of any abundance monitoring technique (Southwell 1989). Where precision is poor, abundance estimates and inferences about populations cannot be made with confidence. In effect, real trends can be masked by large sampling errors. In statistical terms then, it is obviously preferable to design a study from which the most precise measurements possible are obtained. However, in logistical terms this can be very difficult. The amount of effort required to survey even small areas is considerable. In many cases a trade off between precision and effort may be necessary.

The results of this study suggest that the best compromise between precision and effort are plots of 15×1 m. The coefficient of variation (CV) values for this plot size are considerably lower than those calculated for smaller plot sizes.

Conversely, there is little difference between CV values for the 15×1 m plot size and the two larger plot sizes of 20×1 m and 25×1 m. Searching a 15×1 m plot takes approximately 60% of the time required to search a 25×1 m plot.

6.5 CONCLUSIONS

There is a potential that errors made in identifying scats could lead to bias in estimates of animal abundance. While this result may have partly been an artefact of the experimental design, it suggests that observers may over estimate scat density for species present in low numbers and under estimate scat numbers for species in high numbers. Care should therefore be taken when using and interpreting scat count data for these species.

Where scats are present in a scat plot they are not necessarily always counted. The proportion of scats missed varies between sites. This site-specific variation could potentially influence count data. The observed variation was related in part to the vegetation characteristics of a site. However, it also appears that other, as yet unmeasured factors also play an important role in influencing the probability of finding scats. Fortunately, the error observed in this study was low suggesting that adjustment for site variation is unnecessary.

Up to 80 days there was no appreciable amount of decomposition for Bennett's wallaby scats. This indicates that a sampling interval of 80 days was sufficiently small that scats would not decompose between scat counts.

Of the plot sizes tested, those of dimensions 15×1 m appear to be the optimum size for use in counts of herbivore scats on Tasmanian forestry plantations. They are intermediate in size, and hence search time, yet they sacrifice little in the way of precision to the larger plot sizes.

CHAPTER 7

HERBIVORE ABUNDANCE AND DAMAGE

7.1 INTRODUCTION

One of the most basic questions to address in pest control must be : ‘how is the damage caused by a pest related to its abundance ?’ Essentially all pest control programs assume that damage increases as some function of increasing pest numbers. However, the exact nature of such relationships has rarely been studied, and those studies which exist are divergent in their findings. It appears to be inherently appealing for authors to assume that the relationship between pest abundance and damage is linear (e.g. Southwood and Norton 1973). In other words, each unit increase in pest abundance brings about a directly proportional increase in damage. Linear relationships have certainly been demonstrated between damage to crops and the abundance of a range of pests species including rodents (Poche *et al.* 1982; Lefebvre *et al.* 1989), birds (Crabb *et al.* 1986), pigs (Gorynska 1981; Katahira *et al.* 1993), and deer (Conover 1989; Kay 1993). However, the results of a recent study by Hone (1995b) found that the relationship between pig rooting and pig abundance was best described using a logarithmic model.

Successful quantification of the relationship between pest abundance and damage is extremely advantageous for any manager who is trying to control pest damage. Consider the plight of a manager faced with the prospect of trying to control for pest damage as efficiently as possible. Ideally, pest abundance will be reduced to some critical or ‘target’ level, where the effect of damage is negligible, but no further, because any effort expended to reduce abundance beyond this point would be wasted. However, if control measures are costly, it will be preferable to reduce pest abundance to a point where the costs of control begin to outweigh the benefits

obtained (Braysher 1993). It will clearly be impossible to assign target abundance levels if there is no understanding of the relationship between abundance and damage.

In the absence of detailed relationships many pest managers appear to aim to simply reduce pest abundance, reasoning that any reduction in numbers will bring about some reduction in damage. However, this assumption may be inaccurate. Consider the two hypothetical models presented in Figure 7.1, describing the relationship between pest abundance and damage. Model I is a linear relationship as described by Conover (1989) and Model II is a logarithmic function, as described by Hone (1994, 1995b). Assume that a manager is hoping to achieve target damage levels by using some form of control. The control effort substantially reduces the abundance of the pest species from level 'a' to level 'b'. If the relationship between abundance and damage takes the form of Model I then the reduction in abundance will achieve a damage level slightly below the target. However, if the relationship between abundance and damage takes the form of Model II then, despite the reduction in abundance, damage remains largely unaltered. In effect, the control effort has been wasted. This example is, of course, somewhat simplified and, as Braysher (1993) suggests, the costs of control also need to be considered.

The principle technique used by forestry managers to control for browsing damage in Tasmania is 1080 poisoning. The assumption is that poisoning will reduce the local population density of browsers and subsequently reduce the severity of browsing damage. Unlike other pest/crop systems, where population reduction may need to be ongoing (Putman 1989), establishment of seedlings only requires that they be protected during their first year or two of growth. Despite over 35 years of the use of 1080 as a control mechanism (Mollison 1960), there has been

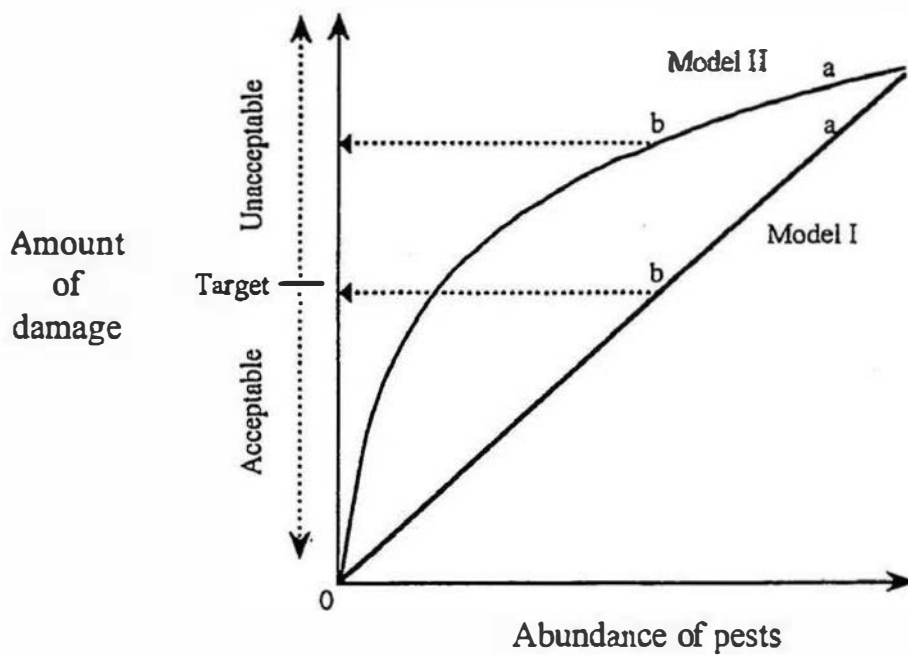


Figure 7.1 : Two theoretical models describing the relationship between pest abundance and pest damage. Model I has been generated from a linear function. Model II has been generated from a logarithmic function. Pest abundance prior to control is marked with the letter 'a'. Pest abundance following control is marked with the letter 'b'. The target damage level is marked on the vertical axis.

essentially no study of the relationships between browser abundance and damage levels. Furthermore, very little work has been carried out to examine the effects of 1080 on browser populations, and assess its effectiveness as a control technique. In this chapter, a study is outlined which aimed to determine :

- i) what relationships, if any, exist between herbivore abundance and browsing damage severity; and
- ii) what effects 1080 poisoning has on herbivore populations.

7.2 METHODS

7.2.1 Field sites

The 35 plantations which were used during this study are described in Chapter 2. Data were collected from 13 of these plantations during 1994-95, 15 plantations during 1995-96 and 7 plantations during 1996-97 (Table 7.1).

7.2.2 Techniques

Measuring damage

The damage data for each plantation, and the method by which they were collected were described in Chapters 2 and 3.

Measuring herbivore abundance

Several methods have been used to study trends in populations of vertebrate herbivores in Tasmania. These include harvest/catch-effort methods (Johnson 1977; Driessen 1992), examination of historical records (Johnson 1977; Barker and Caughley 1990), spotlight counts from transects (Johnson 1977; Gregory 1988), stake-out counts (Johnson 1977; Gregory 1988) and faecal pellet counts (Johnson

Table 7.1 : The year during which data collection commenced at each plantation, the area of each plantation, number of scat plots and calculated sampling intensity (plots.ha⁻¹).

Plantation	Year	Area(ha)	Plots	Plots.ba ₁	Plantation	Year	Area(ha)	Plots	Plots.ba ₁
Bswan	1995	18	16	0.89	Bswan 2	1996	38	25	0.66
Adu29	1994	24	20	0.83	Bwal	1994	21	20	0.95
Alo48	1996	55	25	0.45	Bwee	1995	43	25	0.58
Alo49	1996	12	10	0.83	Bwfr	1995	11	9	0.82
Amark	1996	24	19	0.79	Fba21h	1996	29	25	0.86
Aseol	1996	17	15	0.88	Fhp23g	1995	38	25	0.66
Asx13	1994	16	14	0.88	Fkd13b	1995	52	25	0.48
Asx51	1995	27	25	0.93	Fsolb	1995	64	25	0.39
Atn18	1995	15	18	1.20	Nalans	1996	110	25	0.23
Atn20	1995	30	25	0.83	Nbless	1995	41	25	0.61
Aww55	1995	80	25	0.31	Ncw	1994	43	25	0.58
Brin	1994	14	12	0.86	Ndig	1994	17	16	0.94
Bcam	1995	13	11	0.85	Npear	1994	35	25	0.71
Bsch	1995	41	25	0.61	Nped	1995	20	18	0.90
Bfr	1994	10	7	0.70	Npu	1994	39	25	0.64
Bkui	1995	13	10	0.77	Nrit	1994	24	25	1.04
Brfl	1994	10	10	1.00	Nspill	1994	28	25	0.89
Brf2	1994	13	10	0.77					

1977; Statham 1983; Gregory 1988). During this study it was necessary to gain at least a relative measure of animal abundance across a large number of sites. The only logistically viable method available to achieve this was that of faecal pellet, or scat, counts (for a more detailed discussion refer to Chapter 6).

Scat plots were established at each of the 35 forestry plantations immediately after seedlings were planted. The number of plots placed within a plantation depended on the size of the plantation, ranging from 7 at the smallest (Bfr) up to 25 at the largest (Table 7.1). An attempt was made to keep sampling intensity as constant as possible (around one plot per hectare). However, 25 was the maximum number of plots ever placed on a plantation, as this was considered the most that could be covered by two people in one day. As a result, sampling intensity ranged from 0.23 to 1.20 plots.ha⁻¹. To facilitate finding the plots during subsequent sampling, they were placed out on a plantation according to randomly chosen points of intersection on a one hectare grid overlay. In the smaller plantations, where sampling intensity was around 1 plot per hectare, all intersection points could end up being selected.

Scat plots used during the 1994-95 sampling period were 25 metres long and 1 metre wide. Following experimental work, which revealed that smaller plots could be used without altering density estimates or sacrificing precision (Chapter 6), plot size was reduced to 15 × 1 m for the 1995-96 and 1996-97 sampling periods. A plot was delineated by stretching a tape measure between two wooden pegs placed at the ends of the plot. An observer would move slowly from one end of a plot to the other with a one metre ruler abutting the tape measure. The ground immediately in front of the ruler would be searched for herbivore scats and any found would be identified, counted, and removed from the plot. Individual pellets were counted as opposed to pellet groups (Johnson 1977). If a scat could be identified as coming from a macropod species, but the exact species of origin was

uncertain, the scat was categorised as an ‘unknown macropod’. All plots were cleared when they were established and the number of scats recorded. The scats in each plot were then counted and removed from each plot at 80 day intervals until the study was completed after approximately 320 days (four sampling periods). For each sampling period the rate of scat deposition was calculated by dividing the number of scats which were found by the area of the plots surveyed (m²) and the number of days separating sampling periods :

$$\text{Deposition rate} = \frac{(\text{number of scats found})}{(\text{area of plots}) \times (\text{days between sampling})}$$

As a means of quantifying the average abundance of herbivores on each plantation over time, average scat deposition rate was calculated using the following formula :

$$\text{Average scat deposition rate} = \frac{x_1 + x_2 + x_3 + x_4}{4}$$

where ‘ x_1 ’ is the deposition rate calculated for the 0-80 days after planting sampling period, ‘ x_2 ’ is the deposition rate for the 81-160 day period, ‘ x_3 ’ is the deposition rate for the 161-240 day period and ‘ x_4 ’ is the deposition rate for the 241-320 day period.

7.2.3 Statistics

Data from all 35 forestry plantations (32 planted with *E. nitens* and 3 planted *E. globulus*) were used for the analyses examining trends in scat deposition rates. Twenty-six of these plantations were poisoned with 1080 and the remaining nine were not poisoned (see Chapter 2 for a full description of each plantation). Data from the 32 *E. nitens* plantations were used in the analyses comparing damage severity to average scat deposition rates. A multiple stepwise regression model (Tabachnik and Fidell 1989) was used to model browsing damage (dependent variable) as a function of the average scat deposition rate for each herbivore species

(independent variables). The analysis was performed using the SAS statistics package (SAS Institute 1989a,b). A forward selection procedure was used with the significance level for incorporating variables into the model set at $\alpha < 0.05$.

7.3 RESULTS

7.3.1 Browsing damage

The browsing damage data collected for each plantation has already been described in detail in Chapter 3.

7.3.2 The scats that were found

Scats from five herbivore species were found during this study. These were the Bennett's wallaby, the Tasmanian pademelon, the common brushtail possum, the common wombat and the introduced European rabbit. In Table 7.2 the density of scats found at each plantation at the time of plot establishment is shown for each of these species. Data are also presented for the total for macropods (macropod) which incorporates scats from the Bennett's wallaby, pademelon and those macropod scats which could not be identified to species level (Chapter 6). Table 7.3 shows the average scat deposition rates (see section 7.2.2) calculated for each plantation from the data collected at the four sampling periods following plot establishment.

In Figure 7.2 the frequency of occurrence of scats, expressed as a percentage of the total for scats found from all species, is shown for each species and species group. Scats from the Bennett's wallaby were most frequently found, accounting for 47%

Table 7.2 : The density (scats.m⁻²) of scats found in plots at the time of plot establishment. Data is given for each species and the total for macropod scats (Macropod) incorporating Bennett's wallaby, pademelon and macropod scats which could not be identified to species level.

Plantation	Bennett's	pademelon	macropod	possum	rabbit	wombat
Adu29	0.000	0.022	0.022	0.000	0.000	0.000
Alo48	0.563	0.545	1.219	0.000	0.000	0.002
Alo49	0.789	0.323	1.223	0.000	0.000	0.000
Amark	0.000	0.012	0.013	0.003	0.000	0.005
Aseol	0.126	0.623	0.824	0.001	0.000	0.000
Asx13	0.232	0.323	0.568	0.013	0.000	0.000
Asx51	0.000	0.000	0.005	0.005	0.000	0.000
Atn18	0.018	0.049	0.071	0.000	0.000	0.000
Atn20	0.000	0.003	0.007	0.000	0.000	0.000
Awv55	0.528	1.045	1.856	0.000	0.011	0.000
Bcam	0.000	0.000	0.000	0.000	0.000	0.000
Bfr	0.013	0.011	0.024	0.000	0.008	0.000
Brf1	0.076	0.056	0.140	0.000	0.004	0.000
Brf2	0.000	0.008	0.008	0.000	0.000	0.000
Brin	0.043	0.065	0.117	0.000	0.000	0.000
Bsch	0.005	0.008	0.013	0.000	0.003	0.000
Bswan	0.853	0.053	0.938	0.004	0.021	0.031
Bswan2	0.321	0.203	0.576	0.003	0.000	0.000
Bwa	0.210	0.408	0.636	0.008	0.021	0.000
Bwee	0.167	0.000	0.205	0.005	0.029	0.000
Bwfr	0.053	0.220	0.353	0.000	0.007	0.047
Fha21h	0.512	0.123	0.635	0.000	0.000	0.000
Fhp23g	0.000	0.000	0.000	0.000	0.000	0.000
Fkd13b	0.019	0.013	0.035	0.000	0.008	0.000
Fsolb	0.000	0.000	0.003	0.005	0.000	0.000
Nalans	0.073	0.059	0.146	0.000	0.000	0.001
Nbless	0.056	0.027	0.085	0.005	0.003	0.000
Ncw	2.246	1.442	3.806	0.104	0.010	0.003
Ndig	0.113	0.105	0.238	0.003	0.008	0.000
Npear	0.773	0.582	1.451	0.060	0.000	0.000
Nped	0.040	0.027	0.073	0.060	0.002	0.044
Npu	0.405	0.211	0.636	0.035	0.007	0.000
Nrit	0.142	0.062	0.212	0.000	0.023	0.000
Nsp	0.016	0.004	0.020	0.018	0.007	0.000

Table 7.3 : Average scat deposition rate (scats.m⁻².day⁻¹) calculated for each plantation, each species and the ‘macropod’ grouping incorporating Bennett’s wallaby, pademelon and those macropod scats which could not be identified to species level. Correlation coefficients (r) are given for comparisons between the density of scats found for each species at plot establishment (Table 7.2) and the average scat deposition rate calculated for each species (df=34, p>0.05).

Plantation	Bennett's	pademelon	macropod	possum	rabbit	wombat
Adu29	0.0022	0.0020	0.0045	0.0001	0.0000	0.0001
Alo48	0.0050	0.0007	0.0057	0.0000	0.0000	0.0003
Alo49	0.0033	0.0096	0.0106	0.0012	0.0000	0.0002
Amark	0.0019	0.0005	0.0011	0.0000	0.0000	0.0011
Aseol	0.0119	0.0018	0.0136	0.0013	0.0000	0.0000
Asx13	0.0083	0.0096	0.0187	0.0007	0.0000	0.0000
Asx51	0.0023	0.0031	0.0064	0.0000	0.0000	0.0000
Am18	0.0005	0.0031	0.0042	0.0000	0.0001	0.0000
Am20	0.0037	0.0015	0.0048	0.0005	0.0000	0.0000
Aww55	0.0023	0.0047	0.0078	0.0001	0.0001	0.0000
Bcam	0.0111	0.0044	0.0177	0.0001	0.0000	0.0004
Bfr	0.0020	0.0022	0.0067	0.0000	0.0001	0.0000
Bkui	0.0017	0.0005	0.0020	0.0000	0.0000	0.0004
Brf1	0.0112	0.0099	0.0217	0.0006	0.0000	0.0005
Brf2	0.0053	0.0038	0.0081	0.0004	0.0001	0.0001
Brin	0.0037	0.0072	0.0088	0.0008	0.0001	0.0002
Bsch	0.0118	0.0115	0.0266	0.0002	0.0002	0.0000
Bswan	0.0163	0.0035	0.0216	0.0003	0.0004	0.0010
Bswan2	0.0029	0.0004	0.0032	0.0013	0.0000	0.0000
Bwa	0.0040	0.0067	0.0108	0.0001	0.0002	0.0000
Bwee	0.0027	0.0034	0.0111	0.0004	0.0002	0.0000
Bwfr	0.0013	0.0012	0.0033	0.0001	0.0000	0.0007
Fha21h	0.0035	0.0001	0.0005	0.0004	0.0000	0.0000
Fhp23g	0.0000	0.0000	0.0000	0.0000	0.0001	0.0000
Fkd13b	0.0015	0.0040	0.0064	0.0003	0.0001	0.0000
Fsolb	0.0001	0.0002	0.0003	0.0000	0.0000	0.0000
Nalans	0.0011	0.0002	0.0013	0.0004	0.0000	0.0008
Nbless	0.0041	0.0022	0.0076	0.0006	0.0001	0.0002
Ncw	0.0115	0.0049	0.0233	0.0009	0.0003	0.0001
Ndig	0.0037	0.0033	0.0073	0.0002	0.0001	0.0006
Npear	0.0016	0.0010	0.0052	0.0010	0.0002	0.0004
Nped	0.0050	0.0020	0.0076	0.0000	0.0001	0.0000
Npu	0.0140	0.0074	0.0209	0.0009	0.0002	0.0004
Nrit	0.0025	0.0011	0.0035	0.0006	0.0002	0.0000
Nsp	0.0001	0.0001	0.0002	0.0001	0.0002	0.0001
r =	0.078	0.023	0.092	0.141	0.393	0.042

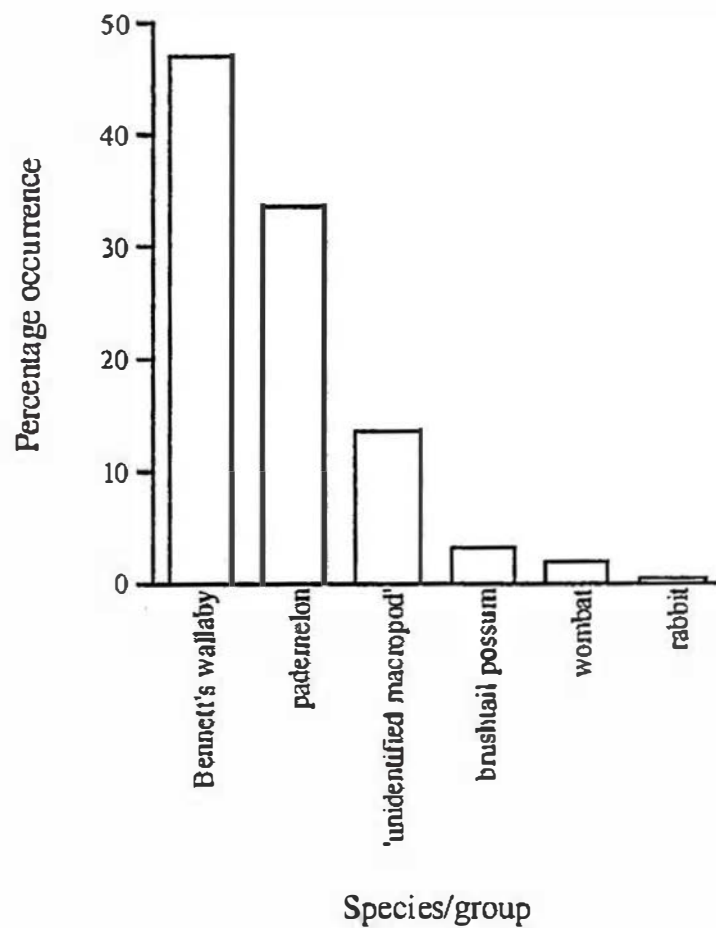


Figure 7.2 : The frequency of occurrence of scats from five herbivore species and macropod scats which could not be identified to species level ('unidentified macropod') presented as a percentage of the total number of scats found at all sites and over all sampling periods.

of all scats. This was followed by scats from pademelons (34%) and macropod scats which could not be identified to species level (14%). Scats from the remaining species comprised only 6% of the total scats found. For all species, there was no significant relationship observed between the density of scats recorded in plots at the time of establishment and the average rate of scat deposition recorded over 320 days (Table 7.3).

7.3.3 Temporal trends in scat deposition

Figures 7.3 to 7.8 show the mean values for the scat deposition rates recorded in each sampling period for plantations poisoned with 1080 and plantations where no poison was used. Data are presented for the Bennett's wallaby, pademelon, macropod grouping (incorporating Bennett's wallaby, pademelons and 'unknown macropod' scats), brushtail possum, rabbit and wombat.

For the Bennett's wallaby (Figure 7.3), there was little difference between poisoned and unpoisoned plantations in mean values for scat deposition rates at either 80 or 160 days after plot establishment. Following this, however, scat deposition rates at non-poisoned plantations climbed steeply to 320 days while scat deposition rates at poisoned plantations increased only gradually and appeared to plateau by 320 days. One-tailed t-tests indicate that there were no significant differences in the mean of scat deposition rates between unpoisoned and poisoned plantations at 80 ($t=1.820$, $df=33$, $p>0.05$), 160 ($t=0.502$, $df=33$, $p>0.05$) or 240 ($t=1.199$, $df=33$, $p>0.05$) days after planting. There was a highly significant difference at 320 days ($t=3.975$, $df=33$, $p<0.01$).

For the pademelon (Figure 7.4), the mean values for scat deposition rates recorded at unpoisoned plantations increased quite quickly to 320 days, while the increase at poisoned plantations was more gradual, with an apparent flattening out by 320 days. One-tailed t-tests indicate that there were significant differences in the mean

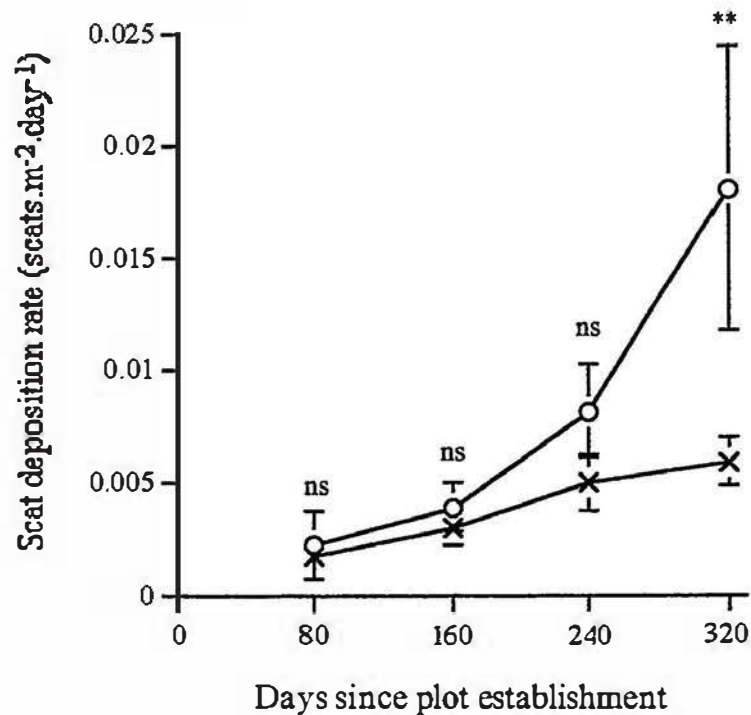


Figure 7.3 : Line plot showing the mean (\pm se) Bennett's wallaby scat deposition rate for plantations where poisoning operations were carried out ($n=28$) and plantations where poisoning was not carried out ($n=7$). Data are given for each of four sampling periods separated by 80 days (time 0 = planting). Non-significant differences ($p>0.05$) are marked with 'ns'. Highly significant differences ($p<0.01$) are marked with '**'.

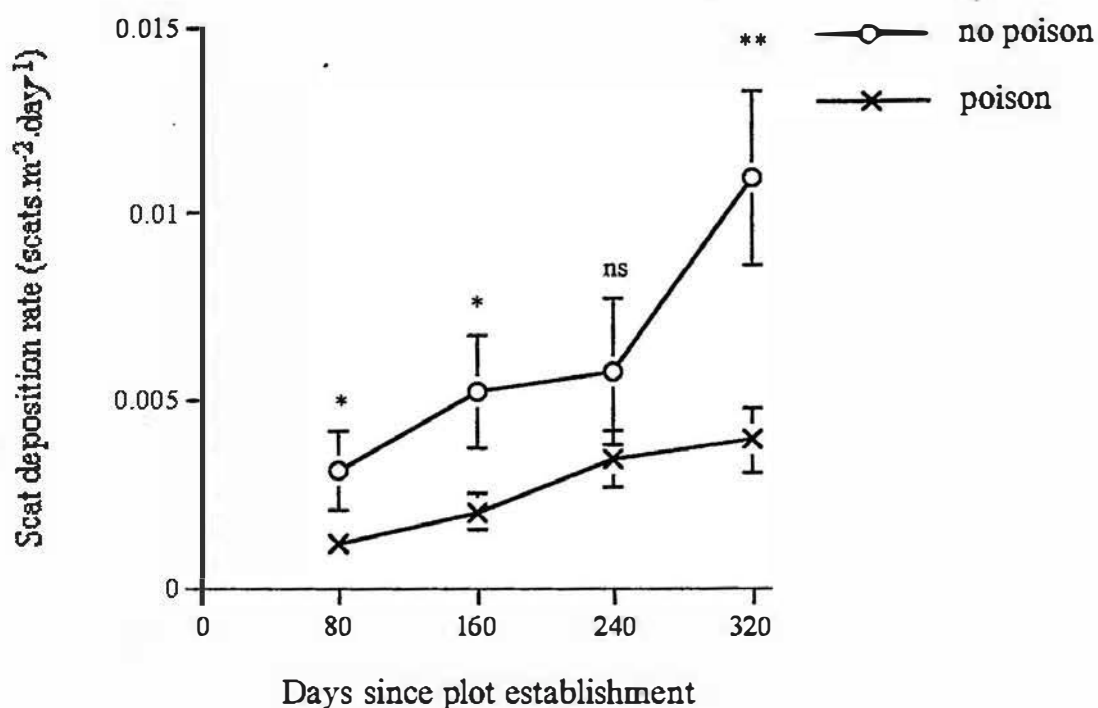


Figure 7.4 : Line plot showing the mean (\pm se) Tasmanian pademelon scat deposition rate for both plantations where poisoning operations were carried out ($n=28$) and plantations where poisoning was not carried out ($n=7$). Data are given for each of four sampling periods separated by 80 days (time 0 = planting). Non-significant differences are marked with 'ns', significant differences ($p<0.05$) with '*', and highly significant differences ($p<0.01$) with '**'.

of scat deposition rates between unpoisoned and poisoned plantations at 80 ($t=3.917$, $df=33$, $p<0.01$), 160 ($t=2.628$, $df=33$, $p<0.05$), and 320 ($t=3.399$, $df=33$, $p<0.01$) days after planting. There was no significant difference at 240 days ($t=1.288$, $df=33$, $p>0.05$).

For the combined macropod data (Figure 7.5), the trend for the poisoned plantations was very similar to that observed for the Bennett's wallaby and pademelon, with a gradual increase in mean values for scat deposition rates to 240 days and then a flattening out to 320 days. For the unpoisoned plantations the mean values for scat deposition rates increased gradually to 160 days and then increased rapidly to 320 days. Throughout, the mean of deposition rates for poisoned plantations were lower than that for unpoisoned plantations. One-tailed t-tests indicate that this difference was not significant at 80 ($t=1.070$, $df=33$, $p>0.05$) or 160 ($t=1.449$, $df=33$, $p>0.05$) days after planting. There was, however, a significant difference at 240 ($t=1.907$, $df=33$, $p<0.05$) and 320 days ($t=4.589$, $df=33$, $p<0.01$).

For the brushtail possum (Figure 7.6), there was quite a marked difference in trends between poisoned and unpoisoned plantations. At unpoisoned plantations the mean value for scat deposition rates appeared stable to 160 days and then there was a rapid increase to 320 days. At the poisoned plantations the mean values for scat deposition rates increased slightly from 80 to 160 days where it reached a plateau until 240 days and then declined to 320 days. One-tailed t-tests indicate that there was no significant difference in the mean values for scat deposition rates between unpoisoned and poisoned plantations at 80 ($t=1.490$, $df=33$, $p>0.05$), 160 ($t=0.296$, $df=33$, $p>0.05$) or 240 ($t=1.203$, $df=33$, $p>0.05$) days after planting. There was, however, a highly significant difference at 320 days ($t=2.503$, $df=33$, $p<0.01$).

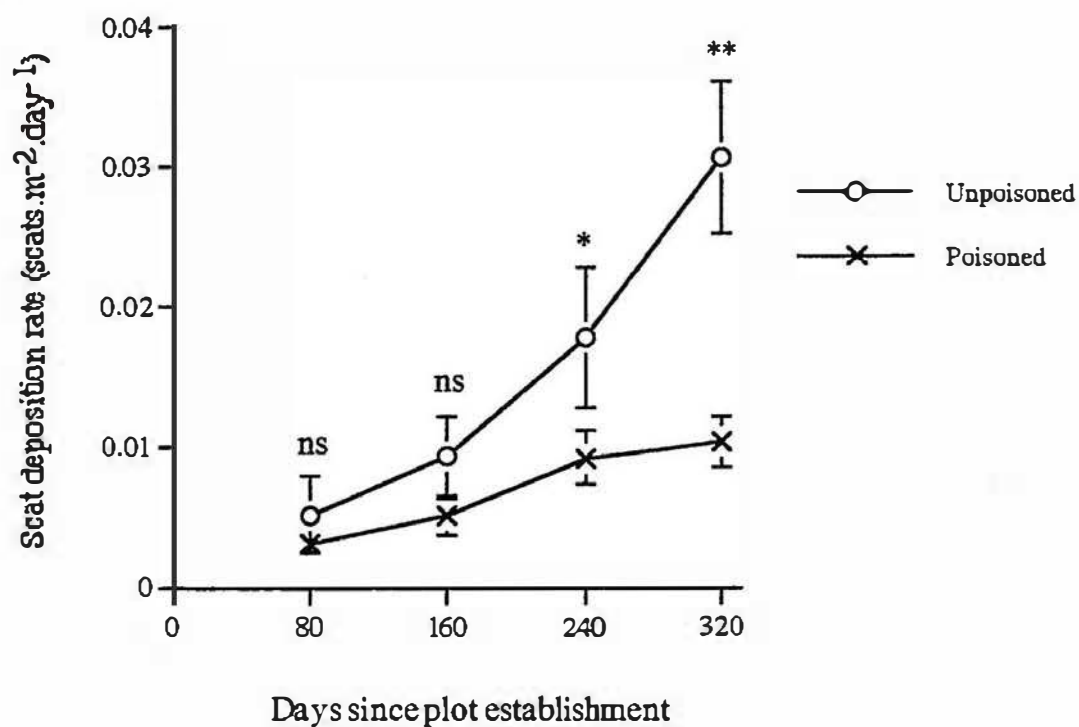


Figure 7.5 : Line plot showing the mean(\pm se) macropod scat deposition rate for plantations where poisoning operations were carried out ($n=28$) and plantations where poisoning was not carried out ($n=7$). Data are given for each of four sampling periods separated by 80 days (time 0 = planting). Non-significant differences ($p>0.05$) are marked with 'ns' and highly significant differences ($p<0.01$) with '**'.

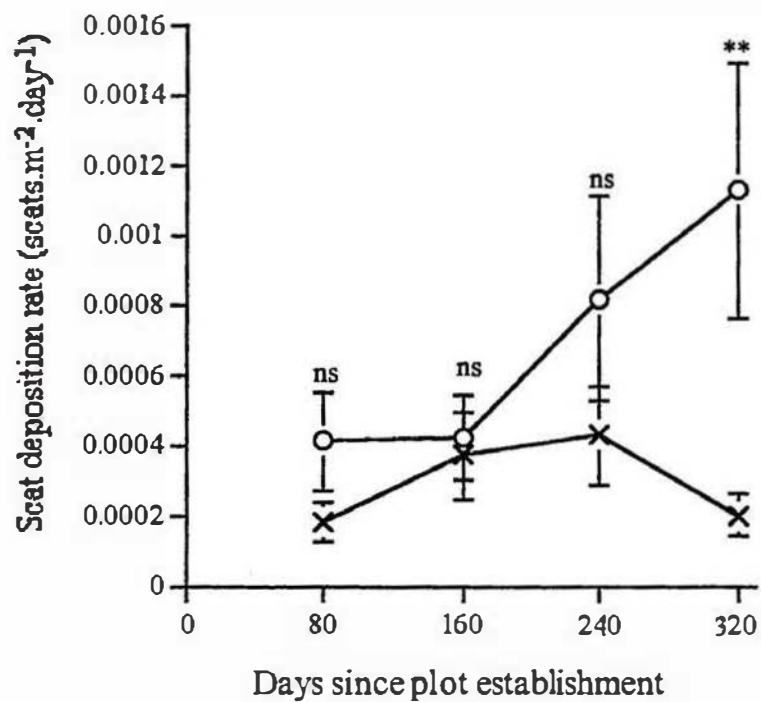


Figure 7.6 : Line plot showing the mean (\pm se) brushtail possum scat deposition rate for plantations where poisoning operations were carried out ($n=28$) and plantations where poisoning was not carried out ($n=7$). Data are given for each of four sampling periods separated by 80 days (time0 = planting). Non-significant differences ($p>0.05$) are marked with 'ns' and highly significant differences ($p<0.01$) with '**'.

For the rabbit (Figure 7.7), remarkably similar trends were observed between poisoned and unpoisoned plantations. The mean values for scat deposition rates declined slightly from 80 to 160 days. Following this, there was quite a rapid increase to 320 days. One-tailed t-tests indicate that there was no significant difference in the mean value for deposition rates between unpoisoned and poisoned plantations at any sampling period (80 days : $t=1.546$, $df=33$, $p>0.05$; 160 days : $t=1.438$, $df=33$, $p>0.05$; 240 days : $t=0.499$, $df=33$, $p>0.05$; 320 days : $t=1.628$, $df=33$, $p>0.05$).

There was considerable variation in the data for the wombat (Figure 7.8) making commenting on trends difficult. For the unpoisoned plantations there appeared to be quite a rapid increase in scat deposition rate to 240 days followed by a levelling out at 320 days. For the poisoned plantations the mean values for scat deposition rates fluctuated considerably and this, coupled with the large variation around the mean, reduce the value of any comparisons in temporal trends between poisoned and unpoisoned plantations. One-tailed t-tests indicate that there was no significant difference in mean deposition rates between unpoisoned and poisoned plantations at any sampling period (80 days : $t=0.573$, $df=33$, $p>0.05$; 160 days : $t=0.746$, $df=33$, $p>0.05$; 240 days : $t=0.810$, $df=33$, $p>0.05$; 320 days : $t=0.325$, $df=33$, $p>0.05$).

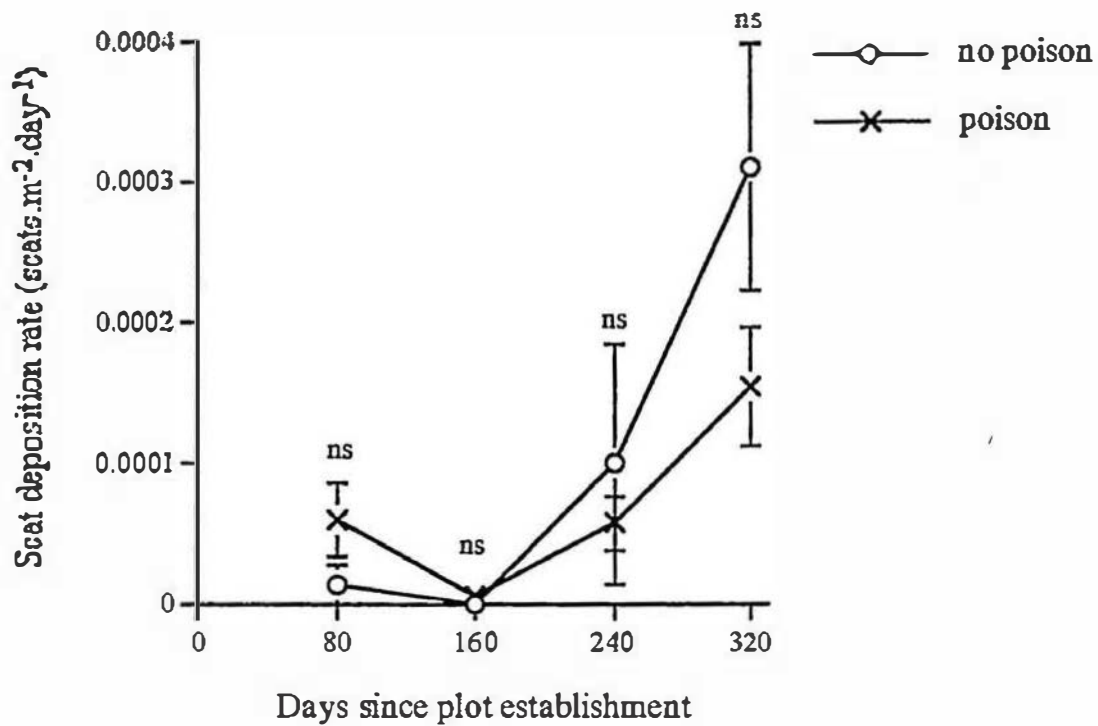


Figure 7.7 : Line plot showing the mean (\pm se) rabbit scat deposition rate for plantations where poisoning operations were carried out ($n=28$) and plantations where poisoning was not carried out ($n=7$). Data are given for each of four sampling periods separated by 80 days (time 0 = planting). Non-significant differences ($p>0.05$) are marked with 'ns'.

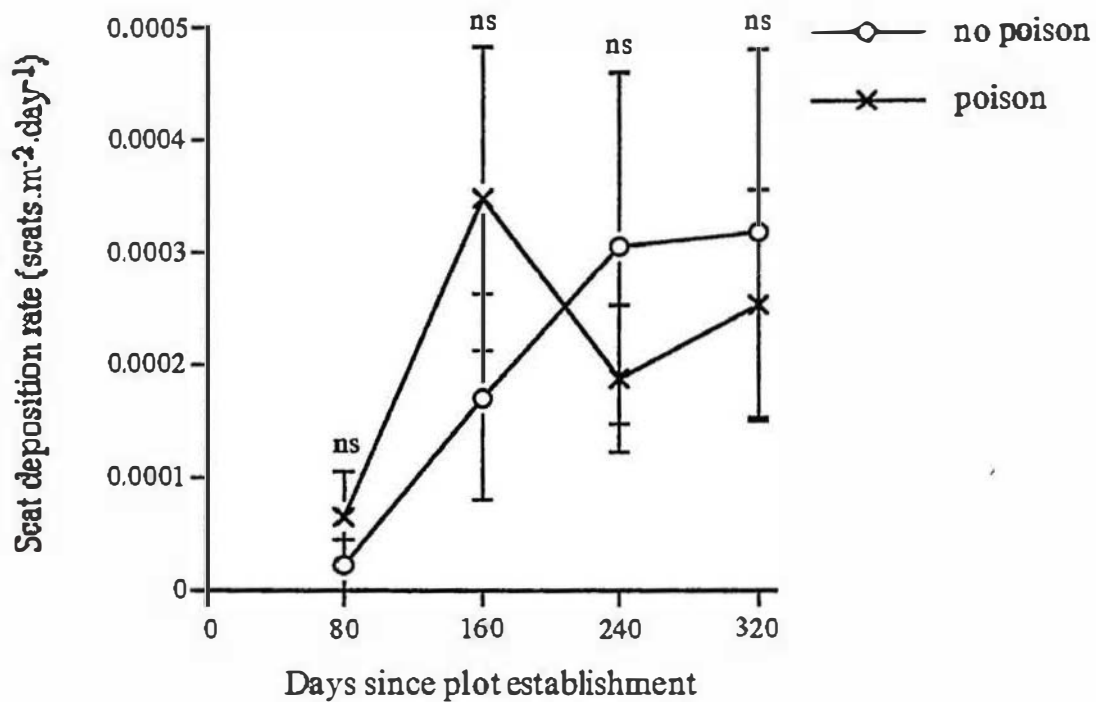


Figure 7.8 : Line plot showing the mean (\pm se) wombat scat deposition rate for plantations where poisoning operations were carried out ($n=28$) and plantations where poisoning was not carried out ($n=7$). Data are given for each of four sampling periods separated by 80 days (time 0 = planting). Non-significant differences ($p>0.05$) are marked with 'ns'.

7.3.4 Scat deposition rate and damage

Figures 7.9 to 7.13 are scatter-plots of mean browse scores recorded at 12 months (Table 3.3) as a function of the average scat deposition rate recorded at each plantation (Table 7.3) for each herbivore species except the wombat. Wombats were excluded from further analyses as they are not considered to browse eucalypt seedlings (Statham 1983). Regression analyses were carried out using linear and curvilinear (logarithmic, exponential, power and polynomial) models. Regression lines are shown where such analysis indicated that a significant relationship existed.

No relationship was observed between mean browse score at 12 months and average scat deposition rate for the Bennett's wallaby (Figure 7.9 : $r^2=0.090$, $df=31$, $p>0.05$), pademelon (Figure 7.10 : $r^2=0.063$, $df=31$, $p>0.05$), macropod grouping (Figure 7.11 : $r^2=0.070$, $df=31$, $p>0.05$) or the rabbit (Figure 7.13 : $r^2=0.004$, $df=31$, $p>0.05$). However, there was a highly significant linear relationship between mean browse score and average scat deposition rate for the brushtail possum (Figure 7.12 : $r^2=0.537$, $df=31$, $p<0.01$). This relationship is described by the following equation :

$$y = 2005.1x + 0.571$$

Equation 7.1

where 'y' is mean browse score at 12 months and 'x' is the average scat deposition rate for the brushtail possum.

Because it is possible that browsing damage severity may be a function of the abundance of not just one, but a combination of herbivorous species, a stepwise multiple regression was carried out comparing damage levels to the average scat deposition rates for all species. No variable other than average scat deposition rate for the brushtail possum met the selection criterion ($p<0.05$) for entry into the

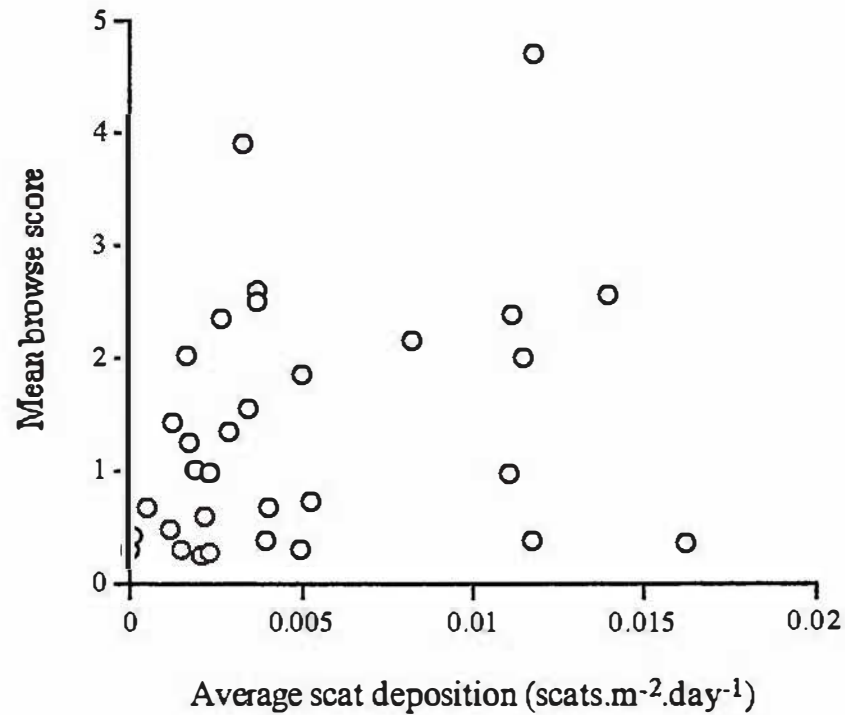


Figure 7.9 : Scatter-plot showing the mean browse score recorded at 12 months after planting for 32 *E. nitens* plantations as a function of average scat deposition rate for the Bennett's wallaby (scats.m⁻².day⁻¹) at each plantation.

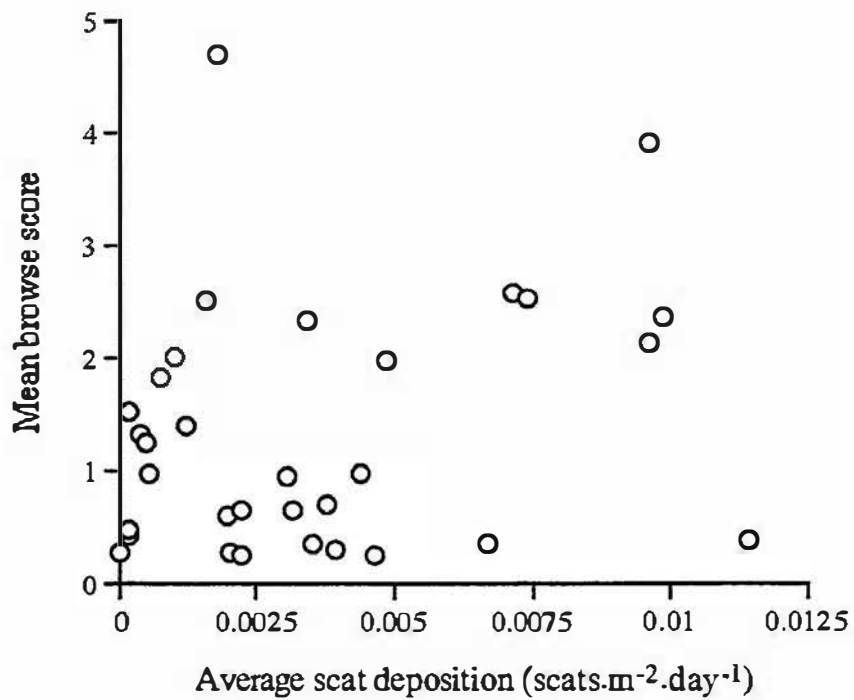


Figure 7.10 : Scatter-plot showing the mean browse score recorded at 12 months after planting for 32 *E. nitens* plantations as a function of average scat deposition rate for the pademelon (scats.m⁻².day⁻¹) at each plantation.

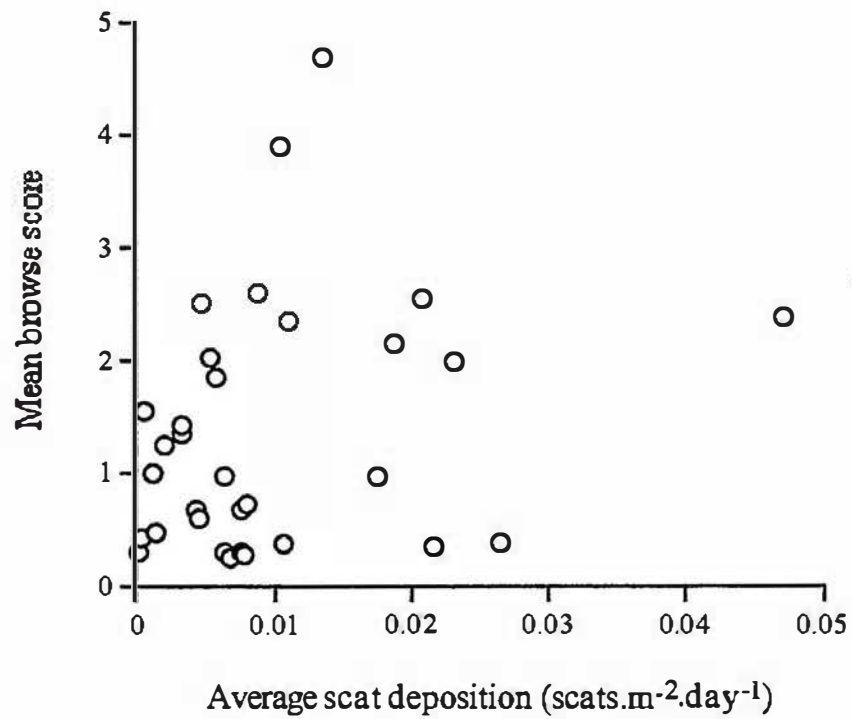


Figure 7.11 : Scatter-plot showing the mean browse score recorded at 12 months after planting for 32 *E. nitens* plantations as a function of the average scat deposition rate for macropods (scats.m⁻².day⁻¹) at each plantation.

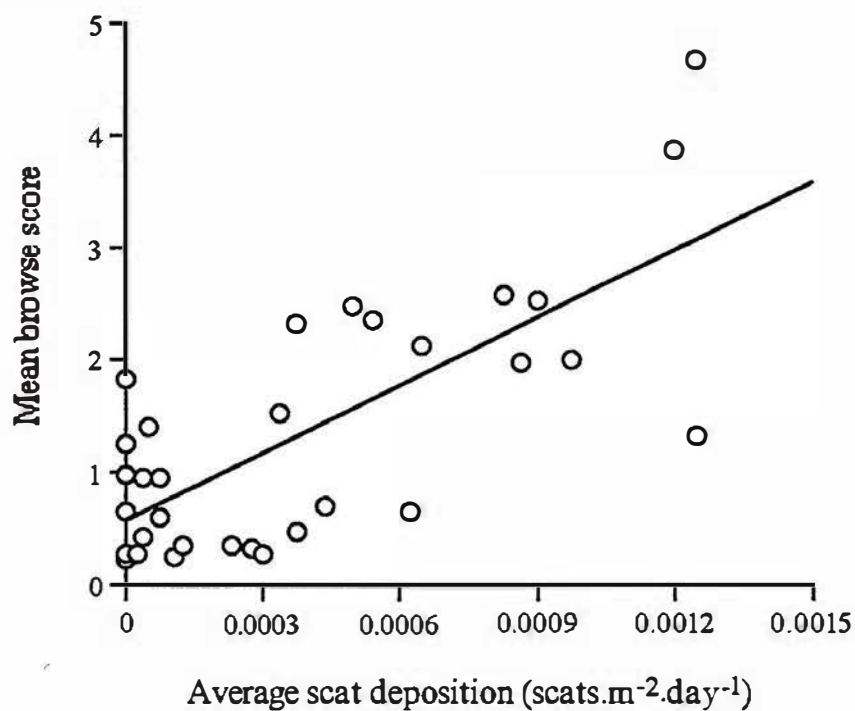


Figure 7.12: Scatter-plot showing the mean browse score recorded at 12 months after planting for 32 *E. nitens* plantations as a function of the average scat deposition rate for the brush-tail possum (scats.m⁻².day⁻¹) at each plantation. A linear regression has been applied to the data.

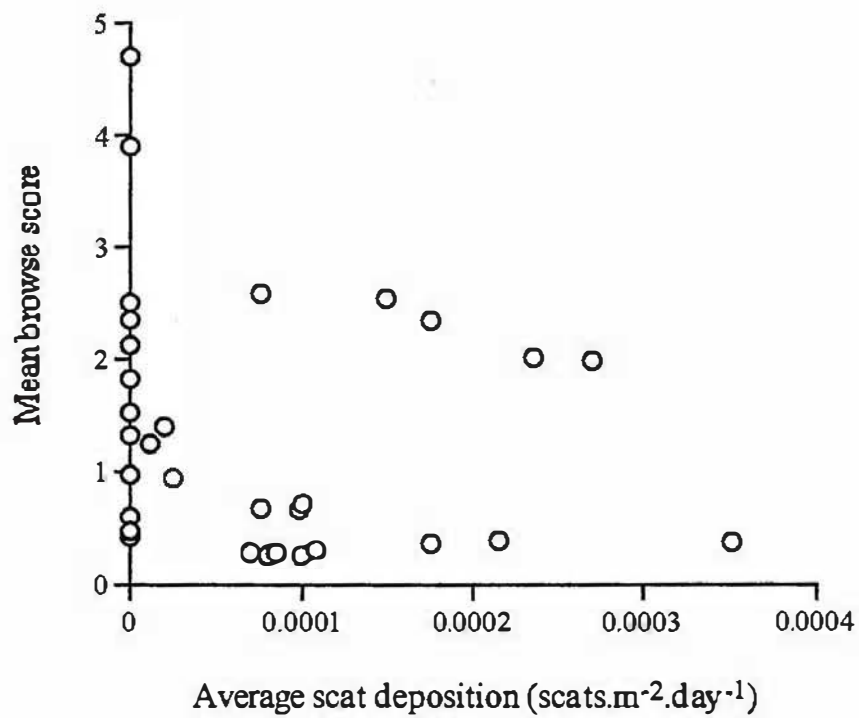


Figure 7.13 : Scatter-plot showing the mean browse score recorded at 12 months after planting for 32 *E. nitens* plantations as a function of the average scat deposition rate for the rabbit (scats.m⁻².day⁻¹) at each plantation.

model. Hence, a multivariate analysis offered no significant improvement on the simple linear model presented above.

Simple linear regressions were used to compare mean browse score at 12 months to the possum scat density recorded at the time of plot establishment and the scat deposition rate recorded at 80 days after establishment. No significant relationship was observed for the former but a highly significant relationship was observed between damage and the possum scat density recorded at 80 days after establishment ($r^2=0.359$, $df=31$, $p<0.01$). This relationship is described by the equation :

$$y = 2038.3x + 0.907$$

Equation 7.2

where 'y' is mean browse score and 'x' is possum scat deposition rate at 80 days.

7.4 DISCUSSION

7.4.1 Relationships between pest abundance and damage

Several models have been proposed to explain the relationships between a pest species' abundance and the level of damage they cause. These include linear models (e.g. Southwood and Norton 1973; Conover 1989; Braysher 1993) and curvilinear models such as the logarithmic and exponential (e.g. Cherrett *et al.* 1971; Braysher 1993; Hone 1994, 1995b). In this study, a significant, positive, linear relationship was observed between damage severity and average scat deposition rate for the brushtail possum ($r^2=0.537$, $df=31$, $p<0.01$). No improvement on this model was achieved by using a curve to describe the relationship.

Similar relationships between herbivore scat count data and browsing damage to plants have been reported by other authors. Conover (1989) examined the factors which influenced the severity of browsing damage to Japanese yew saplings

(*Taxus cuspidata*) planted in Connecticut nurseries. He observed a significant linear relationship between the number of shoots browsed and deer pellet group density. Similarly, Kay (1993) found that damage to regrowth in English woodland blocks was linearly related to deer pellet density. Assuming that scat count data reflect herbivore abundance, these studies suggest that linear relationships might be common between the abundance of herbivores and severity of browsing damage to woody commercial plant species. However, strong conclusions cannot be made based on the small amount of data available.

This discussion will be further expanded in Chapter 9.

7.4.2 Inferences for the relative impact of browsing species

Three native herbivores are considered to be the principle species responsible for causing damage to eucalypt seedlings. These are the Bennett's wallaby, the Tasmanian pademelon and the brushtail possum (Mollison 1960; Gilbert 1961; Cremer 1969; Statham 1983). Additionally, the introduced European rabbit has been considered a problem by some authors (Mollison 1960; Statham 1983; Coleman *et al.* 1997). There has been little work carried out to identify the relative importance of these species as causes of browsing damage to eucalypt seedlings. The one study of note is that of Statham (1983) who used a combination of methods including sand tracking, snow tracking, scat counts, gut content analysis and selective exclosures, to assess the relative impact of these species on *Eucalyptus delegatensis* and *P. radiata* seedlings. Statham (1983) concluded that the Bennett's wallaby was the species responsible for causing most of the browsing damage that she observed. The brushtail possum, pademelon and rabbit were also identified as potentially causing damage, but Statham (1983) considered them to play only a minor role.

Following the findings of Statham (1983), the brushtail possum is often considered to be a minor contributor to damage in comparison with the two macropod species (e.g. Coleman *et al.* 1997). However, recent studies carried out by McArthur and Turner (1997a,b) suggest that the brushtail possum may be a more significant contributor than traditionally thought. They studied the feeding preferences of captive brushtail possums and pademelons for a range of farmed and naturally occurring tree species. Their results indicated that the brushtail possum showed considerable preference for the two main Tasmanian plantation species, *E. nitens* and *E. globulus*. Conversely, pademelons consistently preferred *E. nitens* less than most of the other species offered. Consequently, McArthur and Turner (1997b) suggested that under equivalent numbers and conditions possums may cause more damage than pademelons.

The results of this study add some support to this hypothesis. Despite the fact that possum scats were rarely found, there was a significant relationship between the average scat deposition rate for the brushtail possum and the severity of damage recorded at a plantation. However, there was apparently no relationship for any of the other browsing species. Furthermore, no substantial improvement on the possum scat model was obtained by performing a multivariate analysis incorporating the scat data for all species. This suggests that the brushtail possum was a very important contributor to browsing damage. It should be noted though, that these data do not indicate that the possum is the only species to cause damage. Damage was still evident at some plantations even when no possum scats were found.

The hypothesis that the brushtail possum is, per capita, one of the most significant contributors to browsing damage is further supported by consideration of dietary studies carried out in the field. Statham (1983) studied the gut contents of animals shot on, and around, forestry coupes planted with *E. delegatensis*. As

Statham (1983) herself points out, the results of her dietary study provided no support for the suggestion that the Bennett's wallaby were responsible for causing most of the damage. The stomach contents of shot Bennett's wallabies was primarily composed of monocotyledons, particularly grasses. Tough-leaved dicotyledons comprised only 7-15% of the total diet and eucalypt fragments were rarely found. Pademelons incorporated high proportions of soft leaved dicotyledons and grasses in their diet, but shrub and tree species occurred only at low percentages. At most, eucalypt fragments represented only 1% of total stomach contents. Rabbits included a high proportion of soft leaved dicotyledons and grasses in their diet, but only very rarely ate eucalypt foliage.

In contrast, Statham (1983) and a number of other authors (e.g. Fitzgerald 1984; Green 1984; Owen and Norton 1995) have shown that the brushtail possum feeds on a wide range of tree and shrub species, and these often comprise a major component of the total diet. Statham (1983) observed that *E. delegatensis* comprised up to 27% of total stomach contents during winter. Fitzgerald (1984) studied the diets of brushtail possum in wet sclerophyll forests in Tasmania and found that foliage from a single tree species (*Nothofagus cunninghamii*) made up approximately 40% of the diet. Furthermore, in dry sclerophyll forest areas, eucalypt species comprised up to 29% of fragments separated from faecal pellets, a result that is remarkably similar to that of Statham (1983). All of these data would suggest that it may be necessary to review the importance of the contribution that the brushtail possum makes to browsing damage.

7.4.3 Temporal trends in abundance of herbivores and effects of poisoning

At 80 days after planting there was a significant difference between the deposition rate of pademelon scats at poisoned plantations and the deposition rate of

pademelon scats at non-poisoned plantations. Pademelon scat deposition rates remained lower at poisoned plantations through to the end of sampling (320 days). Though scat deposition data were not collected prior to planting, these results suggest that 1080 poisoning resulted in both an initial and long term reduction in pademelon abundance.

Interpretation of results for the Bennett's wallaby and possum is more difficult. For these species there was little difference between poisoned and unpoisoned plantations in average scat deposition rates at 80 days. However, for both species significant differences were evident at 320 days. The suggestion for these species then, is that while 1080 poisoning did not result in an initial decrease in abundance it did reduce the rate at which the abundance of these species within a plantation increased. This seems counter-intuitive. At all of the poisoned plantations surveyed, poison baits were laid at the time of planting. At five plantations poisoning was again carried out several months later. It would be expected then, that lower scat deposition rates would be recorded at poisoned plantations during the early sampling periods (e.g. 80 or 160 days after planting) and that the magnitude of this difference would decrease as time since poisoning increased. Gregory (1988) observed such a trend in his study of the effects of 1080 poisoning on herbivore populations in Tasmanian agricultural areas. He found that poisoning resulted in an initially quite dramatic drop in macropod scat deposition followed by a general increase. At some plantations this 'resurgence' was extremely rapid and populations appeared to re-establish themselves in as little as three months.

The trends observed for the Bennett's wallaby and possum may be due, in part, to temporal variation in the characteristics of plantations. At establishment, plantations are typically quite devoid of vegetation due to the spraying of herbicide and disturbance caused by heavy machinery. Following this, there tends to be an increase in vegetation cover over time. This represents an increase in food

resources and improved cover for predator-shy species. As a consequence the number of individuals which move out from cover habitat to feed in the plantation may increase over time. This would account for the general increase in scat deposition rates which were observed for all species at both poisoned and unpoisoned plantations. Perhaps, then, Bennett's wallaby and possum scat deposition rates increased more rapidly at unpoisoned plantations because there was a larger pool of individuals available at these sites to respond to the changes in plantation characteristics.

7.4.4 Validity of using scat counts as an abundance measure

The discussion of relationships between browser abundance and damage which have been presented above are largely based on the assumption that scat deposition rates represent at least a relative measure of animal abundance. The validity of this assumption could not be tested here. In Chapter 6 several studies were outlined which aimed to isolate and quantify some of the potential biases involved with the scat counting technique. However, true validation can only be achieved through comparison of scat deposition rate data with the absolute density of animals. Such study has rarely been carried out despite the widespread use of scat count techniques (e.g. Bennett *et al.* 1940; Johnson 1977; Statham 1983). The main reason for this is undoubtedly the considerable amount of difficulty associated with obtaining a 'complete' census count of animals.

Fortuitously, one of the few Australian studies (Johnson and Jarman 1987) which have attempted this was carried out with the red-necked wallaby, an *M. rufogriseus* subspecies closely related to the Bennett's wallaby. In their study, Johnson and Jarman (1987) compared density estimates for the red-necked wallaby and eastern grey kangaroo, obtained from scat counts, to the actual numbers of animals in a New South Wales valley area. Actual counts of animals

could be obtained because the authors were able to recognise all of the individuals in the area by sight. For the red-necked wallaby they found that the estimates based on scat counts were very accurate. For the eastern grey kangaroo, however, scat counts tended to yield a slight underestimate of absolute abundance.

Despite the success of Johnson and Jarman's (1987) study, there would appear to be few instances, outside captive situations (e.g. Eberhardt and Van Etten 1956; Downing *et al.* 1965) where a complete census of animals has been achieved. More frequently, authors have sought to validate scat count techniques via comparison with some other means of estimating abundance. In Tasmania, for example, Johnson (1977) compared absolute density estimates for the brushtail possum based on scat counts and rates of faecal output data (Riney 1957, as cited in Johnson 1977), to those obtained from spotlight counts of animals, and found them to be very similar. For the Bennett's wallaby, however, he found that estimates of absolute density were higher when based on scat count data. This result may be attributable to poor faecal output data ($n=2$) used in the calculation of absolute density. Statham (1983) and Gregory (1988) compared scat deposition data for a range of herbivore species to spotlight and transect count data. Generally their results indicate that trends observed in transect count data are similar to that observed in scat count data. Both authors suggest that scat count techniques are the preferred method of census as they provide a number of logistic advantages and are not subject to the sometimes extreme between-sample variation associated with transect counts. Furthermore, Statham (1983) suggests that pellet count techniques may provide a greater sample size for 'shy' species. Though these findings are not conclusive they do strongly suggest that scat count techniques are a valid means of estimating the relative abundance of large Tasmanian herbivores.

7.5 MANAGEMENT IMPLICATIONS

Herbivore abundance and damage

The results presented in this chapter show that there is a relationship between the average scat deposition rate for the brushtail possum at a plantation and the mean browsing score recorded at 12 months. No relationship could be demonstrated for any of the other surveyed species. The suggestion, then, is that the importance of the brushtail possum as a contributor to browsing damage needs to be reassessed.

Technically, by using Equation 7.1, it might now be possible to begin to predict damage severity based on scat count data. Conversely, this relationship might be used to predict the target abundance levels required to achieve a particular level of damage. Consider, for example, the results presented in Chapter 4, which suggested that seedling growth only starts to be significantly reduced by browsing damage when MBS (12 months) exceeds 0.88. Suppose that a manager is attempting to control for browsing at a particular plantation, so that damage severity is restricted to this level or less. By substituting the desired damage value (0.88) into Equation 7.1 the abundance level of possums at which this amount of damage will occur can be determined. In this case, the target possum abundance level, as measured by average scat deposition rate is 1.8×10^{-4} scats.m⁻².day⁻¹. If the manager has data on scat deposition rates for that plantation, control decisions, such as whether to use poison or not, can be based on comparison between target abundance levels and actual levels. If actual levels greatly exceed target levels then control might be deemed necessary. If, on the other hand, actual levels fall below target levels it may not be necessary to carry out any control at all.

In practice the use of this model as a predictive tool is limited for two reasons. Firstly, the model does not explain enough between-site variation for sufficient confidence to be placed in the accuracy of any derived predictions. Secondly, the

model is retrospective in nature. By the time the data required to make predictions concerning damage are collected the damage has already occurred. In effect, it makes more sense to monitor damage directly. In an effort to improve pre-emptive potential, analyses were performed comparing damage data to the scat densities recorded from plots at the time of plot establishment and during the first sampling period (80 days). No relationship was observed for the former but a significant relationship was found for the 80 day scat density data ($r^2=0.359$, $df=31$, $p<0.01$). This relationship is described by Equation 7.2. Once again, the low correlation coefficient for this model indicates that there is considerable scope for improving its predictive power.

Effectiveness of 1080 poisoning

These data add some support to the contention of Statham (1983) and Gregory (1988) that 1080 poisoning is effective at reducing the abundance of pest species. However, it should be noted that differences in mean scat deposition rates between poisoned and unpoisoned plantations, were not significant for all sampling periods. Furthermore, the data suggest that poisoning has little effect on the abundance of rabbits.

CHAPTER 8

FACTORS INFLUENCING HERBIVORE ABUNDANCE AND DAMAGE

8.1 INTRODUCTION

The results of the large scale damage survey described in Chapter 3 showed that browsing damage severity varies from plantation to plantation. Some plantations are only moderately damaged while others are severely damaged. This spatial variation has long been recognised, but there has been no quantitative investigation of the factors that might explain it. The consequence is that managers are largely unable to predict how badly a particular plantation will be damaged.

The inability to predict damage severity has strongly influenced the way in which control operations are carried out in Tasmania. At present, control efforts tend to be applied more or less uniformly across sites, with poisoning operations being carried out in most newly planted coupes at least once. Results presented in Chapter 3 suggest that this strategy may have some weaknesses. Some sites have an inherently low potential for damage and any control effort expended at these sites is unlikely to bring about significant benefit (Chapter 5). Conversely, some sites appear to be exceptionally prone to damage and it is highly desirable that control effort be concentrated on these sites.

Existing control strategies could be greatly improved if those factors influencing damage severity could be isolated. If, by measuring some key variables, it could be predicted how badly a particular site was going to be damaged, poisoning operations could be restricted to sites where predicted damage exceeded an acceptable level (Chapters 4 and 5). By reducing the frequency of poisoning, the monetary costs and environmental impact of control operations would be reduced. Furthermore, the viability of existing non-lethal control strategies might also be

improved. Electric fences, for example, are successful at preventing herbivores from accessing seedlings (Gregory 1988; Statham 1994), but they are rarely used because they are expensive and can be difficult to maintain (Coleman *et al.* 1997). However, if it can be predicted which sites have a low potential for damage, they can be excluded from control operations, making more resources available to carry out control at 'problem' sites.

The degree to which seedlings within a plantation are damaged must, in part, be a function of :

- i) the number of herbivores which enter the plantation;
- ii) the length of time that each of these individuals spends within the plantation;
- iii) the feeding behaviour of each individual and;
- iv) the environment, through its effects on all of the above (Figure 8.1).

In this chapter a study is outlined which aimed to identify those environmental variables and site characteristics which affect herbivore abundance and damage severity. Some preliminary work is detailed, examining the scope for developing models for use in predicting the degree to which a particular site will be damaged.

8.2 METHODS

8.2.1 Study sites

Data were collected from 35 geographically distinct forestry plantations. These are described in detail in Chapter 2. Further descriptive characteristics of each site are contained in Appendix B.

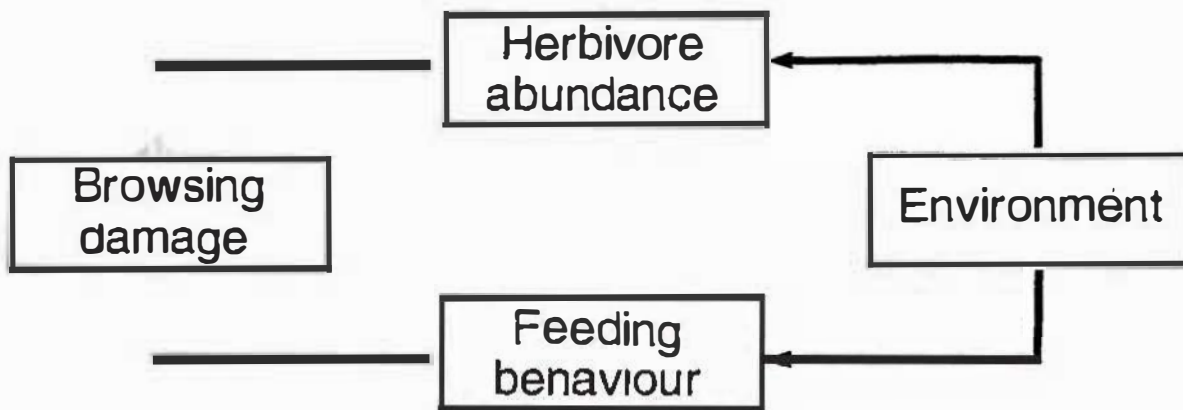


Figure 8.1 : Relationships between damage, herbivore abundance, herbivore feeding behaviour and the environment.

8.2.2 Measuring herbivore abundance

The scat count data described in Chapter 7 were also used here, as a measure of herbivore abundance. Two variables were derived from the data:

- i) average scat deposition rate (six months)¹ - the average scat deposition rate calculated from the deposition rates (scats.m⁻².day⁻¹) obtained at 80 and 160 days after planting. Separate values were calculated for the brushtail possum, Bennett's wallaby, pademelon, macropod grouping and rabbit.
- ii) average scat deposition rate (12 months) - as for (i), except calculated from the deposition rates (scats.m⁻².day⁻¹) obtained at the 80, 160, 240 and 320 days after planting sampling periods (see Section 7.2.2).

8.2.3 Assessing damage to seedlings

Damage to seedlings was assessed at six and 12 months after planting. The method by which these assessments were carried out is described in detail in Chapter 2. The data obtained from these assessments have already been presented in Chapter 3. Two measures of damage were used in the analyses described within this chapter :

- i) mean browse score measured at six months and
- ii) mean browse score measured at 12 months.

8.2.4 Measuring environmental variables

A range of environmental and habitat variables were measured at each site. These variables are broadly divided into three classes :

- i) variables used to describe the characteristics of each plantation;
- ii) variables used to describe the characteristics of adjoining cover habitats; and

¹ To improve readability, all variables mentioned in this chapter have been assigned a simplified descriptive term. This term appears underlined in methods Sections 8.2.2 to 8.2.7 inclusive and then as plain text throughout the rest of the chapter.

iii) climatic variables.

Cover habitats are defined here as areas of bushland or established plantation with a shrub or tree canopy taller than 2 metres. All variables were chosen on the basis that they had been identified as being important in other studies of herbivore-forestry interactions (Home 1975; Barnett *et al.* 1977; Lavsund 1987; Conover 1989; Hannan and Whelan 1989; Kay 1993) and/or because I felt there were good biological arguments to suggest they might influence herbivore abundance and damage.

8.2.5 Variables describing plantation characteristics

Each of the variables used to describe the characteristics of plantations are described below.

Vegetation cover (plantation)

The vegetation cover on each plantation was measured from 50 randomly placed 1m² quadrats. The position of each quadrat was permanently marked using 0.5 m long wooden pegs, wire pigtailed and surveyors tape. Vegetation cover was assessed (Chapter 2) at the time seedlings were planted and then again at 80, 160, 240 and 320 days after planting. The presence and estimated cover of several vegetation components were recorded. A description of these is given in Table 8.1. Coverage estimates were made to the nearest 10% and mean values calculated for each vegetation component at each site. From these data, the following variables were then derived for each site :

- i) mean cover of each vegetation component (six months) - calculated from the cover values obtained at the 0, 80 and 160 days after planting; and
- ii) mean cover of each vegetation component (12 months) - as above, except calculated from the cover values obtained at 0, 80, 160, 240 and 320 days after planting.

Table 8.1 : Description of the various vegetation components assessed on each plantation.

Vegetation component	Description
Grasses	Monocotyledons belonging to the <i>Gramineae</i> family
Ferns	True ferns and related fern-like plants (Dalton 1981a)
Forbs	Herbaceous plants other than grasses and ferns
Plantation species	The eucalypt species planted on the plantation
Shrubs/trees	Woody perennials which had not been deliberately planted on the plantation
Bryophytes	Mosses and liverworts (Dalton 1981b)
Fungi/lichens	Though not plants, the cover of fungi and lichen species (Kantuilas 1981; Mills 1981) was noted where they occurred
Total vegetation	An overall estimate of the vegetation cover within a quadrat, incorporating all the above vegetation components except fungi

Slope, altitude and aspect

The average slope (degrees), altitude (m) and aspect of each plantation was determined from official government 1:25000 topographic maps (Tasmap).

Average slope was calculated using the method of Hays *et al.* (1981). All aspects were ground truthed.

Area and perimeter

The area (ha) and the length of perimeter (km) for each plantation were obtained from forestry company surveying records. The area to perimeter ratio was calculated by dividing the area measure by the perimeter measure.

Proportion of perimeter adjacent to cover

The proportion of a plantation's perimeter that was adjacent to cover habitats was measured by first marking the locations of cover habitat on a 1:10 000 map of

each site. The total length of the perimeter and the length of perimeter adjacent to cover were then measured from this map, using a planimeter with a perimeter scale readout. The proportion of the perimeter that was adjacent to cover was then calculated, by dividing the length of the perimeter adjacent to cover by total perimeter length.

Length of stream-side reserves

Some plantations contained small streams or watercourses. To avoid excessive erosion and siltation, forestry companies leave a narrow strip of bushland adjoining each stream. Whenever one of these stream-side reserves was present within a plantation its length was measured using a 100 m tape measure.

Frequency of poisoning

Forestry company records were consulted to determine how many separate poisoning operations were carried out within each plantation. Data were collected from two months prior to planting to one year after planting.

A summary of the measurements taken at each plantation for each of the above variables is presented in Appendix (B).

8.2.6 Variables describing the characteristics of cover habitats

The nature of cover habitat adjoining plantations was described by measuring variables from five randomly placed 100 × 1 m transects. Transects were run in a straight line from, and perpendicular to, the plantation's edge. The length of a transect was delineated using a 100 metre tape measure. The width of each transect was defined by holding a one metre ruler perpendicular to the tape measure. Transects were established at, or shortly after (<30days), seedlings were planted. The position of transects was marked using 0.5 m wooden pegs and brightly coloured surveyors tape.

Canopy closure

A reading of canopy closure was taken every 10 metres along each transect, starting 10 metres from the plantation's edge. Readings were made from a vertical sighting tube (Flays *et al.* 1981). Percent canopy closure was calculated from :

$$\text{canopy closure (\%)} = \frac{\text{total number of sightings of canopy}}{50} \times 100$$

Vegetation cover (cover habitats)

The cover of each vegetation component described in Table 8.1 was estimated to the nearest 10% (Chapter 2). These assessments were made from the vegetation layer at 0-1 m above ground level. Thus, the shrub and tree layer were not sampled. Cover estimates were made from 1 m² quadrats placed every 10 metres along each transect, starting 10 metres from the plantation's edge. The position of each quadrat was marked using 0.5 m long wooden pegs and brightly coloured surveyor's tape. Vegetation cover was re-assessed at six and 12 months after the plantation was planted. At each sampling period, a mean cover value was calculated for each cover component, by averaging all of the estimates made for that component at that site (Appendix B). From these data, the following variables were then derived for each vegetation component at each site :

- i) mean cover to six months (%) - the average of the cover estimates obtained at the zero and six month sampling periods; and
- ii) mean cover to 12 months (%) - the average of the cover estimates obtained at zero, six and 12 months.

Tree density

Trees were defined as woody perennials taller than 3 metres. The average number of trees per square metre was determined by moving along the transect holding a one metre ruler at waste level and perpendicular to the tape measure marking the edge of the transect. The number of trees which were intercepted by the ruler was recorded. Trees were recorded within a transect if more than half of their diameter, measured at 1 m above ground level, was contained within the transect. Tree density (trees.m⁻²) was calculated from :

$$\text{Tree density} = \frac{\text{Total number of trees in transect}}{500 \text{ m}^2}$$

Shrub density

Shrubs were defined as woody perennials less than 3 metres in height. The number of shrubs per square metre was assessed in the same way as tree density.

Mean tree diameter

For all trees within the transect, measurements were made of diameter at chest height using a diameter tape (m). The average of all tree diameters was then calculated.

A summary of the measurements made for each of these variables, at each plantation, is presented in Appendix (B).

8.2.7 Climatic variables

Rainfall

Bureau of Meteorology monthly rainfall records were obtained for the weather station closest to each plantation. Two measures of rainfall were calculated from these data. The first was the mean annual rainfall for the area, based on an

average of the rainfall data from all recorded years (ranged from 5 to 113 years). The other measure of rainfall was the amount of rain that fell during the 12 month sampling period (rain during sampling).

The average annual rainfall data for each site was presented in Chapter 2. Rainfall during the sampling period is presented for each site in Appendix (B).

8.2.8 Statistics

The SAS statistics package was used for the analysis of the results presented in this chapter. The CORR procedure was used for exploring correlations between sets of variables. One-way analysis of variance (ANOVA), testing for the effects of aspect (the only class variable in the data set) was carried out using the GLM procedure in SAS (SAS Institute 1989b). Multiple regression analyses (Tabachnik and Fidell 1989) were used to model a single continuous dependent variable (e.g. mean browse score) as a function of two or more continuous independent variables (e.g. grass cover, fern cover, average rate of scat deposition for the brushtail possum). Preliminary model development was carried out using the REG procedure in SAS with the MAXR selection option (Sas Institute 1989b). Results for the final model were arrived at using standard multiple regression (SAS' REG procedure).

The residuals calculated for each regression model were inspected for normality, linearity, homoscedasticity and to ensure that the model was not unduly influenced by outlying results (Tabachnik and Fidell 1989). Where appropriate, independent variable data were transformed to improve these attributes. Prior to inclusion into a model, all independent variables were screened to ensure that they were not strongly correlated with each other (Appendix C). Where strong correlation did occur ($r > 0.7$) between two variables, only one out of the pair would be included in any given regression. Several variables had to be omitted from all analyses because their distribution was so strongly skewed toward zero. These were the cover of fungi, measured from both the plantation and adjoining

cover habitats, the cover of shrubs and trees on the plantation, and the length of stream-side reserves. The slope variable was omitted because of two extreme values which appeared to unduly influence the results of any analyses in which slope was included (outlying results with respect to the dependent variable). Further details of the analyses are provided below.

8.2.9 Analyses

1. Identifying variables related to the relative abundance of herbivores

In this series of analyses, regression techniques were used to model average scat deposition rates (Chapter 7), calculated over 12 months, as a function of variables describing the characteristics of the plantation, variables describing the characteristics of adjoining cover habitat and climatic variables (Appendix B). Separate regression analyses were run for the brushtail possum, Bennett's wallaby, Tasmanian pademelon, and rabbit.

2. Investigating relationships between damage severity, herbivore abundance, and vegetation cover

The logic behind this analysis was that the severity of damage observed in a plantation would be related to both the relative abundance of herbivores within the plantation and their feeding behaviour while they were there. Feeding behaviour, or more specifically, the propensity that each herbivore has for feeding on seedlings, may be influenced by the quantity and types of alternative vegetation available for them to eat within the plantation (Figure 8.2). Thus, mean browse score for each site (Chapter 3) was modelled as a function of average rate of scat

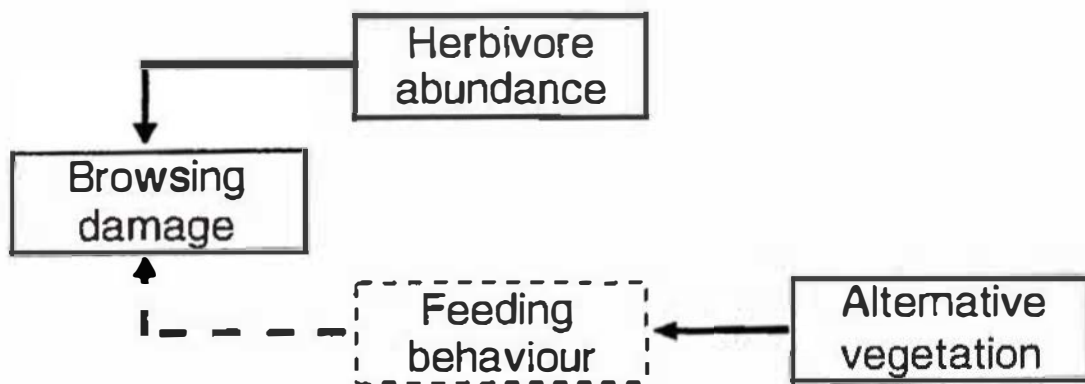


Figure 8.2 : The logic for Analysis 2.

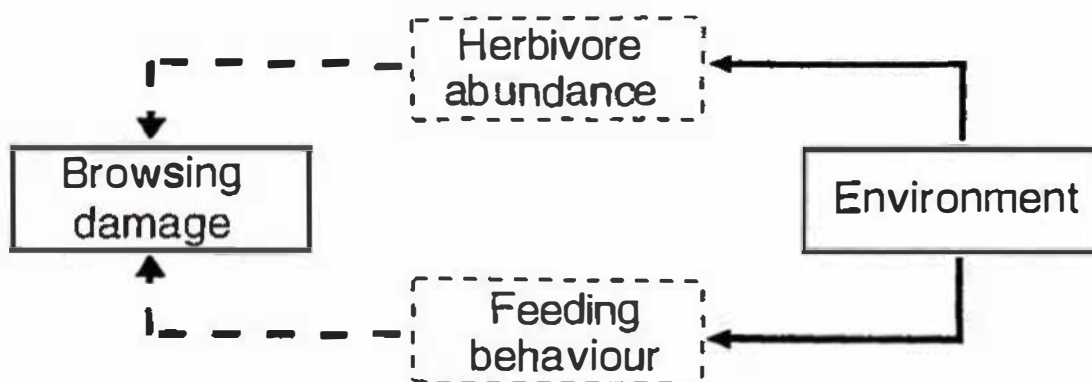


Figure 8.3 : The logic for Analysis 3.

deposition data collected for possums, macropods and rabbits (Chapter 7) and mean vegetation cover values collected from the plantation (Appendix B). Separate regressions were run for data collected at six months and data collected at 12 months from the 32 *E. nitens* plantations.

3. Identifying useful variables for predicting damage severity

The investigation described above incorporates variables which were collected over an extended period of time. For example, mean grass cover to 12 months was determined from vegetation assessments made at the time of planting and then at 80, 160, 240 and 320 days after planting. While such variables may be useful for describing variation in damage, they are not particularly useful for predicting damage. By the time the variable is measured the damage has already occurred. In this analysis, mean browse score data was modelled as a function of a range of environmental variables that could be measured before a plantation is established and that might, therefore, prove useful for predictive purposes (Figure 8.3).

8.3 RESULTS

8.3.1 Identifying variables related to the abundance of herbivores : Investigation using all sites

Relative abundance of the brushtail possum

Results of a one-way analysis of variance indicated that aspect did not have a significant influence on average scat deposition rate (12 months) for the brushtail possum ($F_{(6,28)}=0.73$, $p>0.05$).

Results of a multiple regression, modeling average scat deposition rate (12 months) for the brushtail possum as a function of environmental variables, is shown in Table 8.2. Two independent variables were incorporated in the final

model. In order of decreasing influence these were :

- i) arcsin square root transformed canopy closure ($Sr^2=0.15$, $t=2.538$, $p<0.05$)
- ii) frequency of poisoning a teach site ($Sr^2=0.07$, $t=1.754$, $p=0.09$)

In total, this model explained 25% of the between-site variation in average scat deposition rate for the brushtail possum ($r^2=0.254$, $F_{(2,32)}5.4$, $p<0.01$). The form of the regression is described by the following equation :

$$y = - (6.3 \times 10^{-4} x_1) - (1.8 \times 10^{-4} x_2) + 0.0011$$

Equation 8.1

where 'y' is average scat deposition rate (12 months) for the brushtail possum, 'x₁' is arcsin square root transformed canopy closure and 'x₂' is frequency of poisoning.

Relative abundance of the Bennett's wallaby

Results of a one-way analysis of variance indicated that average scat deposition rate (12 months) for the Bennett's wallaby was not significantly influenced by aspect ($F_{(6,28)}=0.43$, $p>0.05$).

Results of a multiple regression, modelling average scat deposition rate (12 months) for the Bennett's wallaby as a function of environmental variables is shown in Table 8.3. Two independent variables were incorporated in the final model. In order of decreasing influence, these were :

- i) frequency of poisoning ($Sr^2=0.18$, $t=2.671$, $p<0.05$)
- ii) proportion of the perimeter adjacent to cover ($Sr^2=0.12$, $t=2.125$, $p<0.05$).

In total, the regression explained 23% of the between-site variation in average scat deposition rate (12 months) for the Bennett's wallaby ($r^2=0.226$, $F_{(2,32)}=4.67$,

Table 8.2 : Results of a standard multiple regression, modelling average scat deposition rate (12 months) for the brush-tail possum as a function of arcsin square root transformed canopy closure and frequency of poisoning. Parameter estimates (B coefficient), semi-partial correlation coefficients (Sr^2) and t-test statistic values (t) obtained for tests of departure of Sr^2 from zero ('*' for $p < 0.05$) are given for each parameter. Included in the 'regression statistics' section are the correlation coefficient for regression (r^2), F value for analysis of variance, associated probability and adjusted r^2 value.

Variable	B coefficient	Sr^2	t
transformed canopy closure	$- .3 \times 10^{-4}$	0.15	2.538 *
frequency of poisoning	-1.8×10^{-4}	0.07	1.754
intercept	0.0011		
Regression statistics : $r^2 = 0.254$, $F_{(2,32)} = 5.4$, $p < 0.01$ adjusted $r^2 = 0.207$			

Table 8.3 : Results of a standard multiple regression modelling average scat deposition rate (12 months) for the Bennett's wallaby as a function of frequency of poisoning and the proportion (proprn) of perimeter adjacent to cover habitat. Parameter estimates (B coefficient), semi-partial correlation coefficients (Sr^2) and t-test statistic values (t) obtained for tests of departure of Sr^2 from zero ('*' for $p < 0.05$) are given for each parameter. Included in the 'regression statistics' section are the correlation coefficient for regression (r^2), F value for analysis of variance, associated probability and adjusted r^2 value.

Variable	B coefficient	Sr^2	t
frequency of poisoning	-2.98×10^{-3}	0.18	2.671 *
proprn perimeter adjacent cover	5.0×10^{-5}	0.12	2.125 *
intercept	0.0026		
Regression statistics : $r^2 = 0.226$, $F_{(2,32)} = 4.67$, $p < 0.05$ adjusted $r^2 = 0.178$			

$p < 0.05$). The form of the regression is described by the following equation :

$$y = -(2.98 \times 10^{-3} x_1) + (5 \times 10^{-5} x_2) + 0.0026$$

Equation 8.2

where 'y' is average scat deposition rate (12 months) for the Bennett's wallaby, 'x₁' is frequency of poisoning and 'x₂' is proportion of perimeter adjacent to cover.

Relative abundance of the pademelon

Results of a one-way analysis of variance indicate that average scat deposition rate (12 months) for the pademelon was not significantly influenced by aspect ($F_{(6,28)} = 1.74$, $p > 0.05$).

Results of a regression analysis indicated that frequency of poisoning was the only independent variable to explain a significant amount of the between-site variation in average scat deposition rate for the pademelon. A regression incorporating this variable explained 11% of variation ($r^2 = 0.113$, $df = 34$, $p < 0.05$). Addition of further variables to the model did not significantly improve r^2 . The regression form is described by the equation :

$$y = -0.0018x + 0.00475$$

Equation 8.3

where 'y' is average scat deposition rate (12 months) for the pademelon and 'x' is frequency of poisoning.

Relative abundance of macropods

Results of a one-way analysis of variance indicated that average scat deposition rate (12 months) for macropods² was not significantly influenced by aspect ($F_{(6,28)} = 0.77$, $p > 0.05$).

² Includes Bennett's wallaby scats, pademelon scats and macropod scats which could not be identified to species level.

The results of a regression analysis indicated that frequency of poisoning was the only independent variable to explain a significant amount of the between-site variation in average scat deposition rate (12 months) for macropods. A simple regression using this variable explained 15% of between-site variation ($r^2=0.147$, $df=34$, $p<0.05$). Addition of further variables to the model did not significantly improve r^2 for regression. The form of the regression is described by the following equation :

$$y = - 0.0058x + 0.0146$$

Equation 8.4

where 'y' is average scat deposition rate (12 months) for macropods and 'x' is frequency of poisoning.

Relative abundance of the rabbit

Results of a one-way analysis of variance indicated that the average scat deposition rate (12 months) for the rabbit, was not significantly influenced by aspect ($F_{(6,28)}=1.15$, $p>0.05$).

Results of a multiple regression modeling average scat deposition rate for the rabbits as a function of environmental variables is shown in Table 8.4. Two independent variables were incorporated in the final model. In order of decreasing influence these were :

- i) average annual rainfall ($Sr^2=0.12$, $t=2.248$, $p<0.05$)
- ii) arcsin transformed moss cover measured on the plantation ($Sr^2=0.10$, $t=1.990$, $p=0.06$)

In total, the regression explained 23% of the between-site variation in average scat deposition rate for the rabbit ($r^2=0.234$, $F_{(2,32)}=4.88$, $p<0.05$). The form of the

regression is described by the following equation :

$$y = (2.6 \times 10^{-4})x_1 - (1.0 \times 10^{-6})x_2 + (4.6 \times 10^{-5})$$

Equation 8.5

where 'y' is average rate of scat deposition (12 months) for the rabbit, 'x₁' is average annual rainfall and 'x₂' is arcsin square root transformed moss cover measured on the plantation.

Table 8.4 : Results of a standard multiple regression modelling average scat deposition rate (12 months) for the rabbit as a function of average annual rainfall and arcsin square root transformed mean moss cover measured on the plantation. Parameter estimates (*B* coefficient), semi-partial correlation coefficients (*Sr*²) and t-test statistic values (*t*) obtained for tests of departure of *Sr*² from zero ('*' for *p*<0.05) are given for each parameter. Included in the 'regression statistics' section are the correlation coefficient for regression (*r*²), *F* value for analysis of variance, associated probability and adjusted *r*² value.

Variable	<i>B</i> coefficient	<i>Sr</i> ²	<i>t</i>
average annual rainfall	2.6×10 ⁻⁴	0.12	2.248 *
transformed moss cover	-1.0×10 ⁻⁶	0.10	1.990
intercept	4.6×10 ⁻⁵		

Regression statistics : *r*²= 0.234, *F*_(2,32)=4.88, *p*<0.05 adjusted *r*²=0.186

8.3.2 Herbivore abundance, vegetation cover and damage severity

Damage severity at six months after planting

When multiple regression techniques were used to compare mean browse score measured at six months to abundance measures and vegetation cover variables, no model was found which explained a significant amount of between-site variation.

Average scat deposition rate (12 months) for the brushtail possum was the variable which explained most variation ($r^2=0.117$, $df=31$, $p=0.05$).

Damage severity at 12 months after planting

Results of a multiple regression analysis, modelling mean browse score measured at 12 months as a function of measures of herbivore abundance and measures of vegetation cover made on the plantation are presented in Table 8.5. Three independent variables were included in the final model. In order of decreasing influence these were :

- i) average scat deposition rate (12 months) for the brushtail possum
($Sr^2=0.54$, $t=7.168$, $p<0.01$)
- ii) square root transformed mean grass cover on the plantation ($Sr^2=0.09$,
 $t=2.965$, $p<0.01$)
- iii) mean fern cover on the plantation ($Sr^2=0.09$, $t=2.978$, $p<0.01$).

In total, the regression explained 71% of variation, a result which is highly significant ($r^2=0.707$, $F_{(3,28)}=22.6$, $p<0.01$). The form of the regression is described by the following equation :

$$y = 1.998x_1 + 1.613x_2 - 0.128x_3 + 0.461$$

Equation 8.6

where 'y' is mean browse score measured 12 months after planting, ' x_1 ' is average scat deposition rate for the brushtail possum, ' x_2 ' is the square root transformed

mean cover of grass on the plantation and 'x₃' is the mean cover of ferns on the plantation.

Table 8.5 : Results of a standard multiple regression modelling mean browse score measured at 12 months as a function of average scat deposition rate (12 months) for the brushtail possum, square root transformed mean grass cover and mean fern cover. Parameter estimates (*B* coefficient), semi-partial correlation coefficients (*Sr*²) and t-test statistic values (*t*) obtained for tests of departure of *Sr*² from zero ('*' for *p*<0.05, '**' for *p*<0.01) are given for each parameter. Included in the 'regression statistics' section is the correlation coefficient for regression (*r*²), *F* value for analysis of variance, associated probability and adjusted *r*² value.

Variable	<i>B</i> coefficient	<i>Sr</i> ²	<i>t</i>
average scat deposition rate	1998.400	0.54	7.168 **
transformed grass cover	1.613	0.09	2.965 **
fern cover	-0.128	0.09	2.978 **
intercept	0.461		
Regression statistics : <i>r</i> ² = 0.707, <i>F</i> _(3,28) =22.6, <i>p</i> <0.01 adjusted <i>r</i> ² =0.676			

8.3.3 Identifying useful variables for predicting damage severity

Predicting damage severity at six months after planting

Results of a one-way analysis of variance indicated that aspect did not have a significant influence on mean browse score measured at six months (*F*_(6,25)=1.02, *p*>0.05).

Results of a multiple regression modelling mean browse score measured at six months as a function of environmental variables that could be measured at, or before, planting are presented in Table 8.6. Three independent variables were

incorporated in the final model. In order of decreasing influence these were :

- i) arcsin square root transformed canopy closure ($Sr^2=0.15$, $t=2.422$, $p<0.05$)
- ii) area to perimeter ratio ($Sr^2=0.10$, $t=1.931$, $p=0.06$)
- iii) perimeter adjacent to cover ($Sr^2=0.07$, $t=1.604$, $p=0.12$)

In total, the model explained 28% of the between-site variation ($r^2=0.277$, $F_{(3,28)}=3.57$, $p<0.05$). The form of the regression is described by the following equation :

$$y = -1.510x_1 - 0.070x_2 + 0.01x_3 + 2.354$$

Equation 8.7

where 'y' is mean browse score measured at six months, 'x₁' is arcsin square root transformed canopy closure, 'x₂' is area to perimeter ratio and 'x₃' is perimeter adjacent to cover.

Predicting damage severity at 12 months after planting

Results of an analysis of variance indicated that aspect did not have a significant influence on mean browse score measured at 12 months ($F_{(6,25)}=0.68$, $p>0.05$).

Results of a multiple regression modelling mean browse score measured at 12 months as a function of environmental variables that could be measured at, or before, the planting of seedlings are shown in Table 8.7. Four independent variables were incorporated in the final model. In order of decreasing influence these were :

- i) area to perimeter ratio ($Sr^2=0.26$, $t=3.831$, $p<0.01$)
- ii) proportion of perimeter adjacent to cover ($Sr^2=0.10$, $t=2.327$, $p<0.05$)
- iii) arcsin square root transformed canopy closure ($Sr^2=0.08$, $t=2.054$, $p<0.05$)
- iv) arcsin square root transformed mean total vegetation cover measured from cover habitat at the time seedlings were planted ($Sr^2=0.05$, $t=1.747$, $p=0.09$).

Table 8.6 : Results of a standard multiple regression modelling mean browse score measured at six months as a function of arcsin square root transformed canopy closure, area to perimeter ratio and proportion (propn) of perimeter adjacent to cover habitat. Parameter estimates (*B* coefficient), semi-partial correlation coefficients (Sr^2) and *t* values obtained for tests of departure of Sr^2 from zero ('*' for $p < 0.05$) are given. Included in the 'regression statistics' section are the correlation coefficient for regression (r^2), *F* value for analysis of variance, associated probability and adjusted r^2 value.

Variable	<i>B</i> coefficient	Sr^2	<i>t</i>
transformed canopy closure	-1.500	0.15	2.422 *
area to perimeter ratio	-0.070	0.10	1.931
propn perimeter adjacent cover	0.010	0.07	1.604
intercept	2.354		

Regression statistics : $r^2=0.280$, $F_{(3,28)}=3.57$, $p < 0.05$ adjusted $r^2=0.199$

Table 8.7 : Results of a standard multiple regression modelling mean browse score measured at 12 months as a function of area to perimeter ratio, proportion (propn) of perimeter adjacent to cover habitat, arcsin square root transformed canopy closure, and arcsin square root transformed total vegetation cover measured from cover habitat (CH). Parameter estimates (*B* coefficient), semi-partial correlation coefficients (Sr^2) and *t*-test statistic values obtained for tests of departure of Sr^2 from zero ('*' for $p < 0.05$, '**' for $p < 0.01$) are given. Included in the 'regression statistics' section are the correlation coefficient for regression (r^2), *F* value for analysis of variance, associated probability and adjusted r^2 value.

Variable	<i>B</i> coefficient	Sr^2	<i>t</i>
area to perimeter ratio	-0.144	0.26	3.831 **
propn perimeter adjacent cover	0.014	0.10	2.327 *
transformed canopy closure	-1.483	0.08	2.054 *
transformed total vegetation (CH)	2.323	0.05	1.747
intercept	1.619		

Regression statistics : $r^2=0.517$, $F_{(4,27)}=7.21$, $p < 0.01$ adjusted $r^2=0.445$

In total, the model explained 52% of the between-site variation ($r^2=0.517$, $F_{(4,27)}=7.211$, $p<0.01$) in mean browse score. The form of the regression is described by the following equation :

$$y = -0.144x_1 + 0.014x_2 - 1.483x_3 + 2.323x_4 + 1.619$$

Equation 8.8

where 'y' is mean browse score measured at 12 months, 'x₁' is area to perimeter ratio, 'x₂' is proportion of perimeter adjacent to cover, 'x₃' is arcsin square root transformed canopy closure and 'x₄' is arcsin square root transformed mean total vegetation cover measured from cover habitat at the time seedlings were planted.

8.4 DISCUSSION

8.4.1 Identifying variables related to the relative abundance of herbivores

None of the various regression models that were tested were particularly successful at describing between-site variation in average scat deposition rate (12 months) for any of the herbivore species ($r^2 \leq 0.25$). From these analyses, however, several variables were identified as being potentially important in influencing herbivore abundance. The frequency with which poisoning was used at a site appeared to be one of the stronger and more consistent influences. The higher the frequency of poisoning, the lower the average scat deposition rate for the brushtail possum, Bennett's wallaby, pademelon and macropod grouping. These findings are consistent with those reported in Chapter 7. They suggest, once again, that 1080 poisoning does reduce the abundance of these species.

Average scat deposition rate for the brushtail possum was also negatively correlated to a measure of canopy closure, obtained from cover habitats adjoining the plantation. The more closed the canopy, the lower the average scat deposition rate tended to be. One possible explanation for this is that cover habitats with relatively open canopies may support larger numbers of possums than cover

habitats with closed canopies. Open canopies allow a greater amount of light to penetrate to the ground vegetation layer than closed canopies. Correlation analyses (Appendix C) indicated that many of the measures of ground vegetation cover, including total vegetation cover, were negatively correlated with canopy closure. In a Tasmanian dietary study, Statham (1983) observed that the brushtail possum fed predominantly on low lying plant species. Thus, cover habitats with open canopies and an extensive ground vegetation cover may be able to support more possums than cover habitats with closed canopies and sparse ground vegetation cover.

The results presented in this chapter and Chapter 7 indicate that average scat deposition rate for the rabbit is not strongly influenced by frequency of poisoning. This is contradictory to the findings of Statham (1983), who observed that rates of rabbit scat deposition declined by as much as 90% following poisoning operations in Tasmanian forestry coupes. Similarly, Foran *et al.* (1985) observed that rabbit abundance in areas of central Australia, where 1080 poisoning programs were carried out, was approximately half that of unpoisoned control areas. Crosbie *et al.* (1986) observed an 85% reduction in an island population of rabbits following poisoning operations.

The differences between the results of this study and those of Crosbie *et al.* (1986) and Foran *et al.* (1985) are likely to reflect differences in the method of poison delivery and intensity of control operations. Reasons for the differences between this study and that of Statham (1983) are, at first, less obvious. In both studies, the method of poisoning and the intensity of control appear very similar. One startling difference, though, is that both the pre- and post-poisoning scat deposition rates on the coupes that Statham (1983) studied were far higher than on any of the sites surveyed during this study. Her lowest estimate is several times greater than the highest estimate obtained in this study. It is possible then, that rabbit numbers in this study were affected by poisoning, but the scat count method was not sensitive to changes in the abundance of a species already present in very

low numbers (Read and Fox 1990). Alternatively, rabbit numbers simply may not have been affected by poisoning operations. As Braysher (1993) points out, the effort required to reduce pest species abundance increases as the species density decreases.

Average scat deposition rate (for 12 months) for the rabbit was found to be positively related to average annual rainfall, as measured from the nearest weather station, and negatively related to mean moss cover on the plantation. The reasons for the negative relationship between moss cover and rabbit scat deposition remain unclear. However, there is considerable evidence suggesting that rainfall strongly influences the abundance of rabbits, primarily through its effects on food availability (see review by Myers *et al.* 1994). In arid Australia, rabbit numbers increase dramatically following several years of above average rainfall (Cooke 1982). Conversely, rabbits may only be found in low numbers where rainfall levels are very high. In New Zealand, for example, the distribution of rabbits is restricted to areas with an annual rainfall below 2500 mm (Gibb and Williams 1994).

8.4.2 Relationships between damage, herbivore abundance and vegetation

In the introduction to this chapter it was proposed that the amount of damage which occurs within a forestry plantation would be a function of the abundance of herbivores in the plantation, or more specifically how many herbivores enter the plantation and how long they stay; and the feeding behaviour of each of these individuals. Feeding behaviour, including the propensity that each individual has for feeding on eucalypt seedlings, may be influenced by the amount and type of alternative vegetation available within the plantation. These hypotheses were not strongly supported by the results of a multiple regression comparing mean browse score assessed at six months after planting to average rate of scat deposition data and measures of vegetation cover variables made on each plantation. None of the variables tested explained a significant amount of the between-site variation in

mean browse score. An almost significant trend ($p=0.05$) was observed between average brushtail possum scat deposition rate and damage.

An analogous regression carried out with the 12 month data was much more successful. Using three variables : average scat deposition rate for the brushtail possum, square root transformed mean grass cover and mean fern cover, it was possible to explain 71% of the between-site variation in mean browse score measured at 12 months. Damage was negatively related to mean fern cover and positively related to average scat deposition rate for the brushtail possum (also refer to Chapter 7) and grass cover. Thus, damage tended to be highest at those sites with a high average scat deposition rate for the brushtail possum, high grass cover and low fern cover.

Two hypotheses are offered to explain the negative relationship between fern cover and mean browse score. Firstly, it is possible that herbivores feeding within a plantation might choose to feed on fern species in preference to eucalypt seedlings. However, fern cover on plantations tended to be composed almost entirely of thick clumps of bracken (*Pteridium esculentum*), a species which is apparently not preferred by the brushtail possum, Bennett's wallaby or pademelon (Statham 1983; Sprent 1997). A second, more plausible hypothesis is that seedlings growing amongst tall bracken thickets are afforded some kind of physical protection from mammalian herbivores. Both Gilbert (1961) and Cremer (1969) observed that seedlings growing in amongst tall thick weed clumps were subject to less damage than seedlings growing in more open areas. More recently, Flint (1997) found that seedlings were afforded protection by growing amongst tall vegetation. Furthermore, a suite of studies in varying environments have demonstrated that plant species may be afforded protection from herbivory by growing in association with other plant species, giving rise to the 'associational plant refuge hypothesis' (Rausher 1981; Hay 1986; Littler *et al.* 1986; Pfister and Hay 1988; Dannell *et al.* 1991).

The opposite trend was observed for the relationship between grass cover and damage. The higher the level of grass cover, the greater the mean browse score tended to be. Cremer (1969) also observed that grass cover influenced the level to which seedlings are damaged. In his study, he mixed grass seed with eucalypt seed and then scattered this mixture onto a forestry coupe. In his words, “the area sown with grass and eucalypt seeds was browsed so intensively that practically none of the eucalypts survived amongst the grass” (Cremer 1969, p.207).

It is possible that plantations, or areas within a plantation, with high levels of grass cover are more attractive as feeding patches to grazing herbivores than areas which have sparse grass cover. If this is the case, higher levels of damage might be observed in areas with extensive grass cover because more herbivores tend to frequent the area more often or for more prolonged periods of time. However, this hypothesis is not easily reconciled with the results presented in this study. By including scat count data as an independent variable in the analysis the effect of differing herbivore abundance should have already been accounted for.

Furthermore, no strong relationships were observed between the abundance of any herbivore species and the amount of grass cover on the plantation. An alternative explanation may be that grass cover does not influence herbivore abundance but does exert a positive influence on the propensity that individual herbivores have for feeding on eucalypt seedlings.

8.4.3 Identifying variables which might be important for predicting damage

Results of the multivariate analyses, comparing damage data to a range of habitat and plantation characteristic variables, did not yield a model which explained enough between-site variation that it could be used to predict browsing damage levels at either six or 12 months after planting. For the six month data, the best model explained 28% of the between-site variation in mean browse score.

Substantially better results were obtained from an analyses of the 12 month data, with the most successful model explaining 52% of the between-site variation.

While the results of these analyses fall short of effective damage prediction, they do provide valuable insight into the site characteristics and habitat variables which may be important in influencing damage levels. For both the six and 12 month analyses area to perimeter ratio, the proportion of perimeter adjacent to cover habitats and canopy closure in cover habitats were identified as being important influences on damage severity. The negative relationship between canopy closure and mean browse score presumably results from the fact that canopy closure is also negatively correlated with average scat deposition rate (12 months), which is itself positively correlated with mean browse score. Hence, the greater the extent of canopy closure, the lower the average scat deposition rate and the lower the mean browse score.

The area to perimeter ratio of a plantation and the proportion of a plantation's edge that is adjacent to cover habitats have often been suggested as being important in influencing damage levels in Tasmanian eucalypt plantations (Cremer 1969; Coleman *et al.* 1997). Analogous variables have been suggested as potentially exercising an influence on the amount of damage caused to a range of crops throughout the world (e.g. Matschke *et al.* 1984; Conover 1989; Kay 1993). Until now, however, no Tasmanian study has empirically demonstrated their importance. The influence of these variables is likely to be linked to the habitat preferences and movement patterns of the herbivore species that cause browsing damage.

All of the Tasmanian herbivore species, identified as causing problematic damage, are thought to shelter in cover habitats during the day and move into plantation areas to feed throughout the night (Green 1973; Mooney and Johnson 1979; Johnson and Rose 1983; MacLennan 1984; Johnson 1987; Flannery 1994; Le Mar pers. comm. 1998). Predator 'shy' species, such as the Tasmanian pademelon, are not thought to venture far from cover habitats during these feeding forays (Coleman *et al.* 1997). If this is the case, feeding activity would be concentrated at those sections of a plantation's edge that are adjacent to cover habitat. Thus,

plantations with a low proportion of their edge adjacent to cover might be expected to have a lower potential for damage than plantations with a high proportion of edge adjacent to cover. Additionally, plantations with a high area to perimeter ratio may experience low damage because herbivores only have access to a low proportion of the total seedlings planted within the plantation.

8.4.4 Suggestions for improvement, future direction and management

Sample from more unpoisoned sites

A recurrent problem throughout these analyses has been the low sample size obtained for plantations where poisoning was not carried out. In the three years during which field work was carried out, only seven plantations not subject to poisoning operations were provided by the four main forestry companies operating in Tasmania. All efforts were made to incorporate further non-poisoned plantations in the survey, but foresters were exceedingly reticent to risk leaving sites unpoisoned because of anticipated seedling loss. Since poisoning has been shown to affect herbivore abundance on Tasmanian forestry plantations (Statham 1983; see also Chapter 7), this obviously complicates any analyses attempting to determine what environmental factors might be important in influencing herbivore abundance. The development of models for use in predicting damage and making decisions regarding control is severely hampered when the majority of the sites surveyed are already subject to control efforts.

If the goal of developing models that can be used to accurately predict damage is to be realised, sampling will need to be carried out at a substantially greater number of sites where poisoning operations have not been carried out. This will require an undertaking by forestry companies to set aside sites for study at which no control effort is carried out. An alternative, but less satisfactory approach, would be to collect data opportunistically from unpoisoned sites as they become available. For example, some sites are occasionally left unpoisoned because of the concerns of neighbouring land-holders for the welfare of their livestock. In

any event, care should be taken to ensure that any sites that are used for study are chosen randomly and not according to some systematic pattern. It would not be sufficient to leave only those sites unpoisoned that are suspected to have low potential for damage.

Consider other variables for measurement

Improvements on the models presented here might be achieved by measuring a wider range of variables. For example, all of the mammalian herbivore species identified as being significant forestry pests use cover habitats during the day, the amount of this habitat within the vicinity of a plantation might therefore prove to be an important variable for describing between-site variation in herbivore abundance and damage levels. A north American study carried out by Conover (1989), investigating deer damage in nurseries, has already demonstrated that the size of woodlots and the total area of woodlots within a two kilometre radius are important variables influencing damage. Various options were considered, but measures of 'landscape' characteristics were not included in this study because the means for obtaining them were not available. Unlike Conover's (1989) study, aerial photos of the forestry plantations used in this study were several years out of date and forestry records were not sufficiently detailed to make accurate assessments. Future studies should strive to overcome these problems so that a more complete picture of the plantation surrounds is obtained.

Consider other approaches

An alternative, and potentially very productive, approach to that employed in this study, would be to model spatial trends in herbivore abundance and damage using geographic information systems (GIS). This powerful computer-based technique has the potential for modelling spatial and possibly temporal trends in herbivore abundance and damage, based on existing environmental databases (Haines-Young *et al.* 1993; Aspinall 1994; Congalton and Green 1995). Geographic information systems are already widely used for modelling invertebrate pest

species populations and damage patterns (e.g. Liebhold *et al.* 1995; Williams and Liebhold 1995a,b) and there is no technical reason why they could not be used in similar investigations of vertebrate pest species. Indeed, there are existing examples where GIS technology has been used to model herbivore abundance (e.g. Rudis and Tansey 1995) and even grazing patterns (e.g. Pearson and Turner 1995). Unfortunately for this study, sophisticated modelling exercises using GIS require specialist knowledge and attempts at engaging the assistance of a professional in the field proved fruitless.

Investigate the relationships between vegetation and damage more fully

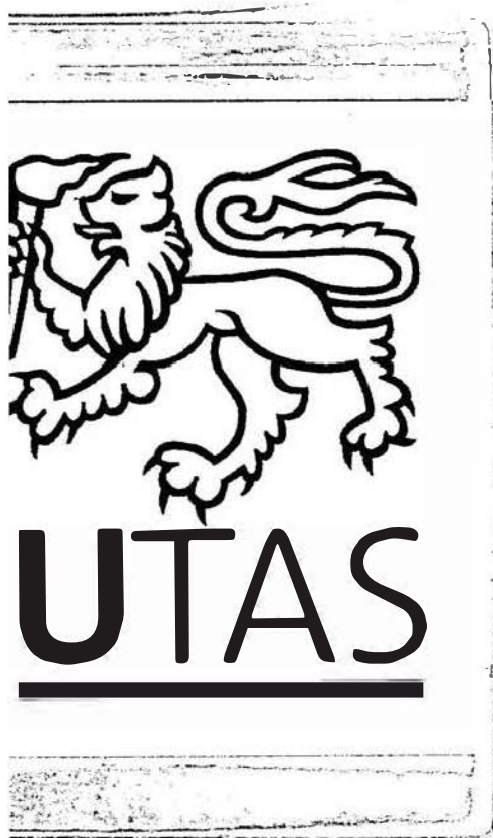
The results of this study, and that of Cremer (1969) and Flint (1997), suggest that the characteristics of the vegetation growing on a plantation have the potential to influence the levels to which seedlings are damaged in plantation areas. If a more detailed investigation were to confirm these results and perhaps elucidate such relationships in greater detail, a spectrum of alternative control strategies might be developed based on the deliberate manipulation of the vegetation community on each plantation. For example, seedlings might be protected from browsing by deliberately planting them amongst a plant crop which increases the crypsis of seedlings or which is, in some way, actually repellent to herbivores.

Alternatively, the judicious use of herbicides might make the plantation less 'appealing' to herbivores by decreasing the cover of preferred plant species.

If the option exists, design a plantation to minimise damage

In accord with anecdotal evidence (e.g. Coleman *et al.* 1997), the results of this study indicate that the area to perimeter ratio of a plantation and the amount of plantation edge which is adjacent to cover habitats are important factors in influencing browsing damage severity. From this, two suggestions for management can be made. If the option is available, managers should design plantations that are large and as close to circular as possible. In this way, area to perimeter ratio is maximised and potential for browsing damage minimised.

Similarly, where a manager has the choice of placing a plantation near to, or away from, cover habitats, the latter option should always be preferred.



CHAPTER 9

GENERAL DISCUSSION

Historically there has been a worldwide tendency for managers to respond to real or perceived 'pest' problems, by attempting to reduce the population of the pest species (see reviews by Lazarus 1989; Richards 1989; Gibb and Williams 1994; Myers *et al.* 1994). In many cases, control efforts have been employed in an *ad hoc* fashion, in isolation to other control techniques, and without any real quantification of the costs of the damage or the benefits of the control (Braysher 1993; Hone 1994; Myers *et al.* 1994; Allen *et al.* 1998). This approach has often achieved only limited or temporary success. Pest populations are able to compensate for the initial removal of individuals because fecundity and immigration are increased, while density dependent mortality is decreased (Stenseth 1981; Begon and Mortimer 1986; Driessen 1992). The pest population re-establishes itself to pre-control levels and further efforts must be expended to maintain a suitably low population.

The shortfalls of exclusive reliance on population reduction as a pest control strategy are well documented in the field of entomology. These shortfalls include the target species developing a resistance to pesticides, the rapid resurgence of populations following control, and many commonly used pesticides being found to be harmful to the environment (Carson 1963). In response, researchers and managers have moved away from the almost complete reliance on chemical pesticides towards 'integrated pest management' (Horn 1988; Zadoks 1993; Dent

1995). Borror *et al.* (1981) have defined the concept of integrated pest management as :

“... the concept of employing the optimum combination of control methods, including biological, cultural, mechanical, physical and/or chemical control measures, to reduce a pest below an economic threshold, with as few harmful effects as possible on the environment and non-target organisms”.

In recent times, many parallels have developed in the field of vertebrate pest control. Evidence has mounted to demonstrate that control efforts that are solely reliant on a single (usually lethal) control technique are ineffective in the long term (see reviews by Lazarus 1989; Richards 1989; Myers *et al.* 1994). Furthermore, animal welfare concerns have meant that many control methods used in the past are now no longer legal for use (e.g. leg hold traps, some poisons). These developments have prompted many researchers and managers to work toward developing more integrated, strategic and long term approaches to the problems of pest management. For example, rabbit control programs in Australia have evolved from small localised attempts at reducing numbers (Mattams 1921) into large and integrated programs which incorporate a range of mechanical, chemical and biological control agents for the long term control of populations over large areas (Burley 1986; Fenner and Ross 1994; Allen *et al.* 1998; Bomford *et al.* 1998; Sinclair *et al.* 1998).

There are also good reasons why forestry managers and researchers should adopt a more holistic approach (Figure 9.1). The most obvious of these is the potential to achieve superior levels of damage control. A less obvious, but perhaps even more important reason, is linked to public perception of control practices. Species that are problematic for forestry enterprise are often native animals (e.g. Heikkila 1990; Welch *et al.* 1991; Gill 1992; Engeman *et al.* 1995a,b)

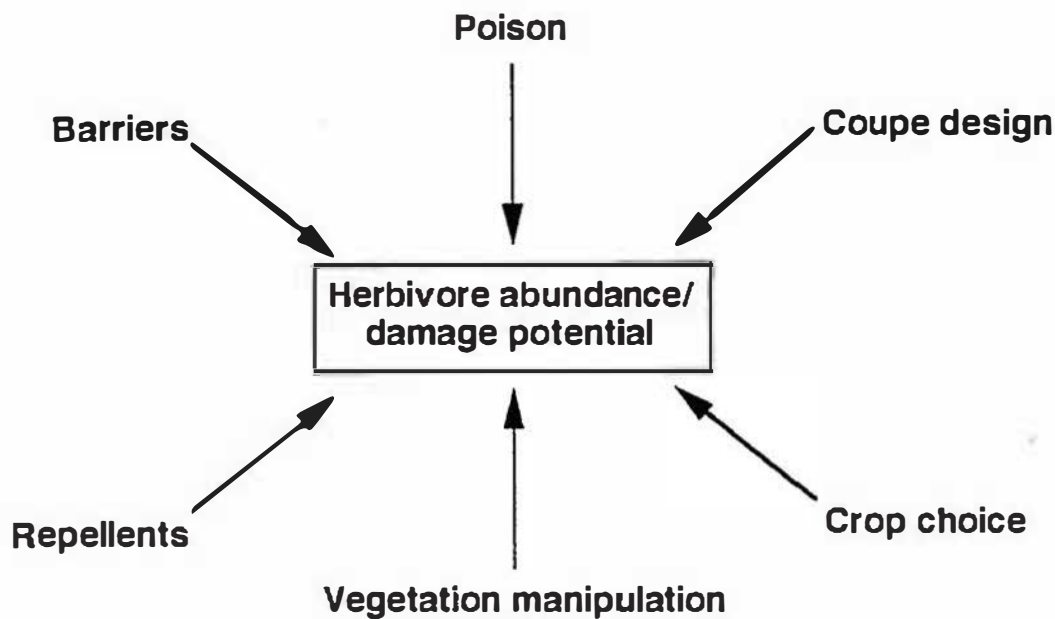


Figure 9.1 : Options for integrated pest management in forestry systems.

and the culling or poisoning of these animals can be heavily criticised by sectors of the community. Under these circumstances strategic and integrated pest management is highly desirable to ensure that population reduction is only carried out where necessary. Furthermore, where managers are required to stop using lethal control entirely, the development of integrated strategies will be of paramount importance, because evidence suggests that existing non-lethal techniques may fail if employed in isolation (Sullivan *et al.* 1987; Andelt *et al.* 1991; Andelt *et al.* 1992; Morgan and Woolhouse 1993).

In context of the vertebrate pest problems faced by forestry managers, I would

suggest that strategic and integrated pest control programs can only be designed where :

- i) the relationship between damage and costs are known, and the damage level at which control is economically justifiable has been identified;
- ii) the relationship between herbivore abundance and damage is known;
- iii) the factors that influence the spatial and temporal distribution of damage are known and the means of predicting damage are available; and
- iv) there is an extensive knowledge of the pest species' feeding behaviour.

This thesis has contributed toward improving our knowledge in each of these areas. Many of the general concepts which were developed are applicable to other forestry pest problems. Following is a discussion of some of the major findings with an integration of work carried out in other forestry/pest systems.

9.1 Economic damage level

One of the more challenging questions for vertebrate pest researchers and managers to answer is : 'At what level of damage or pest abundance is control effort economically justifiable ?' To answer this question, it is necessary to have detailed information on the relationship between damage level and cost.

Describing this relationship can be particularly challenging in forestry systems because of the long rotation length for forestry crops. It is difficult, for example, to determine how damage caused to a seedling just after it is planted will effect the final timber yield when the tree is harvested many years later. Furthermore, actually quantifying damage can be difficult because, in many forestry-pest systems, browsing attacks of varying severity may occur throughout a tree's life.

Perhaps in deference to these problems, the focus of many past studies has been on describing herbivore damage and demonstrating that it has an effect on the crop species, rather than empirically establishing the relationship between damage level and loss in productivity (e.g. Gilbert 1961; Cremer 1969; Heinen and Sharik 1989). More recent research has aimed to improve our understanding in this area (e.g. Abbot *et al.* 1993; Osman and Sharrow 1993; Wilkinson and Nielsen 1995). However, there have still been very few studies which have been carried out with naturally damaged plants (Strauss 1988).

The studies outlined in this thesis aimed to quantify the effect and cost of natural, as opposed to simulated, browsing damage for *E. nitens* seedlings. Damage and growth data were collected during the first year of seedling growth at a range of plantations subject to a variety of damage intensities. The results indicated that browsing damage had two principle effects : promotion of the development of multiple leading shoots and reduced growth rates. Both effects were highly dependent on damage levels. Development of multiple leading shoots was only promoted at plantations with intermediate levels of browsing damage. Growth rate was significantly reduced only when mean browse score measured at 12 months exceeded 0.9. The amount by which average seedling height growth at a plantation (calculated to 12 months) was reduced by herbivore damage followed a positive linear relationship with mean browse score measured at 12 months. Using the mathematical function that describes this relationship, it is now possible to associate meaningful measures of cost (loss in first year growth) to damage severities observed in the field. For example, from the damage assessments obtained in this study, it is estimated that seedling growth was significantly reduced at 56% of *E. nitens* plantations.

9.2 Herbivore abundance and damage

Vertebrate pest control programs inherently assume that there is a positive relationship between pest abundance and damage. In support of this, a number of studies have demonstrated good relationships between the abundance of a pest and damage (e.g. Gorynska 1981; Poche *et al.* 1982; Lefebvre *et al.* 1989). However, few studies have been able to demonstrate strong links for forestry systems (see, for example, the review by Gill 1992). At face value this seems quite surprising, but a closer consideration of the ecology of these systems suggests that there may be good reasons why the link between abundance and damage might be poor, particularly where the pest species has a generalist diet.

Damage is a function not only of the number of pests but also of the amount of damage caused by each individual (i.e. damage per individual). Where damage is measured across space, a strong relationship between abundance and damage will only be evident where damage per individual remains constant across space. Similarly, if damage is measured across time, then strong relationships will only be observed where damage per individual remains constant across time.

In the context of agricultural and forestry systems, the amount of damage caused per individual can be considered to be a function of two separate components of an individual's feeding behaviour :

- i) the frequency with which the individual chooses to feed on the crop; and
- ii) the amount of the crop which is ingested during each of these feeding events.

Both may be influenced by the environment surrounding the herbivore (see Figure 8.1). Consider the work of Saether and Andersen (1990) who studied the foraging behaviour of moose (*Alces alces*) in Norway. They observed that moose fed almost exclusively on birch (*Betula pubescens*) when total food availability was

low. However, when food availability was high moose diets included a much wider range of species and total browsing pressure on birch was lower. Thus, the per individual contribution to birch damage made by the moose was influenced by the availability of alternative food sources.

In a similar way, the amount of damage caused by a herbivore foraging within a plantation may be influenced by the type and amount of alternative vegetation that is available. The magnitude of this influence will be dictated by the feeding preferences of the herbivore. For example, if the crop species occurs in association with alternative plant species of lower palatability then the herbivore might frequently feed on the crop. If, on the other hand, the crop species occurs in association with alternative species of high palatability, then the herbivore might rarely feed on the crop. Studies by Roy (1960), Campbell and Evans (1978), Huss and Olberg-Kalfass (1982, as cited in Gill 1992) and Welch *et al.* (1991) have all found that damage to forestry crops by deer is negatively related to the amount of preferred alternative vegetation that is available.

Because of the potential influences that the environment may have on damage caused per individual, I would suggest that good relationships between pest abundance and damage will only be observed in those instances where :

- i) the pest species has a strong preference for the crop, and this preference is not influenced by the abundance of the crop species; or
- ii) the environment exhibits temporal and/or spatial homogeneity.

The studies outlined in this thesis are the first to attempt to relate the abundance of herbivore species to levels of browsing damage in Tasmanian forestry plantations. No significant relationships were evident for three species : the Bennett's wallaby, pademelon and rabbit. Assuming that scat count data are an

accurate reflection of the abundance of these species, there are two possible explanations for this result. The first is that these species do not contribute to damage. This finding is divergent to the opinions of earlier authors (e.g. Gilbert 1961; Statham 1983) and not wholly supported by the data presented in Figure 7.12, which indicates that damage can still occur at plantations despite the apparent absence of possums. A second explanation is that the damage caused per individual for these species is not constant from plantation to plantation. This seems plausible when the diet of these herbivore species are considered. All three appear to exhibit low preference for eucalypt foliage (Statham 1983; McArthur and Turner 1997a; Sprent 1997) so that the amount of eucalypt foliage which is eaten per individual might be highly influenced by the availability of other plant species.

In contrast to the poor relationships observed for these three species, there was a significant positive linear relationship observed between the average rate of scat deposition for the brushtail possum and mean browse score measured at 12 months ($r^2=0.537$, $df=31$, $p<0.01$). This result would suggest that the brushtail possum is, per capita, one of the more significant contributors to browsing damage. It would also suggest that the damage caused per individual for this species remains relatively consistent across space. This reasoning is supported by dietary studies which indicate that the brushtail possum exhibits very strong preferences for eucalypt foliage (Statham 1983; McArthur and Turner 1997a,b).

Despite the strong preference of possums for eucalypt foliage it is still possible that, on occasion, alternative vegetation characteristics do exert some influence on the amount of damage each individual causes. The regression relating possum abundance to damage was not perfect and some between-site variation in damage remained unexplained. The fact that damage occurred at some sites despite being apparently free from possums, suggests that this variation may in part reflect the

contributions to damage made by other species. It is also possible, however, that this variation is a result of spatial variation in the amount of damage caused per individual possum.

A knowledge of the factors that influence damage per individual would allow the development of an exciting new array of management options. For example, it might be possible to reduce damage to seedlings by manipulating the vegetation community on a plantation so that highly preferred plants are available for herbivores to feed on in preference to seedlings. Campbell and Evans (1978) demonstrated that it was possible to reduce black-tailed deer (*Odocoileus hemionus*) damage to Douglas fir plantations by deliberately seeding them with native herbs. However, management programs must remember that the system shown in Figure 8.1 is a complex and interacting one. Alterations in the environment may affect not only damage per individual, but also the number of individuals in the area. Indeed, Engemen *et al.* (1995a,b) have demonstrated that actually reducing the forage value of an area is an effective means of reducing gopher numbers and hence levels of gopher damage. Attempting to reduce damage through provision of preferred plant species would have proven catastrophic in this situation.

9.3 Factors influencing the spatial distribution of damage

Fundamental to adopting an integrated and strategic pest control program is the ability to either predict the level of damage that will occur at a given site/time and/or the ability to monitor damage levels and initiate control strategies before damage reaches an unacceptable level. Accurate prediction is the ideal strategy since it allows managers more planning time and more options for damage mitigation (e.g. avoid planting crops in damage prone areas). However, there have been surprisingly few studies that have tried to determine what factors influence

the spatial distribution of damage in forestry systems. Fewer still have developed the means for predicting damage. The studies that exist have had mixed success and few, if any, have developed models accurate enough for confident damage prediction.

Welch *et al.* (1991) found that up to 51% of between-site variation in red (*Cervus elaphus*) and roe deer (*Capreolus capreolus*) damage to sitka spruce (*Picea sitchensis*) leaders could be explained using three variables : deer density, ericoid shrubs and the number of trees. Andren and Angelstam (1993) were able to explain up to 40% of between-site variation in moose damage to Scots pine (*Pinus sylvestris*) using two variables : density of pine and density of moose. Hannan and Whelan (1989) were able to develop a moderately successful model ($r^2=0.46$) for describing between-site variation in deer density, but they were unsuccessful in developing models for describing between site variation in damage. In contrast, Kay (1993) was able to explain up to 74% of the between-site variation in damage to one year old coppiced woodland in England, using measures of perimeter length, deer pellet density and presence/absence of fencing.

In the studies outlined in this thesis, a regression model was developed which explained 71% of the between-site variation in mean browse score measured at 12 months. This model incorporated three independent variables. In order of decreasing influence these were cumulative possum scat density, mean grass cover and mean fern cover. An alternate model, formulated using only independent variables that could be measured at the time of planting, was able to explain 52% of between-site variation. The variables incorporated in this model were canopy closure in cover habitats, area to perimeter ratio of the plantation, the proportion of the plantation's edge that was adjacent to cover habitat and mean total vegetation cover, measured from cover habitats at the time of planting.

While these results fall short of actual damage prediction they do provide valuable insight into variables that might be important in influencing damage levels. Furthermore, given the difficulties faced by researchers attempting to explore such complex relationships and their sometimes limited success (e.g. Hannan and Whelan 1989), these results provide very positive encouragement for further attempts at damage prediction.

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APPENDIX A

RESULTS ASSOCIATED WITH CHAPTER 6

Table A.1 : Results of Test 1, the percentage of Bennett's wallaby (B. misid %) and pademelon (P. misid %) scats which were misidentified. Results are presented for each subject and container along with mean and standard deviation values . 'Experienced' subjects are marked with an asterisk.

	Container							
	1	2	3	4	5	6	Mean	sd
Subject 1								
B. misid (%)	-	50.0	25.0	50.0	25.0	16.0	33.2	15.8
P. misid (%)	8.0	22.5	30.0	5.0	0.0	-	13.1	9.0
Subject 2								
B. misid (%)	-	20.0	30.0	33.3	37.5	12.0	26.6	10.4
P. misid (%)	16.0	40.0	43.3	0.0	10.0	-	21.9	19.0
Subject 3								
B. misid (%)	-	30.0	15.0	30.0	22.5	32.0	25.9	7.1
P. misid (%)	30.0	32.5	43.3	10.0	50.0	-	33.2	15.3
Subject 4								
B. misid (%)	-	10.0	10.0	30.0	17.5	16.0	16.7	8.2
P. misid (%)	0.0	20.0	10.0	15.0	60.0	-	21.0	23.0
Subject 5								
B. misid (%)	-	20.0	20.0	26.7	25.0	20.0	22.3	3.3
P. misid (%)	20.0	25.0	23.3	25.0	20.0	-	22.7	2.5
Subject 6								
B. misid (%)	-	20.0	10.0	50.0	20.0	22.0	24.4	15.1
P. misid (%)	16.0	22.5	40.0	50.0	30.0	-	31.7	13.6
Subject 7 *								
B. misid (%)	-	20.0	15.0	26.7	5.0	26.0	18.5	8.9
P. misid (%)	12.0	12.5	10.0	0.0	40.0	-	14.9	14.9
Subject 8 *								
B. misid (%)	-	30.0	20.0	23.3	15.0	16.0	20.9	6.1
P. misid (%)	20.0	27.5	20.0	40.0	10.0	-	23.5	11.1
Subject 9 *								
B. misid (%)	-	10.0	35.0	6.7	42.5	6.0	20.0	17.4
P. misid (%)	18.0	5.0	33.3	40.0	20.0	-	23.3	13.7

Table A.1 (continued) : Results of Test 1, the percentage of Bennett's wallaby (B. misid %) and pademelon (P. misid %) scats which were misidentified. Results are presented for each subject and container along with mean and standard deviation values . 'Experienced' subjects are marked with an asterisk.

	Container							
	1	2	3	4	5	6	Mean	sd
Subject 10 *								
B. misid (%)	-	60.0	55.0	63.3	42.5	56.0	55.4	7.9
P. misid (%)	22.0	37.5	33.3	25.0	50.0	-	33.6	11.1
Subject 11 *								
B. misid (%)	-	30.0	45.0	13.3	27.5	16.0	26.4	12.7
P. misid (%)	54.0	20.0	13.3	15.0	10.0	-	22.5	18.0
Subject 12 *								
B. misid (%)	-	20.0	5.0	16.7	5.0	0.0	9.3	8.6
P. misid (%)	0.0	32.5	56.7	20.0	60.0	-	33.8	25.2
Mean B. misid	-	27.5	23.8	30.8	23.8	19.8		
		sd=14.8	sd=15.1	sd=16.5	sd=12.6	sd=14.2		
Mean P. misid	18.0	24.8	29.7	20.4	30.0	-		
	sd=14.3	sd=10.1	sd=14.8	sd=16.3	sd=21.3			

Table A.2 : Results of Test 2, the percentage of Bennett's wallaby (B. misid %) and pademelon (P. misid %) scats which were misidentified. Results are presented for each subject and container along with mean and standard deviation values . 'Experienced' subjects are marked with an asterisk.

	Container						Mean	sd
	1	2	3	4	5	6		
Subject 1								
B. misid (%)	-	30.0	25.0	46.4	35.0	33.3	33.9	8.0
P. misid (%)	9.1	19.4	26.7	6.3	30.0	-	18.3	10.5
Subject 2								
B. misid (%)	-	30.0	11.1	50.0	22.2	8.7	24.4	16.7
P. misid (%)	20.0	18.8	32.1	5.3	12.5	-	17.7	9.9
Subject 3								
B. misid (%)	-	20.0	5.6	23.3	8.6	11.4	13.8	7.6
P. misid (%)	20.0	12.5	11.1	5.0	11.1	-	11.9	5.4
Subject 4								
B. misid (%)	-	0.0	15.8	16.7	12.5	8.2	10.6	6.8
P. misid (%)	22.2	28.2	42.9	20.0	50.0	-	32.7	13.2
Subject 5								
B. misid (%)	-	10.0	10.5	7.4	13.2	6.7	9.6	2.6
P. misid (%)	14.9	34.3	13.8	5.9	22.2	-	18.22	10.7
Subject 6								
B. misid (%)	-	60.0	36.4	39.3	30.3	42.9	41.8	11.2
P. misid (%)	22.7	15.8	26.1	33.3	0.0	-	19.6	12.6
Subject 7 *								
B. misid (%)	-	20.0	12.5	16.7	17.6	21.1	17.6	3.3
P. misid (%)	15.8	7.7	16.7	41.1	0.0	-	16.3	15.5
Subject 8 *								
B. misid (%)	-	20.0	5.0	15.0	21.7	25.5	17.4	7.9
P. misid (%)	11.6	12.9	28.6	0.0	50.0	-	20.6	19.3
Subject 9 *								
B. misid (%)	-	30.0	26.3	17.9	35.9	6.1	23.2	11.6
P. misid (%)	10.4	2.7	15.4	35.0	44.4	-	21.6	17.5

Table A.2 (continued) : Results of Test 2, the percentage of Bennett's wallaby (B. misid %) and pademelon (P. misid %) scats which were misidentified. Results are presented for each subject and container along with mean and standard deviation values . 'Experienced' subjects are marked with an asterisk.

	Container							
	1	2	3	4	5	6	Mean	sd
Subject 10 *								
B. misid (%)	-	0.0	0.0	37.5	23.1	44.4	21.0	20.7
P. misid (%)	7.0	7.7	21.7	11.1	0.0	-	9.5	7.9
Subject 11 *								
B. misid (%)	-	16.7	16.7	25.9	13.3	41.2	22.8	11.3
P. misid (%)	24.3	31.0	56.5	26.7	75.0	-	42.7	22.2
Subject 12 *								
B. misid (%)	-	10.0	0.0	22.7	2.5	6.0	8.2	8.9
P. misid (%)	0.0	17.5	42.1	17.6	0.0	-	15.4	17.3
Mean B. misid	-	17.0	13.7	26.6	19.7	21.3		
		sd=11.0	sd=11.1	sd=13.6	sd=10.4	sd=15.6		
Mean P. misid	14.8	17.4	27.8	17.3	24.6	-		
	sd=7.4	sd=9.7	sd=13.7	sd=13.9	sd=25.2			

Table A.3 : The results of Test 2, the percentage of Bennett's wallaby (B. unid %) and pademelon (P. unid %) scats which could not be confidently identified. Results are presented for each subject and container along with mean and standard deviation values. 'Experienced' subjects are marked with an asterisk.

	Container							
	1	2	3	4	5	6	Mean	sd
Subject 1								
B. unid (%)	-	0.0	0.0	6.7	0.0	10.0	3.3	4.7
P. unid (%)	12.0	10.0	0.0	20.0	0.0	-	14.0	5.3
Subject 2								
B. unid (%)	-	0.0	10.0	20.0	10.0	8.0	9.6	7.1
P. unid (%)	10.0	20.0	6.7	5.0	20.0	-	12.3	7.2
Subject 3								
B. unid (%)	-	0.0	10.0	0.0	12.5	12.0	6.9	6.4
P. unid (%)	10.0	20.0	10.0	0.0	10.0	-	10.0	7.1
Subject 4								
B. unid (%)	-	20.0	5.0	0.0	0.0	2.0	5.4	8.4
P. unid (%)	10.0	2.5	6.7	0.0	0.0	-	3.8	4.4
Subject 5								
B. unid (%)	-	0.0	5.0	10.0	5.0	10.0	6.0	4.2
P. unid (%)	6.0	12.5	3.3	15.0	10.0	-	9.4	4.7
Subject 6								
B. unid (%)	-	0.0	45.0	6.7	17.5	30.0	19.8	18.1
P. unid (%)	12.0	5.0	23.3	25.0	20.0	-	17.1	8.4
Subject 7 *								
B. unid (%)	-	0.0	20.0	40.0	15.0	24.0	19.8	14.5
P. unid (%)	24.0	35.0	20.0	15.0	50.0	-	28.8	14.0
Subject 8 *								
B. unid (%)	-	0.0	0.0	33.3	17.5	6.0	11.4	14.2
P. unid (%)	14.0	22.5	6.7	0.0	40.0	-	16.6	15.5
Subject 9 *								
B. unid (%)	-	0.0	5.0	6.7	2.5	2.0	3.2	2.6
P. unid (%)	4.0	7.5	13.3	0.0	10.0	-	7.0	5.2

Table A.3 (continued) : The results of Test 2, the percentage of Bennett's wallaby (B. unid %) and pademelon (P. unid %) scats which could not be confidently identified. Results are presented for each subject and container along with mean and standard deviation values. 'Experienced' subjects are marked with an asterisk.

	Container							
	1	2	3	4	5	6	Mean	sd
Subject 10 *								
B. unid (%)	-	70.0	50.0	46.7	35.0	46.0	49.5	12.8
P. unid (%)	14.0	35.0	23.3	10.0	10.0	-	18.5	10.7
Subject 11 *								
B. unid (%)	-	40.0	26.7	16.7	62.5	32.0	35.6	17.3
P. unid (%)	26.0	27.5	23.3	25.0	60.0	-	32.4	15.5
Subject 12 *								
B. unid (%)	-	0.0	0.0	26.7	0.0	0.0	5.3	11.9
P. unid (%)	0.0	0.0	36.7	15.0	0.0	-	10.3	16.1
Mean B. unid	-	10.8	14.7	17.8	14.8	15.2		
		sd=22.3	sd=17.3	sd=15.7	sd=18.2	sd=14.5		
Mean P. unid	11.8	16.5	14.4	10.8	19.2	-		
	sd=7.4	sd=12.1	sd=10.9	sd=9.7	sd=20.2			

Table A4 : The mean density (scats/m²) estimates of pademelon and Bennett's wallaby scats obtained from plots of varying sizes. Data is presented for each of the ten sites. Standard deviations are given in parenthesis.

Bennett's wallaby

Size (m ²)	Site									
	Adu29	Bfr	Bf1	Bf2	Bri	Bwa	Ndi	Npu	Nrit	Nsp
5	0.45 (0.44)	0.00 (0.00)	1.02 (0.64)	0.66 (0.64)	0.68 (0.43)	0.28 (0.33)	0.39 (0.28)	1.05 (0.50)	0.23 (0.25)	0.00 (0.00)
10	0.49 (0.34)	0.55 (0.61)	0.92 (0.83)	0.60 (0.41)	0.77 (0.45)	0.24 (0.29)	0.48 (0.29)	0.84 (0.39)	0.39 (0.56)	0.00 (0.00)
15	0.54 (0.33)	0.40 (0.45)	0.86 (0.76)	0.77 (0.28)	0.75 (0.50)	0.27 (0.26)	0.53 (0.37)	0.87 (0.37)	0.35 (0.41)	0.00 (0.00)
20	0.54 (0.34)	0.43 (0.52)	0.85 (0.69)	0.86 (0.40)	0.65 (0.40)	0.28 (0.22)	0.51 (0.35)	0.86 (0.36)	0.35 (0.32)	0.01 (0.02)
25	0.57 (0.31)	0.44 (0.53)	1.00 (0.76)	0.82 (0.28)	0.62 (0.37)	0.30 (0.22)	0.46 (0.28)	0.87 (0.32)	0.37 (0.30)	0.01 (0.02)

pademelon

Size (m ²)	Site									
	Adu29	Bfr	Bf1	Bf2	Bri	Bwa	Ndi	Npu	Nrit	Nsp
5	0.41 (0.37)	0.30 (0.28)	1.42 (1.20)	0.70 (0.60)	1.18 (0.92)	0.65 (0.59)	0.22 (0.21)	0.34 (0.23)	0.09 (0.01)	0.00 (0.00)
10	0.34 (0.32)	0.22 (0.22)	1.04 (0.98)	0.60 (0.40)	1.11 (0.85)	0.63 (0.49)	0.23 (0.16)	0.35 (0.19)	0.10 (0.08)	0.00 (0.00)
15	0.32 (0.27)	0.21 (0.19)	0.82 (0.73)	0.81 (0.25)	1.03 (0.70)	0.67 (0.47)	0.31 (0.14)	0.36 (0.15)	0.10 (0.08)	0.00 (0.00)
20	0.29 (0.23)	0.37 (0.30)	0.74 (0.64)	0.89 (0.40)	0.89 (0.57)	0.65 (0.43)	0.29 (0.13)	0.34 (0.14)	0.11 (0.08)	0.01 (0.02)
25	0.29 (0.22)	0.41 (0.26)	0.77 (0.64)	0.78 (0.31)	0.84 (0.51)	0.63 (0.40)	0.34 (0.16)	0.38 (0.16)	0.09 (0.07)	0.01 (0.02)

Table A4 (continued) : The mean density (scats/m²) estimates of total macropod (includes Bennett's wallaby, pademelon and macropod scats not identified to species level) and 'other' herbivore scats obtained from plots of varying sizes. Data is presented for each of the ten sites. Standard deviations are given in parenthesis.

total macropod

Size (m ²)	Site									
	Adu29	Bfr	Brfl	Brf2	Bri	Bwa	Ndi	Npu	Nrit	Nsp
5	0.89 (0.82)	0.00 (0.28)	2.44 (1.70)	1.12 (0.59)	1.97 (1.36)	0.96 (0.71)	0.61 (0.35)	1.23 (0.52)	0.30 (0.25)	0.01 (0.02)
10	0.87 (0.70)	0.15 (0.20)	1.98 (1.74)	1.07 (0.46)	1.97 (1.34)	0.89 (0.61)	0.72 (0.28)	0.99 (0.42)	0.47 (0.56)	0.01 (0.01)
15	0.89 (0.61)	0.58 (0.19)	1.69 (1.46)	1.37 (0.40)	1.88 (1.20)	0.95 (0.55)	0.81 (0.35)	1.03 (0.44)	0.45 (0.43)	0.00 (0.01)
20	0.86 (0.57)	0.51 (0.20)	1.63 (1.30)	1.44 (0.49)	1.63 (0.96)	0.96 (0.49)	0.81 (0.38)	1.02 (0.45)	0.46 (0.35)	0.01 (0.02)
25	0.89 (0.53)	0.69 (0.18)	1.80 (1.36)	1.35 (0.39)	1.49 (0.83)	0.95 (0.46)	0.81 (0.32)	1.08 (0.44)	0.47 (0.32)	0.02 (0.02)

other herbivores

Size (m ²)	Site									
	Adu29	Bfr	Brfl	Brf2	Bri	Bwa	Ndi	Npu	Nrit	Nsp
5	0.00 (0.00)	0.00 (0.00)	0.24 (0.69)	0.06 (0.18)	0.00 (0.00)	0.06 (0.19)	0.01 (0.05)	0.00 (0.00)	0.01 (0.04)	0.02 (0.09)
10	0.00 (0.00)	0.00 (0.00)	0.13 (0.34)	0.04 (0.09)	0.00 (0.00)	0.04 (0.11)	0.11 (0.46)	0.00 (0.00)	0.01 (0.02)	0.01 (0.04)
15	0.00 (0.00)	0.00 (0.00)	0.09 (0.23)	0.03 (0.06)	0.04 (0.14)	0.03 (0.07)	0.07 (0.31)	0.02 (0.10)	0.01 (0.01)	0.02 (0.05)
20	0.00 (0.00)	0.02 (0.02)	0.07 (0.17)	0.05 (0.09)	0.12 (0.23)	0.03 (0.06)	0.05 (0.23)	0.08 (0.18)	0.01 (0.03)	0.01 (0.04)
25	0.01 (0.01)	0.01 (0.01)	0.06 (0.14)	0.08 (0.12)	0.12 (0.18)	0.04 (0.11)	0.06 (0.21)	0.07 (0.15)	0.01 (0.02)	0.01 (0.03)

Table A5 : The coefficients of variation calculated for each plot size at each site.

Bennett's wallaby

		Site								
Size (m ²)	Adu29	Bfr	Brfl	Brf2	Bri	Bwa	Ndi	Npu	Nrit	Nsp
5	30.78	-	19.72	30.48	20.09	37.27	22.87	15.00	33.83	-
10	22.07	35.2	28.46	21.82	18.48	38.74	18.98	14.68	45.25	-
15	19.33	35.6	28.02	11.58	21.00	30.22	22.20	13.52	37.41	-
20	19.68	38.0	25.67	14.78	19.27	24.40	21.58	13.24	29.28	75.90
25	16.98	38.0	24.10	10.88	18.67	22.77	19.39	11.56	25.64	75.90

pademelon

		Site								
Size (m ²)	Adu29	Bfr	Brfl	Brf2	Bri	Bwa	Ndi	Npu	Nrit	Nsp
5	28.69	29.10	26.73	27.11	24.60	28.61	30.19	21.77	33.73	-
10	30.14	31.90	29.74	20.87	24.27	24.70	21.45	17.35	24.67	-
15	26.68	28.00	28.23	9.84	21.55	22.37	14.08	13.18	26.57	-
20	24.86	25.60	27.18	14.29	20.25	20.73	13.74	12.84	24.15	75.90
25	23.56	19.90	26.37	12.34	19.20	19.88	14.51	13.48	25.30	75.90

Table A5 : The coefficients of variation calculated for each plot size at each site.

Total macropods

Size (m ²)	Site									
	Adu29	Bfr	Bfl	Brf2	Bri	Bwa	Ndi	Npu	Nrit	Nsp
5	29.00	-	22.09	16.77	21.77	23.32	18.04	13.42	25.93	75.90
10	25.30	43.00	27.79	13.66	21.48	21.53	12.12	13.42	37.95	38.00
15	21.53	10.10	27.28	9.28	19.98	18.18	13.59	13.63	30.36	-
20	20.96	12.70	25.26	10.67	18.63	16.01	14.99	13.95	24.34	75.90
25	18.98	8.25	23.93	9.14	17.57	15.18	12.42	13.00	21.80	38.00

other herbivores

Size (m ²)	Site									
	Adu29	Bfr	Bfl	Brf2	Bri	Bwa	Ndi	Npu	Nrit	Nsp
5	-	-	94.28	94.88	-	71.10	109.00	-	97.50	97.47
10	-	-	90.56	72.46	-	30.77	109.00	-	97.50	97.47
15	-	-	84.35	72.46	95.96	16.67	109.00	95.10	97.50	68.56
20	-	91.31	85.07	60.01	52.34	12.50	109.00	53.70	67.10	68.56
25	97.02	91.31	70.67	47.44	43.58	11.13	85.50	45.30	67.10	68.56

APPENDIX B

SITE CHARACTERISTICS

Table B.1 : Mean cover (%) of vegetation components measured on each plantation to six months.

Plantation	Vegetation component					Total vegetation
	Grasses	Ferns	Forbs	Shrubs & trees	Bryophytes	
Adu29	0.2	0.1	1.0	0.4	0.0	3.8
Alo48	25.6	0.0	27.2	0.8	4.1	37.2
Alo49	28.6	0.0	30.8	0.8	2.8	43.7
Amark	45.7	5.7	12.4	0.5	4.2	61.4
Aseol	41.7	1.9	20.2	0.0	2.6	63.8
Asx13	1.5	3.3	5.6	0.2	0.7	16.3
Asx51	2.3	0.6	10.5	0.6	0.1	10.7
Atn18	8.2	0.0	4.3	0.0	3.6	17.3
Atn20	4.0	0.0	2.9	1.4	8.0	15.6
Aww55	16.0	2.5	6.5	0.0	2.0	26.2
Bcam	0.0	2.0	15.3	0.0	0.0	16.5
Bfa	0.9	0.8	1.6	0.2	0.0	6.8
Bkui	1.1	1.8	4.0	0.5	0.7	9.3
Brfl	2.1	0.5	2.2	0.2	0.0	9.7
Brf2	0.3	1.2	0.7	0.0	0.0	3.9
Brin	0.1	0.6	3.7	0.0	0.0	6.7
Bsch	3.0	4.3	12.5	0.1	0.0	19.5
Bswan	7.5	5.1	11.0	2.9	0.6	27.6
Bswan2	9.2	7.9	9.3	3.4	3.6	30.2
Bwal	7.3	1.6	4.4	0.0	0.0	14.3
Bwee	10.7	4.7	15.9	0.1	0.2	31.6
Bwfr	32.0	0.0	29.3	0.0	0.0	61.0
Fha21h	15.7	3.3	4.1	0.8	10.6	31.0
Fhp23g	11.7	1.0	3.6	0.6	0.3	17.5
Fkd13b	0.7	0.1	3.1	2.2	3.4	11.0
Fsolb	5.0	5.1	2.0	0.5	0.6	14.3
Nalans	0.9	7.1	1.0	0.1	0.0	14.6
Nbless	6.5	2.8	9.0	0.2	0.0	19.5
Ncw	0.7	0.0	3.6	0.0	0.3	4.7
Ndig	0.7	1.8	1.0	0.1	0.0	8.3
Npear	0.4	0.3	7.4	0.1	0.4	9.8
Npedd	0.1	0.6	10.4	1.7	0.0	12.7
Npug	0.1	2.0	1.9	0.2	0.0	7.3
Nrit	1.0	2.9	0.3	0.0	0.0	8.3
Nspill	2.2	0.7	18.5	0.0	0.0	22.2

Table B.2 : Mean cover (%) of vegetation components measured on each plantation to 12 months.

Plantation	Vegetation component					Total vegetation
	Grasses	Ferns	Forbs	Shrubs & trees	Bryophytes	
Adu29	0.6	0.4	6.4	1.2	0.2	12.8
Alo48	31.3	0.0	25.9	0.5	5.2	50.8
Alo49	32.9	0.0	31.1	0.5	3.6	54.4
Amark	50.2	6.5	12.5	0.6	5.2	66.7
Aseol	45.0	1.6	22.1	0.0	4.3	68.9
Asx13	3.3	3.5	5.2	0.2	5.8	20.5
Asx51	3.2	0.5	10.9	0.5	0.1	13.5
Atn18	10.5	0.0	6.1	0.0	8.5	26.1
Atn20	4.6	0.0	3.6	1.6	11.4	19.6
Aww55	23.3	5.5	9.8	0.0	2.2	38.0
Bcam	0.4	2.8	19.6	0.8	0.7	23.3
Bfia	2.4	2.4	4.7	0.1	0.0	12.8
Bkui	2.7	2.9	8.0	2.1	3.1	19.0
Brf1	4.1	1.3	2.4	0.2	0.0	12.3
Brf2	1.0	2.4	1.2	0.0	0.2	7.0
Brin	1.1	2.8	6.0	0.6	0.0	13.0
Bsch	5.7	5.4	15.7	0.1	0.0	27.3
Bswan	12.2	6.0	14.9	4.3	2.1	36.6
Bswan2	10.4	8.5	11.1	4.5	4.0	35.6
Bwal	10.3	3.9	9.3	0.2	0.1	24.8
Bwee	18.2	6.0	19.0	0.4	1.0	40.8
Bwfr	35.5	0.0	30.7	0.0	0.4	65.7
Fha21h	18.3	3.6	5.3	1.0	13.2	36.4
Fhp23g	19.9	1.3	9.5	1.7	2.0	33.3
Fkd13b	1.8	0.1	4.8	5.2	5.2	18.4
Fsolb	8.2	7.7	4.2	1.8	5.2	26.8
Nalans	1.4	8.4	1.6	0.2	0.1	17.4
Nbless	12.1	3.3	12.8	0.3	0.7	30.2
Ncw	1.1	0.0	5.1	0.0	0.2	6.5
Ndig	1.1	3.1	3.1	0.9	0.1	13.0
Npear	1.0	0.4	12.7	0.1	0.5	15.7
Npedd	0.5	0.8	17.3	4.3	0.2	22.8
Npug	0.2	3.5	2.7	1.7	0.1	12.4
Nrit	0.8	4.7	0.5	0.0	0.1	10.4
Nspill	5.5	1.8	21.9	0.4	0.0	31.4

Table B.3 : Measures of environmental variables made on each plantation including average slope, altitude, aspect, area, perimeter length, the proportion of the perimeter adjacent to cover habitat (P. adjacent to cover), and the number of 1080 poisoning operations that were carried out (Poisoning frequency).

Environmental variable							
Plantation	Slope (degrees)	Altitude (m)	Aspect	Area (ha)	Perimeter (km)	P. adjacent to cover	Poisoning frequency
Adu29	12	490	NW	24.0	2.4	0.74	1
Alo48	1	365	E	59.6	5.3	0.99	1
Alo49	1	395	NE	12.7	2.2	0.99	1
Amark	4	1305	E	10.4	2.3	0.92	1
Aseol	2	385	SW	17.8	2.4	1.00	1
Asx13	3	460	NE	16.0	2.0	1.00	0
Asx51	14	475	W	26.7	3.0	0.55	1
Atn18	14	375	S	14.7	1.8	0.49	1
Atn20	6	405	E	30.4	2.9	1.00	2
Awv55	5	600	NE	80.0	3.6	1.00	2
Bcam	11	690	NE	12.9	1.5	0.99	1
Bfra	25	345	S	9.6	1.5	0.31	1
Bkui	10	450	SE	12.5	1.6	0.91	2
Brf1	11	560	NE	9.5	1.5	0.45	0
Brf2	11	165	NW	12.7	1.7	0.35	0
Brin	5	225	W	13.5	2.0	0.85	0
Bsch	14	505	S	40.5	2.8	1.00	1
Bswan	13	255	W	18.0	2.5	0.99	1
Bswan2	11	255	E	37.0	2.7	1.00	1
Bwal	5	130	NW	21.0	2.6	0.29	1
Bwee	16	160	SW	43.1	3.8	1.00	1
Bwfr	2	175	W	10.5	1.5	0.57	1
Fha21b	8	45	SW	27.2	2.3	1.00	1
Fhp23g	12	195	SW	38.0	3.1	1.00	1
Fkd13b	13	175	NE	52.0	3.2	1.00	2
Fsolb	9	135	W	64.0	4.0	0.94	2
Nalans	5	250	NE	108.8	4.7	0.66	1
Nbless	16	510	SW	41.0	3.2	0.24	1
Ncw	2	630	NW	42.9	2.9	0.79	0
Ndig	9	250	S	17.0	2.0	0.77	0
Npear	1	645	NE	34.9	2.8	0.54	1
Npedd	9	365	NE	20.0	2.1	0.99	1
Npug	10	340	W	39.0	3.8	0.98	0
Nrit	12	330	NE	24.0	3.5	0.89	0
Nspill	13	275	SE	28.0	2.7	0.51	0

Table B.4 : Measures of environmental variables made from cover habitats adjoining each plantation and the nearest weather station; including canopy closure, tree density (trees.m⁻²), shrub density (shrubs.m⁻²), mean tree diameter (cm), annual rainfall (mm) calculated from data for all recorded years, and the amount (mm) of rain to fall during the one year sampling period.

Environmental variable						
Plantation	Canopy closure (%)	Tree density	Shrub density	Mean tree diameter	Annual rainfall	Rain during sampling
Adu29	82	0.14	0.14	50.2	520	751
Alo48	10	0.03	1.40	5.8	942	1145
Alo49	10	0.05	1.60	4.3	942	1145
Amark	75	0.17	0.05	36.3	520	683
Aseol	25	0.04	1.70	4.9	942	1145
Asx13	80	0.03	0.09	119.7	943	1451
Asx51	75	0.05	0.80	38.9	783	1199
Atn18	76	0.19	0.36	16.1	976	1477
Atn20	86	0.33	0.84	33.6	836	1218
Awv55	36	0.08	0.70	13.1	1145	1534
Bcam	65	0.20	0.50	8.3	1131	1749
Bfra	86	0.09	0.14	53.1	1176	1145
Bkui	78	0.45	0.80	5.8	903	1901
Brf1	70	0.04	0.15	89.1	695	876
Brf2	22	0.03	0.80	38.3	695	876
Brin	72	0.03	0.30	102.5	1176	1217
Bsch	82	0.12	1.00	11.5	1131	1749
Bswan	60	0.02	0.09	12.3	819	1180
Bswan2	63	0.01	0.05	15.2	819	784
Bwal	65	0.06	0.04	23.5	929	867
Bwee	30	0.04	0.90	6.22	950	1353
Bwfr	72	0.04	0.09	11.5	1059	2248
Fha21h	68	0.17	1.90	11.6	1348	1491
Fhp23g	88	0.18	1.70	8.7	838	1294
Fkd13b	68	0.27	1.20	9.8	838	1280
Fsolb	64	0.16	3.60	12.3	1348	1912
Nalans	42	0.01	0.02	15.2	819	784
Nbless	54	0.02	0.40	25.9	883	1067
Ncw	38	0.11	0.47	11.8	1772	2022
Ndig	66	0.04	0.47	101.7	1059	1829
Npear	70	0.35	1.02	7.8	2154	2208
Npedd	88	0.11	0.34	16.4	1176	1461
Npug	40	0.06	1.91	36.6	819	768
Nrit	46	0.04	0.02	102.9	928	777
Nspill	78	0.04	0.06	67.6	1160	1002

Table B.5 : Cover (%) of vegetation components measured from cover habitat adjoining each plantation, shortly after (<30 days) seedlings were planted.

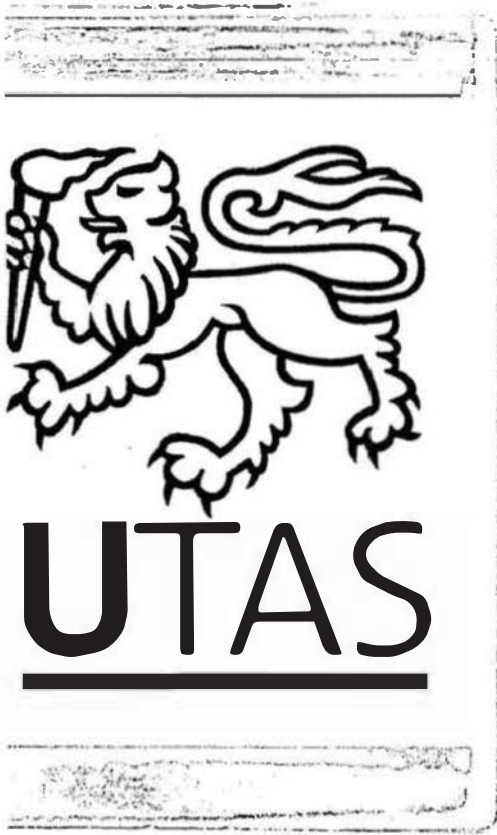
Plantation	Vegetation component					Total vegetation
	Grasses	Ferns	Forbs	Shrubs & trees	Bryophytes	
Adu29	0.0	12.0	0.6	0.4	11.8	28.0
Alo48	12.7	5.3	18.3	10.2	3.8	43.8
Alo49	10.3	6.2	17.9	13.3	0.0	46.2
Amark	5.2	10.3	10.2	0.8	8.2	33.3
Aseol	28.5	5.2	20.7	5.6	2.2	60.3
Asx13	0.2	12.4	0.8	0.8	16.0	29.6
Asx51	18.7	17.3	8.2	6.2	4.2	52.0
Atn18	0.0	8.4	0.8	4.0	10.4	24.2
Atn20	1.2	10.7	2.3	8.3	22.1	40.3
Aww55	20.2	20.6	6.6	0.0	4.8	49.0
Bcam	3.2	2.7	6.3	5.2	0.0	16.7
Bfra	0.2	3.0	1.8	2.2	3.4	13.0
Bkui	0.4	8.2	0.4	3.4	8.4	20.6
Brf1	0.0	13.2	5.4	0.6	0.0	21.2
Brf2	3.0	8.0	20.0	8.4	3.4	42.2
Brin	0.0	6.2	1.0	3.6	3.6	17.0
Bsch	7.8	24.6	6.6	3.4	3.4	43.8
Bswan	5.3	2.7	10.2	7.8	7.7	34.7
Bswan2	11.4	0.0	7.2	10.7	0.0	27.3
Bwal	3.0	5.2	10.6	17.3	5.1	39.8
Bwee	18.2	9.8	21.0	11.6	0.6	56.2
Bwfr	17.1	5.1	20.1	8.9	2.5	52.7
Fha21h	2.7	18.3	7.2	4.2	0.0	30.7
Fhp23g	3.0	19.8	5.4	3.8	7.2	37.0
Fkd13b	0.0	19.6	5.8	1	9.4	36.0
Fsolb	1.2	19.0	8.8	17.4	6.2	47.4
Nalans	10.3	10.3	20.2	10.1	0.0	47.6
Nbless	16.2	12.4	14.6	10.0	1.2	48.2
Ncw	11.0	6.2	20.0	8.6	12.6	54.6
Ndig	0.2	7.8	6.4	0.4	1.0	18.8
Npear	1.6	7.6	5.4	7.4	35.6	54.4
Npedd	0.0	4.8	0.6	0.4	7.4	11.4
Npug	1.6	3.2	22.8	19.8	0.2	38.2
Nrit	11.4	8.2	5.6	6.2	1.6	32.2
Nspill	11.0	27.4	12.6	4.8	1.2	54.8

Table B.6 : Mean cover (% calculated to 12 months) of vegetation components measured from the cover habitat adjoining each plantation.

Plantation	Vegetation component					Total vegetation
	Grasses	Ferns	Forbs	Shrubs & trees	Bryophytes	
Adu29	1.5	11.1	0.9	0.5	10.0	24.6
Alo48	11.4	6.0	20.1	9.8	5.3	49.4
Alo49	11.6	6.4	18.5	14.1	0.0	48.8
Amark	5.3	9.8	9.1	1.2	8.9	32.4
Aseol	25.0	6.0	20.7	5.9	3.9	59.7
Asx13	0.1	11.9	1.3	1.0	14.6	27.4
Asx51	18.7	16.9	8.2	5.8	6.0	52.3
Atn18	1.0	8.9	1.3	4.2	7.9	22.5
Atn20	1.2	11.5	3.1	8.4	20.0	41.4
Awv55	18.6	18.7	5.9	0.3	6.2	46.5
Bcam	4.1	3.4	7.1	5.3	0.9	20.3
Bfra	0.5	3.4	2.6	2.3	3.8	12.8
Bkui	0.5	8.5	1.0	3.7	7.9	19.3
Brf1	0.4	12.5	5.4	1.1	1.3	19.8
Brf2	2.2	8.4	18.4	7.8	3.6	36.1
Brin	0.0	7.2	1.9	4.0	4.2	16.2
Bsch	7.3	22.7	7.4	4.1	4.1	41.8
Bswan	5.5	3.2	11.1	8.3	8.3	33.5
Bswan2	11.8	0.0	8.0	10.4	0.5	27.6
Bwal	4.3	5.1	11.6	16.2	6.4	40.6
Bwee	17.2	9.3	20.1	11.5	1.4	55.2
Bwfr	17.5	5.7	19.1	9.6	3	51.6
Fha21h	3.6	18.5	7.6	5.8	0.9	34.5
Fhp23g	3.6	19.2	6.0	6.3	7.7	38.5
Fkd13b	0.4	19.4	5.1	2.7	9.4	35.8
Fsolb	1.7	18.0	8.7	15.5	3.0	44.6
Nalans	8.8	12.5	20.1	11.2	0.6	50.5
Nbless	14.8	11.1	14.8	10.0	4.5	48.9
Ncw	10.8	7.0	19.7	10.5	11.9	56.3
Ndig	0.7	8.2	6.6	1.0	1.9	18.8
Npear	1.7	7.9	6.0	6.2	32.6	52.1
Npedd	0.4	5.5	3.3	1.1	7.8	14.6
Npug	1.6	4.1	21.7	13.7	0.4	37.5
Nrit	10.9	7.9	5.9	7.3	1.9	31.2
Nspill	11.5	26.3	11.5	5.4	1.7	53.6

APPENDIX C

CORRELATION ANALYSES ASSOCIATED WITH CHAPTER 8



[illegible]

[illegible]

Table C.3 : Correlation matrix showing the results of pair-wise correlation analyses for the independent vegetation cover variables measured from the coupe and from cover habitat. Included are grass cover (grass), fern cover (fern), forb cover (forb), shrub and tree cover (S&T), moss cover (moss) and total vegetation cover (total). The abbreviation 'ch' has been used to denote those variables measured from cover habitat. Some of the variables have been transformed (denoted by the abbreviation 'T') and Tables C.1 and C.2 should be consulted for details. Values for Pearson's correlation coefficient (r) are given for each pair-wise comparison.

[illegible]

Table C.4 : Results of correlation analyses relating coupe characteristics to cumulative scat density for the brushtail possum, macropod grouping, Bennett's wallaby and Tasmanian pademelon. Arcsin square root transformations have been performed on those variables preceded by '(T)'. A single asterisk is used to indicate results significant at the $p < 0.05$ level and '**' for $p < 0.01$.

Coupe characteristic	Cumulative herbivore scat density				
	Brushtail Possum	Bennett's wallaby	Tasmanian pademelon	Macropods	Rabbit
Altitude	0.425	0.650	0.457	0.746 *	0.153
Slope	-0.589	-0.368	-0.489	-0.194	-0.063
Area	0.380	0.365	-0.258	-0.081	0.860 **
Length of perimeter	0.354	0.155	-0.356	-0.269	0.675 *
Area to perimeter ratio	0.224	0.331	-0.208	-0.012	0.809 **
(T) Proportion of perimeter adjacent to cover	0.580	0.302	0.321	-0.060	-0.009
Average annual rainfall	0.234	-0.007	-0.235	-0.153	0.682 *
(T) Mean grass cover	-0.543	-0.283	0.036	0.182	-0.301
(T) Mean fern cover	-0.189	-0.368	-0.072	-0.407	-0.512
(T) Mean forb cover	-0.373	-0.345	-0.244	-0.260	0.223
(T) Mean shrub and tree cover	0.023	0.167	0.252	0.015	-0.247
(T) Mean moss cover	0.149	0.188	0.403	0.046	-0.342
(T) Mean total vegetation cover	-0.507	-0.425	-0.145	-0.263	-0.196

Table C.5 : Results of correlation analyses relating cover habitat characteristics and rainfall data to cumulative scat density for the brushtail possum, Bennett's wallaby, Tasmanian pademelon, macropod grouping and rabbit. Vegetation cover variables are an average of cover measures taken at 0, 6 and 12 months after planting. Arcsin square root transformations have been performed on those variables that are preceded by '(T)'. A single asterisk is used to indicate results significant at the $p<0.05$ level and '**' for $p<0.01$.

Variable measured	Cumulative herbivore scat density				
	Brushtail Possum	Bennett's wallaby	Tasmanian pademelon	Macropods	Rabbit
Canopy closure	-0.291	-0.294	0.228	0.059	-0.512
Tree density	0.456	0.529	-0.042	0.262	0.769 *
Mean tree diameter	-0.257	-0.443	0.168	-0.106	-0.745 *
Shrub density	0.402	0.582	0.189	0.099	0.218
Mean grass cover	-0.183	-0.242	-0.761 *	-0.344	0.824 **
Mean fern cover	-0.711 *	-0.524	-0.326	-0.163	-0.072
Mean forb cover	0.084	0.357	-0.316	0.009	0.700 *
Mean shrub/tree cover	0.439	0.253	-0.323	-0.213	0.816 **
Mean moss cover	0.294	0.130	0.272	0.015	0.055
Mean total vegetation cover	-0.018	0.077	-0.467	-0.193	0.837 **