Methods for Assessment and Techniques for Management of Striped Legless Lizard *Delma impar* Populations in South-eastern Australia

By

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Statement of Originality

This thesis is my original work and has not been submitted in whole or in part, for a degree at this or any other university. Nor does it contain, to the best of my knowledge and belief, any material published or written by another person, except as acknowledged in the text.

Megan Bernadette O’Shea

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Abstract

Keywords: Striped Legless Lizard, *Delma impar*, Pygopodidae, Grasslands, Western Basalt Plains Grasslands, Pit-fall trapping, artificial shelters, roof tiles, individual recognition, harmonic direction-finder, biomass reduction, fire.

The Striped Legless Lizard *Delma impar* has been studied over a five year period at Iramoo Wildlife Reserve, some 18 kilometres west of Melbourne, Victoria. This site contains the largest documented population of *D. impar* in Australia.

The population appeared robust over the study period and was present in relatively undisturbed grassland habitat dominated by Kangaroo Grass *Themeda triandra* and also in disturbed grassland habitat dominated by the weeds Serrated Tussock *Nassella trichotoma* and Chilean Needle Grass *Nassella neesiana*. *Delma impar* also persisted well in survey plots where the grassy habitat was burnt in each of two or one years and on unburnt sites.

The first two years of data were collected by O'Shea (1996) and O'Shea & Hocking (1997a & b) and was assessed in order to form a basis upon which the data collected in subsequent years could be analysed. Efforts to develop alternative methods for conducting surveys on the species included modifications to the pit-fall trap and drift fence system; an assessment of the seasonal timing of surveys; and an evaluation of the potential use of artificial shelters.

Pit-fall trapping with drift-fences was found to be a useful tool for conducting habitat manipulation experiments on the species. However, the use of artificial shelters, roof tiles in this case, is cheaper, less labour intensive and less disturbing and therefore provides an adequate survey tool. Surveys conducted in Spring yielded a larger number of *D. impar* over a shorter survey period.

An objective pictorial method was devised to identify individual *D. impar* specimens. This enabled the rate of recapture to be assessed and can be applied to both mature and immature animals. Attempts were made to adapt harmonic direction-finder technology for use on *D. impar*. This technology has previously been used to monitor the movements of juvenile and small snake species. This attempt was not successful.

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Photo: Matthew Cattanach

Chapter 1

Conservation biology of the Striped Legless Lizard

1.1 Conservation biology

1.1.1 Conservation biology

The Striped Legless Lizard *Delma impar* is internationally recognised as a vulnerable species (IUCN, 2002). Efforts to ensure the species' long-term survival and maintain its potential for evolutionary development in the wild across its natural geographic range (Smith & Robertson, 1999) provide an example of approaches to the relatively new discipline of conservation biology (Soulé & Wilcox, 1980). An initial investigation into the conservation biology of *D. impar* was conducted by Coulson in 1990.

Conservation biology is the applied science of maintaining the earth's biological diversity (Hunter, 1996). It extends beyond the traditional sciences to form a cross-disciplinary field which takes into account the overlapping spheres of the basic biological sciences, sciences of the physical environment, applied management sciences, social factors and modes of implementation (Jacobson, 1990). In the case of *D. impar*, the Striped Legless Lizard Working Group oversees and co-ordinates recovery efforts between universities (research), management agencies, zoos (captive populations and education) and community awareness and volunteer groups (Banks, 1992; Kutt *et al.*, 1995; Webster *et al.*, 1995). Recovery plans for the species take these overlapping factors into account (Webster *et al.*, 1992; Working Party, 1994; ACT Government, 1997; Coulson, 1995; Smith & Robertson, 1999).

Biodiversity is the totality of life forms and is generally considered at the levels of genetic diversity, population diversity, species diversity and the diversity of ecological communities (Clark *et al*., 1990; New, 2000). Hunter (1996) suggested that functional diversity should also be taken into account, including the ecological and evolutionary processes that link biotic factors with each other and with abiotic factors, such as soil, air and water. In the case of *D. impar*, the main focus is at the species diversity level. However, since the initial investigation into the conservation biology of *D. impar* (Coulson, 1990), there have also been efforts to understand the ecology of the species at the genetic level (Osmond, 1994), at the population level (O'Shea, 1996) and at the ecological community level (Dorrough, 1995; Hadden, 1995; Nunan, 1995).

1.1.2 Extinction processes

Conservation biology is often a crisis discipline (Soulé, 1985) with a common focus on the prevention of extinctions of populations, species, communities and ecosystems. Increasingly, the focus is at the population level, with efforts to develop an understanding of the relationship between a population's size and the probability of its extinction. That is, the science of population viability assessment or PVA (Shaffer, 1981; Gilpin & Soulé, 1986; Clark *et al*., 1990).

Extinction is the irrevocable loss of a biological element such as a population, species, community or ecosystem (Clark *et al.*, 1990). Population elements are regulated by deterministic (press) factors and stochastic (pulse) factors (Shaffer, 1981; New, 2000). Deterministic extinction arises from some inexorable change or force from which there is no hope of escape, that is, a catastrophic event. Stochastic extinctions result from normal random changes or environmental perturbations (Shaffer, 1981). Furthermore, deterministic (press) events have been shown to result in an increased risk of stochastic (pulse) events leading to species or population extinctions (Gilpin & Soulé, 1986).

Deterministic events and four sources of stochastic events (demographic stochasticity, genetic stochasticity, environmental stochasticity, and natural catastrophes, which may occur at random intervals through time) are considered when developing population viability assessments (Shaffer, 1981; Gilpin & Soulé, 1986). For *D. impar* natural catastrophes, demographic stochasticity, environmental stochasticity and deterministic events were modeled and found to influence the probability of population viability (Webster *et al.*, 1991; ARAZPA, 1996), especially in the case of environmentally stochastic and catastrophic events such as fire. Although much of the data incorporated into the *D. impar* PVA model was based on estimates of baseline population values, it was apparent that the main factor influencing the conservation status of the species was a substantial contraction in its distribution since European settlement, largely as a result of habitat destruction. As with many species, small population sizes and biogeographic isolation, as a result of habitat fragmentation and alteration (deterministic events), were the over-arching factors which appeared to negatively influence *D. impar* population viability as a result of stochastic events, especially natural catastrophes such as drought and fire, and further deterministic events.

1.1.3 Habitat loss and degradation

Fragmentation, alteration and destruction of habitat is one of the major threats to biodiversity and is a common focus for practitioners of conservation biology (Wilcox & Murphy, 1985; Soulé & Kohm, 1989). Habitat degradation is the means by which habitat quality is diminished although physical habitat may still remain. Habitat loss refers to changes whereby a given species or suite of taxa can no longer be supported (New, 2000). The process of habitat loss occurs as a series of stages, initiated by habitat dissection, followed by habitat perforation, fragmentation and attrition (Hunter, 1996). These are deterministic (press) factors in the extinction process.

Habitat fragmentation enables the survival of more or less small remnant patches of the original habitat (Wilcove *et al.*, 1986; New, 2000). A remnant is a patch of native vegetation around which most or all of the original vegetation has been removed (Saunders *et al*., 1987). Typically, remnants are small and not necessarily representative of the original plant or animal communities (Saunders *et al*., 1987). For example, Hadden (1995) found that the majority of sites supporting *D. impar* populations were less than 50 hectares in size and that many were isolated from other grassland habitat areas.

The two primary effects of habitat fragmentation are an alteration of the microclimate within and surrounding the remnant, that is physical changes, and the isolation of each area from other remnant patches in the surrounding landscape, that is biogeographic changes (Saunders *et al*., 1991). Both physical and biogeographic effects of fragmentation have been noted as having an impact on the distribution and abundance of *D. impar*. In an example of physical change, Dorrough and Ash (1999) found that the species had been locally excluded from habitat which had been recently ploughed. They also suggested that bare ground, as a result of overgrazing, severely limits *D. impar* dispersal (biogeographic effect as a result of physical change). Yet they found that the species was able to colonise adjacent secondary habitat (grassy woodland from which the woody overstorey had been removed) at an estimated rate of less than 12 metres per year, if it was continuous with primary habitat.

Generally, the persistence of a species in a remnant depends on the retention of enough suitable habitat to maintain sufficient numbers to withstand the risk of extinction (Saunders *et al*., 1991). The larger the remnant, the more likely it is that populations will be large and more likely to resist chance (stochastic) extinctions (Gilpin & Soulé, 1986). Landscape fragmentation reduces the total area of habitat available for biota.

Furthermore, the remaining habitat is broken up into remnants that are isolated to varying degrees (Saunders *et al*., 1991; Wilcove *et al*., 1986). Isolated remnants are modified by the factors of size, shape and position in the landscape. The risks of stochastic extinctions in remnant habitats are influenced by a species' ability to colonise a remnant, which depends to some extent on the distance of the remnant from other areas of native vegetation (Ewens *et al.*, 1987; Gilpin, 1987). Although some animal species may have the physical ability to disperse long distances, success may be limited by a lack of behavioural repertoire to traverse the matrix surrounding the remnant (Saunders *et al*., 1991). Species most vulnerable to fragmentation are those that occur in low population densities, with poor dispersal abilities, sedentary habits or specialised needs and those endemic to an area (Andrews, 1990) – *D. impar* meets all of these points*.* The dispersal of *D. impar* appears to be limited by both its inability to traverse large distances and its inability to traverse between remnants of preferred habitat. Thus, *D. impar* now has a patchy distribution which appears to be the result of habitat history (Dorrough & Ash, 1999), whereby the grassland habitat has undergone a process of disturbance (to the exclusion of the species), followed by fragmentation from the surrounding landscape. In some instances, the habitat appears to have recovered due to a cessation or a reduction in disturbance events, yet despite apparent suitability it is likely that *D. impar* is no longer able to utilise this habitat due to the species' poor dispersal abilities.

1.1.4 Ecosystem functions

The scale of change arising from habitat fragmentation and degradation is not only at the level of populations or species, but is also at the level of functional units of interacting organisms (Recher & Lim, 1990) and ecosystem properties (Hobbs & Hopkins, 1990). Ecosystem processes that are likely to be influenced by fragmentation include energy flow within and between habitat patches, nutrient cycling, hydrology, soil properties, patch dynamics (succession and distribution of habitat patches in space and time) within fragments, and the movement of materials across landscape boundaries (Soulé & Kohm, 1989; Hobbs & Hopkins, 1990). Impacts of a change in the regime of any of these factors is poorly understood for the survival of *D. impar* and the role of *D. impar* in the maintenance of any of these processes has not been evaluated.

Remnants have to serve the purpose that was previously performed by the whole landscape, including preservation of landscape and ecosystem functions and conservation of biota. Unlike very large or contiguous areas in which natural process may continue to function as before fragmentation, remnants are subject to a variety of exogenous disturbances which may disrupt normal processes and often result in a decline in the number of species present. In such cases, management will always be necessary to maintain the conservation value of remnants (Saunders *et al.*, 1987; Hobbs & Hopkins, 1990). For *D. impar*, the event of fire has been viewed as having a high catastrophic potential (Webster *et al*., 1991; ARAZPA, 1996; Coulson, 1999). Yet fire is a natural ecosystem function, serving a regenerative purpose, in the grassland habitat of this species (Morgan & Lunt, 1999). Prior to habitat fragmentation and population isolation, *D. impar* would most likely have recolonised burnt habitat patches from the surrounding landscape. However, the potential for this to occur is extremely limited for most extant populations of the species, given their degree of isolation (Hadden, 1995) and the species' apparently poor dispersal abilities. The extent to which the species can survive fire *in situ* is unknown, however there is speculation that Summer and Autumn fires are likely to result in lower mortality rates, as animals are able to seek refuge in soil cracks which appear in the clay substrate at these times (Coulson, 1990; Webster *et al.*, 1992; ARAZPA, 1996).

1.2 The temperate lowland grasslands of south-eastern Australia

1.2.1 Grasslands around the world

Changes to habitat constitute the paramount threat to biodiversity (New, 2000) and this has become apparent in grassland environments around the world. Native grasslands are found on every continent, cover approximately 40 percent of the world's land area, comprise 40 of the world's 234 centres of plant diversity, 23 of the world's 217 endemic bird areas and 35 of the world's 136 terrestrial ecoregions identified as outstanding examples of diverse ecosystems (White *et al*., 2000). Generally, the extent of global native grasslands have decreased as a result of major modifications, including extensive conversion to croplands and human settlement, changes in fire regimes, grazing of domestic livestock, fragmentation and the introduction of non-native species. The biodiversity of some of the world's grassland areas is considerably altered (White *et al*., 2000), with considerable losses observed in the prairies of North America, the pampas of South America and the chalk grasslands of Europe (Lunt, 1991).

Within Australia, habitat modification is recognised as the most significant cause of loss of biodiversity, with nearly 70 percent of all native vegetation removed or significantly modified by human activity since 1788 (State of the Environment Advisory Council, 1996; National Land & Water Resources Audit, 2002). The most extensive clearing has been in regions that are both topographically and climatically suitable for large-scale cropping or improved grazing (Hobbs & Hopkins, 1990; Recher & Lim, 1990; State of the Environment Advisory Council, 1996). In south-eastern Australia, this has included vast areas of temperate lowland grassland, an ecosystem of which less than 0.5 percent of the original remains intact (Kirkpatrick *et al.*, 1995). *Delma impar* is endemic to, and a specialist of, the temperate lowland grasslands of south-eastern Australia and its conservation status is reflective of the habitat loss, modification, degradation and fragmentation of this ecosystem (Cogger *et al*., 1993).

1.2.2 The Western Basalt Plains Grassland Community

In Victoria, one of the strongholds of *D. impar* is across the Western Basalt Plains, an area covering approximately 21,000 km^2 (Stuwe, 1986). The Plains were formed by a series of lava flows, the Newer Volcanics, which are overlain by fertile, heavy grey or red cracking clay soils (Rosengren, 1987; McDougall *et al*., 1994). The topography is low-lying, undulating and expansive, with occasional rocky outcrops and other geological features at the edges of larva flows (Savio, 1993; McDougall, 1987; Rosengren, 1987;

McDougall *et al.*, 1994). The climate is temperate with generally mild winters and warm to hot summers. Rainfall is generally evenly spread throughout the year with a Spring maximum, and ranges between 400 millimetres and 900 millimetres (McDougall, 1987; McDougall *et al.*, 1994).

The vegetation of the Western Basalt Plains is influenced by water availability and fire regimes. Water availability to plants is not only affected by the moderately low levels of rainfall but also by the clay soil structure. Clay soils swell and become readily waterlogged but also dry out rapidly forming hard cracking horizons that resist root penetration. Although large amounts of water may be periodically locked up in the soil, the clay structure results in only small amounts being available to plants (McDougall, 1987*).* Frequent fire regimes are also considered an important factor contributing to the vegetation characteristics of the plains (Stuwe, 1986).

The complex interaction of soils, water availability and fire have given rise to a diversity of habitats across the Western Basalt Plains of Victoria, ranging from grassy woodland, to chenopod rocky open scrub to plains grassland and grassy wetland (Savio, 1993). In areas where these factors are especially limiting, the establishment of deep rooted perennial trees and shrubs are inhibited (Willis, 1964; Stuwe, 1986) and the Western Basalt Plains Grassland Community is prevalent.

The Western Basalt Plains Grassland Community once occupied a vast area of the Western Basalt Plains region, with Kangaroo Grass *Themeda triandra* generally being the dominant species (Stuwe, 1986; McDougall, 1989). Sub-dominant grasses such as wallaby grasses *Austrodanthonia* spp., spear grasses *Austrostipa* spp. and tussock grasses *Poa* spp. and a variety of perennial herbs such as Lemon Beauty Heads *Calocephalus citreus*, Pink Bindweed *Convolvulus erubescens*, Tufted Bluebells *Wahlenbergia communis* and Creamy Candles *Stackhousia monogyna* are also common features of this community. Composite species are particularly well represented in the suite of forbs occupying inter-tussock spaces (Muir, 1994; Stuwe, 1986).

The Western Basalt Plains offered ideal grazing lands and rapidly attracted European settlement, agriculture and associated development in the 1830's (Stuwe, 1986; Lunt, 1991). A new range of exogenous disturbances were exerted which differed to those experienced over evolutionary time, such as grazing, trampling by hooved animals and pasture improvement (McIntyre & Lavorel, 1994). This resulted in the introduction of weed species, changes in grassland composition and function (Moore, 1964; Moore, 1970) and the near-total disappearance of indigenous vegetation from pastoral regions (Frood & Calder, 1987). Other novel disturbances included cropping, urban development, slashing, tree planting, overgrazing, soil disturbance for fire prevention purposes and inappropriate fire regimes (Stuwe, 1986; Frood & Calder; 1987; Muir, 1994).

The Western Basalt Plains Grassland Community, a sub-component of the lowland temperate grasslands of south-eastern Australia, has been reduced to about 0.1 percent of its former extent (Stuwe, 1986; Craigie & Stuwe, 1992) and the decline is continuing. It is considered one of the most endangered vegetation communities in Victoria *(*Stuwe, 1986; Kirkpatrick *et al.*, 1995) and is listed as a threatened community under Section 2 of the *Flora and Fauna Guarantee Act* 1988 (Muir, 1994).

Furthermore, the grasslands and grassy woodlands, once covering 34 percent of Victoria (Department of Conservation and Environment, 1992), contain many of the State's rare or threatened plants, animals and invertebrates. Of the 866 plant taxa considered rare or threatened in Victoria, 131 (15 percent) are confined or largely confined to grasslands and grassy woodlands. Twenty (95 percent) of the 21 species of mammals, one of the two species of bird and one of the two species of reptile which are now extinct in Victoria, formerly occupied these ecosystems (Department of Conservation and Environment, 1992). Hadden (1998) refers to the invertebrates of Victoria's temperate lowland grasslands as 'the forgotten fauna', reflecting the paucity in knowledge of the invertebrate taxa that inhabit these ecosystems and their conservation status (Yen, 1995; Yen *et al.*, 1995; Yen, 1999).

With much less than one percent of the former extent of the Western Basalt Plains Grassland Community remaining, remnants and the species they support are typically fragmented and thus subject to the pressures of isolation. The possibility of connecting remnants is almost impossible (Muir, 1994). These pressures have led to a breakdown in ecological processes, and the remnants and the species they contain, such as *D. impar*, now require active management.

1.2.3 Ecological processes in lowland grasslands

Today, the vegetation of individual lowland, temperate grassland remnants of south-eastern Australia are considered to be the result of environmental factors (such as lithology, slope and landscape position, and altitude) combined with management histories and the effects of soil disturbance, grazing, nutrient enrichment and fire regimes (Stuwe & Parsons, 1977; McIntyre & Lavorel, 1994; Morgan & Lunt, 1999). Recent developments in the understanding of some of these ecological processes indicates that active management is now required for most Western Basalt Plains Grassland Community remnants.

In particular, the importance of regular biomass reduction for the prevention of *T. triandra* senescence, increased nutrient loads and weed invasion has become apparent, giving rise to management prescriptions including fuel reduction burns, grazing and slashing regimes (Stuwe & Parsons, 1977; Stuwe, 1986; Craigie & Stuwe, 1992; Morgan & Lunt, 1999; Henderson, 2001).

Further efforts to understand the ecological roles of nutrient cycles, soil processes and mycorrhizal associations (Ford, 1997; Wijesuriya, 1999) are in the early stages of development. Work has also commenced on factors affecting native forb recruitment and survival (Morgan, 1995; Morgan, 1998; Henderson, 2001; Robinson, 2003). The unique associations between species, especially threatened species, are also being explored. For example, it has been found that the Sunshine Diuris *Diuris fragrantissima* requires the fungal symbiont *Tulasnella calospora* for germination (Webster &

McKay, 1993) and that the Golden Sun Moth *Synemon plana* is dependent on wallaby grass *Austrodanthonia* sp. for feeding and reproduction (Britton *et al*., 1995; ACT Government, 1998).

The current poor understanding of many of these ecological processes poses difficulties for managing small remnants which often contain more than one species of conservation significance and which may have conflicting management requirements. For example, the Plains Wanderer *Pedionomus torquatus* has a marked preference for sparse grassland (Baker-Gabb *et al*., 1990), whereas the Eastern Barred Bandicoot *Perameles gunnii* prefers habitat with a medium to high density of grass (Minta *et al*., 1990).

Delma impar provides further example of potential conflict that can arise when developing management prescriptions. The use of fire is often seen as a preferable tool to grazing or slashing for the purposes of vegetation management and biomass reduction. Yet the observed death of animals immediately following fire has restricted or prevented the use of this method at sites where the species is known to occur (Craigie & Stuwe, 1992; McDougall *et al*., 1992).

1.3 Ecology of grassland reptiles

Rawlinson (1980) points out that according to the concept of limiting factors there is a range of physical and chemical factors in each habitat, and that every species has its own characteristic range of tolerance for each factor. Due to the ectothermic nature of reptiles, temperature is one of the most important single physical factors, with a great portion of the daily activity of many species being devoted to responding to the thermal environment (Heatwole & Taylor, 1987). The utilisation of habitat is an important factor for thermoregulation with three modes being employed, heliothermy (utilisation of radiant heat), thigmothermy (utilisation of conductive heat) and behavioural patterns (such as shuttling, breezing, burrowing and postural changes).

In addition to thermoregulation, reptile habitat has important implications for locomotion, shelter, foraging and breeding requirements (Rawlinson, 1980;

Hadden, 1998). Reptile species richness has been correlated with structural attributes of the vegetation. That is, a greater number of reptile species are found in structurally diverse heterogeneous habitats which offer a range of opportunities for individual reptile species to meet requirements (Pianka, 1966; Pianka, 1967; Kitchener *et al*., 1980; Heatwole, 1981).

In temperate lowland grasslands of northern and western Victoria which have been grazed, Hadden (1998) found that structural and floristic diversity were the overwhelming factors influencing reptile species richness. That is, fewer and more generalist reptile species were found at sites which were floristically depauperate and dense in vegetation cover, whereas a greater species richness, which also contained specialist reptile species, was found at sites which had a high floristic diversity and were more open in vegetation cover. It is postulated that a more open vegetation cover provides greater opportunities for thermoregulation and foraging activities (Hadden, 1998).

More specifically, in Western Basalt Plains Grasslands, Hadden (1998) found that the cover of *T. triandra* combined with the richness of invertebrate species were the most important factors influencing herpetofaunal species richness, and further suggested that invertebrate species richness was a product of floristic diversity and thus grassland quality.

In comparison to many other habitat types, grasslands are generally lacking in vertical structure. For many grassland reptiles a fossorial or semi-fossorial habit may provide opportunities for thermoregulation and other requirements, which may not otherwise be available due to a lack of habitat heterogeneity and structural diversity, in that the soil layer provides a sheltered and stable environment , with only slight changes in temperature and humidity (Osmond, 1994). For Western Basalt Plains Grasslands Hadden (1992) found a positive correlation between the species richness of reptiles and amphibians and the percentage of rock cover, suggesting that this terrestrial environment is an important component for herpetofauna in this grassland environment.

Overall, the greatest threat affecting the conservation status of reptile species in Australia is habitat clearance (Cogger *et al.*, 1993), presumably because it removes opportunities for reptiles to meet their biological requirements. The 'Action Plan for Australian Reptiles' (Cogger *et al.*, 1993) found that tussock grassland habitats have a high number (nine) of threatened reptile species in comparison to most other habitat types.

1.4 The family Pygopodidae and genus *Delma*

The lizard family Pygopodidae consists of 38 species in six to eight genera (Shea, 1993a; Jennings *et al.*, 2003). The family is endemic to Australia and two species are found in Papua New Guinea (Greer, 1989). All species have elongate and attenuated bodies which lack any external trace of front limbs and the hind limbs have been reduced to small flap-like appendages (Greer, 1989).

The Pygopodidae is derived from a stock of gekkonid lizards, sharing a large number of derived anatomical, behavioural and reproductive characters (Kluge, 1974; Greer, 1989). Examples include similarities in the structure of the inner ear (Wever, 1974), the use of the tongue to wipe the face and eye spectacle (Greer, 1989), the ability to vocalise (Weber & Werner, 1977; Annable, 1983) and a constant clutch size of two (Patchell & Shine, 1986). The pygopodids are considered to be a sister group to the diplodactyline geckos. This relationship is characterized by a condition in which the meatal closure muscle forms an O-shape in both groups (Kluge, 1987) and is also supported by molecular phylogenetic evidence (Donnellan *et al.*, 1999).

It has been suggested that pygopodids are derived from a surface-active ancestor which became elongate and subsequently limbless, enabling it to move more efficiently through litter and vegetation and enter confined spaces such as soil crevices and spider burrows in search of prey (Shine, 1986; Greer, 1989). Pygopodids probably evolved in warm and open habitats and occur in the arid, semi-arid and seasonally dry parts of Australia, including woodlands, shrublands and grasslands (Greer, 1989). Although some genera utilise the vertical structure of low vegetation, it seems that none are truly arboreal and most are surface-active, semi-fossorial or fossorial (Greer, 1989; Shea, 1993a).

All genera except one, *Lialis*, feed on arthropods, exhibiting a range of generalist to specialist and active to sit-and-wait feeding strategies (Patchell & Shine, 1986; Greer, 1989; Shea, 1993a). Activity periods also vary between genera and species (Greer, 1989).

Approximately 25 percent of pygopodid species were recognised as threatened in the 'Action Plan for Australian Reptiles' (Cogger *et al.*, 1993). However it is uncertain whether their cryptic habits contribute to these classifications or whether the group as a whole is poorly known (Jones, 1999). Most species of conservation concern are poorly represented in museum collections. Generally, the main threats to pygopodids are habitat destruction and modification (Jones, 1999).

Delma is the largest genus of the Pygopodidae, comprising 17 species (Shea, 1991; Cogger, 1996). It is one of the more primitive genera of the Pygopodidae and was considered to be a poorly defined group, with uncertainty as to whether the genus is a true lineage or an assemblage of similar species (Kluge, 1974; Greer, 1989; Shea, 1991; Shea, 1993a). However, a recent study on the systematics of the Pygopodidae has found *Delma* to be a monophyletic group (Jennings *et al*., 2003). Species in the genus *Delma* exhibit a range of ecological characteristics in terms of diet, activity periods and habitat use (Shine, 1986). *Delma impar* is the only species in this genus which is restricted to temperate grassland habitats (Smith & Robertson, 1999).

1.5 The Striped Legless Lizard – *Delma impar*

To date there has been very little literature published on the Striped Legless Lizard. This literature review draws on a sizeable body of unpublished work which has not been peer reviewed. These documents can be obtained upon request from the Victorian Striped Legless Lizard Working Group which is located within the Victorian Department of Sustainability and Environment.

1.5.1 Taxonomy and external morphology

The type specimen for the Striped Legless Lizard *Delma impar* (Fischer 1882) was collected in Melbourne (Coulson, 1990). It is distinguished from other species of the genus *Delma* by the partly fused condition of the nasal and first supralabial scales. The nostril lies between these two scales, which are separated by a narrow suture posterior to the nostril and are fused anterior to the nostril (Kluge, 1974). However, Coulson (1990) found this to be an unreliable diagnosis for the species and Shea (1991) noted that the feature also occurred in another species, *Delma plebeia*.

Delma impar was given its vernacular name by Coulson (1990) due to the series of dark brown or blackish parallel dorso-lateral and lateral longitudinal stripes, which commence at the tympanum and run the length of the body and tail (Cogger, 1996). The background colour of the dorsum is pale grey-brown above, the head is dark brown and the ventrum is white. Many individuals have a sulfur yellow colouration around the face. Hatchlings lack any stripes and have a very dark brown or black head (Coulson, 1990).

The species is slow growing (Kutt, 1992), with a maximum size being attained by 10 to 12 years (Kutt, 1993b). It has been postulated that individuals could live up to 20 years (Banks *et al.*, 1999). The snout-vent length ranges from 33 millimetres to 116 millimetres and the total length ranges from 74 millimetres to 306 millimetres. Weights range between 0.6 grams to 7.6 grams (Coulson, 1990).

1.5.2 Distribution

Delma impar has the southern-most distribution of all pygopodids (Coulson, 1990). It is restricted to the lowland tussock grassland habitats of south-eastern Australia (Coulson, 1990) and historically, appears to have been widespread throughout this habitat, ranging from the extreme south-east of South Australia, through Victoria and into south-east New South Wales west of the Great Dividing Range and the Australian Capital Territory (ACT) (Coulson, 1990; Working Party, 1994; Hadden, 1995; ACT Government, 1997) (figure 1.1).

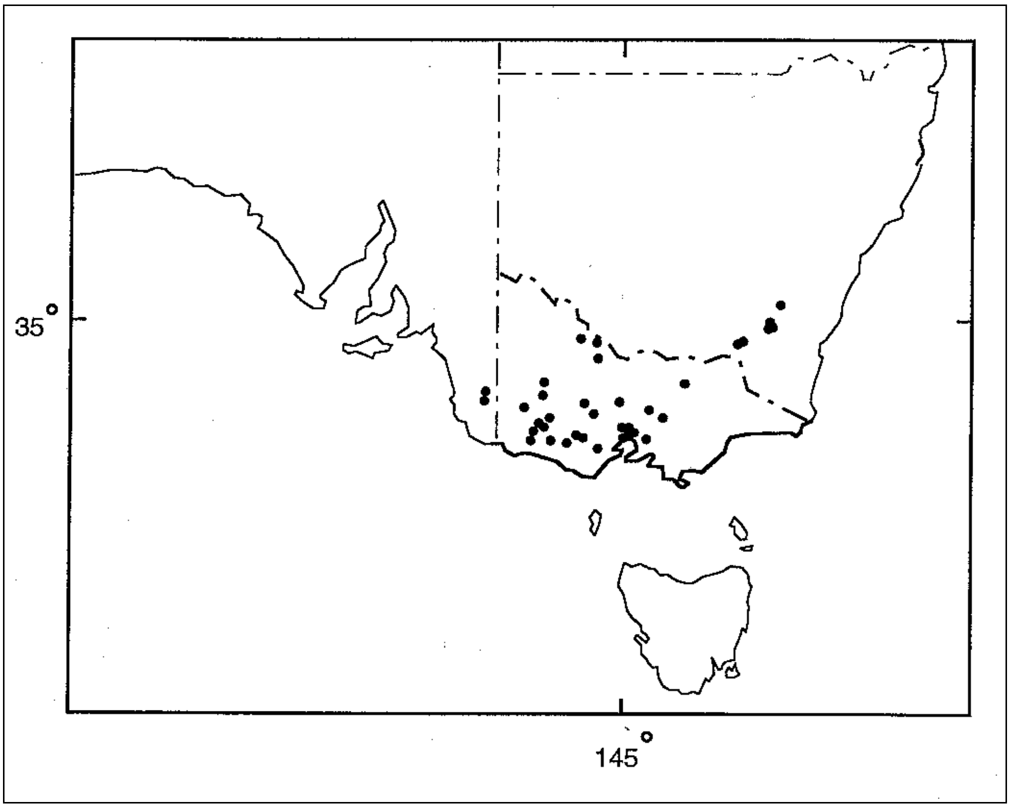


Figure 1.1: The distribution of *D. impar*. (Source: Dorrough, 1995).

Historically recorded at 125 sites across this range, the species now has a patchy distribution limited to an estimated 45 sites (Hadden, 1995). These are concentrated mostly around Ballarat, Bendigo, Melbourne (Coulson, 1990; Hadden, 1995) and the ACT (Kukolic, 1992; Osborne *et al.*, 1993; Rauhala *et al.*, 1995) with historical records and recent discoveries of the species at sites in New South Wales within 100km of the ACT (Kluge, 1974; Shea, 1993b; Husband, 1995; Dorrough *et al.*, 1996) and rediscoveries near Bool Lagoon in South Australia (Banks *et al.*, 1999).

The previously widespread distribution of the species is supported by patterns of genetic variation among populations which reflect levels of gene flow that occurred sometime in the past when migration may have been comparatively easy, even though a genetic distinction can now be made between Victorian and ACT populations (Osmond, 1994).

1.5.3 Habitat

Delma impar is adapted to a cool and relatively homogeneous thermal regime (Osmond, 1994), thus accounting for its distribution in low altitude tussock grasslands of south-eastern Australia. Generally, it is considered more likely to be present at a site and have a higher abundance when the vegetation has been relatively undisturbed and forms a well developed dense sward of native grasses (*T. triandra* in Victoria and *T. triandra* and Tall Spear-grass *Austrostipa bigeniculata* in the ACT) (Coulson, 1990; Williams & Kukolic, 1991; Kukolic, 1992; Kukolic, 1993; Osborne *et al.* 1993; Kukolic, 1994; Kukolic *et al.*, 1994; Working Party, 1994; Hadden, 1995; Rauhala *et al.*, 1995; Rauhala, 1997; Dorrough & Ash, 1999; Rauhala, 1999).

However, Hadden (1998) found that the species was equally likely to occur in both dense swards of closed grassland and the widely spaced tussocks of open grassland and Dorrough (1995) found no difference in the detection of animals in tall or short grass. Alternatively, Moro (1990) and Kutt (1992) found that *D. impar* presence was characterised by low vegetation density. Hadden (1998) concluded that the species could be supported in either open or closed grassland, providing that tussock species comprised a component of the vegetation. Therefore, the current understanding of structural requirements is limited.

Delma impar has also been frequently recorded at sites which have a large, if not predominant component of exotic tussock species such as Toowoomba Canary-grass *Phalaris aquatica*, Serrated Tussock *Nassella trichotoma* and Chilean Needle Grass *Nassella neesiana* (Kukolic, 1994; Kukolic *et al.*, 1994; Dorrough, 1995; Hadden, 1995; Husband, 1995; Rauhala *et al.*, 1995; Corrigan *et al.*, 1996; O'Shea & Hocking, 1997a; Rauhala, 1997; Kimber & Timewell, 2001; Dunford *et al.*, 2001; Michael, 2001). As a result, Dorrough and Ash (1999) suggest that the species is not dependent on any particular grassland plant species or community and that floristic measures previously used as predictors for presence and abundance of *D. impar* are actually indices of past management, rather than the direct effects of the floristics. *Delma impar* is very rarely recorded at sites dominated by annual exotic species (Moro, 1990; Hadden, 1998).

Past management appears to be an important factor in determining *D. impar* presence. Although the species has been recorded at sites which have been subject to grazing, ploughing, slashing and fire disturbances (Williams & Kukolic, 1991; Kukolic, 1994; Kukolic *et al.*, 1994; Dorrough, 1995; Husband, 1995; Nunan, 1995; Rauhala *et al.*, 1995; Whitby, 1995; Dorrough *et al.*, 1996; Dorrough & Ash, 1999), it has been found that it has also been excluded by some of these practices. In particular, fertiliser application (as a possible indicator of heavy grazing) and ploughing (as a function of time) have been found to restrict the species presence at a site, especially when no refugia remains (such as vegetation which is unpalatable to stock or rock cover) and sites are isolated, both preventing recolonisation (Dorrough, 1995; Hadden, 1998; Dorrough & Ash, 1999). Sites subject to low levels of disturbance may also provide refugia for the species during periods of drought, high level disturbance and catastrophe (Husband, 1995; Rauhala *et al.*, 1995).

Delma impar habitat is found on flat areas of geological formations derived from basalt in Victoria or Silurian/volcanic origins in the ACT. Soil types include bleached-red duplex, black cracking clays and red podzolic and have a moderate to high clay content, often producing cracks in Summer (Hadden, 1995). Generally, there is a low (less than 10 percent) cover of lightly embedded surface rocks (Coulson, 1990; Hadden, 1995; Kimber & Timewell, 2001). Cracking clays and surface rocks are not a feature in the ACT (Osborne *et al.*, 1993; Working Party, 1994; Hadden, 1995) and although Dorrough & Ash (1999) found that all sites supporting the species were rich in clay, they did not consider that clay content was a significant predictor of either presence or abundance of *D. impar*. Most sites have less than 10 percent bare ground (Dorrough, 1995; Hadden, 1995; Rauhala *et al.*, 1995; Hadden, 1998; Rauhala, 1999; Dorrough & Ash, 1999).

Generally, remnants are small and fragmented, ranging in size from 0.25 hectares to 400 hectares, with the majority falling within the less than one hectare to 50 hectare size class. Extant sites are equally divided between public and private ownership, with the majority under some form of grazing management (52 percent of sites) and a large proportion (33 percent of sites) are under no form of management. Other sites have experienced various levels of burning and a few have been cultivated (Hadden, 1995).

1.5.4 Diet

Delma impar is a selective arthropod feeder which utilises surface active and sedentary prey types (Nunan, 1995; Kutt *et al.*, 1998). Spiders, Lepidoptera larvae, field crickets *Teleogryllus commodus* and cockroaches have been recorded as the most frequently taken prey items, with a range of readily available invertebrate prey items avoided (Coulson, 1990; Wainer, 1992; Nunan, 1995; Kutt *et al.*, 1998; O'Shea & Hocking, 2000).

The diet of *D. impar* appears to vary across native dominated and exotic dominated habitat types. For example, O'Shea and Hocking (2000) found that *T. commodus* were more frequently consumed by *D. impar* captured in native dominated grassland than in exotic dominated grassland, corresponding to an increase in the abundance of *T. commodus* in native habitat types. They also found that cockroaches were more frequently consumed by *D. impar* captured in exotic dominated habitat. In some instances, Nunan (1995) found a significant positive correlation between *D. impar* abundance and the availability of Lepidoptera larvae at each site. However, Nunan (1995) concluded that no consistent trend between native and non-native sites could be found and that diet did not appear to be a limiting factor, allowing *D. impar* to persist and reproduce over a range of habitat types.

Delma impar displays flexibility in its degree of dietary specialisation, apparent foraging strategy and seasonal preferences. It is able to exploit food resources within the preferred range in patchy habitats, a trait which is considered useful for inhabiting sub-optimal or highly disturbed grassland sites (Nunan, 1995). Although it is unclear what sense(s) are used to detect prey (Nielsen & Underwood, 1997), it appears to utilise both sit-and-wait and widely foraging strategies, taking both mobile and sedentary prey items (Wainer, 1992; Osmond, 1994; Coulson, 1995; Nunan, 1995; Kutt *et al*., 1998; Banks *et al.*, 1999).

There appears to be some level of seasonal variation in the diet which possibly reflects the life history and therefore the abundance and availability of prey, with caterpillars being most frequently consumed in Spring and early Summer and *T. commodus* being most frequently consumed in late Summer (Kutt *et al.*, 1998; O'Shea & Hocking, 2000). O'Shea and Hocking (2000) also found cockroaches to be more frequently consumed in late Summer but could not relate this to availability. Spiders were consistently consumed throughout the *D. impar* active period, despite fluctuations in abundance (Kutt *et al*., 1998; O'Shea & Hocking, 2000).

1.5.5 Behaviour

Delma impar is considered a diurnal semi-fossorial species, being surface active during the day and spending nights and brummation periods within the soil layer or at the base of tussocks (Martin, 1972; Coulson, 1990; Anderson, 1992; Kutt, 1992; Osmond, 1994; Araya, 1997; Banks *et al*., 1999; Rauhala, 1999). Osmond (1994) related this to the species preferred temperature range (24.5 °C to 27.5°C) which could be achieved mostly by thigmothermic behaviour on the surface during the day (Kutt, 1992; Hadden, 1995; Banks *et al*. 1999) and retreating to the insulated and less thermally fluctuating soil layer at night. Incidentally, the species also displays heliothermy behaviour (Martin, 1972; Hadden, 1995; Araya, 1997). *Delma impar* has a low thermal dependence of metabolism at low temperatures, which is similar to other surface active species, and is considered to be an adaptation to a cold climate (Osmond, 1994).

Delma impar is considered to be highly seasonal in its activity periods. Animals enter brumation in late March to April and re-emerge in September (Banks *et al*., 1999). Highest catch rates have been recorded in November and December (Kukolic, 1994; Kukolic *et al*., 1994), but on some occasions low catch rates have been recorded for these months during seasons of extremely high temperatures and low rainfall and when the ground cover is sparse. It is thought that the species remains relatively immobile in the soil layer during these harsh conditions and that low catch rates do not necessarily reflect declines in abundance (Rauhala, 1999).

1.5.6 Movement

Relatively little is understood of the movement behaviour of *D. impar*. It appears to be only a moderately active species (Kukolic *et al*., 1994; Osmond, 1994), yet it is also capable of considerable surface activity.

Despite this, there are no records of long-distance dispersals (Dorrough, 1995). The greatest distance traveled by an individual has been recorded at 60 metres over a two day period (Kukolic *et al*., 1994) and Kutt (1992) recorded a movement of 19 metres in a single night. The very limited recapture data that exists for the species suggests that there has been no movement of individuals between sites in the ACT (Rauhala, 1997). For example, one individual was recaptured in the original 50 metre by 50 metre cross-arm trapping array after six years and nine months (Rauhala, 1997). Osmond (1994) suggests that the species specific thermal requirements may be one of the impediments to movement between populations that are divided by areas that are exposed to wide temperature fluctuations, in comparison to the relatively homogeneous thermal gradients of native grasslands.

Dorrough and Ash (1999) found that over 150 years, populations of *D. impar* in the ACT had migrated distances of up to two kilometres from primary grassland habitat into adjacent secondary grassland habitat. On this basis they suggest that populations of the species are relatively slow moving, with an estimated rate of less than 12 metres per year.

1.5.7 Reproduction

The cycle of reproduction of *D. impar* appears to be consistent with that described by Patchell and Shine (1986) for other pygopodid species. Females are suspected to ovulate in spring and gravid individuals have been recorded in November and early December (Kutt, 1992; Kukolic, 1993; Kukolic, 1994; Osmond, 1994; Whitby, 1995; Rauhala, 1997). Eggs are deposited under rocks and in soil cavities in mid to late December (Mills, 1992; Kutt, 1992; Banks *et al.*, 1999). Generally two eggs per female are laid and several clutches have been located in the one site, suggesting communal oviposition (Mills, 1992; Banks *et al.*, 1999). Incubation time is variable, ranging from 30 days to 60 days in captivity (Kutt, 1992; Coulson, 1995; Banks *et al.*, 1999). The period of incubation appears to be longer in the wild, as hatchlings are only trapped in the field in early March (Kukolic, 1994).

The mating system is assumed to be polygynous (Coulson, 1995). Reproductive maturity is thought to be three years for females and two years for males (ARAZPA, 1996). A female recaptured after six years and nine months and subsequently aged at about 10 years was observed to be gravid, suggesting that females remain reproductively viable at least up until this age (Rauhala, 1997).

At the ARAZPA (1996) workshop convened for the purposes of *D. impar* population viability modeling, female reproductive success was estimated to be between 50 percent to 100 percent, juvenile mortality was estimated to be 50 percent in the first year and 20 percent in the second year and sex ratios were assumed to be even, for the baseline model. Trapping programmes have recorded varying sex ratios, these have included a roughly equal ratio, ratios of more males to females and ratios of more females to males (Rauhala, 1997; Rauhala & Andrew, 1998; Banks *et al*., 1999; Rauhala, 1999). Rauhala (1999) suggested that skewed sex ratios probably represent differential activity patterns between the sexes and that these may vary from year to year depending on environmental conditions experienced during the season or period of trapping. There is no information regarding patterns of movement or microhabitat requirements for the sexes (Rauhala & Andrew, 1998).

1.5.8 Conservation status

Delma impar is internationally recognised by the IUCN (2002) and ANZECC (1999) as a vulnerable species which is likely to be in danger of extinction in the future, as a result of declining numbers due to destruction and loss of habitat. It is also considered vulnerable under the Australian Commonwealth *Environment Protection and Biodiversity Conservation Act* 1999, the Australian Capital Territory *Nature Conservation Act* 1980 and the New South Wales *Threatened Species Conservation Act* 1995. In South Australia and Victoria it is listed as an endangered species under the *National Parks and Wildlife Act* 1972 and the *Flora and Fauna Guarantee Act* 1988, respectively.

Arising from these declarations are a number of documents, which outline programmes of recovery for *D. impar*. These include:

- *Striped Legless Lizard* Delma impar *National Recovery Plan 1999 – 2003* (Smith & Robertson, 1999);
- *Striped Legless Lizard (*Delma impar*): A Vulnerable Species. Action Plan No. 2* (ACT Government, 1997);
- *Management Directions for the Striped Legless Lizard (*Delma impar*) in the Australian Capital Territory* (Coulson, 1995); and
- *Striped Legless Lizard* Delma impar*. Flora and Fauna Guarantee Action Statement No. 17* (Webster *et al.*, 1992).

1.5.9 Threats

The decline in distribution of *D. impar* can be attributed to the loss and fragmentation of its native grassland habitat (Hadden, 1995), as a result of urban residential, industrial and agricultural development (Webster *et al*, 1992). At sites where the species persists, threats range from pasture improvement to crop production, prolonged heavy grazing by stock, rock removal, inappropriate fire regimes, introduced species, introduced predators, increased nutrient levels, changed hydrological regimes, soil compaction and the installation of infrastructure such as roads and footpaths (Williams & Kukolic, 1991; Webster *et al.*, 1992; Cogger *et al.*, 1993; Kukolic, 1993; Kukolic *et al.*, 1994; Working Party, 1994; Dorrough, 1995; Hadden, 1995; ARAZPA, 1996; Rauhala, 1997; Hadden, 1998).

Fragmentation of extant populations is considered to be a serious issue, as the species appears unlikely to colonise new areas or exchange genetic material between disjunct populations (Webster *et al.*, 1992). Osborne *et al.* (1993) suggested that efforts should be made to reserve the most extensive sites that contain the species. Recently a trial translocation project has been implemented in Victoria, however the success of this is yet to be determined.

1.6 Thesis overview

In summary of the above, there is now a reasonable level of knowledge relating to the distribution of *D. impar*, the types of habitat in which it is likely to be found and associations with past management practices and aspects of its dietary requirements. This is not to say that our understanding in each of these areas is absolute and does not require further investigation or pose further question.

Areas of knowledge relating to the species biology, life history and population demographics are notably lacking, as was made apparent in the PHVA workshop (ARAZPA, 1996). It has been perceived that one of the main reasons for this paucity of knowledge is due to *D. impar's* cryptic nature and difficulties associated with observing, capturing and recapturing the species. The location of what is considered to be Australia's largest population of *D. impar*, at Iramoo Wildlife Reserve Victoria, has provided the opportunity to closely study the species in a scientific manner, with selective replicated treatments over time.

This study aims to:

- provide some baseline information on the population size, structure and demography of *D. impar* at Iramoo Wildlife Reserve;
- suggest improved methods for monitoring and obtaining more accurate methods for collecting biological and life history information on *D. impar*;
- suggest methods for monitoring the movement of individual *D. impar* within the grassland habitat;
- provide a preliminary assessment on the use of fire as a habitat management tool and its potential impacts on *D. impar* populations.

Chapter 2 Characteristics of the Striped Legless Lizard Population at Iramoo Wildlife Reserve

Source: Urban and Regional Land Corporation

Chapter 2

Characteristics of the Striped Legless Lizard population at Iramoo Wildlife Reserve:

A Point of Reference for the Assessment and Management of Striped Legless Lizard Populations

2.1 Introduction

The Primary Recovery Criterion identified in the Recovery Plan for the Striped Legless Lizard *Delma impar* is to ensure that viable populations or clusters of populations are represented and maintained in reserves or appropriately managed sites across the known distribution of the species (Smith & Robertson, 1999). Because small isolated populations continue to be vulnerable to stochastic extinction forces, even after the original causes of decline are removed (Lacy , 1993), an approach of metapopulation management has been adopted for *D. impar*. Seventeen clusters (including one theoretical cluster based on BIOCLIM predictions) of *D. impar* populations have been identified (ARAZPA, 1996; Smith & Robertson, 1999). These may consist of a single population but more commonly are comprised of two or more populations. Clusters have been grouped on the basis of geographic and environmental criteria to represent areas of similar habitat and thus ensure that specific habitat adaptations and genetic variation across the species' range are preserved, enabling the continuation of the species' evolutionary viability (ARAZPA, 1996).

An operational objective for adopting and managing this cluster approach was to 'ensure the survival of all metapopulations with less than one percent risk of extinction of each metapopulation in 100 years' (ARAZPA, 1996). Population Viability Analysis (PVA) provides an estimation of extinction probabilities by incorporating identifiable threats to population (or cluster) survival into models of the extinction process (Lacy, 1993) and may be extended to use as a tool for the guidance of wildlife management (Lacy, 1993; Lindenmayer *et al*., 1993). PVA does not predict overall what will happen to a population (or cluster) but assists in forecasting the likely effects only of those factors incorporated into the model (Lacy, 1993). Thus, the model is only as good as the data and assumptions which are put into it (Lindenmayer *et al*., 1993).

The computer simulation model VORTEX has been used to construct PVA's for *D. impar* (ARAZPA, 1996). This model is appropriate for long-lived, low fecundity species and operates by modeling population dynamics as discrete, sequential events that occur according to probabilities that are random variables, following user-specified distributions (Lacy, 1993). In the PVA process for *D. impar* overall, and for individual populations, much of the demographic input information was assumed or inferred.

Based on previous studies (Larwill *et al*., 1993; O'Shea, 1996; O'Shea & Hocking, 1997a), the population of *D. impar* at Iramoo Wildlife Reserve appears to be large and robust and is therefore able to provide opportunities to develop base-line demographic information which may in turn be used as a guide to the health, viability and impacts of management for other populations and clusters. Although direct comparisons with other populations or clusters is not necessarily appropriate, due to inherent variabilities in factors such as site productivity, climate and competition from congeners, an understanding of the influences of stochastic processes on the biology and long-term viability of a population existing under conditions which appear to be highly favourable should provide some insight as to how similar processes may affect populations existing under less optimal conditions.

This chapter provides a description of the population of *D. impar* at Iramoo Wildlife Reserve and provides an indication as to which factors are likely to be important for the long-term viability of this population. The outcomes of this study provide a baseline by which to plan management activities at other sites, or clusters of sites, supporting *D. impar* populations, including goals for population density, population structure, sex ratios and levels of predation. The study also aims to provide insight as to how population data might be best collected and interpreted.

2.2 Methods

2.2.1 Study site

Iramoo Wildlife Reserve, formerly the Albion Explosives Factory Site, is a 37.5 hectare Western Basalt Plains Grassland remnant, located on the Keilor Plains, approximately 18 kilometres west of Melbourne, Victoria (37°44’S, 144°47’E) (figure 2.1), at about 60 metres above sea level. The mean annual rainfall of 500 millimetres is spread evenly throughout the year, with a slight rise in Spring. Mean daily maximum temperatures range from 26°C in February to 13 °C in July, with corresponding mean daily minimum temperatures ranging from 14 °C to 5 °C. Soils are a medium to heavy clay, derived from the basalt flows of the Newer Volcanics, and vary across the site in their depth, colour, structure, tendency to crack and form gilgai, and in the exposure of surface rock (Ross & Todd, 1998).

Prior to colonisation Aboriginal people of the Marin Bulluk clan lived in the area for over thirty thousand years, utilising local resources for food, water, shelter and other materials. Evidence of this is still extant at Iramoo Wildlife Reserve (Ross & Todd, 1998). Explorers Charles Grimes and James Fleming first described the region surrounding Iramoo Wildlife Reserve in 1803 as ‘an open grassy plain – without timber – as far as the eye could see’ (Ford & Vines, 1997). In the mid 1800’s the area was subdivided for sheep grazing purposes. In 1939 the Australian Commonwealth Government established the Albion Explosives Factory, covering approximately 500 hectares (Ross & Todd, 1998).

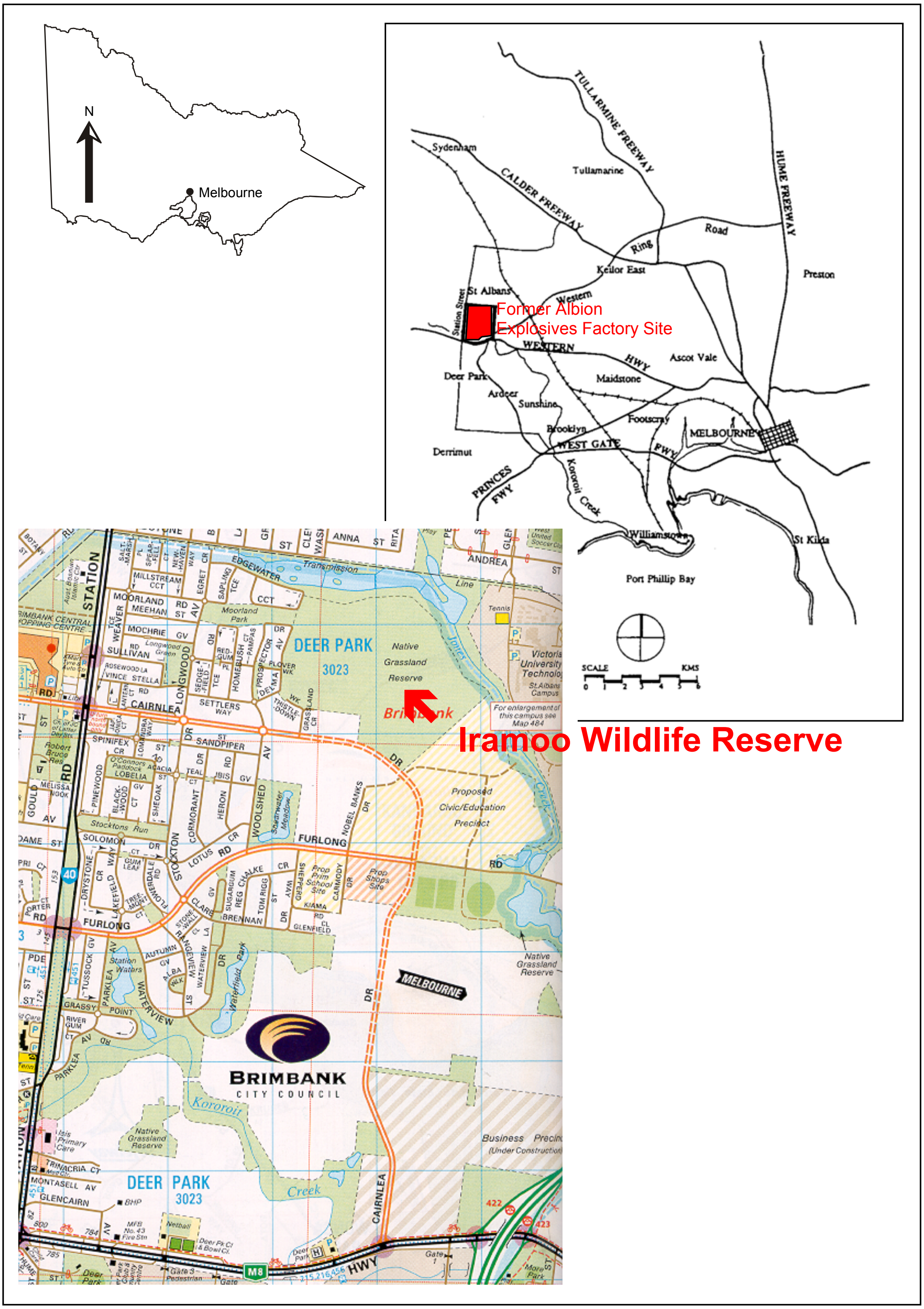


Figure 2.1: The location of Iramoo Wildlife Reserve in relation to Melbourne and Victoria. Sources: Parliamentary Standing Committee on Public Works (1997); Melway (2001).

The explosives factory remained in operation until 1989 (Parliamentary Standing Committee on Public Works, 1997). During this time sheep were grazed across the site to reduce fire hazards and following decommission were removed in 1991. Very little management of the vegetation has occurred since.

In 1993 a flora and fauna assessment of the Albion Explosives Factory Site located the largest of a series of remnant native grasslands in the northern section, which was also found to support a population of *D. impar* (Larwill *et al.*, 1993) and was subsequently recognised as being of national conservation significance (Costello & Larwill, 1996). This northern section formed the area of study for the duration of this project and will officially be named Iramoo Wildlife Reserve, following the current development of the new housing estate, Cairnlea, on the remaining portions of the Albion Explosives Factory Site. Other smaller grassland remnants, some supporting *D. impar*, have also been designated as conservation reserves within the Cairnlea estate.

At the commencement of this study, Iramoo Wildlife Reserve was bounded by Jones Creek to the east and the Gladstone Street Drain to the north (Ross & Todd, 1998) and was contiguous with the explosives factory to the south and west. During the progress of the study substantial subdivision development works were undertaken to the south, effectively isolating the reserve from the rest of the explosives factory site. In addition, substantial earth works were undertaken to create a series of lakes along the extent of the Gladstone Street Drain. As a result of further residential development Iramoo Wildlife Reserve is now surrounded by suburbia, with tenuous open space links to nearby grassland reserves. The reserve has been described as an open grassland, gently sloping towards Jones Creek, with a shelterbelt of Sugar Gums *Eucalyptus cladocalyx* running east-west across the site and a stony knoll in the western part of the site (Ross & Todd, 1998)

2.2.2 Habitat vegetation

At the commencement of this study, the vegetation of the study area was characterised by a dense stand of Kangaroo Grass *Themeda triandra* and adjacent areas of grassland dominated by the exotic species Serrated Tussock *Nassella trichotoma* and Chilean Needle Grass *Nassella neesiana*. The boundaries between the native and exotic vegetation types were considered to be generally well defined (O'Shea, 1996) (figure 2.2). The habitat was described as being in moderately good condition, in terms of species numbers and weed levels, with very high levels of vegetation cover (75 percent to 100 percent) (Larwill *et al.*, 1993).

Although the boundaries between the native and exotic vegetation types became less well defined throughout the duration of this project, these boundaries have remained discernible. However, through the process of *T. triandra* senescence (described by Stuwe and Parsons, 1977; Morgan and Lunt, 1999 and summarised in Chapter 6 of this dissertation) both species of *Nassella* and several Asteraceae species have ingressed and dispersed throughout the *T. triandra* dominated areas, making the distinction between native and exotic dominated areas less sharp. Unfortunately, the progress of this change was not mapped, as the invasive potential of these weeds was not apparent at the time of commencement.

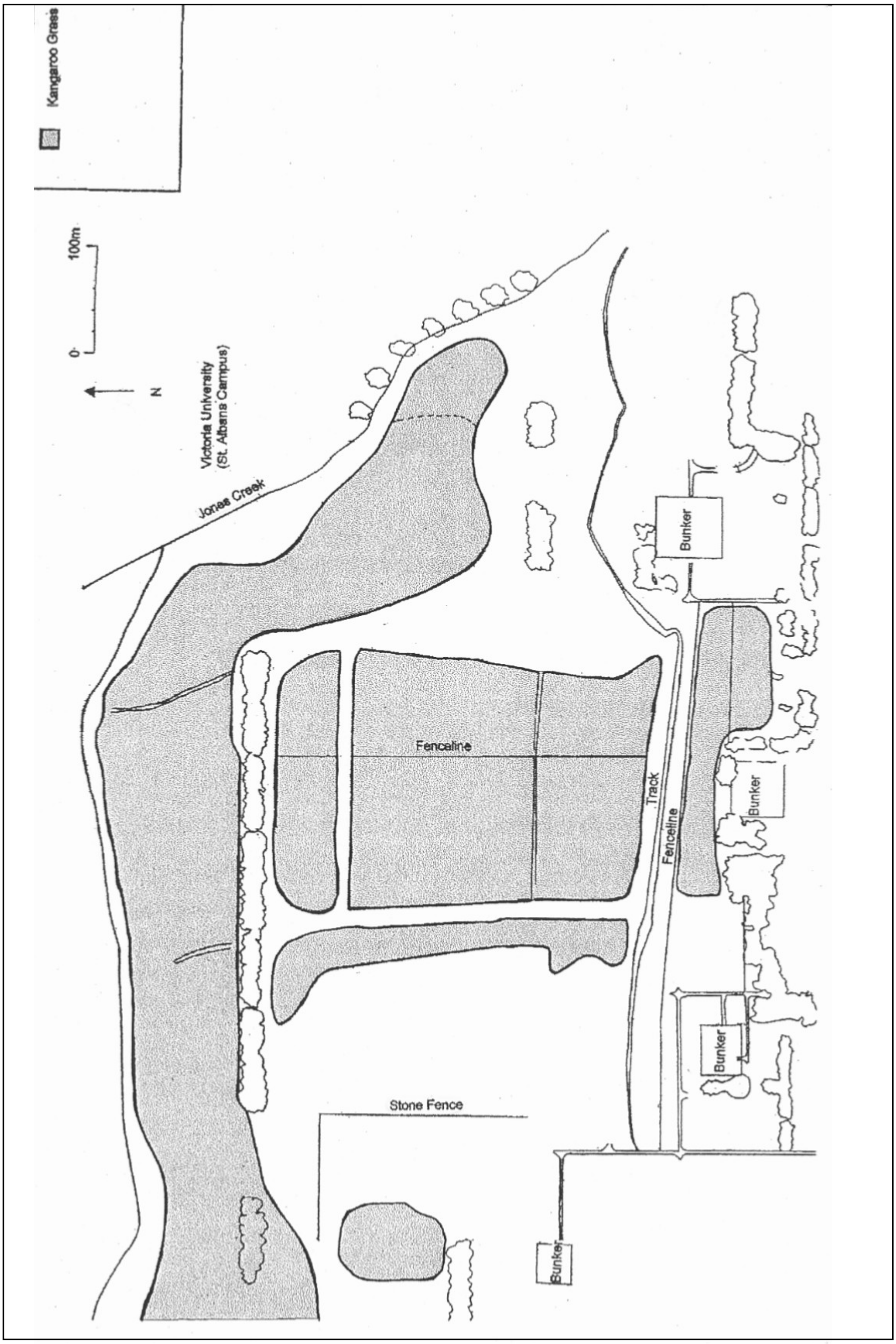


Figure 2.2: The distribution of *Themeda triandra* across the Iramoo Wildlife Reserve study site.

2.2.3 Standard Delma impar survey methods

Surveys for *D. impar* were conducted using pit-fall traps and drift-fences arranged to form survey plots. Each survey plot consisted of eight pit-fall traps and drift-fences, installed to form two 10 metre by 10 metre mini-grids, located 20 metres apart (figure 2.3). Each pit-fall trap consisted of an 11 litre tapered ice-cream bucket buried in the soil, so that the mouth of the bucket was flush with the soil surface. Lids were placed over the pit-fall traps during non-survey periods. Drift fences consisted of fly-wire mesh (five metres by 30 centimetres) which was held upright with metal loop pegs and the bottom two centimetres of the mesh was buried below the soil surface. Drift-fences were removed between survey periods.

Survey plots were initially established by O’Shea (1996) along transect lines which intersected with the native/exotic vegetation boundary in areas dominated by tussocking grassy vegetation. Three transect lines were established, with survey plots located at 20 metre intervals.

Further survey plots were established by O’Shea and Hocking (1997a). These were located at a minimum distance of approximately 50 metres from each other and in areas dominated by native or exotic tussocking grassy vegetation. In order to achieve a minimum distance of 50 metres between survey plots established in 1996 (O’Shea, 1996), several survey plots were removed. Any remaining survey plots which were located within 50 metres of each other were never operated simultaneously.

Surveys were conducted continuously (traps were closed during rainfall events) between January and March (hereafter referred to as ‘Summer’) for the years 1998, 1999 and 2000. In addition, data obtained from O’Shea (1996) and O’Shea and Hocking (1997a) were also used in the data analyses below. The timing of these surveys was used to provide a basis for assessing alternative techniques of capture or detection, including seasonal timing.

Table 2.1 shows the trapping dates, number of days trapping and the number of plots surveyed for each of these years. Figure 2.4 provides a base map of all established survey plots. Figures 2.5 – 2.9 indicate the distribution of survey plots used in each of the survey periods Summer 1996 – Summer 2000.

Year	Survey dates	Number of days	Number of survey plots
1996	4 Jan – 15 Mar	56	18
1997	11 Jan – 18 Mar	49	30
1998	13 Jan – 23 Mar	57	12
1999	10 Jan – 26 Mar	50	24
2000	14 Jan – 17 Mar	50	16

Table 2.1: The trapping programme for each Summer survey period.

Traps were cleared of all vertebrate species on at least a daily basis. For all *D. impar* that were captured, the following data were recorded:

- date;
- pit-fall trap location;
- snout-vent length (measured against a ruler to the nearest millimetre);
- total length (measured as above);
- presence/absence of tail autotomy;
- weight (measured on a top-loading balance to the nearest 0.01 gram); and
- sex was recorded for the years 1999 and 2000 only (using the cloacal spur technique described by Rauhala and Andrew, 1998).

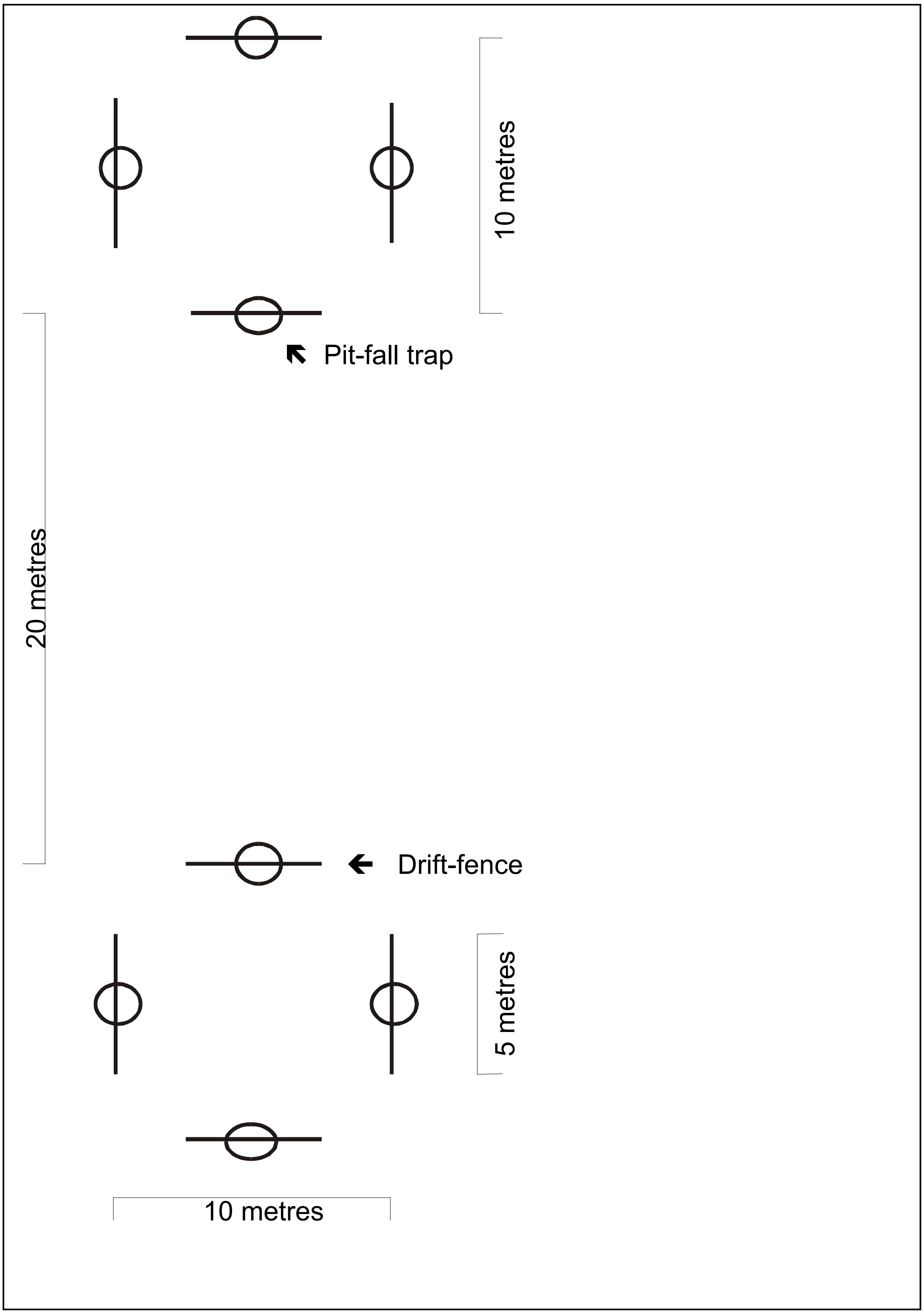


Figure 2.3: The arrangement of pit-fall traps and drift-fences to form a survey plot.

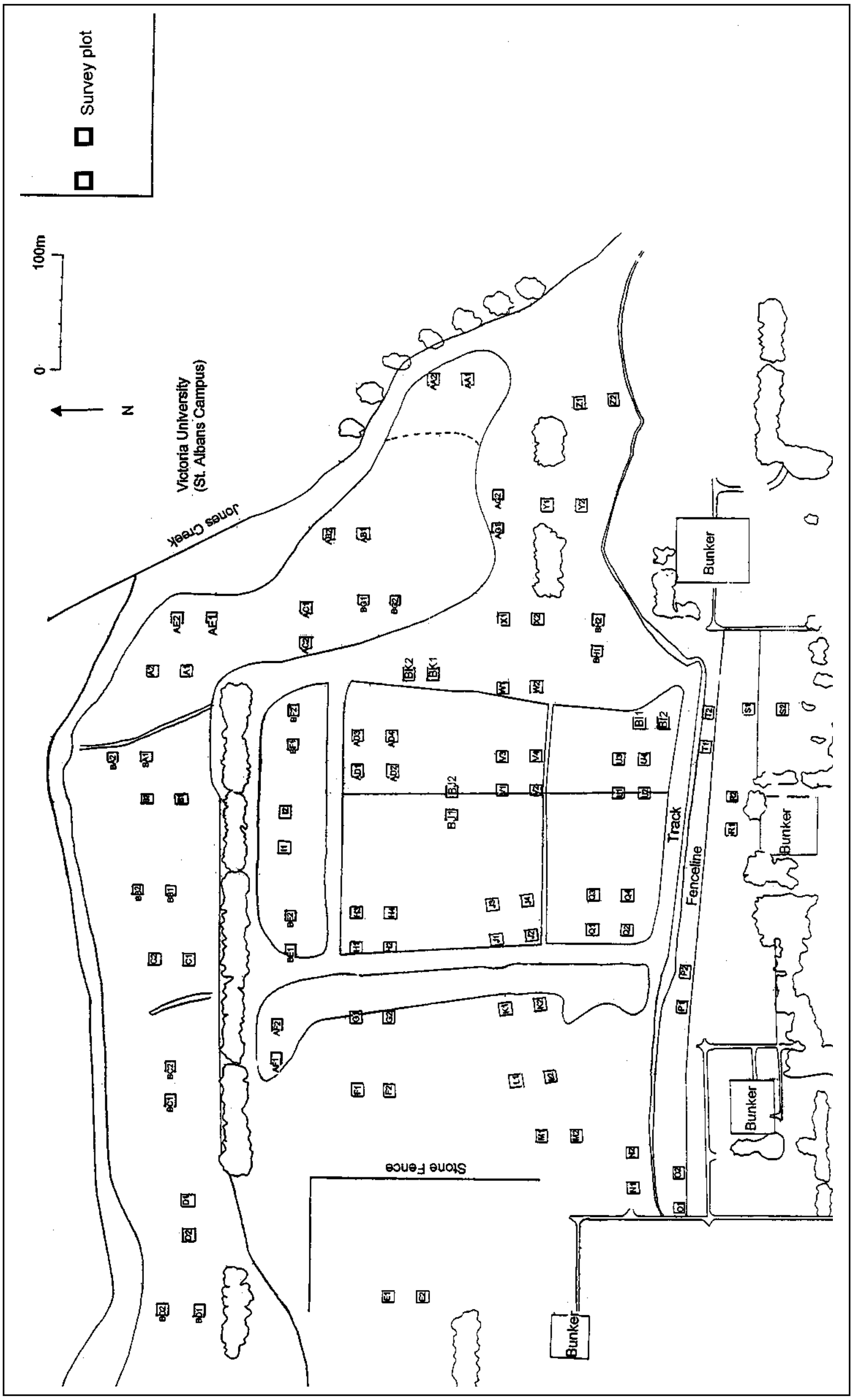


Figure 2.4: The location of survey plots at Iramoo Wildlife Reserve. A pair of boxes represents one survey plot.

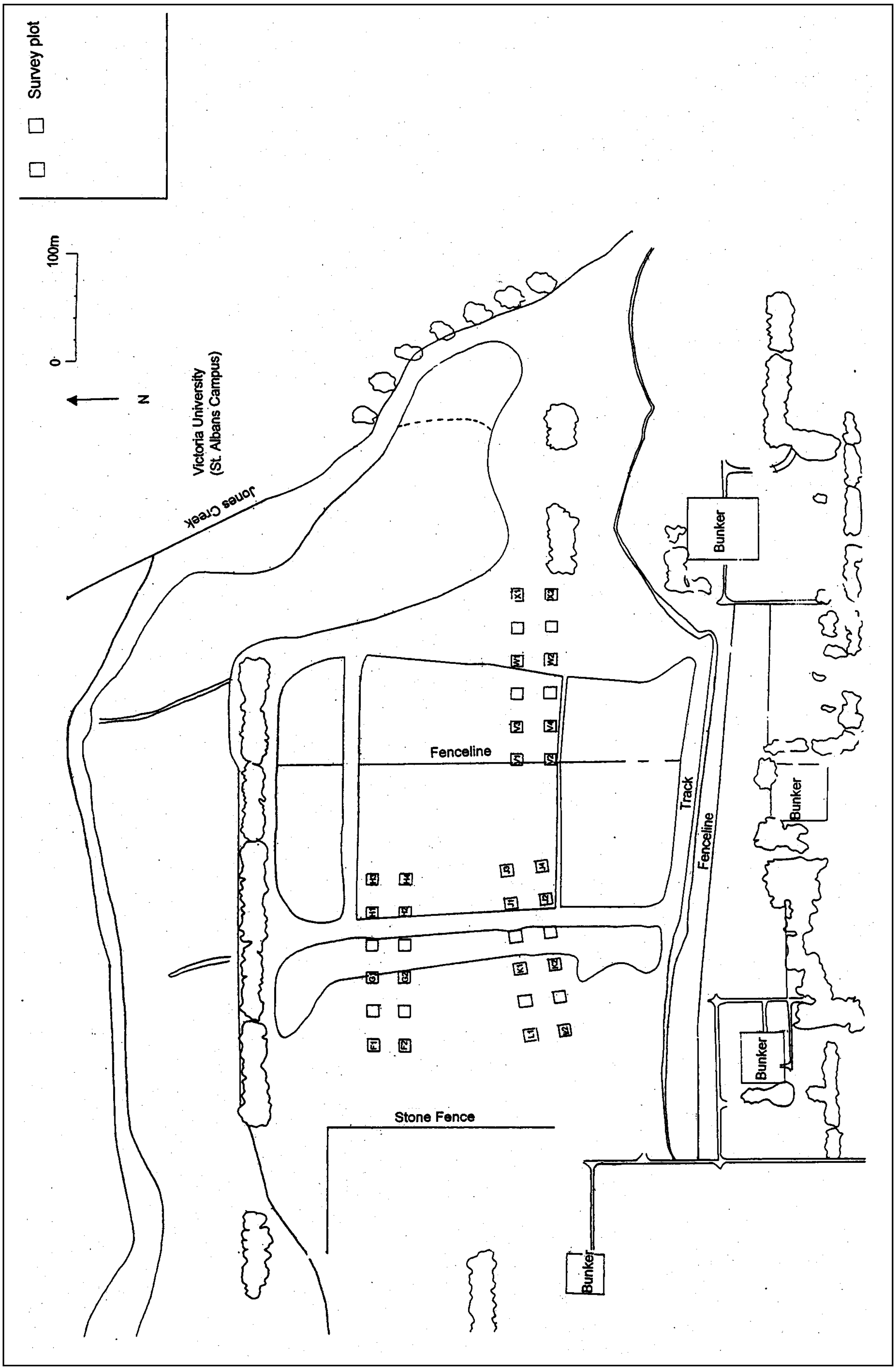


Figure 2.5: The location of survey plots for the Summer 1996 survey period. A pair of boxes represent one survey plot.

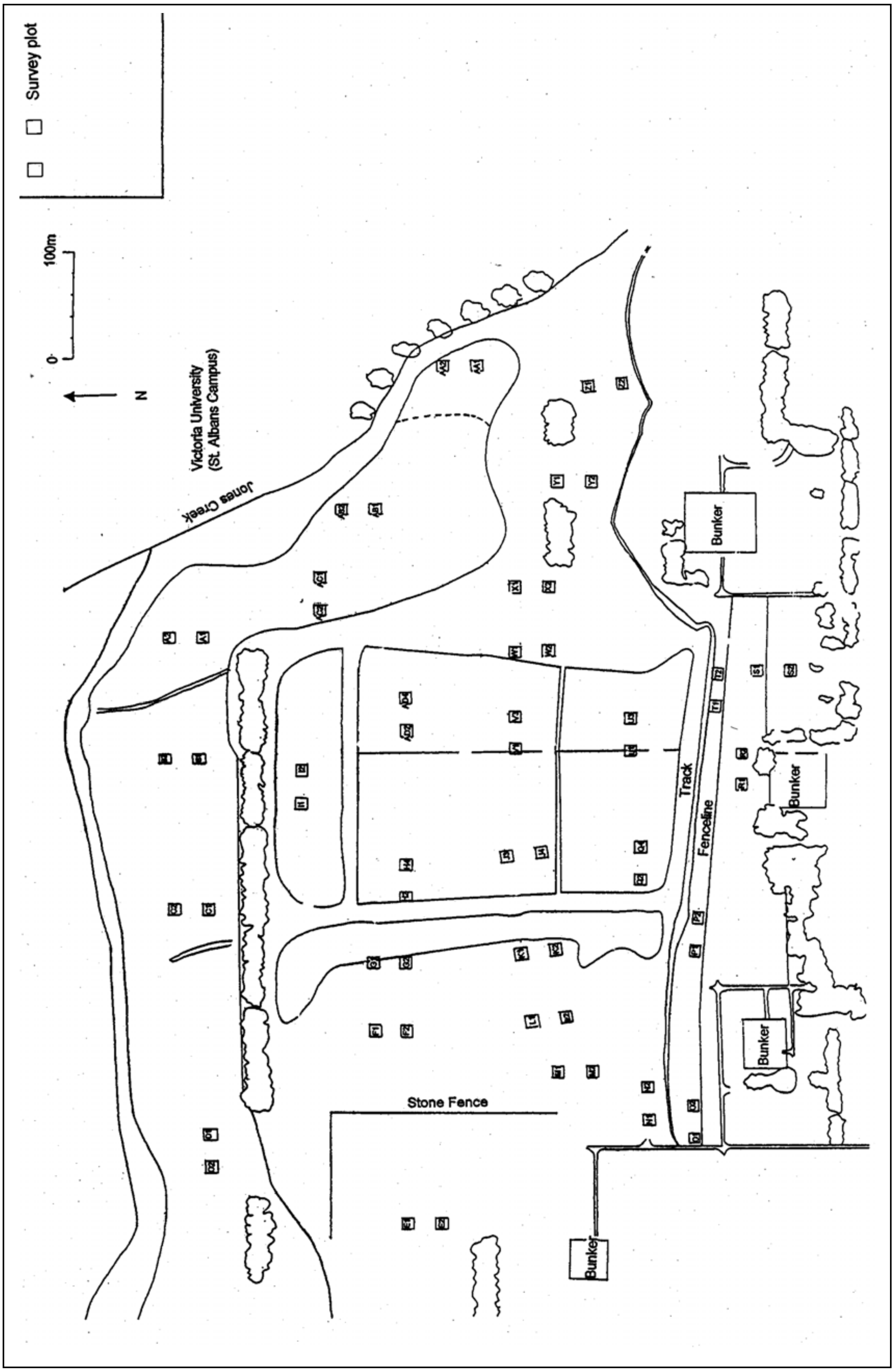


Figure 2.6: The location of survey plots for the Summer 1997 survey period. A pair of boxes represent one survey plot.

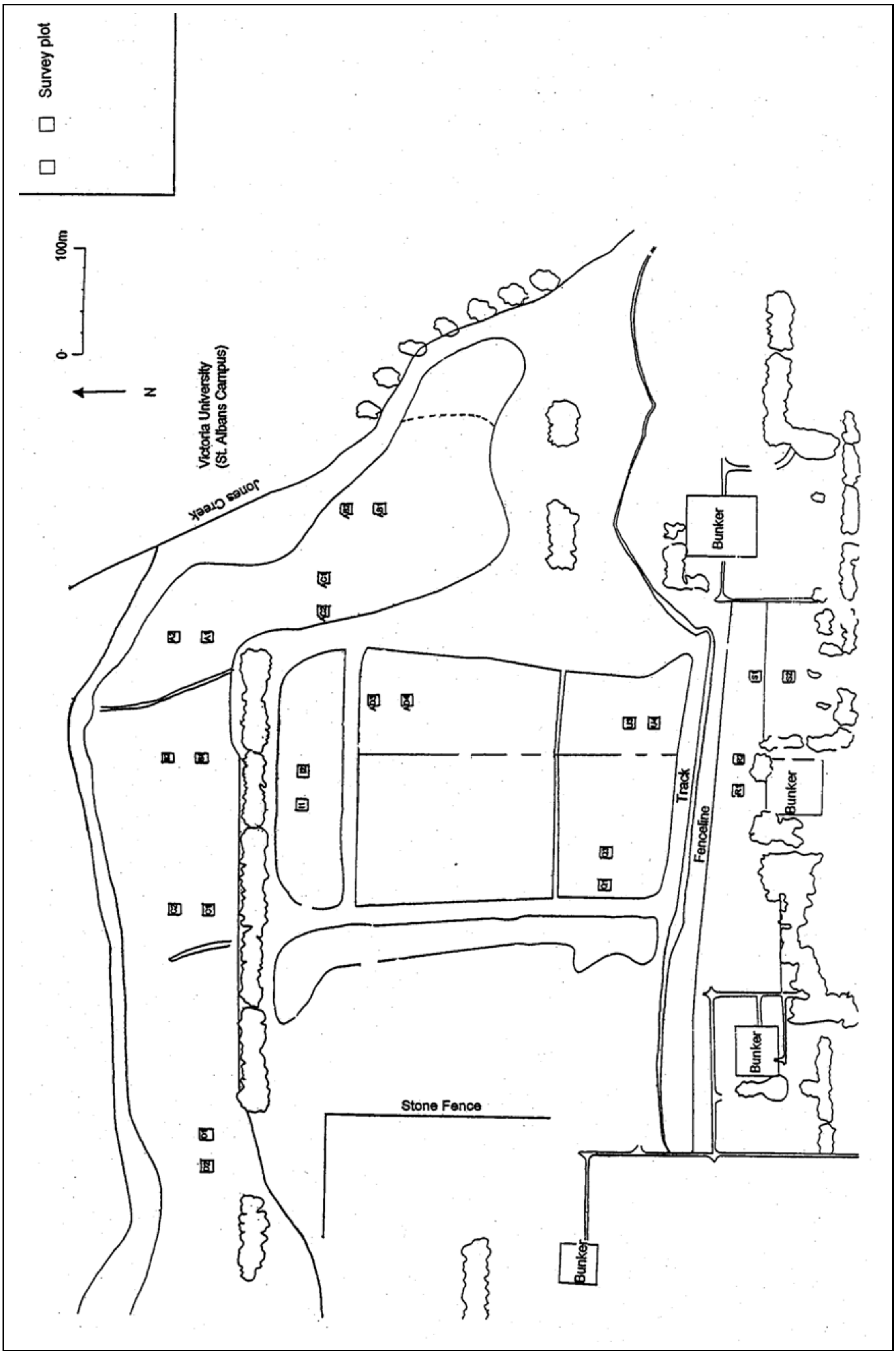


Figure 2.7: The location of survey plots for the Summer 1998 survey period. A pair of boxes represent one survey plot.

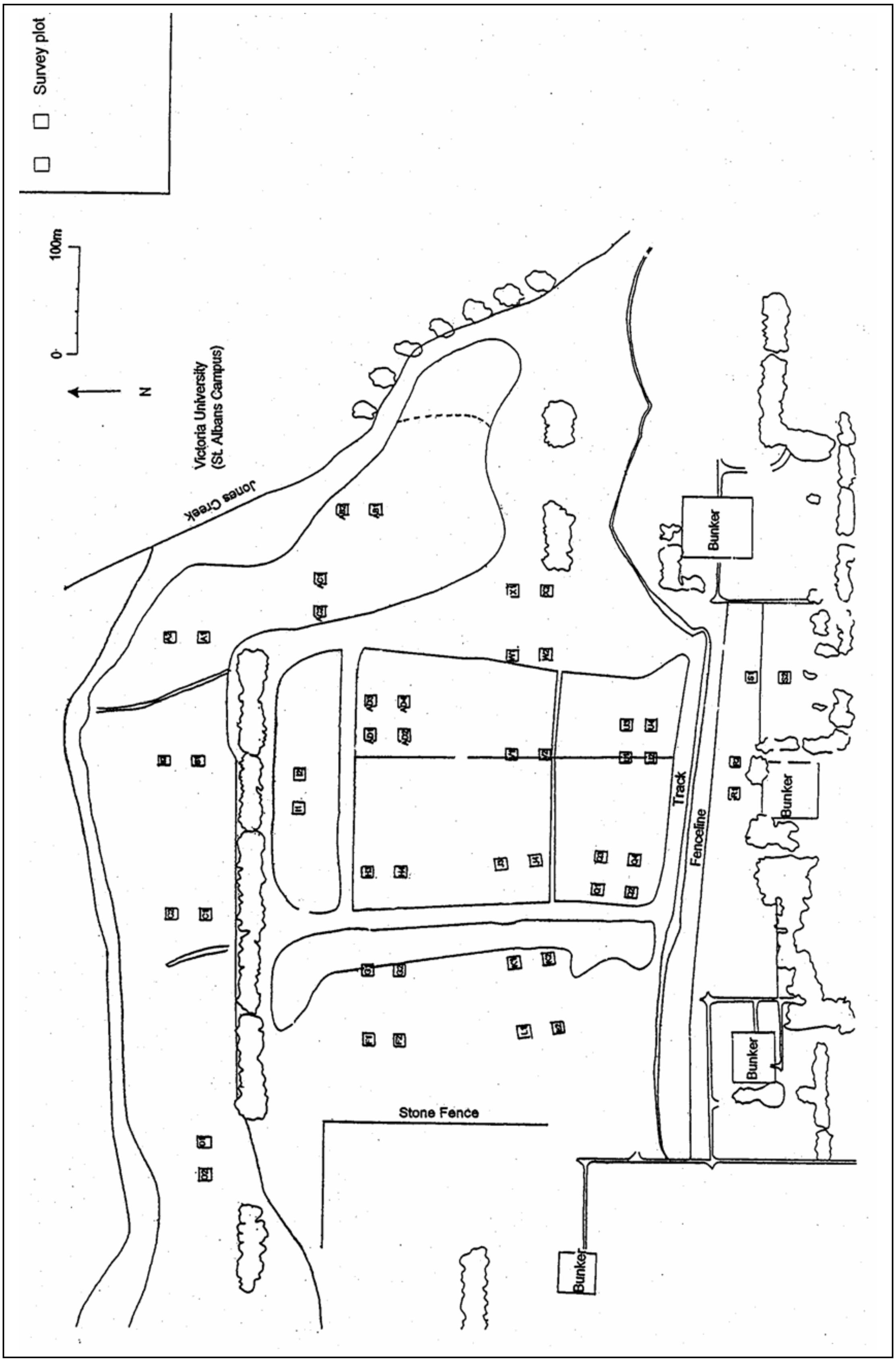


Figure 2.8: The location of survey plots for the Summer 1999 survey period. A pair of boxes represent one survey plot.

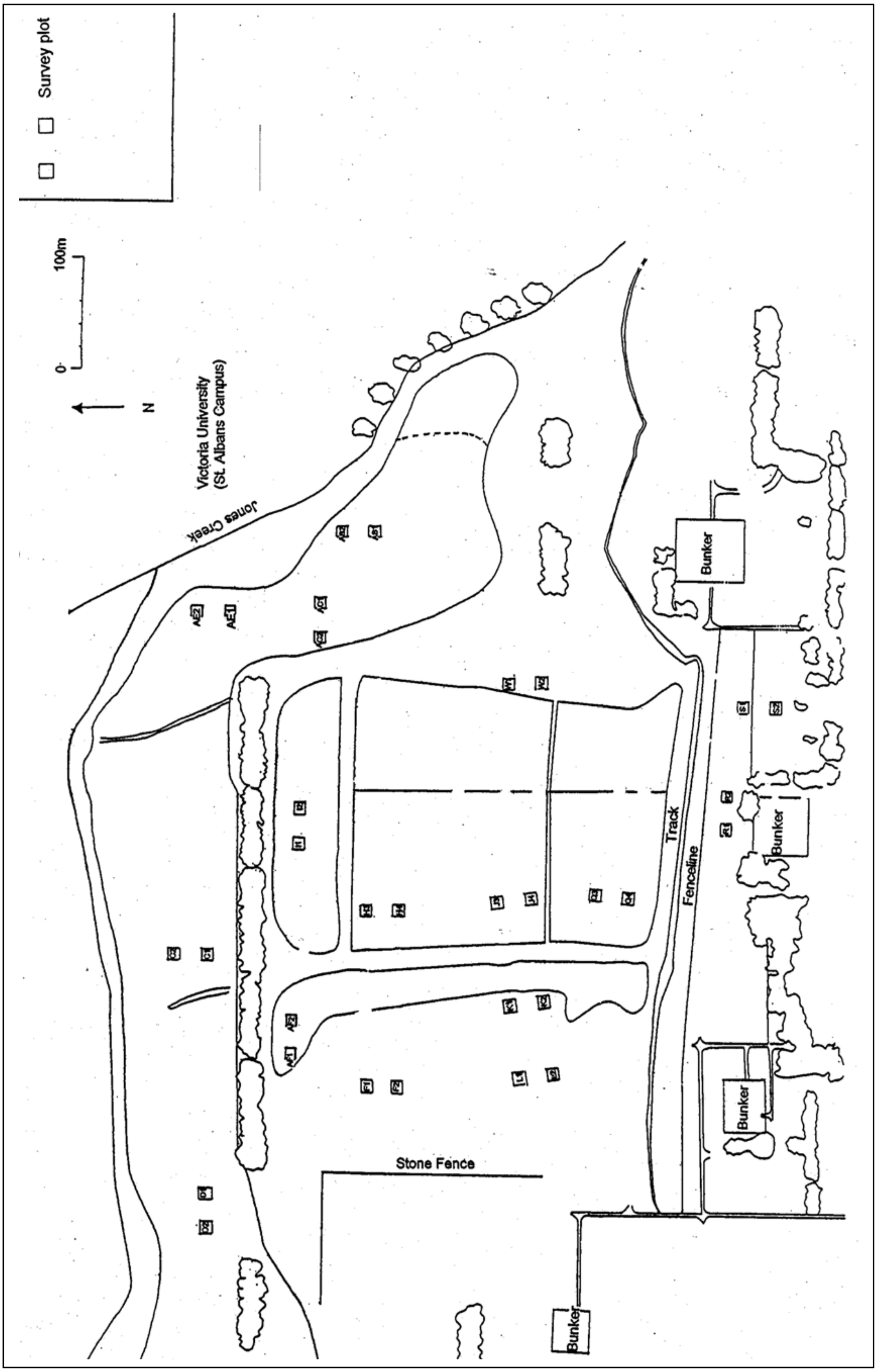


Figure 2.9: The location of survey plots for the Summer 2000 survey period. A pair of boxes represent one survey plot.

In addition, each animal was individually pyrobranded for identification purposes. Briefly, this involved assigning the right-hand row of paired ventral scales as units and assigning the left-hand row as tens, enabling a different combination for each animal. For example individual 37 would receive a brand on the 3rd left-hand scale (tens) and on the 7th right-hand scale (units). Further description of this technique is provided in Chapter 4 of this dissertation. The sexing and pyrobranding techniques could not be used accurately or humanely on individuals with a snout-vent length less than 70 millimetres. These animals were considered to be juveniles. Animals less than 50 millimetres snout-vent length were considered to be hatchlings (refer to Section 2.4.2 for further size class details). *Delma impar* were released within one metre of the point of capture, within 24 hours of the time of capture.

2.3 Preliminary analyses of 1996 data

Preliminary analyses to determine an appropriate scale of data analysis and assess any potential variations in data collected from native and exotic habitat types, were performed on the O'Shea (1996) data. As 1996 was the first year that any pit-fall trapping had been conducted, there were no possible influences from previous trapping events. The location of traps in 1996 provided for a clear analysis of the *D. impar* populations in native versus exotic vegetation.

2.3.1 Choosing a scale

In order to determine a scale at which trapping data could be analysed, the dispersion (spatial distribution) of *D. impar* captures in Autumn 1996 was assessed using the standardised Morisita index of dispersion. This index provides a measure of how populations of plants or animals are spaced across a landscape and is considered to be one of the best measures of dispersion because it is independent of population density and sample size (Krebs, 1989). The index ranges from –1.0 to +1.0, with 95 percent confidence limits at +0.5 and –0.5. Random patterns have an index of zero, clumped patterns above zero and uniform patterns below zero (Krebs, 1989).

The standardised Morisita index of dispersion was assessed at the following scales:

- number of *D. impar* captured per individual pit-fall trap;
- number of *D. impar* captured per mini-grid (4 pit-fall traps); and
- number of *D. impar* captured per paired mini-grid (survey plot = 8 pit-fall traps).

Following the selection of an appropriate scale (see Results, Section 2.5.1), the standardised Morisita index of dispersion was applied at this scale (ie. the scale of ‘survey plot’) to data for each survey period, in order to determine whether the pattern of dispersion altered over time.

2.3.2 Native and exotic vegetation

The trapping survey performed by O’Shea in 1996 was conducted in native and exotic tussock grassland vegetation. It was found that there was no significant difference between the number of *D. impar* captured in native dominated vegetation and the number of *D. impar* captured in exotic dominated vegetation. As part of the preliminary analysis of this data a standardised Morisita index of dispersion was derived for each of the two vegetation types and the dispersion of *D. impar* was found to be aggregated in each type. Thus, the lack of a significant difference in the number of *D. impar* captured in native and exotic vegetation and generally similar trends in aggregation for both vegetation types suggest that it is appropriate to pool capture data irrespective of the vegetation type.

2.4 Analyses

2.4.1 Pooling data

For the following analyses, the data collected by O'Shea (1996) was firstly analysed to detect any differences between the population of *D. impar* captured in native dominated tussock grassland and the population of *D. impar* captured in exotic dominated tussock grassland. Because no major differences were found, it was assumed that animals were from the same population and trapping and data analyses in subsequent survey periods were conducted across the study site, irrespective of native/exotic habitat types.

2.4.2 Population structure

As there was no developed method for determining the age of individual *D. impar*, individuals were assigned to one of eight size classes, based on snout-vent length measurements, enabling a description of the structure of *D. impar* populations. Based on captive (Banks *et al*., 1999) and field recapture observations (Rauhala, 1997), the *D. impar* snout-vent length measurement provides an approximate indication of age. The following size classes and age assumptions are used in this study (table 2.2):

Snout-vent length (mm)	Size class (mm)	Age class
40 - 49	45	Hatchling
50 - 59	55	Juvenile (1st year)
60 - 69	65	Juvenile (2nd year)
70 - 79	75	Mature
80 - 89	85	Mature
90 - 95	95	Mature
100 - 109	105	Mature
>110	115	Mature

Table 2.2: Assumptions of *D. impar* age classes, based on snout-vent length measurements.

To assess for possible differences in *D. impar* population structure between native dominated grassland and exotic dominated grassland, a Mann-Whitney U-test (Zar, 1996) was performed on *D. impar* snout-vent length data recorded for each of these habitat types in the Summer 1996 survey period.

Data for each survey period was pooled irrespective of habitat type. Possible differences in the population structure between survey periods was assessed using the Kruskal-Wallis independent samples test (Zar, 1996) for snout-vent length data recorded for each survey period. A nonparametric Tukey-type multiple comparison (Zar, 1996) was performed to detect differences between survey periods.

2.4.3 Sex ratios

The sex of individual *D. impar* was only recorded for the survey periods Summer 1999 and Summer 2000 and therefore no assessment was made of the sex ratios for native and exotic habitat types. For each of these survey periods, the homogeneity of sex ratios for all survey plots was assessed using a heterogeneity chi-square analysis (Zar, 1996). Departures from a 1:1 sex ratio were then assessed using chi-square analysis (Zar, 1996). In order to further quantify the sex ratio and provide a comparison with the results of the chi-square analysis, the mean ratio (Krebs, 1989) of females to males was also calculated for the above survey periods.

2.4.4 Condition

A condition factor for *D. impar* was derived from methods previously developed to assess the gross nutritional state of different age groups or populations of fish (Weatherly, 1972). Weatherly (1972) described a relationship such that any material body in which, with increasing size, the linear proportions remain constant, weight (if density remains constant) and any bodily linear dimension are related as follows:

$$K = W/L^3$$

where K is the condition factor, W is the weight in grams and L is the length in centimetres.

In order to determine the constancy of linear proportions for *D. impar*, the total length of individuals was regressed against the snout-vent length. Data from all survey periods was used in the analysis, however individuals which displayed obvious autotomised tails were removed from the data set.

Using the formula derived from the linear regression, predicted total lengths were determined from recorded snout-vent lengths. Any individuals with a measured total length which was greater or less than one standard deviation from the predicted total length were also removed from the data set. This accounted for five individuals, which all had much smaller measured total lengths than predicted total lengths, suggesting that tail autotomy had not been detected in these cases.

Weatherley's (1972) condition factor was adapted by multiplying the weight of each individual by a factor of 1,000. The condition factor was then derived for each *D. impar*, excluding those with autotomised tails (recorded and predicted as above), for all survey periods. The condition factor was regressed against the total length, to check that it was a suitable index for use over a range of total lengths. That is, for example, if the development of gonads or fat bodies with age (as measured by total length) significantly added to the weight of individuals, it would then be expected that longer animals were more likely to have higher condition factors. This would render the use of the condition factor unsuitable, as comparisons in general condition could not be made between individuals of differing total lengths. Indeed, the results of this regression indicated that the condition factor is only suitable for use with mature *D. impar* (refer to Results, Section 2.6.3).

Differences in the condition factor for *D. impar* captured in native dominated vegetation and *D. impar* captured in exotic dominated vegetation, in Summer 1996, were assessed using an independent samples t-test. Hatchlings, juveniles and individuals with evidence of tail autotomy were removed from the data set. The requirements of normality and homoscedasticity were assessed using the Kolmogorov-Smirnov goodness-of-fit (Zar, 1996) and Levene's tests (SPSS Inc., 2000), respectively.

Variation in the condition factor of *D. impar* captured in the 1996, 1997, 1999 and 2000 Summer survey periods was assessed by one-way ANOVA with a post-hoc Tukey's test (Zar, 1996). The condition factor of *D. impar* captured in the Summer 1998 survey period was not assessed due to a paucity of animals which met the requirements for calculating the condition factor value. Data was natural log transformed to meet the requirements of normality and homoscedasticity. Hatchlings, juveniles and individuals with evidence of tail autotomy were removed from the data set prior to analysis.

In order to determine whether the condition factor was significantly different between male and female *D. impar*, a factorial analysis of variance (Zar, 1996) was performed on capture data for the survey periods 1999 and 2000. The homogeneity was assessed using Levene's test of equality of error variances. Hatchlings, juveniles and individuals with evidence of tail autotomy were removed from the data set.

2.4.5 Predation

Tail autotomy is utilised by lizards as a method for avoiding predation. The frequency of tail autotomy has previously been used to comment on the possible causes of differences in the structures of populations of another pygopodid, the Pink-tailed Legless Lizard *Aprasia parapulchella* (Jones, 1999). Although the frequency of tail autotomy may provide some indication of the level of predation, it is not possible to determine whether this is an indication of predator efficiency or predation intensity, with efficient predators less likely to leave lizards with tail autotomy (Schoener, 1979; Jones, 1999).

Delma impar displaying tail autotomy were placed into the category of having been subject to predation and animals with no tail breakage were not considered to have undergone any form of predation. Log-likelihood ratio contingency tables (Sokal & Rohlf, 1995) were used to determine if there were any significant differences in predation for the following:

- native versus exotic (based on O'Shea's 1996 data);
- between survey periods;
- immature versus mature individuals (all survey periods combined); and
- females versus males (1999 and 2000 survey periods combined with immature individuals removed from data set).

2.4.6 Population size

The size of the *D. impar* population at Iramoo Wildlife Reserve was estimated for each survey period. The procedure for this was as follows:

- the average distance of movement of all *D. impar* recaptured over the five survey periods was calculated;
- the average distance of movement was used to estimate the survey area for each survey plot;
- the estimated survey area was multiplied by the number of survey plots operated for each survey period;
- because the duration of trapping varied for each of the survey periods, the shortest survey period was chosen as a cut-off point for all other survey periods and the number of animals captured within the cut-off time frame was used to calculate the density;
- the density was calculated as the total number of individual *D. impar* captured (within the cut-off time frame) over the total area sampled and converted into the number of *D. impar* per hectare;
- the density was multiplied by the total reserve size (37.5 hectares).

2.5 Preliminary results from analyses of 1996 data

2.5.1 Choosing a scale

Based on the standardised Morisita index of dispersion the dispersion of *D. impar* showed an aggregated or clumped distribution, with 95 percent confidence, at all three scales of measurement (table 2.3). The scale of 'survey plot' was therefore decided upon as the scale at which all subsequent analyses of *D. impar* trapping data would be based, as this provided the broadest scale at which data could be analysed and reduced the large number of zero counts that were recorded at the smaller pit-fall trap and mini-grid scales. When the index was applied at this scale to each survey period, captures of *D. impar* were found to be in a clumped/aggregated pattern of dispersion for all survey periods except 1998 (table 2.4).

Scale of capture	Standardised Morisita index
Individual pit-fall traps (n = 144)	0.503
Mini-grid (n = 32)	0.500
Paired mini-grid 'survey plot' (n = 18)	0.501

Table 2.3: The standardised Morisita index of dispersion at three different scales of trapping for *D. impar*, based on capture data for the 1996 survey period. SMID of 0 = random dispersion; SMID of 1 = maximum aggregation.

Survey period	Standardised Morisita index
1996	0.501
1997	0.513
1998	0.137
1999	0.513
2000	0.449

Table 2.4: The standardised Morisita index of dispersion, measured for each survey period at the 'survey plot' scale.

2.5.2 Native and exotic vegetation

Based on 1996 data the dispersion of *D. impar* in native dominated grassland vegetation had a standardised Morisita index dispersion of 0.210 and in exotic dominated grassland vegetation the index was 0.500. In both cases *D. impar* displays a degree of spatial aggregation and subsequent analyses are based on data which has been pooled irrespective of vegetation type.

2.6 Results

Overall, 302 individual *D. impar* were captured in pit-fall trap and drift-fence plots surveyed in late Summer for the years 1996 to 2000, at an average rate of 5.79 individuals per 100 survey plot days (table 2.5). An average of 6.3 percent of individuals were recaptured. The snout-vent length measurements for *D. impar* ranged from 40 millimetres to 113 millimetres and weights, for animals with full tails, ranged from 0.51 grams to 12.87 grams. Hatchlings consistently appeared in traps in mid-February, with the exception of the 1996 survey period when no hatchlings were captured (table 2.6).

Survey period	**Total individuals captured**	**Trap rate**[1]	**Trap rate**[2]	**Recaptures** (% individuals)
1996	68	6.87	8.59	4 (5.9)
1997	78	5.31	6.63	5 (6.4)
1998	12	1.75	2.19	1 (8.3)
1999	72	6.00	7.50	3 (4.2)
2000	72	9.00	11.25	5 (6.9)
Total	302	Average = 5.79	Average = 7.23	Average = 6.3%

Table 2.5: Overall survey results for the Summer survey periods 1996 – 2000.
Trap rate[1] = No. *D. impar* captured per 100 survey-plot nights.
Trap rate[2] = No. *D. impar* captured per 1,000 pit-fall trap nights.

Survey period	**Number of hatchlings**	**Hatchlings as percentage of total captures**	**Earliest trap date**
1996	0	0	N/A
1997	9	11.5	21 February
1998	3	25	13 February
1999	7	9.5	23 February
2000	12	16	16 February

Table 2.6: The first appearance of hatchlings in pit-fall traps for the Summer survey periods 1996 – 2000.

2.6.1 Population structure

There was no significant difference in the snout-vent lengths of *D. impar* captured in native dominated vegetation and the snout-vent lengths of *D. impar* captured in exotic dominated vegetation (U=520.5; n=41, 26; p=0.872) (figure 2.10a & b). The first and third quartiles of snout-vent lengths of *D. impar* captured in each of these vegetation types lies between the 85 millimetre and 105 millimetre size classes. Thus, over 50 percent of the *D. impar* captured in each of these vegetation types were mature animals. The median *D. impar* snout-vent length was in the 95 millimetre size class. The 55 millimetre size class was absent in the exotic dominated vegetation, however as this size class was represented by only one individual in native dominated vegetation, such a result could have occurred by chance.

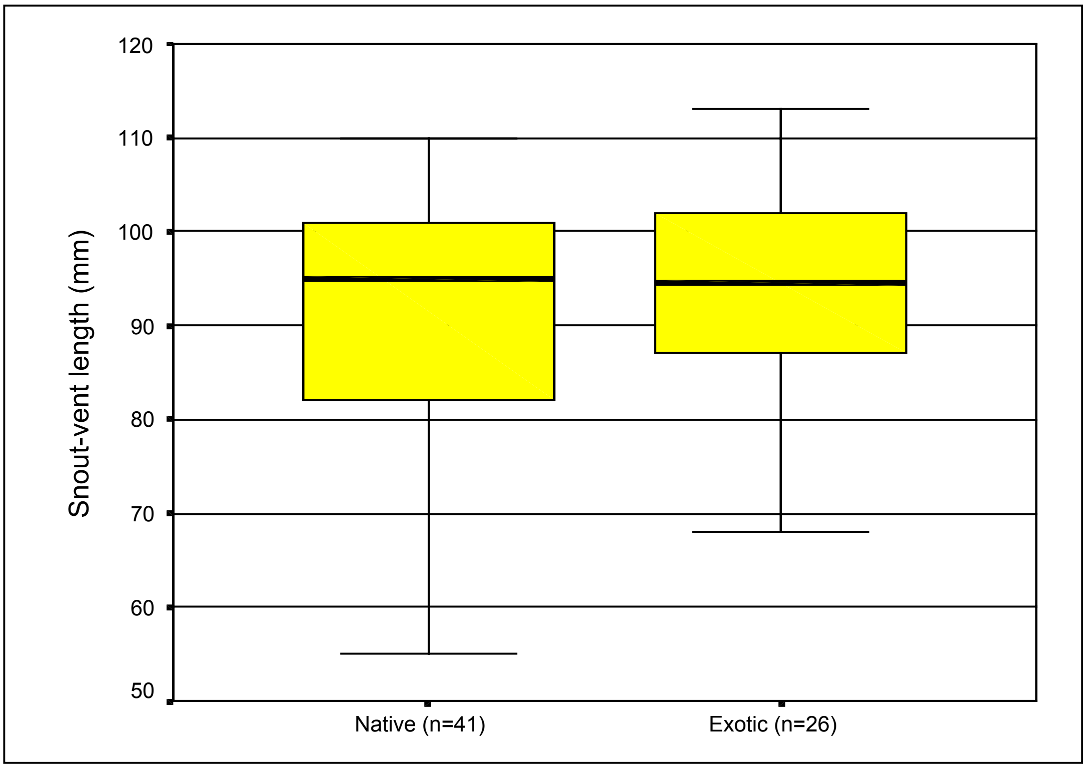


Figure 2.10a: The snout-vent length distribution of *D. impar* captured in native and exotic vegetation types in Summer 1996. Horizontal bar = median; box = first quartile (lower edge) to third quartile (upper edge); whiskers = range.

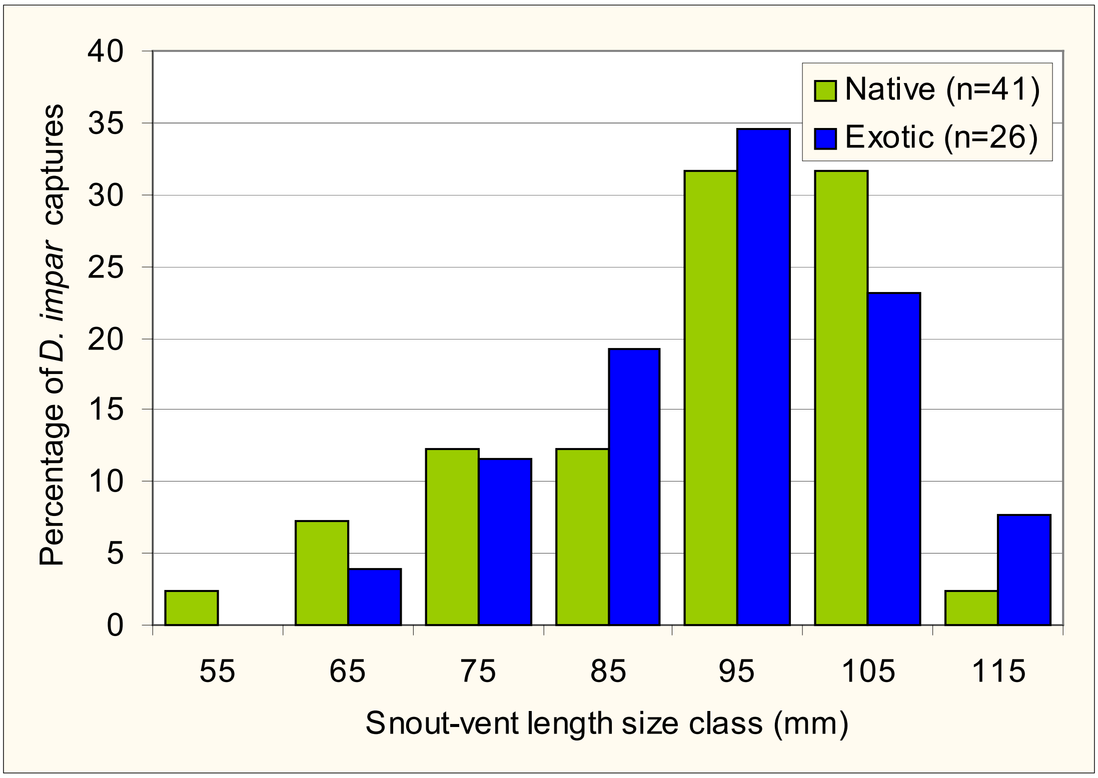


Figure 2.10b: The distribution of size classes of *D. impar* captured in native (n=41) and exotic (n=28) vegetation types, Summer 1996.

There was a significant difference in the snout-vent lengths of *D. impar* between most survey periods (H=43.112; df=4; p<0.001) (figure 2.11), with only the 1997 and 1999 survey periods not being significantly different to each other. For each survey period the location of the first and third quartiles varied, however, excluding the 1998 survey period, *D.impar* between these quartiles were animals within the mature age size classes (ie. greater than 70 millimetres snout-vent length). The median snout-vent length was greater than 75 millimetres for all survey periods.

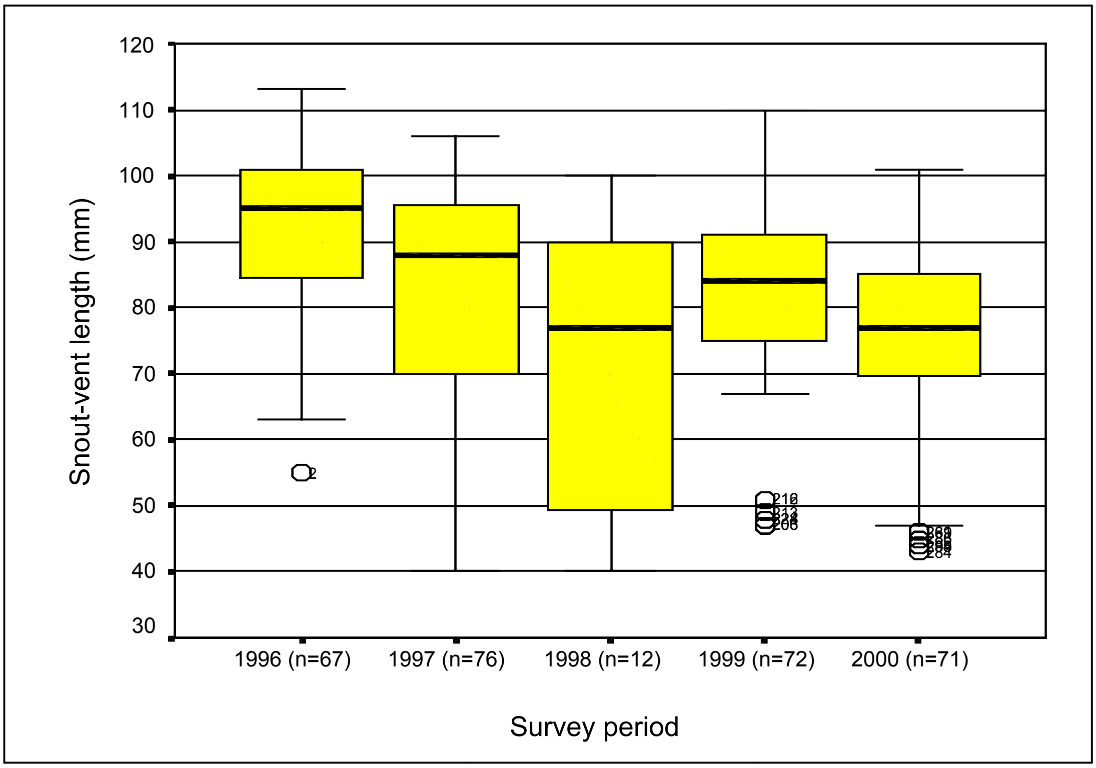


Figure 2.11: The distribution of snout-vent lengths for *D. impar* captured in the Summer survey periods 1996 – 2000. Horizontal bar = median; box = first quartile (lower edge) to third quartile (upper edge); whiskers = range; circles = possible outliers.

The hatchling age class was represented in all survey periods except Summer 1996. However, inconsistencies in the presence of hatchling cohorts, from hatchlings to first year juveniles to second year juveniles, over consecutive survey years (table 2.7), suggests that survey methods do not provide an adequate assessment of these size classes in all survey periods. For example, the presence of individuals from the 1996 hatchling cohort in subsequent survey periods suggests that a hatching event did occur for the 1995 - 1996 activity period, even though no hatchlings were recorded in the Summer 1996 survey period. Furthermore, the presence of hatchlings in Summer 1998, the absence of first year juveniles in Summer 1999 and subsequent reappearance of second year juveniles in Summer 2000 suggests that the trapping method did not always detect these cohorts in individual trapping seasons. Alternatively, the classification of age classes may be flawed as a result of differential growth rates over time, due to seasonal influences.

Hatchling cohort	Hatchlings as a % of total captures		First year juveniles as a % of total captures		Second year juveniles as a % of total captures
1996 cohort	0% (1996)	➔	6.4% (1997)	➔	8.3% (1998)
1997 cohort	11.5% (1997)	➔	8.3% (1998)	➔	6.8% (1999)
1998 cohort	25% (1998)	➔	0% (1999)	➔	8.3% (2000)

Table 2.7: Changes in the proportion of a cohort represented in the total number of captures over Summer survey periods.

2.6.2 Sex ratios

The sex ratios for data collected in Summer 1999 were homogeneous across all survey plots (χ^2= 11.829; df=19, 63; p=0.893) (92.5 percent of cells had expected values less than five) and were not significantly different from 1:1 (χ^2=0.397; df=1, 63; p=0.529). The sex ratios of data collected in Summer 2000 were not homogeneous across all survey plots (χ^2=22.449; df=12, 56; p=0.033) (100 percent of cells had expected values less than five), with two survey plots accounting for the significant difference. When the two significantly different survey plots were removed from the data set homogeneity was achieved and the resultant sex ratio was not significantly different from 1:1 (χ^2=0.581; df=1, 43; p=0.446).

In Summer 1999 the mean ratio was 0.85 females to each male. In Summer 2000 the mean ratio was 1.34 females to each male. However, when values from the two survey plots which caused a significant departure from homogeneity (as determined by chi-square analysis, above) were removed from the data set, the mean ratio was found to be 0.79 females to each male, in Summer 2000. In general terms, this suggests that perhaps there are slightly fewer females than males in the Iramoo Wildlife Reserve population.

2.6.3 Condition

A significant positive linear relationship was found between the snout-vent length and total length (r^2=0.902; b=3.127; t=41.378; n=187; p<0.001), indicating that there is a constancy of linear proportions for *D. impar* without autotomised tails (figure 2.12). Prior to calculating the condition factor data was screened (as outlined in Section 2.4.4) to remove individuals with measured total lengths greater than one standard deviation from the predicted total length. These individuals were considered to have autotomised tails which were not visually identifiable.

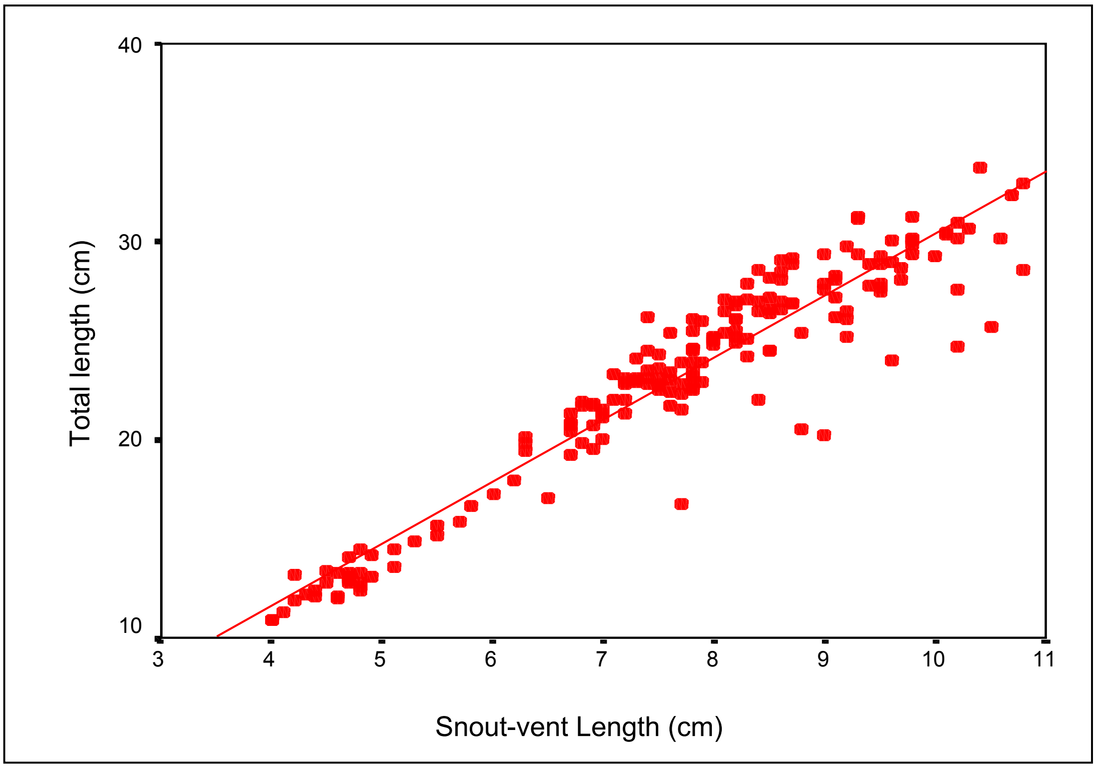


Figure 2.12: A positive linear relationship exists between the snout-vent length and the total length of individual *D. impar* (n=187) (TL= -0.836 + 3.127(SVL); r^2=0.902; p<0.001).

Regression of the condition factor against total length for all *D. impar*, excluding those with autotomised tails, revealed that although there was a small r^2 value the slope of the regression was significantly different to zero (r^2=0.264; b=-0.0005; t=-7.852; n=174; p<0.001). However, when only mature *D. impar* were included in the regresssion, no significant difference was found (r^2=0.025; b=-0.0002; t=1.784; n=128; p=0.077) (figure 2.13), indicating that use of the condition factor is only suitable for comparing populations of mature *D. impar*.

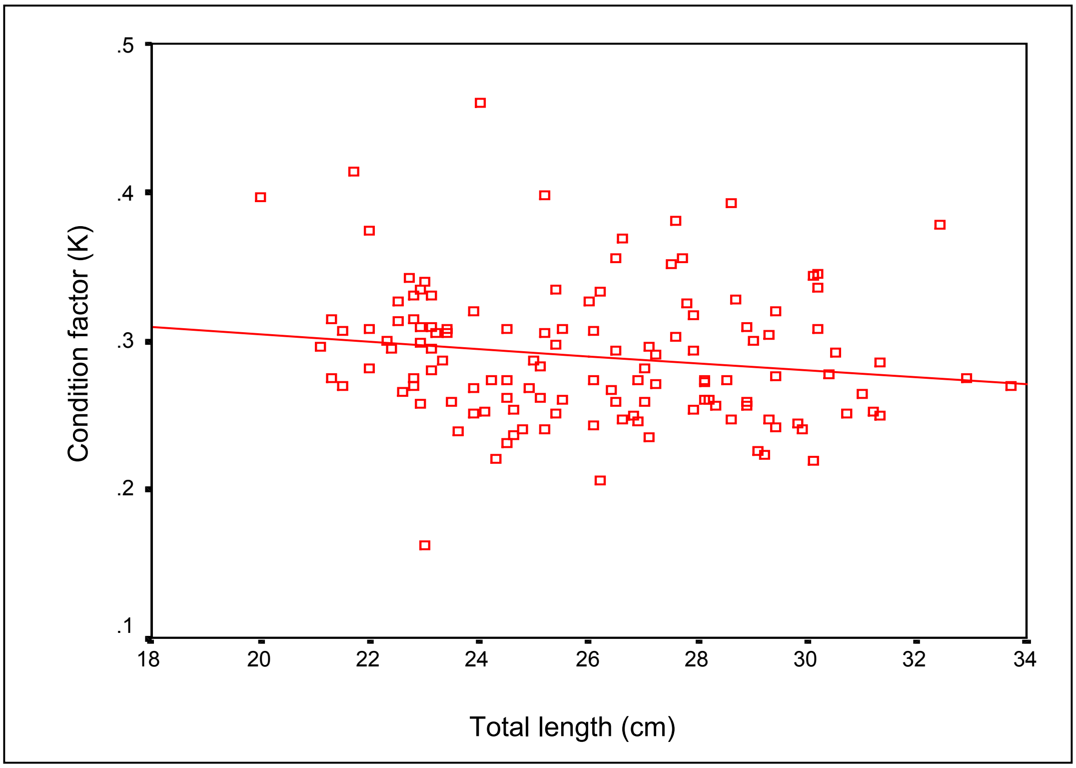


Figure 2.13: There is no significant relationship between total length and condition factor for mature *D. impar* (n=128) (K=0.425+(-0.0005)(TL)).

No significant difference was found in the condition factor for *D. impar* captured in native dominated vegetation and *D. impar* captured in exotic dominated vegetation (t=1.119; df=30; p=0.272) (figure 2.14), although the range was greater in exotic vegetation.

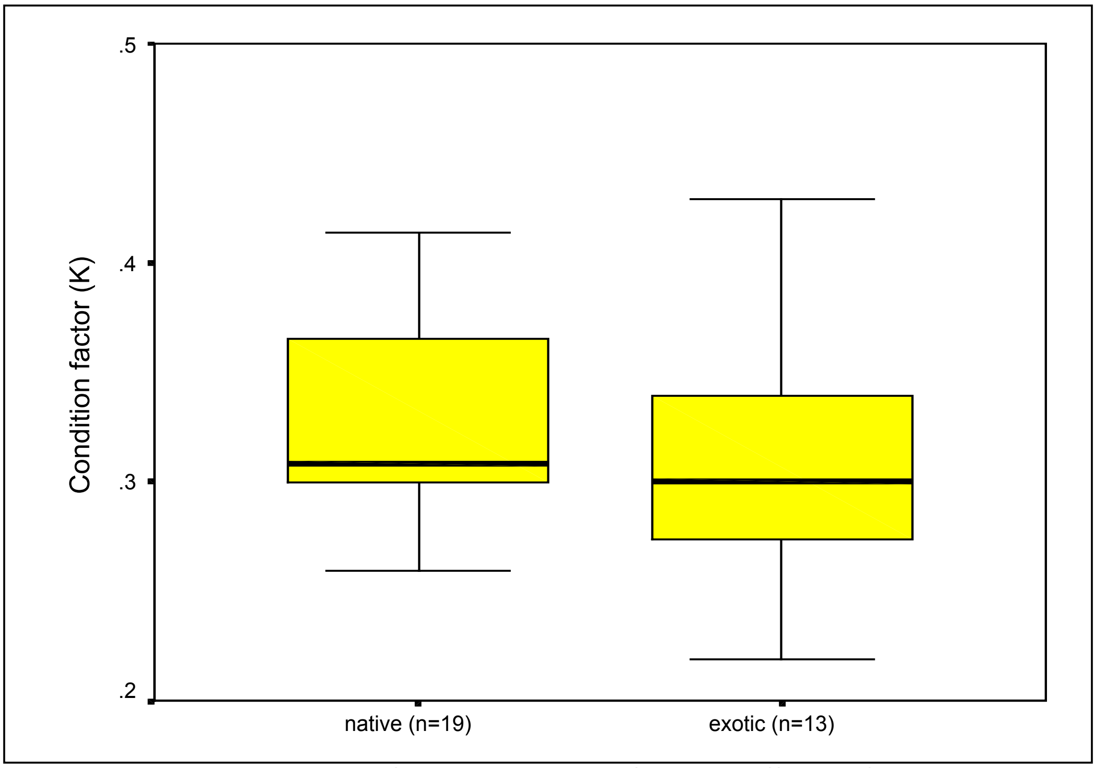


Figure 2.14: The condition factor is not significantly different for *D. impar* captured in native and exotic vegetation, Summer 1996 (p=0.272). Horizontal bar = median; box = first quartile (lower edge) to third quartile (upper edge); whiskers = range.

A significant difference in the condition factor for *D. impar* was found for Summer survey periods between years (F=5.257; df=3, 123; p=0.002), with a Tukey's test identifying the survey period Summer 1996 as being significantly different to survey periods Summer 1999 and Summer 2000. Figure 2.15 suggests that condition factors for Summer 1996 may also be different to those for the survey period Summer 1997 and this was found to be the case when the possible outlier was removed from the data set. In all cases, the condition factor of individuals for the Summer 1996 survey period was greater than for other survey periods (table 2.8).

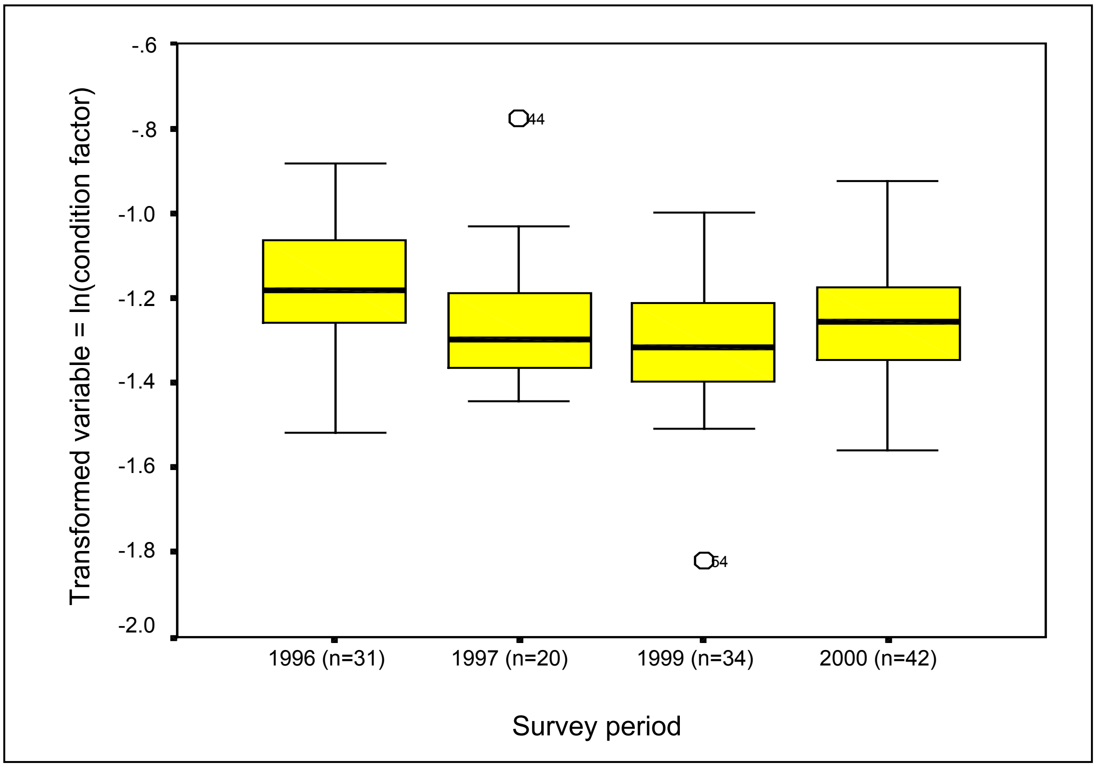


Figure 2.15: The condition factor for *D. impar* captured in survey period Summer 1996 is significantly different to those captured in Summer 1999 and Summer 2000 (p=0.002). Horizontal bar = median; box = first quartile (lower edge) to third quartile (upper edge); whiskers = range; circles = possible outliers.

Survey period	Number of *D. impar*	Average condition factor
Summer 1996	31	0.315
Summer 1997	20	0.289
Summer 1999	34	0.273
Summer 2000	42	0.286
Average for all survey periods		0.291

Table 2.8: The average condition factor for *D. impar* captured in the Summer survey periods 1996, 1997, 1999 and 2000.

A factorial analysis of variance to test for differences in the condition of male and female *D. impar* over the survey periods Summer 1999 and Summer 2000 found no significant interaction between the factors of 'survey period' and 'sex' (F=2.002; df=1,69; p=0.162; *partial* η^2=0.03). In agreement with the above results, no significant difference was found in the condition of *D. impar* between the survey periods Summer 1999 and Summer 2000 (F=1.754; df=1, 69; p=0.190; *partial* η^2=0.025). Furthermore, no significant difference was detected in the condition of male and female *D. impar*

(F=3.650; df=1, 69; p=0.06; *partial* η^2=0.05), suggesting that the condition factor can be pooled for the sexes.

2.6.4 Predation

The level of predation as determined by the proportion of tail breakages to complete tails was not found to be significantly different between native dominated vegetation types (n=42) and exotic dominated vegetation types (n=26) (*G*=0.755; df=1; p=0.541). Data for each survey period was pooled irrespective of vegetation type and a significant difference was found in the level of ‘predation’ between survey periods (*G*=15.686; df=4; p=0.003) (figure 2.16). Pooling data across all survey periods, a significant difference was also found in the level of ‘predation’ between immature (n=59) and mature (n=238) *D. impar* (*G*=23.460; df=1; p<0.001), with mature animals displaying a greater proportion of tail breakages (figure 2.17). There was no significant difference in the level of ‘predation’ between females (n=59) and males (n=54) for the combined Summer 1999 and Summer 2000 survey periods (*G*=0.456; df=1; p=0.499) (figure 2.18).

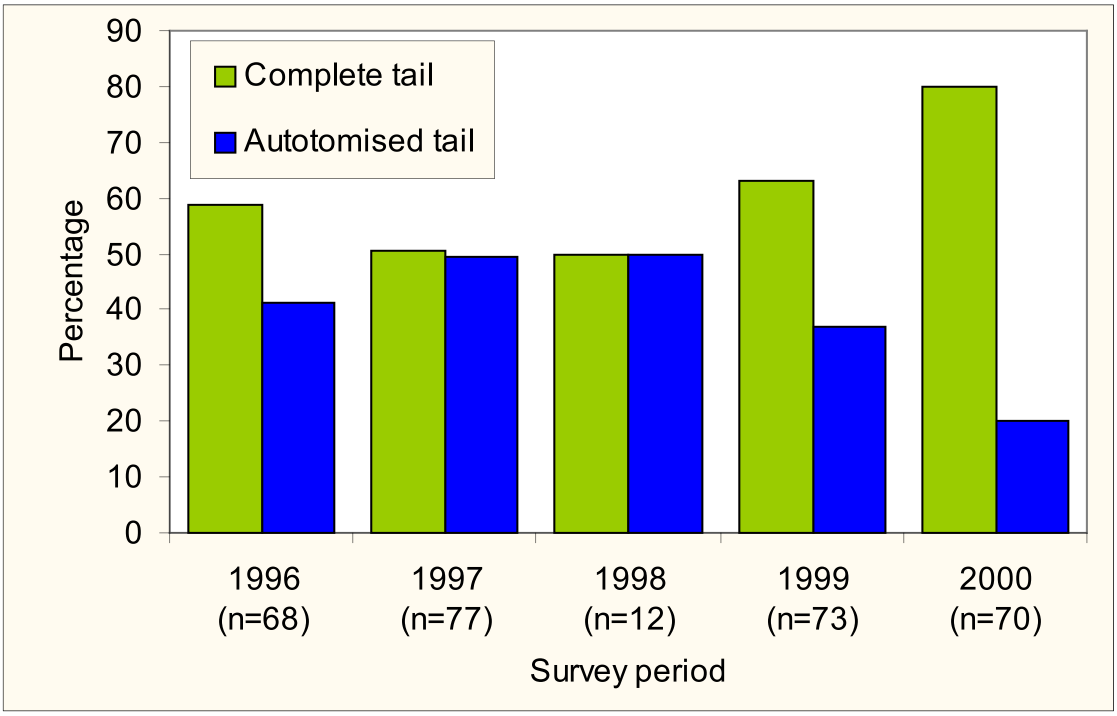


Figure 2.16: The proportion of *D. impar* displaying autotomised tails to complete tails for the Summer survey periods 1996 – 1999.

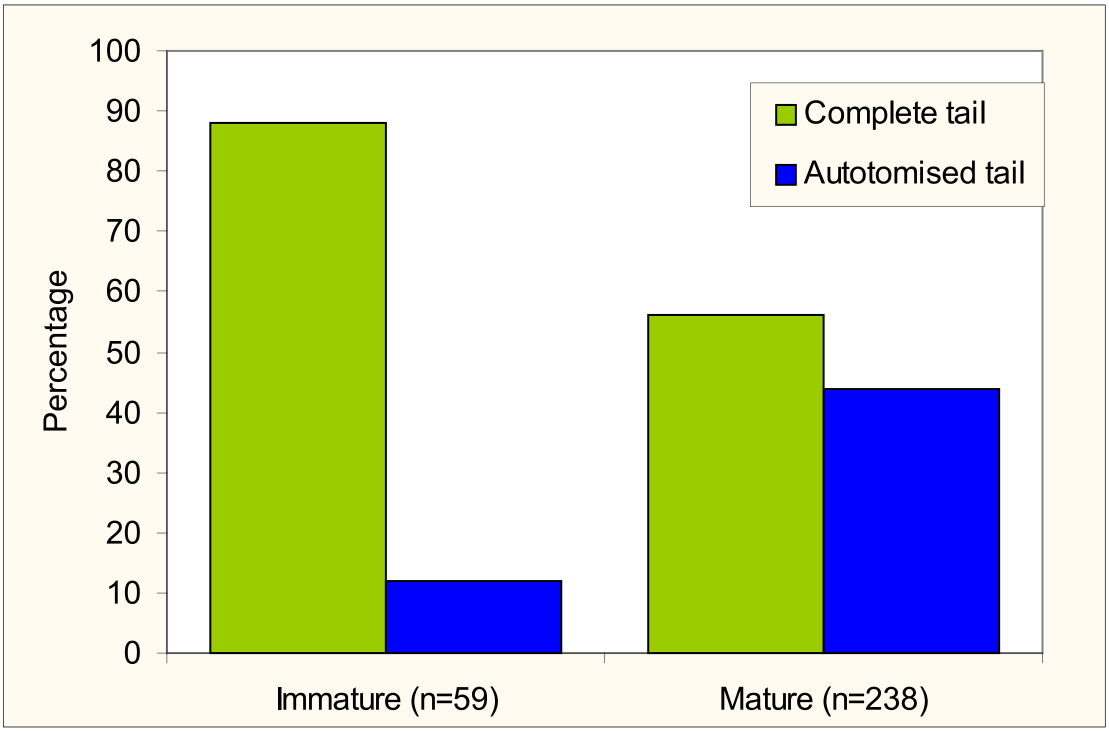


Figure 2.17: The proportion of *D. impar* displaying autotomised tails is much greater for mature individuals than immature individuals (p<0.001).

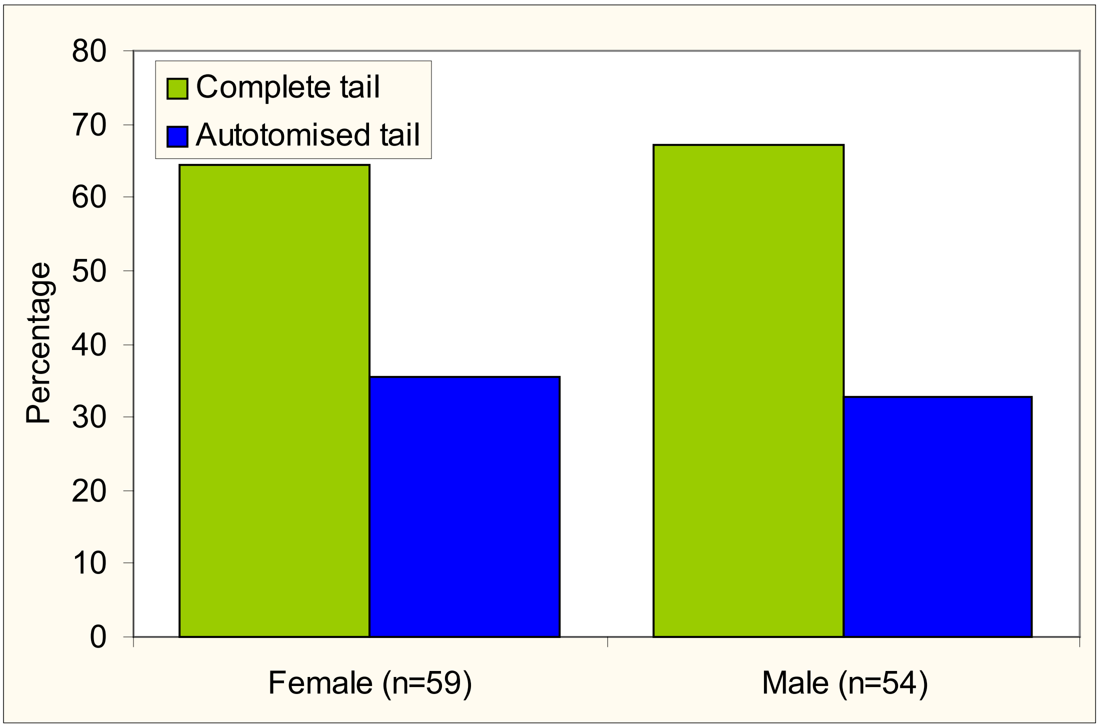


Figure 2.18: There is no difference in the proportion of female and male *D. impar* displaying tail autotomy (p=0.499) in the Summer survey periods 1999 and 2000 combined.

2.6.5 Population size

The average distance of movement of individual *D. impar* (n=20) between recapture events over the Summer survey periods 1996 – 2000 was 10.68 metres. This was approximated to 10 metres and added to each side of the drift-fence perimeter of the survey plot. Thus, it was assumed that each survey plot was capable of surveying an area of 0.18 hectares for *D. impar*. The shortest trapping duration was 49 days in the Summer 1997 survey period and capture data after this number of days was not considered in the population estimate for the Summer survey periods 1996, 1998, 1999 and 2000. Using this method for assessing the size of the *D. impar* population at Iramoo Wildlife Reserve, the estimate ranged from 208 *D. impar* in Summer 1998 to 912 *D. impar* in Summer 2000, with an overall average of 602 *D. impar* (table 2.9). The average density was 16.05 *D. impar* per hectare.

Survey period	**Number of survey plots**	**Total survey area** (hectares)	**Number of *D. impar* captured** (over 49 trapping days)	**Density** (no. *D. impar* per hectare)	**Population estimate for IRW**
1996	18	3.24	64	19.75	740.7
1997	30	5.40	78	14.44	541.7
1998	12	2.16	12	5.56	208.3
1999	24	4.32	70	16.20	607.6
2000	16	2.88	70	24.31	911.5

Table 2.9: Estimates of the size of the *D. impar* population at Iramoo Wildlife Reserve for the Summer survey periods 1996 – 2000.

These population estimates are considered to be an underestimate. It is apparent from the pattern of capture that, with the exception of the Summer 1998 survey period, the number of *D. impar* captured in the assessed survey plot trapment areas over the defined 49 day trapping duration did not approach 100 percent (figure 2.19).

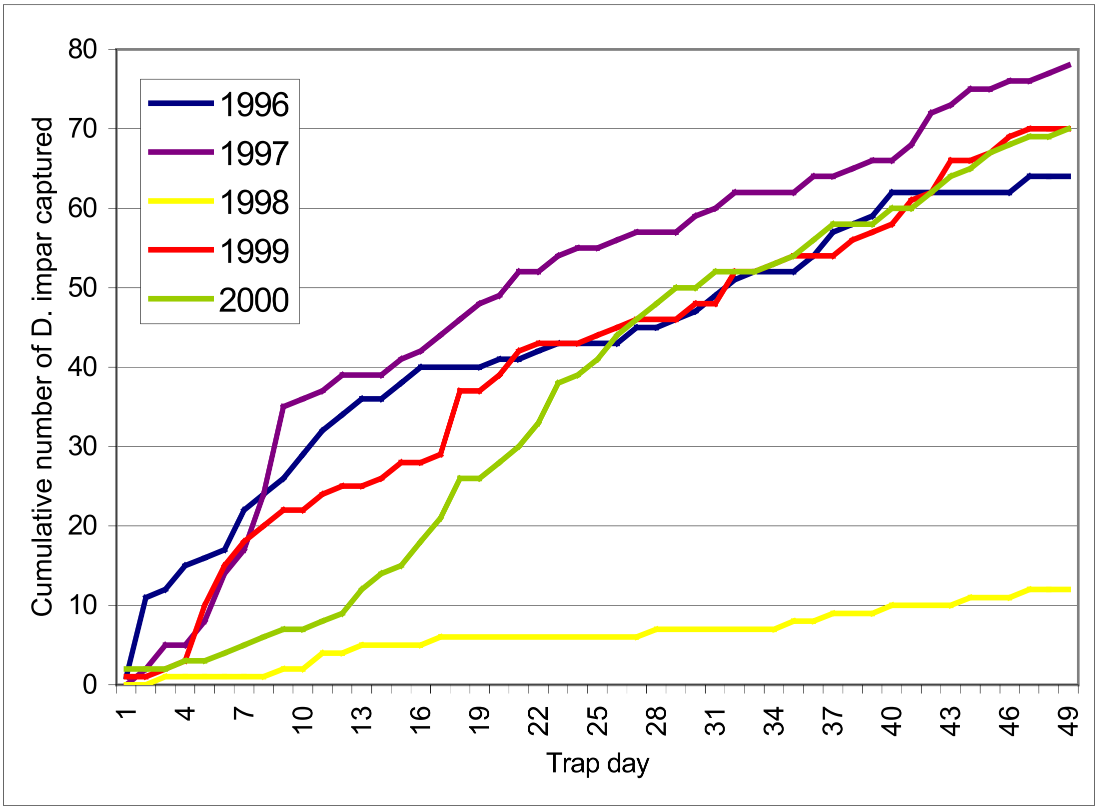


Figure 2.19: The pattern of capture of *D. impar* over 49 days trapping in the Summer survey periods 1996 – 2000.

2.7 Discussion

The average overall Summer survey period trap rate for *D. impar* at Iramoo Wildlife Reserve and the trap rate for each of the survey periods are comparable with rates reported at other sites in Victoria (O'Shea, 1996) and the Australian Capital Territory (Rauhala, 1999). Variability in year to year trap rates appear to be a feature of pit-fall trapping for *D. impar* (Kukolic *et al.*, 1994; Rauhala, 1999; Dunford *et al.*, 2001). Rauhala (1999) suggests that trap rates may be influenced by factors such as variability in weather conditions, habitat management and vegetation cover across survey periods, and that variation in trap rates does not necessarily reflect variations in *D. impar* abundance over time. Trap rates are also influenced by the trapping programme itself, in that the pattern of capture appears to be erratic at the commencement of trapping, with high initial capture rates which eventually level out to a more stable trapping pattern (Coulson, 1990; Corrigan *et al.*, 1996; O'Shea, 1996). Thus, trapping programmes which are conducted over short periods of time are more likely to observe higher capture rates than trapping programmes conducted over longer periods of time.

The snout-vent lengths of *D. impar* captured at Iramoo Wildlife Reserve were within the range reported by Coulson (1990), however extremes of weights were outside both the minimum and maximum weights reported by Coulson (1990). Hatchlings consistently appeared in the traps in mid-February which is in contrast to the emergence of hatchlings in late January in captivity (Banks *et al.*, 1999) and in early March in the wild in the Australian Capital Territory (Kukolic, 1994). Differences in the timing of hatching are most likely a result of differences in prevalent climatic conditions between the field locations and also between conditions in the wild and conditions in captivity. It is unknown whether climatic conditions have an influence on the timing of mating or the duration of incubation.

2.7.1 Dispersion

An aggregated spatial dispersion of *D. impar* has been noted in previous studies (O'Shea & Hocking, 1997a), with truncated Poisson (Rauhala, 1997) and geometric distribution (Dorrough & Ash, 1999) models fitted to capture data. Whether the cause of such aggregation is behavioural or due to heterogeneity of the environment is unknown. Similar increases in local densities have been noted in other species of pygopodid and Greer (1989) comments that while this suggests a degree of social behaviour beyond simple mating encounters, the phenomenon could also simply be due to the accumulation of individuals in favourable parts of the habitat. No apparent differences have been observed at Iramoo Wildlife Reserve between habitat with high densities of *D. impar* and habitat with low densities of *D. impar*. Furthermore, the location of aggregations appears to move over time (O'Shea & Hocking, 1997a). The more random pattern of distribution recorded in the Summer 1998 survey period is thought to arise from the poor capture result which was most likely due to extreme drought conditions.

2.7.2 Native and exotic vegetation

Exotic dominated grassland tussock vegetation appears to provide an extension of usable habitat for *D. impar* at Iramoo Wildlife Reserve. No significant difference was found in the number of captures, size class distribution, condition factor or predation levels (as indicated by tail autotomy) for *D. impar* captured in native dominated tussock habitat and *D. impar* captured in exotic dominated tussock habitat. This is consistent with Nunan's (1995) findings that the species is able to reproduce in highly disturbed and non-native sites, with no evidence of a reduction in body condition. Further support is lent to the work of Dorrough & Ash (1999) and Hadden (1998) who concluded that *D. impar* is not dependent on any particular plant species or community or habitat structure, providing that tussock vegetation forms a component of the habitat and sites are within the geographical range of *D. impar* and temperate lowland native grasslands of south-eastern Australia.

Although it is accepted that *D. impar* can occur in exotic dominated tussock grassland habitats, it is generally thought that such sites are marginal or sub-optimal (Coulson, 1990; Hadden, 1995). Exotic dominated tussock habitat at Iramoo Wildlife Reserve maintains a clear link with areas of native dominated tussock habitat, however the importance of this connectivity remains unclear. It would be useful to investigate how the parameters measured in this study are affected at sites dominated by exotic tussock vegetation which are isolated from native dominated tussock vegetation, in order to determine whether exotic habitat is in fact marginal or whether such sites should be considered suitable for supporting viable populations of *D. impar* and managed accordingly. Until such evidence is obtained, the ongoing survival of *D. impar* in exotic dominated habitats should not be discounted.

2.7.3 Population structure

The population structure of *D. impar* at Iramoo Wildlife Reserve, based on size classes, is consistent with that of a long-lived species, in that there is a larger proportion of mature individuals than juveniles or hatchlings. This is in agreement with previous longevity estimates based on recapture data (Rauhala, 1997), captive observations (Banks *et al.*, 1999) and comparisons with the sister family of lizards Gekkonidae.

It is interesting to note that despite no records of hatchlings in the 1996 survey period, first year juveniles were recorded in the 1997 survey period. Perhaps this suggests that egg deposition was later or that incubation took longer than normal in the 1995/1996 reproductive season and that hatchlings did not emerge until after trapping was complete. Seasonal climatic factors might be responsible for such a delay.

Alternatively, hatchlings may have been present but inactive and thus avoided capture in the 1996 survey period. It is apparent that not all size classes are successfully recorded in all survey periods. The causes of such apparent variability and the potential impacts on overall population structure and thus viability are unknown. Such variability in the detection and abundance of hatchlings and juveniles between survey periods has been noted in other studies (Rauhala, 1999) and may not necessarily be a true reflection of the structure of the population for each survey period. Further work needs to be undertaken to develop appropriate methods that reliably sample hatchlings and juveniles and provide a true reflection of the overall population structure.

Overall viability in the apparent structure of the *D. impar* population and a progressive decline in the median snout-vent lengths across survey periods is difficult to interpret and may not provide a true reflection of any changes which might impact on population size and reproductive success. The consistency of the median snout-vent length being greater than 75 millimetres for all survey periods and the location of the first and third quartiles being within the mature age size classes (with the exception of the 1998 survey period when a low sample size of *D. impar* was recorded), suggests that any real changes in population structure did not have a dramatic impact, over the duration of the study.

2.7.4 Sex ratios

For the purposes of population viability assessments (ARAZPA, 1996), it was assumed that *D. impar* populations have an even sex ratio, due to a lack of consistency in ratios recorded in previous studies (Rauhala & Andrew, 1998; Banks *et al*., 1999; Rauhala, 1999) and suggestions that this is the case for most species of pygopodid (Patchell & Shine, 1986; Greer, 1989).

The use of chi-square analysis to develop an understanding of *D. impar* sex ratios at Iramoo Wildlife Reserve is problematic. The small sample sizes available for chi-square analysis resulted in a very high proportion of cells (in some cases 100 percent of cells) having expected frequency values of less than five. It is a requirement of the chi-square test that fewer than 20 percent of cells have expected frequencies of less than five (Zar, 1996), otherwise the test is considered to be violated and the likelihood of a non-significant result (Type II error) is increased.

Given the high proportion of violating cells in the chi-square analyses, the result of a non-significant deviation from an even sex ratio should be questioned. Although it does not seem unlikely that sex ratios are roughly even, the need to question this notion is further supported by the mean ratios which suggest slightly fewer females than males, and also by the highly skewed ratios which were recorded for some survey plots.

2.7.5 Condition

The condition factor provides an indication of the health of *D. impar* within a population. It is useful for assessing both sexes after mating and egg laying has occurred but is not recommended as a tool for assessing potentially gravid females. The condition factor can only be applied to *D. impar* which have complete tails and is not useful for assessing immature animals.

No difference in the condition of mature individuals was detected between *D. impar* in native dominated habitat and *D. impar* captured in exotic dominated habitat. Given that the species is considered to be relatively sedentary with a small home range, this suggests that exotic dominated habitat at Iramoo Wildlife Reserve is a viable habitat and that animals are able to meet at least their dietary and thermoregulatory requirements.

It is speculated that fluctuations in the condition factor of the *D. impar* population at Iramoo Wildlife Reserve across Summer survey periods are most likely due to the prevalent seasonal conditions and fluctuations in dietary resources. Further investigation of this is required.

2.7.6 Predation

Based on the proportion of *D. impar* displaying tail autotomy, it appears that the level of predation on the species fluctuates from year to year at Iramoo Wildlife Reserve. However, based on the information available it is not possible to interpret in which survey periods animals were subject to more or less predation. This is also the case for the difference detected in the proportion of autotomised tails between immature *D. impar* and mature *D. impar*. In both cases, a high proportion of autotomised tails could indicate a high level of predation but poor predator efficiency. That is, a high level of tail autotomy may indicate a high level of predation on *D. impar* but that many predation attempts are unsuccessful and *D. impar* manage to escape predators by use of tail autotomy. Alternatively, a low proportion of autotomised tails could also indicate a high level of predation but with a high predator efficiency, thus leaving only animals which have not been subject to predation and therefore do not display tail autotomy.

Although tail loss has been attributed to predation attempts in this study, it is possible that some proportion of tail loss may be a result of intraspecific aggression. Such interactions have been recorded for other lizard species (eg. Jennings & Thompson, 1999), however no such observations have been made and documented for *D. impar* or other pygopodid lizards.

2.7.7 Population size

Yearly estimates of the size of the population of *D. impar* at Iramoo Wildlife Reserve were all within the same magnitude of order. However, these are considered to be an underestimate. For each survey period except Summer 1998, it was apparent from the pattern of cumulative captures over time that the trapping programme had not come close to capturing 100 percent of *D. impar* within the defined 49 day trapping period.

Although the pattern of trapping for *D. impar* in the Summer 1998 survey period suggests that the trapping programme was approaching the capture of 100 percent of the individuals within the trapment area, this seems unlikely. The trap rate and estimate of the population size for the Summer 1998 survey period were both very low. In subsequent Summer survey periods, both the trap rate and the population estimate figures were greater than the average. Given the species' low fecundity, it is unlikely that the length of time between the Summer 1998 survey period and the subsequent survey periods would have been sufficient for the population to recover from such a substantial crash, if such a population crash had occurred as is suggested by the trap rate data. Fluctuations in yearly trapping rates have previously been noted (Rauhala, 1999). Rauhala (1999) suggests that such fluctuations are possibly a result of adverse environmental and habitat condition, which affect trappability, and do not necessarily reflect true changes in the abundance of *D. impar* at a site. Further investigation of the causes of these observed fluctuations in trap rates across survey periods is required.

Given the longevity of the species, it would be expected that some proportion of individuals captured in one survey period would still be present at the site in subsequent survey periods. The apparent trap-shyness of the species following initial capture means that these individuals are not included in yearly population estimates, despite the likelihood that many are still present at the site and potentially even within the same trapping area as previously observed. Thus population size estimates are further considered to be underestimated as a result of this.

2.8 Conclusion

The population of *D. impar* at Iramoo Wildlife Reserve occurs in an aggregated spatial distribution in both native and exotic tussock habitat, with no observable differences in abundance, population structure, condition of individuals or patterns of predation.

The average population size estimate was 602 *D. impar* over the entire reserve area. However, this figure is considered to be an underestimate due to the apparent trap-shyness of the species and fluctuations in yearly trapping rates and patterns which did not appear to reflect the true abundance of the species. The average trap rate was 5.79 *D. impar* per 100 survey plot days or more conventionally, 7.23 *D. impar* per 1,000 pit-fall trap days. The average density was 16 *D. impar* per hectare, which is higher than the average densities estimated for any of the cluster populations at the Population and Habitat Viability Assessment (PHVA) Workshop (ARAZPA, 1996). On this basis it may be valuable to reconsider estimates of *D. impar* densities for each cluster population for any future PHVAs that might be performed.

Pit-fall trapping with drift-fences in Summer survey periods was useful for collecting data to assess overall sex ratios, condition and patterns of predation for *D. impar*. In relation to the population structure, the information gained using pit-fall trapping in the Summer survey periods was variable and does not appear to be reliable in terms of examining the hatchling and juvenile components of the population. Although some information was gleaned about the mature component of the population, it is difficult to know whether this is a true reflection of fluctuations within the population or simply a result of variability in trapping success.

The data collected using pit-fall traps with drift-fences in Summer suggests that the sex ratio of the Iramoo Wildlife Reserve population is close to even, possibly with slightly fewer females than males. Although the population structure varied across survey periods, mature *D. impar* were the most commonly trapped component of the population. The median snout-vent length was greater than 75 millimetres for each survey period. The overall condition of the population also fluctuated across survey periods, with an overall average condition factor of 0.291.

The *D. impar* population at Iramoo Wildlife Reserve is the largest known population and the most closely studied. The results of pit-fall trapping with drift-fences over five Summer survey periods at this site provides a basis for assessing alternative techniques of capture or detection, including seasonal timing and also provides a basis for the assessment of other *D. impar* populations, especially those within the West Melbourne, Keilor, Werribee Plains cluster (Smith & Robertson, 1999).

Chapter 3

Pit-fall Trapping and an Alternative Method for Assessing Striped Legless Lizard Populations

Chapter 3

Pit-fall trapping and an alternative method for assessing Striped Legless Lizard populations

3.1 Introduction

Drift-fences with pit-fall traps are commonly used to inventory, monitor and quantitatively sample populations of amphibians, reptiles, small mammals and invertebrates (Gibbons & Semlitsch, 1981; Corn, 1994). Although there have been several variations in configuration, pit-fall trapping with drift-fences has been the conventional method for monitoring Striped Legless Lizard *Delma impar* populations (for example: Coulson, 1990; Kutt, 1992; Dorrough, 1995; O’Shea, 1996; Rauhala, 1997; Rauhala, 1999; Dunford *et al.*, 2001). Occasionally systematic searching such as rock-rolling has also been used (Whitby, 1995; Hadden, 1992).

Pit-fall trapping with drift-fences has enabled the collection of useful information on *D. impar*, such as the species’ distribution, broad habitat requirements and potential threatening activities, with some insights into aspects of the reproduction, behaviour and movement. However, it was recognised in 1995 that the monitoring techniques employed were not providing sufficient information about the biology of the species to assist with recovery efforts (Webster *et al.*, 1995). This lack of information was further highlighted at the Population and Habitat Viability Assessment Workshop (ARAZPA, 1996) where many of the inputs to the model were simply estimated or assumed.

Although pit-fall trapping with drift-fences is reasonably effective at initially capturing individual *D. impar* (albeit at apparently low capture rates), the level of recaptures within and between survey periods is poor and restricts the collection of follow-up information on individuals within a given population, and the estimation of population sizes or densities. Presumably, in some cases low capture and recapture rates are a result of reduced population densities at a given site. However, *D. impar* is considered to be a cryptic species (Webster *et al.*, 1995) and there is speculation and some evidence that animals becomes trap shy and learn to avoid or overcome pit-fall trap and drift-fence capture systems. For example, Coulson (1990) has observed *D. impar* springing over drift-fences and Kutt (1992) tracked individuals moving quite close to pit-falls without becoming captured.

In addition to the limited data that has been collected on the species' basic biology using pit-fall trap and drift-fence methods, these techniques are costly in terms of equipment, require a substantial amount of time and labour for installation, and the installation process can be damaging to the grassland habitat (Gibbons & Semlitsch, 1981; Moro, 1990; Corn, 1994; Coulson, 1995) and potentially influence the behaviour and therefore capture success of the animals.

It has been recommended that efforts be made to improve the pit-fall and drift-fence trapping technique or devise alternative methods of detection, to enable the collection of data on the basic biology of *D. impar* (Coulson, 1995; ARAZPA, 1996; Smith & Robertson, 1999). This has been assigned as a high priority under the Striped Legless Lizard National Recovery Plan (Smith & Robertson, 1999).

3.1.1 The effect of leaving drift-fences in place or removing them between survey periods

In most studies for *D. impar*, drift-fences are removed at the end of each survey period and re-installed immediately prior to the recommencement of trapping (pit-fall traps are generally left in place with lids secured to prevent any captures). However, some surveys have been conducted using pit-fall traps and drift-fences which have been left in place between survey periods

(Moro, 1990; Mills, 1992; Whitby, 1995; O'Shea & Hocking, 1997a; O'Shea & Hocking, 1997b). This reduces the amount of labour required to commence a pit-fall trapping operation and also reduces the amount of disturbance to the grassland habitat.

Although Whitby (1995) attempted to test the effect of permanent and temporary drift-fences between survey periods on *D. impar* capture rates, his results were inconclusive due to a lack of captures in either treatment. Despite this, he went on to suggest that the removal of nets between survey periods would not guarantee captures. Others have suggested that permanent drift-fences and the duration of time over which traps are operated could increase the opportunities for *D. impar* to learn to avoid the traps (Coulson, 1990; Kutt, 1992; O'Shea & Hocking, 1997a) and thus reduce both initial capture rates and recapture rates. Friend *et al.* (1989) further suggested that the soil disturbance created by the recent installation of drift-fences may serve to attract semi-arid fossorial vertebrate species and found a significantly greater number of snakes and legless lizards (treated as a single group) in traps with temporary drift-fences.

The study reported in this chapter attempted to clarify whether permanent drift-fences compared with temporary drift-fence have any influence on capture and recapture rates and whether such a slight alteration to the trapping method would add to the knowledge gained in trapping programmes and decrease the cost, labour and habitat disturbance involved.

3.1.2 The effect of season

Pit-fall trap and drift-fence surveys for *D. impar* have generally been conducted during the warmer months of the year, ranging from November to March (for example: Coulson, 1990; Kutt, 1992; Rauhala, 1997). Generally it is thought that a greater capture success can be achieved in the November and December periods (Kukolic, 1994), due to increased activity as a result of mating. However, it is also possible that *D. impar* is more surface active during Spring in order to meet thermoregulation requirements, whereas animals may be able to adequately maintain preferred body temperatures below the soil surface in warm Summer periods. Kukolic *et al.* (1994) compared catchability indices for November and December and found that highest catchabilities were achieved in November. However, this data was based on a regime of continuous trapping and did not take into account the pattern of trapping as described in Chapter 2, where a high number of *D. impar* are captured in the initial stages of the trapping programme before a more stable pattern of trapping is achieved.

The study reported in this chapter attempted to determine whether the timing of pit-fall and drift-fence trapping at Iramoo Wildlife Reserve could optimise capture success, using replicated sampling conducted in Spring and Summer survey periods.

3.1.3 Roof tiles as artificial shelters and as an alternative to pit-fall trapping

Delma impar have been located under rocks (Whitby, 1995; Hadden, 1992), timber (Michael, 2001), asphalt and corrugated iron sheeting, suggesting that artificial refuges may provide an alternative method for detecting and monitoring *D. impar* populations. Artificial refuges have previously been used to successfully monitor other reptile and amphibian populations and communities (for example: Schulz *et al.*, 1995; Barker & Hobson, 1996; Heinze, 1997; Sutton *et al.*, 1999; Lowson & Avery, 2000; Minahan *et al.*, 2002). Initial studies to assess whether roof tiles and paving bricks could be used as an alternative monitoring technique for *D. impar* were discouraging due to a paucity of individuals detected and a lack of evidence of use of these artificial refuges (O’Shea & Hocking, 1997b; Rauhala, 1997).

However, subsequent observations of *D. impar* under roof tiles of a different style, randomly located at Iramoo Wildlife Reserve, suggested that perhaps the style of roof tile and the method in which they were placed in the habitat were important factors. It was also speculated that there may be a delay period in setting up roof tile monitoring grids and their subsequent use by *D. impar*. Delay periods between the lay out of artificial refuges and their subsequent use by reptiles has been suggested by other authors (Barker & Hobson, 1996; Schulz *et al.*, 1995).

The study reported in this chapter attempted to determine whether roof tiles as artificial shelters could be used as a technique for monitoring, describing and estimating the *D. impar* population at Iramoo Wildlife Reserve and the potential application of this methodology for detecting and assessing other *D. impar* populations.

3.2 Methods

3.2.1 The effect of leaving drift-fences in place or removing them between survey periods

Pit-fall and drift-fence trapping procedures, as described in Chapter 2, were used to assess the effect of leaving drift-fences in place or removing them between survey periods, on capture success for *D. impar*. Traps with permanent drift-fences had the drift-fence left in place and erect between the end of one survey period and the commencement of the next survey period (approximately nine months). Before trapping recommenced the permanent drift-fences were checked to ensure that they were intact and properly tensioned, with the lower edge buried below the soil surface. Traps with removed and replaced drift-fences had the drift-fences removed within a few weeks of the cessation of one survey period and were re-installed immediately prior to the commencement of the next survey period (approximately nine months). This series of experiments were conducted in conjunction with standard *D. impar* surveys which commenced in January and ended in March (as described in Chapter 2 and are hereafter referred to as ‘Summer’ trapping) for three survey periods. In addition, the experiments were conducted for an additional survey period in Spring (hereafter referred to as ‘Spring’ trapping).

The duration between the end of one survey period and the commencement of trapping for the Spring survey period was approximately seven months. Table 3.1 (next page) provides a summary of the survey periods in which the experiments were performed and the number of replicates for permanent drift-fences and removed/replaced drift-fences. Figures 3.1 – 3.4 indicate the distribution of survey plots used in each of the experimental survey periods.

Trap data was assessed for normality and homoscedasticity using the Kolmogorov-Smirnov goodness-of-fit (Zar, 1996) and Levene's tests (SPSS Inc., 2000) respectively. Data not meeting these requirements were transformed by Ln(x+1), as indicated in table 3.1. An independent samples t-test (Zar, 1996) was used to analyse the data for each survey period. It was not feasible that hatchlings (as described in Chapter 2) could have been influenced by the treatments in this experiment, as they would have had no prior experience of the pit-fall trapping system, thus they were removed from the data sets prior to analysis.

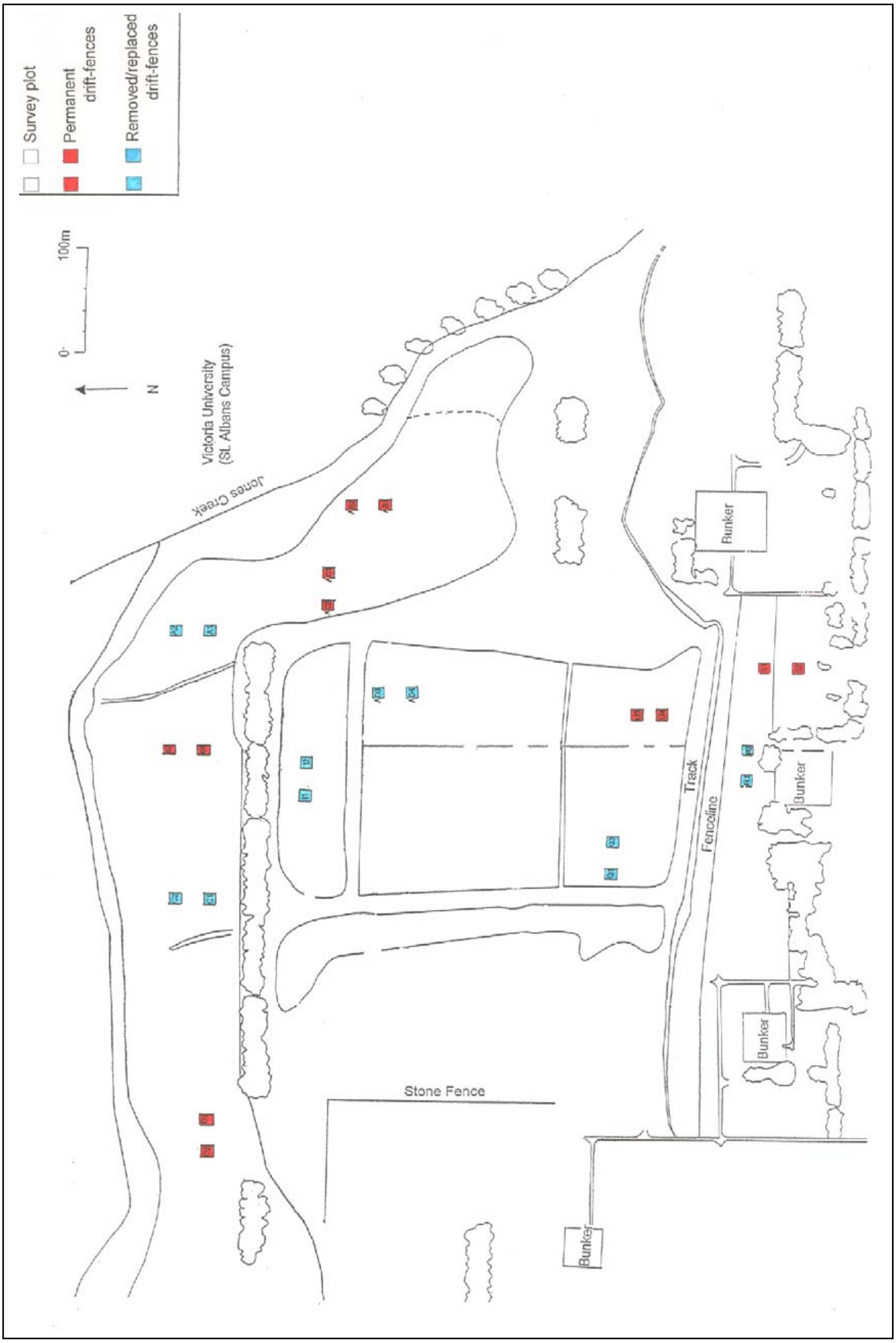


Figure 3.1: The location of survey plots with permanent drift-fences and removed/replaced drift-fences, Summer 1998. A pair of boxes represent one survey plot.

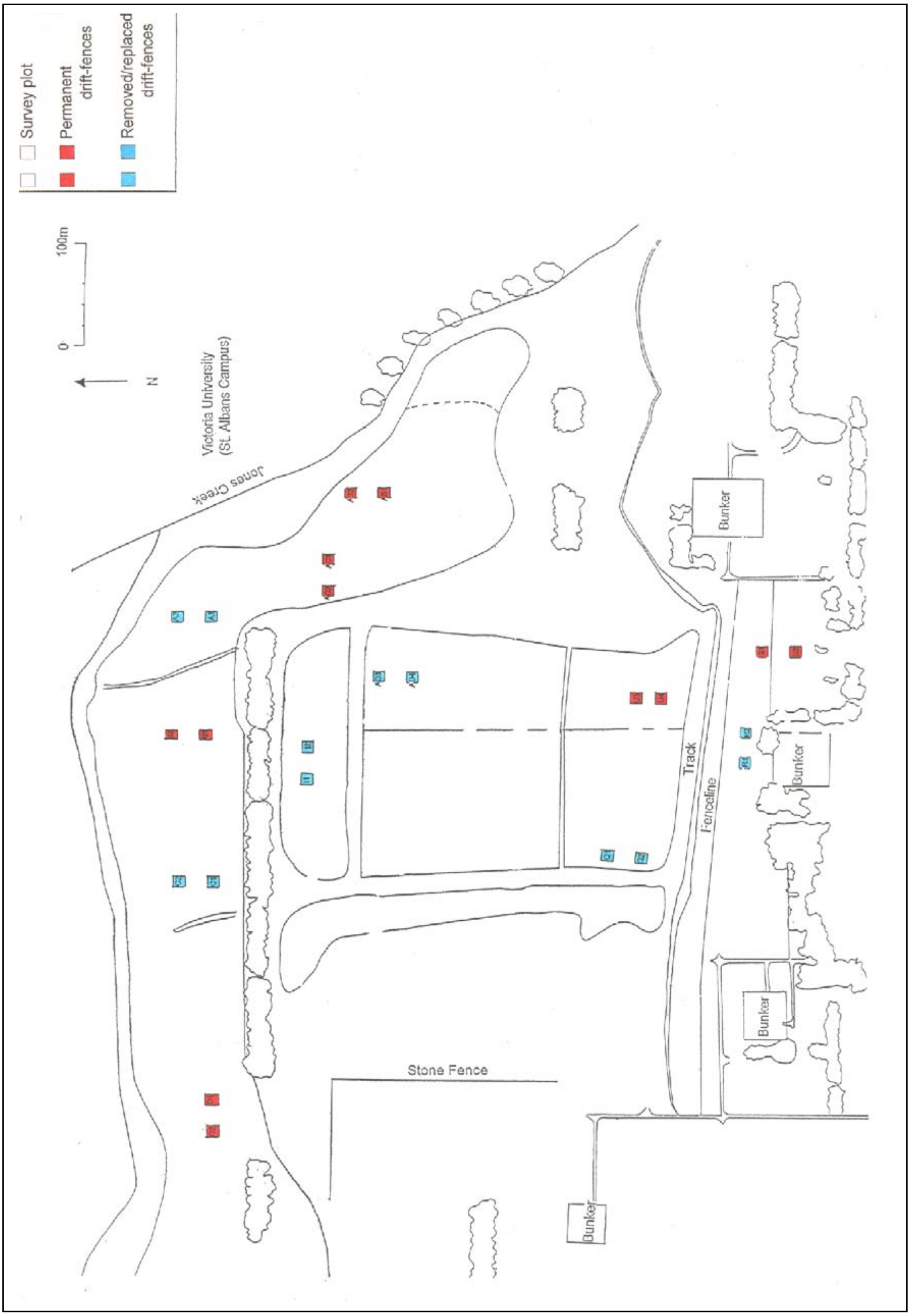


Figure 3.2: The location of survey plots with permanent drift-fences and removed/replaced drift-fences, Summer 1999. A pair of boxes represent one survey plot.

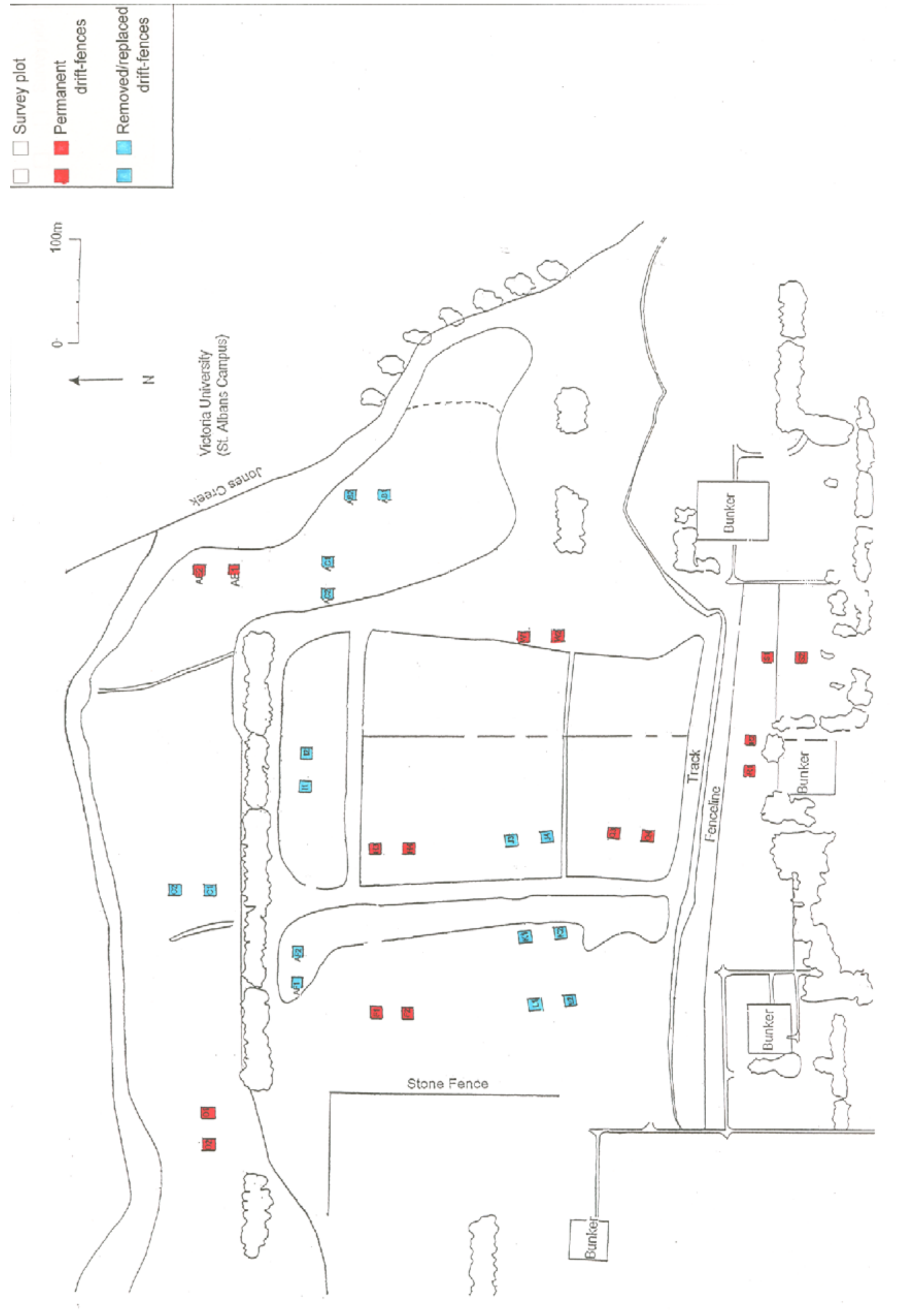


Figure 3.3: The location of survey plots with permanent drift-fences and removed/replaced drift-fences, Summer 2000. A pair of boxes represent one survey plot.

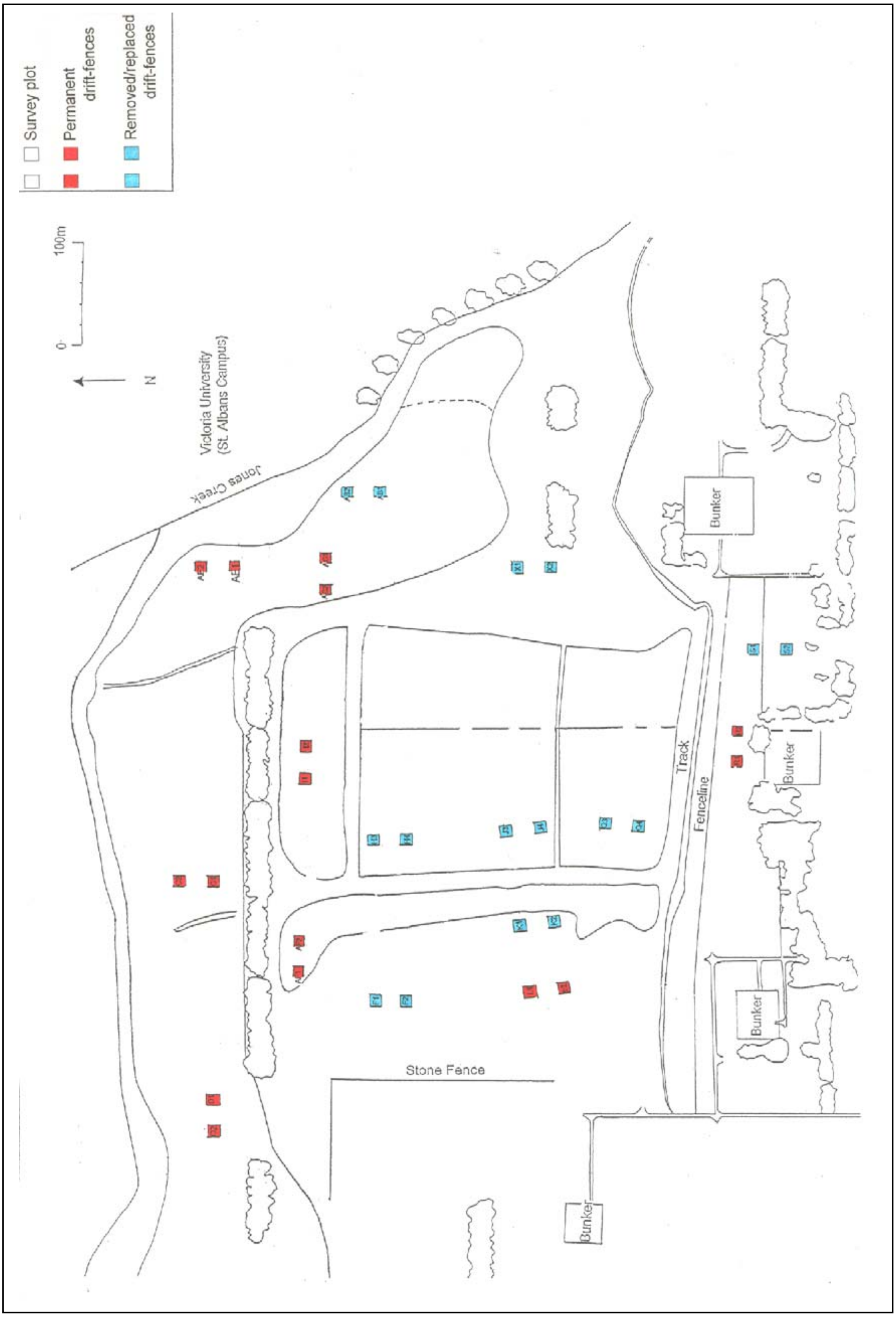


Figure 3.4: The location of survey plots with permanent drift-fences and removed/replaced drift-fences, Spring 2000. A pair of boxes represent one survey plot.

Survey period	Dates	No. days	No. survey plots (permanent)	No. survey plots (removed/ replaced)	Transformation of data for normality
Summer 1998	13 Jan – 23 Mar	57	6	6	None
Summer 1999	10 Jan – 26 Mar	50	6	6	Ln(x+1)
Summer 2000	14 Jan – 17 Mar	50	8	8	None
Spring 2000	15 Nov – 19 Dec	25	8	8	Ln(x+1)

Table 3.1: A summary of the survey methods for assessing the comparative influence of permanent drift-fences and removed/replaced drift-fences on the efficiency of trapping of *D. impar* at Iramoo Wildlife Reserve.

3.2.2 The effect of season

In order to assess the effect of the timing of surveys on the success of capture of *D. impar*, surveys were conducted in the Spring and Summer periods, over two years. Pit-fall trapping was conducted for each of these time periods using the procedures described in Chapter 2. Table 3.2 provides a summary of trapping dates, number of days trapping and the number of plots surveyed for each season and year. Figures 3.5 and 3.6 indicate the distribution of survey plots for the 1998/1999 and 1999/2000 seasonal comparisons, respectively.

	Spring			**Summer**		
Year	**Survey dates**	**No. days**	**No. survey plots**	**Survey dates**	**No. days**	**No. survey plots**
1998/1999	23 Nov – 24 Dec	27	6	10 Jan – 26 Mar	50	6
1999/2000	12 Nov – 23 Dec	26	8	14 Jan – 17 Mar	50	8

Table 3.2: A summary of the survey methods for assessing the effect of season on the efficiency of trapping for *D. impar*.

Because the pattern of trapping for *D. impar* is not constant (see Chapter 2) only captures recorded in the first 25 days of trapping were included in the analyses. In order to compare capture data for Spring and Summer, data were pooled for the Spring 1998 and Spring 1999 survey periods and also for the Summer 1999 and Summer 2000 survey periods and analysed using an independent samples t-test. Data were found to meet the requirements of normality and homoscedasticity using the Kolmogorov-Smirnov goodness-of-fit and Levene’s tests respectively. The standardised Morisita index of dispersion (Krebs, 1989) (refer to Chapter 2) was calculated for each survey period.

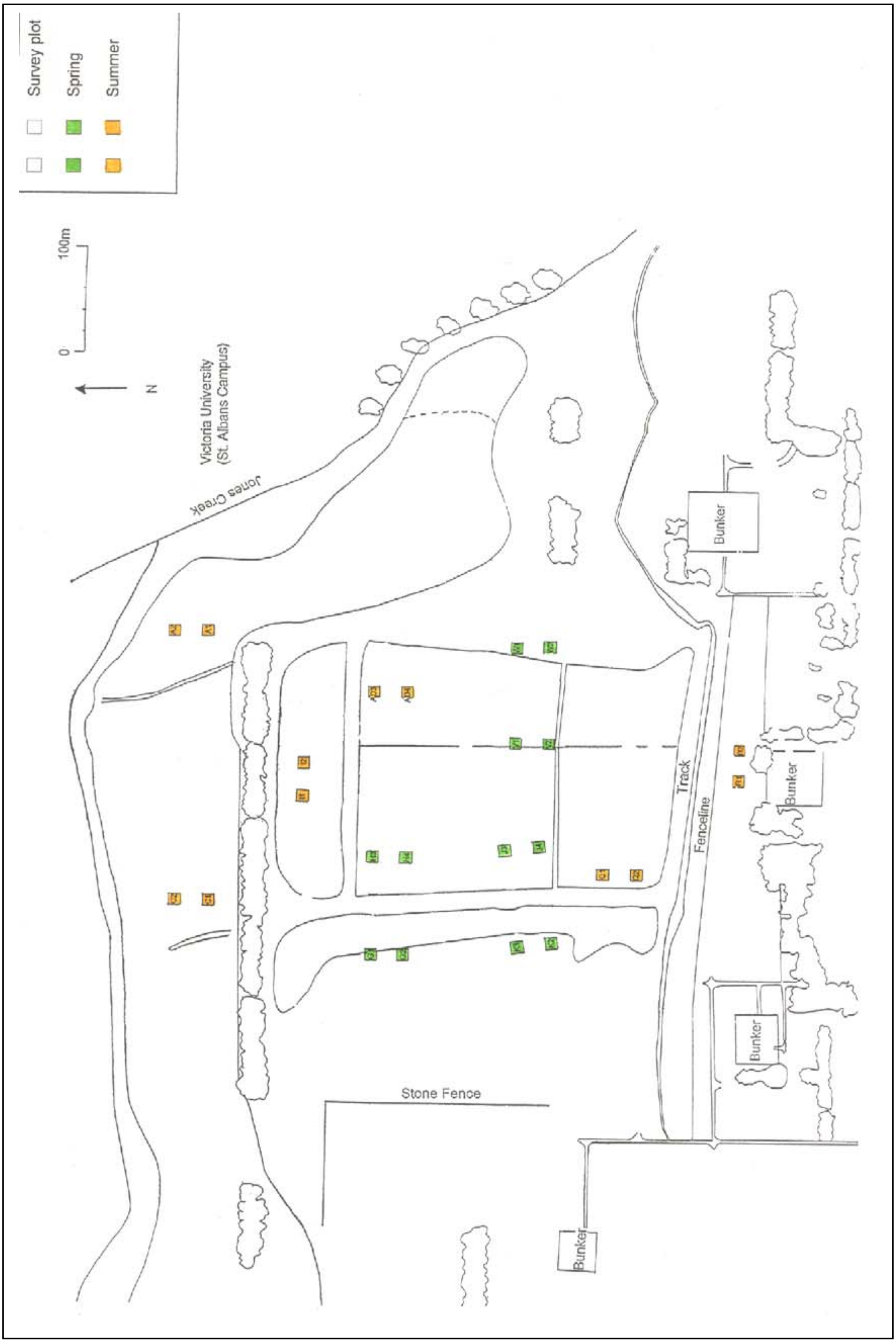


Figure 3.5: The location of survey plots in Spring 1998 and Summer 1999. A pair of boxes represent one survey plot.

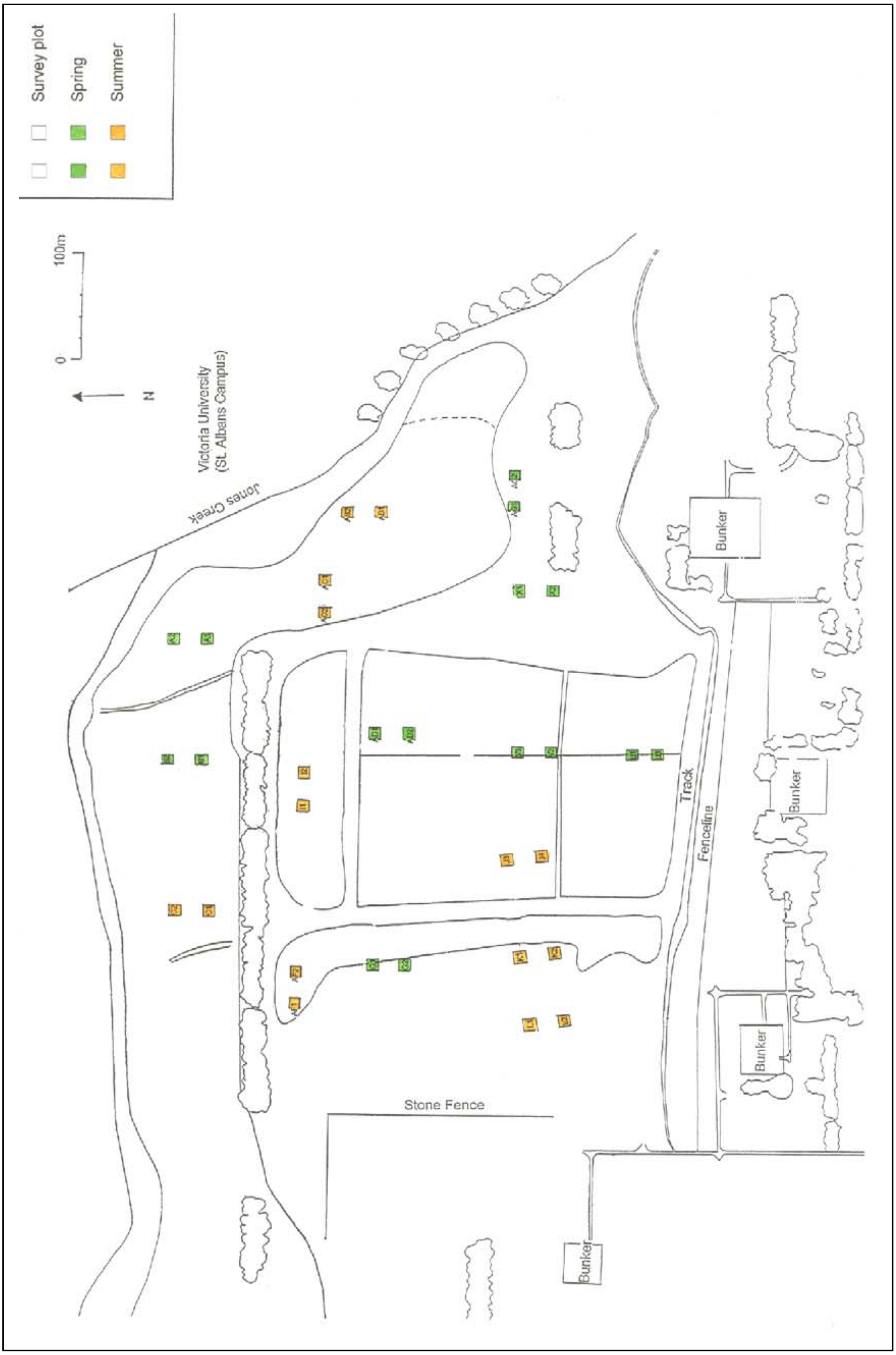


Figure 3.6: The location of survey plots in Spring 1999 and Summer 2000. A pair of boxes represent one survey plot.

3.2.2.1 Population structure

The snout-vent length data of *D. impar* captured in Spring survey periods was compared to the snout-vent length data of *D. impar* captured in Summer survey periods using a Mann-Whitney U-test (Zar, 1996).

3.2.2.2 Sex ratios

Potential variations in *D. impar* sex ratios across the Spring and Summer survey periods were assessed using heterogeneity chi-square analysis (Zar, 1996), and chi-square analysis (Zar, 1996) and mean ratios (Krebs, 1989).

3.2.2.3 Condition

The condition factor was calculated for males only for each of the Spring and Summer survey periods, using the criteria and formula described in Chapter 2. The condition factor was not calculated for females, due to the potential influence that increased mass as a result of gravidity during the Spring reproductive period would have on the calculation. In this study it was not possible to accurately determine whether or not females were gravid and thus the effect of gravidity on the calculation of a condition factor was not assessed. The condition factors of male *D. impar* for each of the Spring survey periods and each of the Summer survey periods were compared using independent samples t-tests. Following no significant differences in the condition factor between survey periods within each of these seasons, Spring and Summer condition data for each survey period were pooled. The condition factor of male *D. impar* in Spring was compared to the condition factor of male *D. impar* in Summer, using an independent samples t-test.

3.2.3 Roof tiles as artificial shelters and as an alternative to pit-fall trapping

Monier ‘Elabana’ roof tiles (42 centimetres by 32 centimetres) and paving bricks (23 centimetres by 12 centimetres) were originally set out as described in O’Shea and Hocking (1997b) on 20 November 1996. In summary, six replicate plots were located in Kangaroo Grass *Themeda triandra* dominated vegetation at Iramoo Wildlife Reserve. Each plot was arranged as a four by four grid, with a one metre interval between each grid point. A roof tile or paver was randomly assigned to each grid point with a total of eight roof tiles and eight pavers per plot. The grass beneath each tile/paver was clipped to the soil surface and removed, prior to placement (O'Shea & Hocking, 1997b). Each grid was located approximately 50 metres from the nearest neighbouring grid. Monitoring of these tiles and pavers between February and July 1997 failed to detect the presence of *D. impar* (O'Shea & Hocking, 1997b).

In September 1997, several *D. impar* were inadvertently observed under a different style of roof tile ('Standards' cement tiles measuring 43 centimetres by 28 centimetres) which were incidentally located at various random points of Iramoo Wildlife Reserve. Twenty-eight of these tiles had simply been tossed into the grassland in no specified order and grass mats had formed underneath. In response to this, a further six replicate grids were set out using the 'Standards' tiles, in January 1998, to determine whether this style of tile would provide a more successful means for studying the *D. impar* population at Iramoo Wildlife Reserve.

Grids of 'Standards' tiles were located 20 metres adjacent to the original tile/paver grids (figure 3.7). Sixteen 'Standards' tiles were arranged in four by four grids, allowing a 2.5 metre space between each tile. In order to determine whether the removal of vegetation prior to placement was a factor influencing *D. impar* use of these artificial shelters, each of the 16 tiles were randomly assigned as either mown (8 tiles) or unmown (8 tiles). Mown tiles had the vegetation beneath clipped and removed prior to placement. Unmown tiles were simply placed on top of the vegetation, with efforts to ensure that each tile lay flat on the ground, without large gaps between the substrate (as might occur if a tile was placed directly on a tussock).

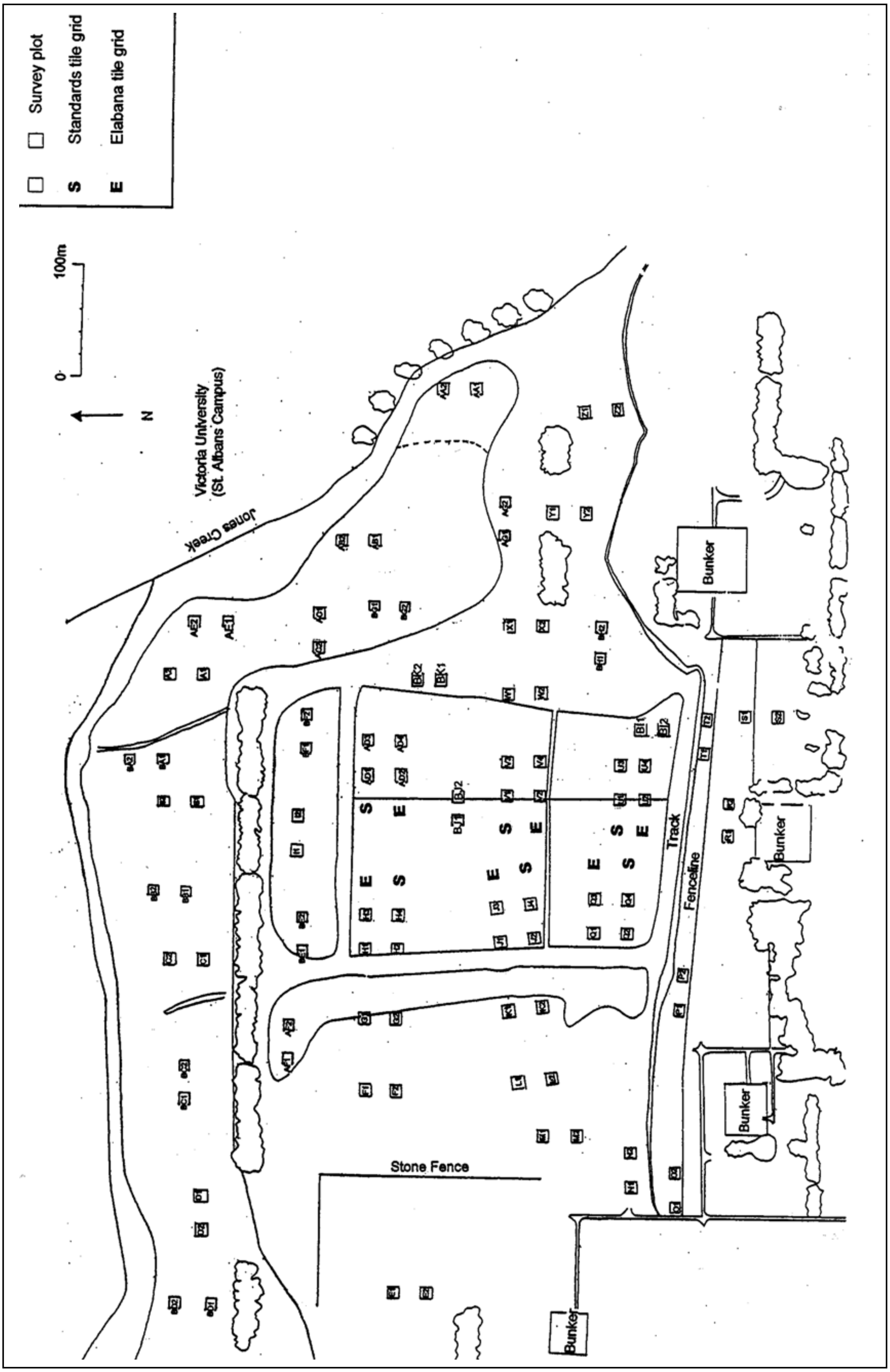


Figure 3.7: The location of Elabana and Standards roof tiles grids.

In order to assess the length of settling time required between tile placement and utilisation by *D. impar*, a further eight 'Standards' tiles were placed within each of the mown/unmown grids on 19 September 2000 (figure 3.8). Tiles were placed as for unmown treatments (see above).

Both the 'Elabana' and 'Standards' roof tile grids were regularly monitored at approximately monthly intervals between February 1998 and December 2001, with the frequency of monitoring increasing to no greater than once per fortnight in the warmer months. A short time after this part of the study commenced, a general sense of weather conditions which were favourable for observing *D. impar* under the tiles was acquired. In an effort to optimise the number of observations of *D. impar* under tiles, efforts were made to conduct tile monitoring on days when the ambient air temperature at ground level was close to 20°C, there was a cool breeze and approximately 50 percent cloud cover. Despite these efforts, monitoring was conducted over a range of times and conditions and the following protocols were followed in an effort to maximise the success of roof tile monitoring:

1. Half (14 tiles) of the randomly located 'Standards' tiles were checked (i.e. tiles not located in formal monitoring grids). If two or more *D. impar* were observed under these tiles then the survey proceeded to check all tiles in formal monitoring grids. If less than two *D. impar* were observed, then the survey was suspended;
2. Three days or more later, the other half of the randomly located 'Standards' tiles were checked. Again, the survey would only proceed if two or more *D. impar* were observed under these tiles;
3. Three or more days later, all roof tiles arranged in formal monitoring grids were checked for the presence of *D. impar*, if these had not already been checked.

At least two weeks were allowed to pass between surveys, in the period following this procedure.

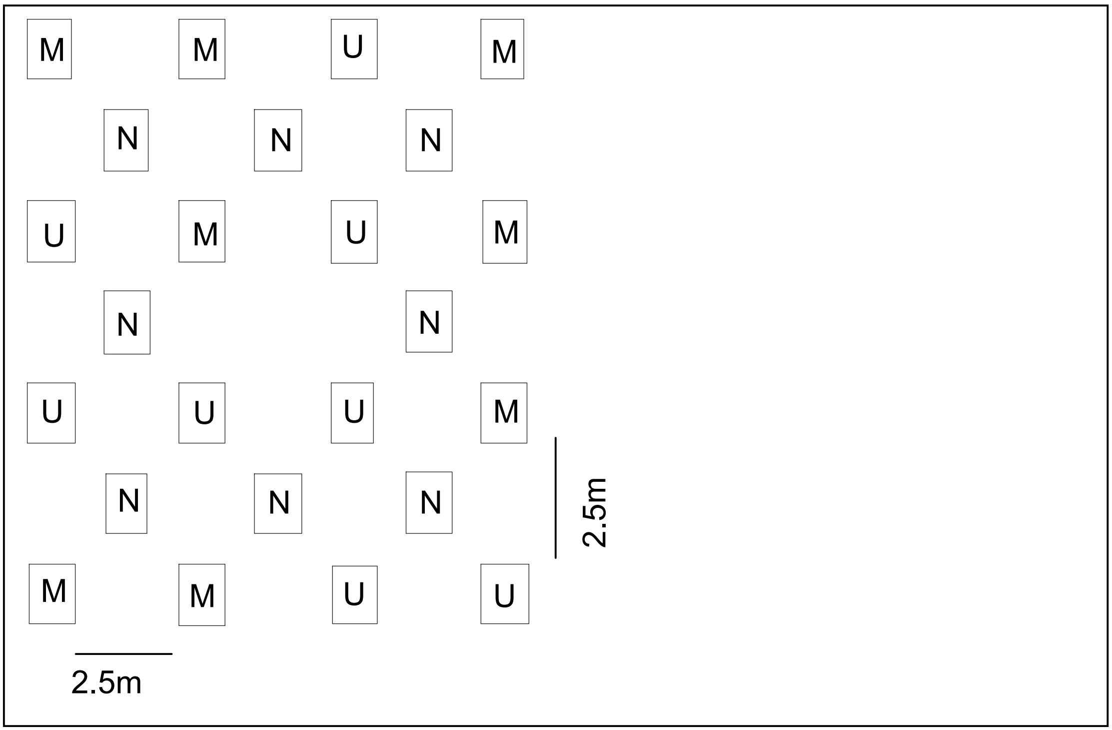


Figure 3.8: The placement of new tiles (N) amongst ‘unmown’ (U) and ‘mown’ (M) tiles. The position of unmown and mown tiles was randomised for each tile grid.

In order to minimise the number of *D. impar* escaping capture during roof tile surveys, a bottomless and lidless box was placed around each tile immediately prior to lifting. The box had a thick layer of sponge foam attached along the undersides, to ensure a closer fit with the undulations of the terrain, thus removing any small gaps through which *D. impar* could escape (figure 3.9). It was also found that a second pair of hands to catch the animal, whilst the first pair lifted the tile also helped to minimise escape events. Roof tiles were replaced to their exact location after lifting.

Figure 3.9: The roof tile monitoring box, with sample roof tile inside.

On each monitoring occasion, the following information was recorded:

- date;
- commencement and completion time;
- ambient air temperature at ground level at commencement and completion;
- temperature under ‘Standards’ and ‘Elabana’ tiles;
- wind direction;
- wind strength (still, breezy, windy, strong wind, gusty); and
- cloud cover (clear, partial cloud, greater than 50 percent cloud, full cloud).

Morphometric, identification and location data were recorded for each *D. impar* captured (as described in Chapter 2). The location of any escaped *D. impar* was also noted. Captured individuals were returned to under the roof tile where they were located.

3.2.3.1 Analyses

3.2.3.2 The influence of tile treatment

In order to assess whether *D. impar* sightability (and potential catchability) could be improved by any of the tile treatments, paired samples t-tests (Zar, 1996) were performed on the number of *D. impar* sighted under roof tiles, for the following sets of data (a sighting includes both animals which were captured and animals which escaped capture):

- 'Elabana' tiles versues 'Standards' tiles;
- 'unmown' tiles versus 'mown' tiles (ie. vegetation mown or not mown before tile placement); and
- old (unmown) tiles versus newly placed (unmown) tiles.

3.2.3.3 The influence of time and weather

In order to determine which conditions had the most influence on the number of sightings of *D. impar* under roof tiles, and therefore enabling an estimate of the best survey conditions to be selected for monitoring, the following data (collected at each survey period) were entered into a stepwise multiple regression (SPSS Inc., 2000):

- count (number of *D. impar* captures (including recaptures) and sightings);
- month;
- time of day that survey commenced;
- ambient air temperature;
- temperature under tiles;
- wind speed;
- wind direction; and
- amount of cloud cover.

3.2.3.4 Sex ratios

Capture data from roof tile grids was assessed for homogeneity using a heterogeneity chi-square analysis and analysed for a 1:1 sex ratio using chi-square analysis. The mean ratio was also calculated. Data (including recaptures) was divided into month of capture and further analysed for homogeneity across survey months.

3.2.3.5 Condition

An average condition factor for male *D. impar* captured under roof tiles was derived following the procedure outlined in Chapter 2.

3.2.3.6 Population size

An estimate of the overall Iramoo Wildlife Reserve population size based on roof tile monitoring was derived using the Jolly-Seber method and the Leslie, Chitty, and Chitty Test of Equal Catchability (Krebs, 1989). This method provided a single estimate for all tile grids on each monitoring occasion when *D. impar* were captured. Occasions when *D. impar* were only sighted were not included in the population assessment, as it was not possible to determine whether individuals were newly sighted or recaptures. Based on an estimate of the survey area covered by roof tile grids, populations estimates were extrapolated for the entire Iramoo Wildlife Reserve.

3.3 Results

3.3.1 The effect of leaving drift-fences in place or removing them between survey periods

No significant difference was evident between the number of *D. impar* captures recorded in survey plots with permanent drift-fences and the number of *D. impar* captures recorded in survey plots with drift-fences removed at the cessation of one survey period and re-installed immediately prior to commencement of the next survey period, for either the Spring or Summer survey periods (table 3.3). There was no consistent trend in recapture rates between treatments. Generally, recapture rates were low (average = 8.0 percent recaptures) and were not inconsistent with rates reported for standard trapping methods, as in Chapter 2.

Survey period	**Permanent drift-fences**		**Removed/replaced drift-fences**		**Statistical findings**
	Total captures	Recaptures	Total captures	Recaptures	
Autumn 1998	4	0%	5	20%	t=0.286; d.f.=10; p=0.780
Autumn 1999	19	15.8%	29	0%	t=0.776; d.f.=10; p=0.456
Autumn 2000	29	10.3%	31	6.5%	t=0.170; d.f.=14; p=0.867
Spring 2000	54	9.4%	32	9.4%	t=1.268; d.f.=14; p=0.225

Table 3.3: The influence of permanent and removed/replaced drift-fences on the efficiency of *D. impar* captures.

3.3.2 The effect of season

Overall, 151 individual *D. impar* were captured in survey plots in the survey periods Spring 1998, Summer 1999, Spring 1999 and Summer 2000 (table 3.4). The number of captures within a 25 day trapping period were found to be significantly higher in Spring than in Summer (t=2.713; df=26; p=0.012) (figure 3.10). Although a similar number of individuals can be successfully captured during Summer survey periods (for example, the total numbers captured in Spring 1998 (32) and Summer 1999 (33)), higher trap rates per unit of time in Spring survey periods suggest that Spring trapping will enable a greater number of *D. impar* to be captured over a shorter trapping duration, than trapping in the Summer of the same activity season (figures 3.11a & b).

Recapture rates were typically low for all survey periods and hatchlings were not recorded in either of the Spring survey periods. A clustered spatial aggregation of *D. impar* was displayed in both seasons, as indicated by the standardised Morisita index of dispersion.

	Spring 1998	Summer 1999	Spring 1999	Summer 2000
Total individuals captured (total days in brackets)	32 (27)	33 (50)	49 (26)	37 (50)
Trap rate (*D. impar*/100 survey plot days)	19.75	11	23.56	9.25
Captures over initial 25 days	32	19	49	21
Trap rate over initial 25 days (*D. impar*/100 survey plot days)	21.33	12.67	24.5	10.5
Recaptures (% total captures)	0 (0)	0 (0)	3 (6.1)	3 (8.1)
Hatchlings (% total captures)	0 (0)	4 (12.1)	0 (0)	6 (16.2)
Dispersion index	0.295	0.543	0.200	0.243

Table 3.4: Survey results for Spring and Summer pit-fall trapping 1998/1999 and 1999/2000.

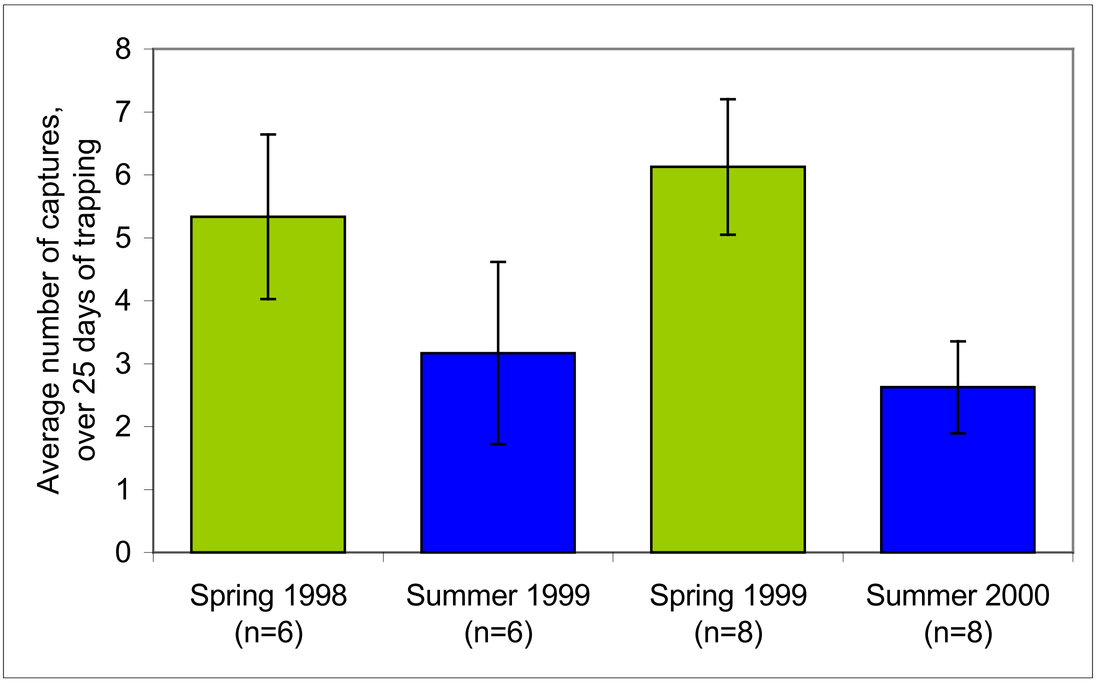


Figure 3.10: The number of captures of *D. impar* over a 25 day trapping period in Spring and Summer.

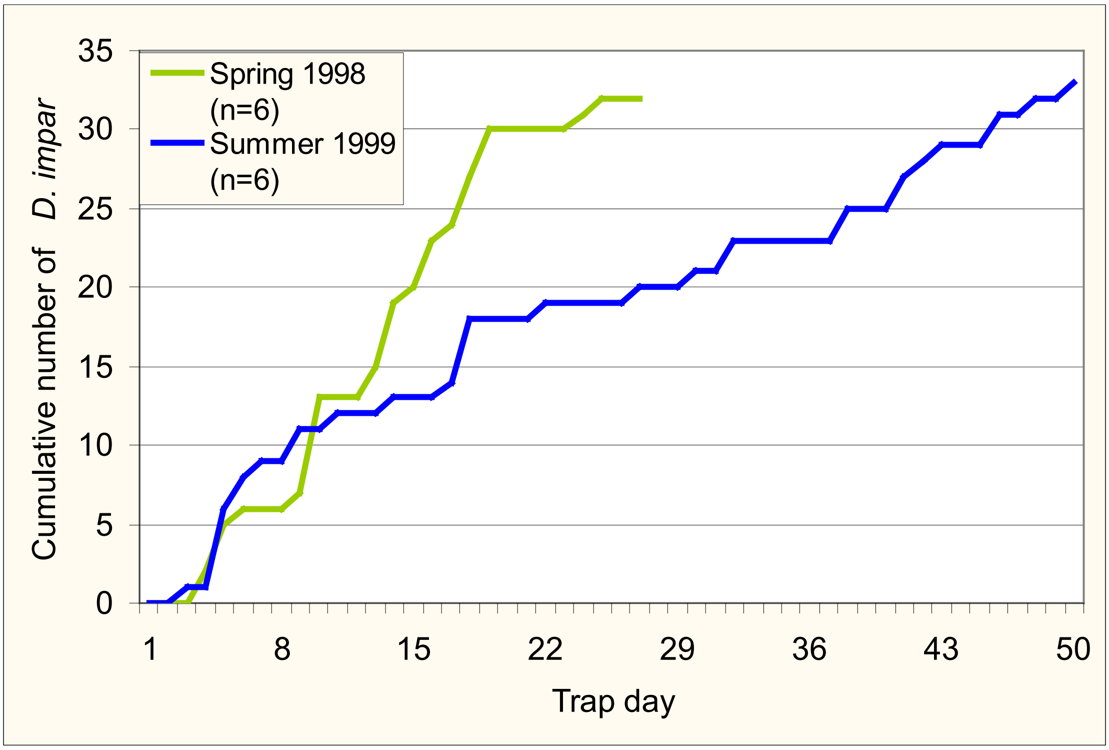


Figure 3.11a: Cumulative captures of *D. impar* over time for Spring 1998 and Summer 1999 surveys.

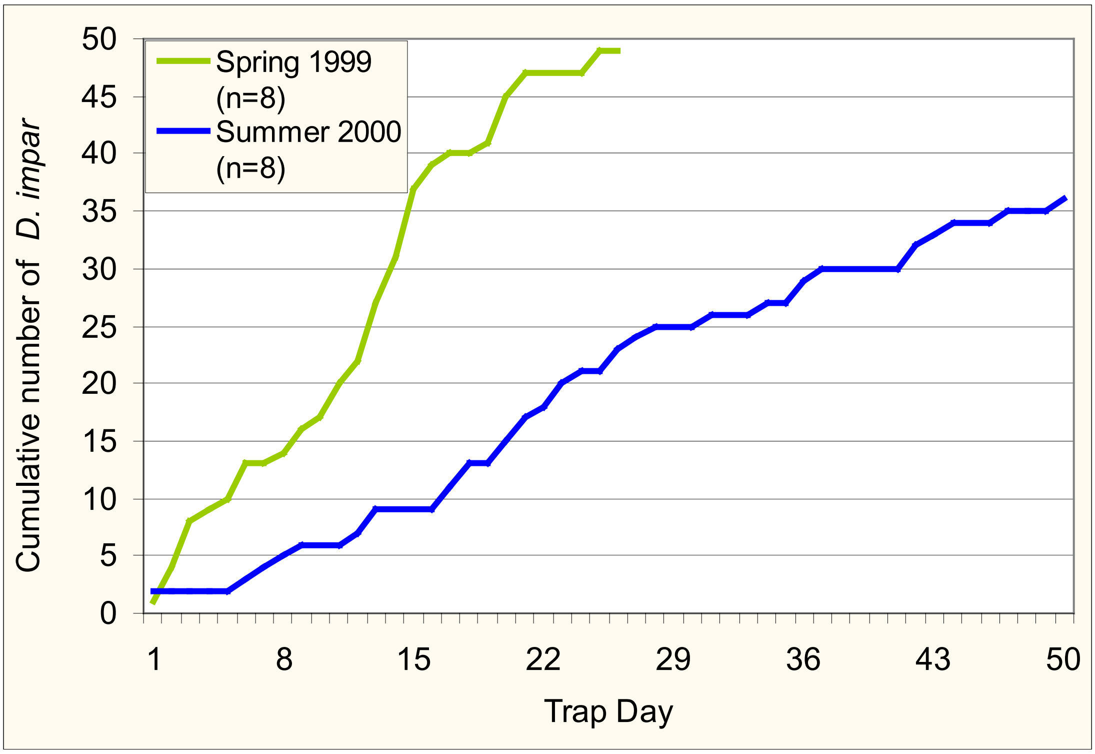


Figure 3.11b: Cumulative captures of *D. impar* over time for Spring 1999 and Summer 2000 surveys.

3.3.2.1 Population structure

The snout-vent lengths of *D. impar* captured in Spring survey periods was significantly different to the snout-vent lengths of *D. impar* captured in Summer survey periods (U=2276.5; n=83, 70; p=0.021). However, this difference was due to the presence of hatchlings in the data set for the Summer survey periods. When hatchlings were removed from the data set, there was no significant difference in *D. impar* snout-vent lengths for the two seasons (U=2276.5; n=83, 61; p=0.302). In keeping with the snout-vent length distribution of *D. impar* described in Chapter 2, the location of the first and third quartiles were within the mature age classes (ie. greater than 70 millimetres snout-vent length) and the median snout-vent was greater than 75 millimetres, for both Spring and Summer survey periods (figure 3.12).

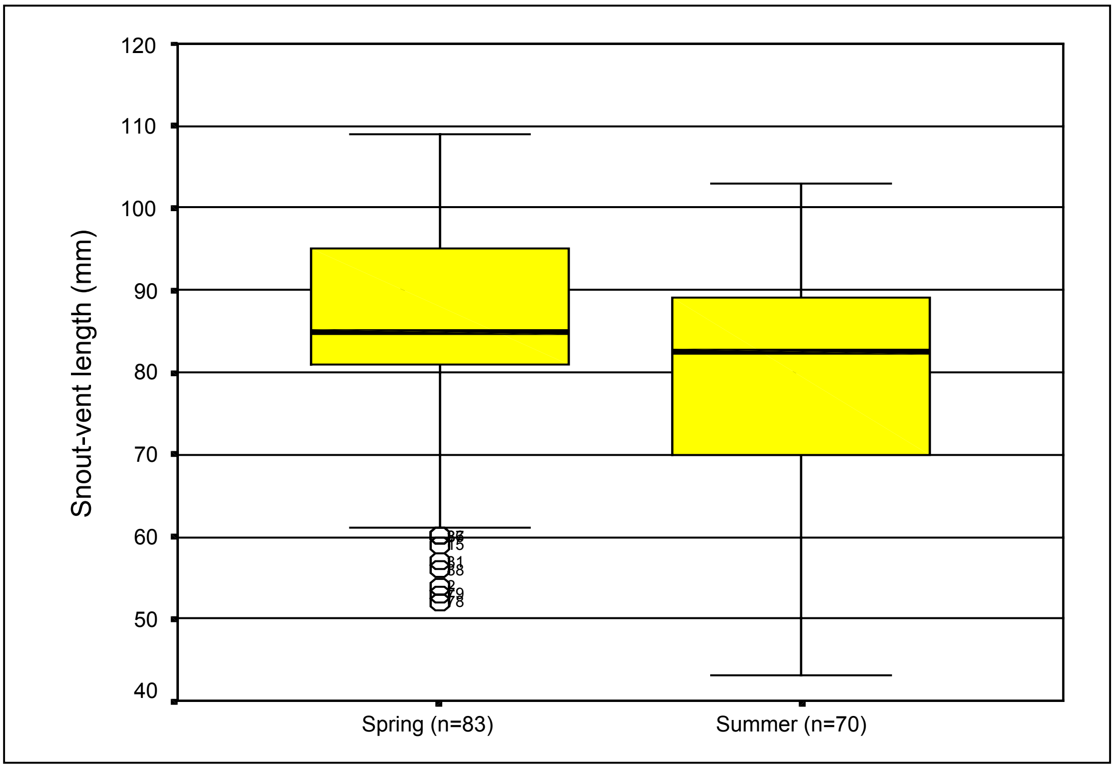


Figure 3.12: The distribution of snout-vent lengths for *D. impar* captured in Spring and Summer survey periods. Horizontal bar = median; box = first quartile (lower edge) to third quartile (upper edge); whiskers = range; circles = possible outliers.

3.3.2.2 Sex ratios

A total of 60 female and 60 male *D. impar* were captured over the four survey periods Spring 1998, Summer 1999, Spring 1999 and Summer 2000, suggesting an overall 1:1 sex ratio. Despite this, sex ratios across survey plots were not homogeneous (χ^2=39.673; df=25; p=0.032), due to extreme values for three of the total 28 survey plots. When these three extreme values were removed from the data set, homogeneity was achieved (χ^2=21.163; df=21; p=0.449; 97.7 percent of expected values were less than five) and sex ratios were found to be even across all survey periods (χ^2=0.041; df=1; p=0.840).

Excluding the three values from non-homogeneous survey plots, the overall mean ratio was 0.9 female/male *D. impar*, which is appreciably close to even. There was, however, some variation in the mean ratio for each of the survey periods, with a ratio as low as 0.73 females/males in Spring 1999 (table 3.5).

	No. captures		**n**	**Mean ratio**
	Female	**Male**		
Overall	46	53	25	0.9
Spring 1998	9	10	5	0.9
Summer 1999	13	14	6	0.93
Spring 1999	16	22	8	0.73
Summer 2000	8	7	6	0.8

Table 3.5: The ratio of females to males in Spring and Summer survey periods and overall, following the removal of non-homogeneous data points.

3.3.2.3 Condition

Following the removal of hatchlings, juveniles, females, males with autotomised tails and any individuals with an observed total length greater or less than one standard deviation of the predicted total length (refer to Chapter

2), a total of 32 individual *D. impar* males were available for a comparison of the condition factor between Spring and Summer. No significant difference was found in the condition factor of males captured in Spring 1998 and Spring 1999 (t=0.205; df=17; p=0.840) or in the condition factor of males captured in Summer 1999 and Summer 2000 (t=0.367; df=11; p=0.721). When Spring data was pooled and Summer data was pooled, male *D. impar* were found to have a significantly higher condition factor in the Summer survey periods (t=4.142; df=30; p<0.001) (figure 3.13). The average condition factor of male *D. impar* in Spring was 0.233, compared to the average condition factor of male *D. impar* in Summer which was 0.280.

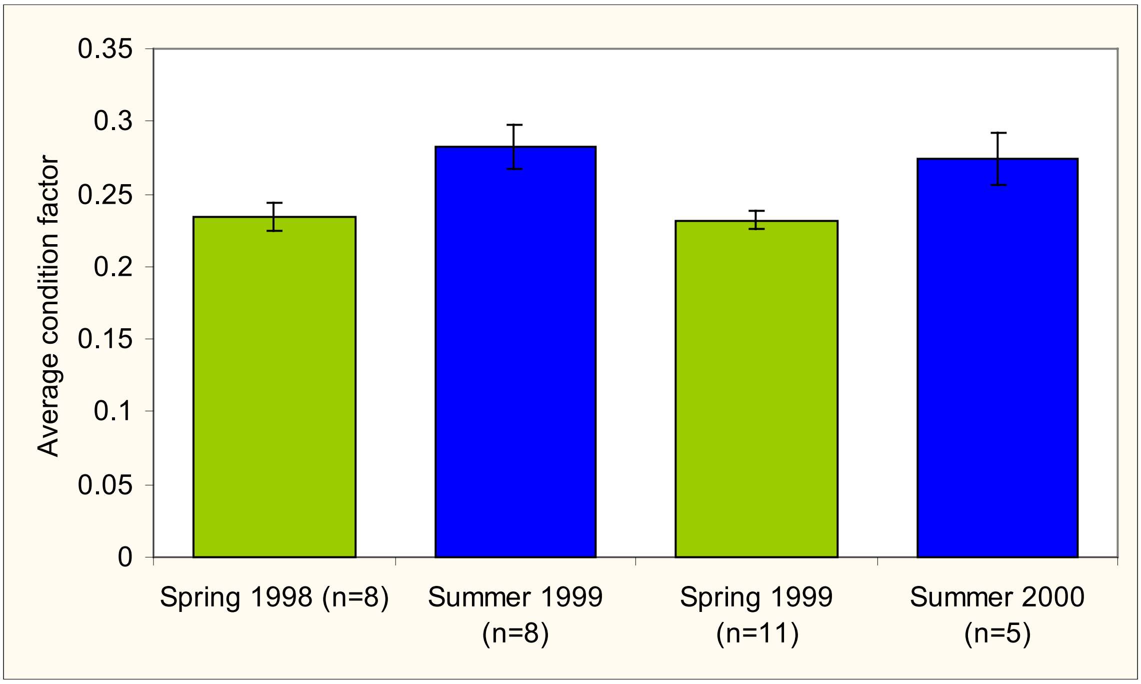


Figure 3.13: The condition factor of male *D. impar* in Spring and Summer, over four survey periods. Individuals with suspected autotomised tails were removed from the data set.

3.3.3 Roof tiles as artificial shelters and as an alternative to pit-fall trapping

Throughout this study roof tiles in monitoring grids were turned on 38 occasions giving a total of 5952 tile turns. In all 151 *D. impar* sightings were recorded, with an overall rate of 2.5 *D. impar* sightings per 100 tile turns and a minimum overall recapture rate of 25 percent. A sighting includes both animals which were captured and animals which escaped capture.

In total, 70 individual *D. impar* were captured, with 26 (37 percent) of these individuals recaptured at least once (18 individuals recaptured once, 4 individuals recaptured twice, 4 individuals recaptured 3 times; a total of 38 recapture events). The interval between which an individual was captured and subsequently recaptured ranged from less than one month up to 15 months. Movement distances were small, with the maximum overall straight-line distance that any individual moved being 14.30 metres and the average overall straight-line distance moved by individuals being 4.05 metres (figure 3.14). All recaptures were recorded within the same tile grid as previously captured, that is no animal was found to move between tile grids. There is evidence for some level of tile fidelity in that 11 (29 percent) of the 38 recapture events were recorded under one of the tiles where the individual had previously been found.

In addition to captures and sightings, a total of 56 *D. impar* sloughs were found under the tiles grids during the survey period. Sloughs were removed to prevent any double counting.

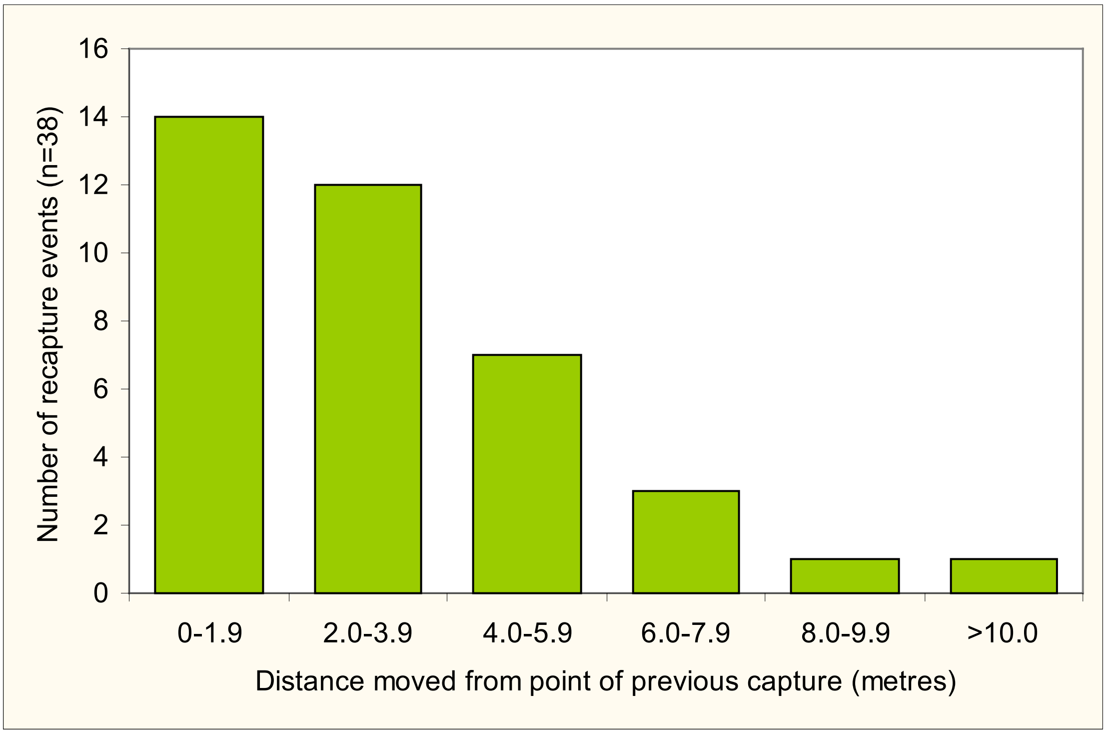


Figure 3.14: A summary of the distances moved by individual *D. impar* between capture and recapture events under roof tiles.

3.3.3.1 The influence of tile treatment

The number of *D. impar* sighted under roof tiles was not significantly different for the treatment comparisons between 'Elabana' and 'Standards' tiles, or between tiles which had the grass underneath clipped and tiles under which the grass was unmown, or between old and newly placed tiles (table 3.6).

Comparison	**Result**
'Elabana' vs 'Standards'	t=0.435; df=20; p=0.668
Unmown vs Mown	t=0.114; df=20; p=0.910
Old vs New	t=1.789; df=7; p=0.117

Table 3.6: Results of paired sample t-tests for comparisons of roof tile treatments.

The following analyses using data derived from roof tile monitoring were performed by pooling *D. impar* sighting and/or capture data, irrespective of the style of tile, whether or not the grass underneath tiles had been mown and whether or not the tiles were old or newly placed.

3.3.3.2 The Influence of time and weather

Using stepwise multiple regression, the variables 'temperature under tile' (R^2=0.217) and 'month' (R^2=0.226) were found to be the best predictors for the number of *D. impar* sightings under roof tiles, accounting for 44 percent (R^2=0.443) of variation explained by the model (F=9.555; df=2,24; p=0.001). The tile temperatures under which *D. impar* were most frequently sighted were from the mid-20°C to the high-30°C range (figure 3.15). *D. impar* were also most frequently sighted under roof tiles in the months of September and October (figure 3.16).

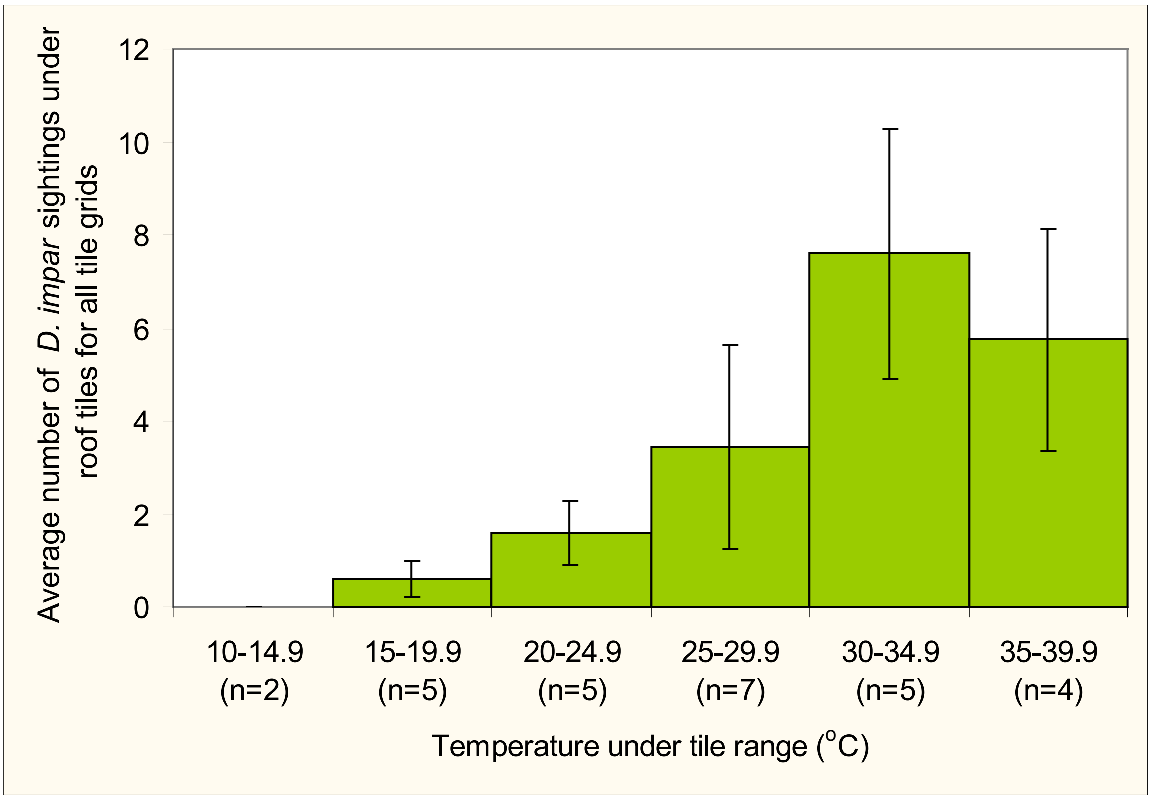


Figure 3.15: The number of *D. impar* sighted under roof tiles over a range of temperature conditions.

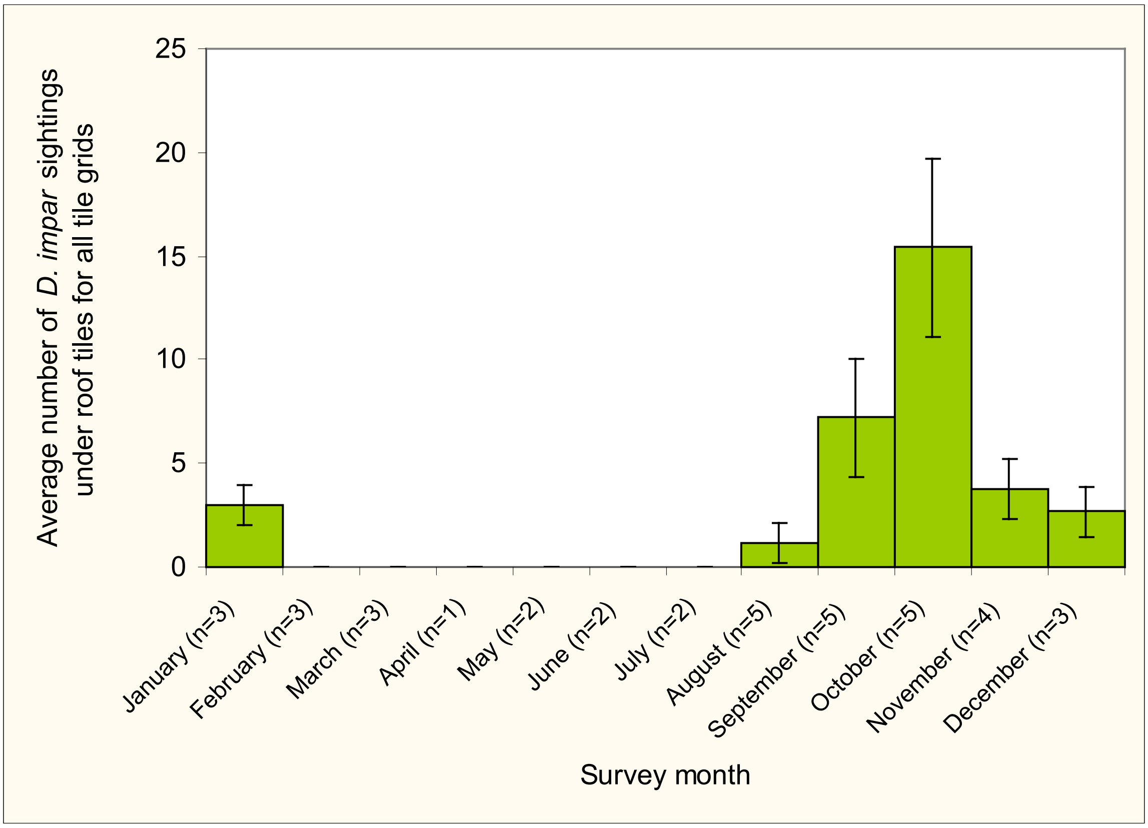


Figure 3.16: The number of *D. impar* sighted under roof tiles over the months of the year.

Given that *D. impar* were only sighted under roof tiles for five months of the year, if monitoring occasions outside this time frame are removed from the data set, the number of monitoring occasions is reduced to 22 with a total of 3552 tile turns. On this basis the rate of sightability for *D. impar* under roof tiles is increased to 4.3 per 100 tile turns.

3.3.3.3 Population structure

Almost all *D. impar* captured under roof tiles were mature animals. The presence of immature *D. impar*, particularly hatchlings and first year juveniles was noticeably lacking (figure 3.17).

Figure 3.17: The distribution of size classes of *D. impar* captured under roof tiles.

3.3.3.4 Sex ratios

Sex ratios were homogeneous across roof tile survey grids (χ^2=4.001; df=11;p=0.272) (91.7 percent of cells had expected values less than five) and the overall sex ratio was not significantly different from 1:1 (χ^2=1.209; df=1; p=0.272). The mean ratio was 0.76 female/male and thus, in conjunction with the large proportion of cells with an expected frequency of less than five, the validity of the results of the chi-square analyses are again questionable.

The sex ratio of animals over the months of the roof tile surveys was not homogeneous (χ^2=11.944; df=5; p=0.036) (58.3 percent of cells had expected values less than five). Figure 3.18 indicates a swap over between the proportion of the sexes using roof tiles over time, with males being more commonly observed in early Spring (August – October) following emergence from brumation and females being more commonly observed in late Spring and early Summer (November – December) during the egg development and deposition periods.

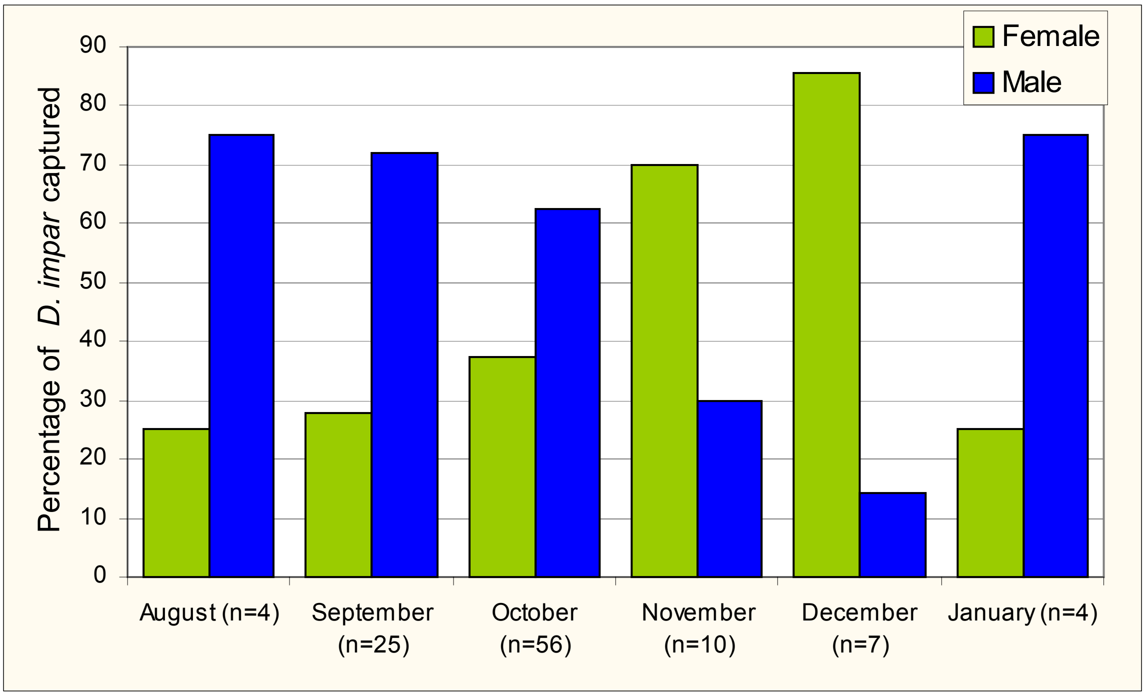


Figure 3.18: The percentage of female and male *D. impar* captured under roof tiles for the months August – January, over all survey years.

3.3.3.5 Condition

The condition factor was estimated for a total of 22 male *D. impar*, following the removal of hatchlings, juveniles, females, males with autotomised tails and individuals with an observed total length greater or less than one standard deviation of the predicted total length. Condition factors of male *D. impar* captured under roof tiles ranged from 0.199 to 0.319, with an average condition factor of 0.259.

3.3.3.6 Population size

Estimates for the size of the *D. impar* population under all roof tile grids for monitoring occasions when individuals were captured is provided in table 3.7

and ranged from five *D. impar* under all tile grids to 63 *D. impar* under all tile grids, with an average of 22.8 *D. impar* under all tile grids. To calculate the overall area surveyed by tile grids, twice the average overall straight-line distance movement by *D. impar* (ie. 4.05 metres) was added to both the length and width of the space occupied by roof tile survey grids. Thus, it was estimated that overall roof tile grids surveyed a total of 2,476.80 square metres or roughly 0.25 hectares.

On this basis, an average density of 91.2 *D. impar* per hectare were estimated to be present in roof tile survey areas on occasions when animals were captured. This extrapolates to a total of 3,420 individual *D. impar* at Iramoo Wildlife Reserve, assuming that animals are evenly distributed across the site. Given that the success of monitoring improved with time, due to a decline in the number of escaped animals (improved capture success) and probably also due to a heightened understanding of the weather conditions under which to successfully conduct monitoring, population estimates derived towards the end of the study are probably closer to the true population size. This view is supported by the results of the Leslie, Chitty and Chitty test of equal catchability which predicted that a total of 121.7 new animals would enter the study population during the survey period. However, only 52 new animals were observed to enter the study population during this time.

Date	Total marked	Total caught	Population estimate	LCI	UCI	Number escaped
21-11-1998	0	2	9 (1,350)	2.1	129.8	1
14-12-1998	0	2	9 (1,350)	2.1	129.8	0
13-1-1999	1	1	3 (450)	1.2	10.5	0
21-9-1999	2	3	5.3 (795)	3.1	14.1	6
5-10-1999	2	7	5.3 (795)	7.1	30.9	4
1-11-1999	1	1	5 (750)	1.6	18.3	1
1-9-2000	0	2	6 (900)	2.3	27.1	0
18-9-2000	2	6	11.2 (1,680)	6.5	25.3	3
28-9-2000	4	14	21.3 (3,195)	14.4	41.5	2
11-10-2000	4	11	44.1 (6,615)	18.3	104.7	5
22-11-2000	2	7	48 (7,200)	15.9	140.4	1
16-1-2000	2	4	28.3 (4,245)	10.2	69.6	0
23-8-2001	0	1	44 (6,600)	6.6	349.5	0
1-10-2001	7	18	39.2 (5,880)	22.0	73.3	0
25-10-2001	9	20	63 (9,450)	22.9	285.5	9

Table 3.7: Estimates of the size of the *D. impar* population under all roof tile monitoring grids (and for the entire Iramoo Wildlife Reserve in brackets), on survey occasions when individuals were captured.
LCI = Lower Confidence Interval.
UCI = Upper Confidence Interval.

3.4 Discussion

3.4.1 The effect of leaving drift-fences in place or removing them between survey periods

There is no evidence that the removal and replacement of drift-fences between survey periods serves to increase the number of *D. impar* captures or recaptures. Likewise, there is no evidence that permanently installed drift-fences have a negative effect on the number of *D. impar* captures and recaptures. It is probable that factors most influencing the number of *D. impar* captures are population density, seasonal fluctuations and levels of spatial aggregation in relation to the placement of survey plots, as discussed in Chapter 2.

The cause of the low recapture rates, recorded when using pit-fall trapping with drift-fences remains unknown. This may be related to previously captured individuals avoiding the trap arrangement, regardless of whether drift-fences are permanent or temporary, or may be related to some other aspect of the behaviour of this species, such as extensive movements away from trapping areas. It seems that trap avoidance is most likely, given that there is a relatively comprehensive coverage of pit-fall traps across Iramoo Wildlife Reserve and that most recaptured individuals are recorded within or near the survey plot of initial capture (average movement = 10.68 metres based on data from Chapter 2). If animals were making extensive movements away from areas of initial capture, it would be expected that a higher percentage of recaptures would be in survey grids away from the initial capture survey grid and that average movement distances would be much greater.

If *D. impar* learn to avoid or overcome the obstacle created by pit-fall traps with drift-fences, it may be possible to modify the trap design and compensate for the behaviour of the species. As *D. impar* have been observed springing over drift-fences (Coulson, 1990) an obvious method for preventing this would be to raise the height of drift-fences. Noting the species ability to spring over drift-fences Kutt (1992) installed drift-fences which were 10 centimetres higher than those previously used by Coulson (1990) (total = 25 centimetres), however the trap rates recorded for *D. impar* were characteristically low and no individuals were recaptured. O'Shea & Hocking (1997b) also experimented with increased drift-fence height (total = 60 centimetres) but no conclusive results were formed due to a paucity of captures in both the unmodified traps and traps with increased drift-fence height. Further investigation of such modifications to the drift-fence height may be warranted, although Mengak and Guynn (1987) found that drift-fence height had no effect on the trapping efficiency for herpetofauna communities inhabiting pine stands in the United States of America.

Although the recapture rates for *D. impar* have been low for most surveys, primary capture rates for surveys conducted in the Australian Capital Territory

(ACT) appear to be generally higher than surveys conducted in other states. This may be due to some characteristic inherent to the ACT populations. Alternatively, higher primary trap rates in the ACT may be attributed to the survey methodologies employed. Pit-fall trap and drift fence configurations have varied considerably between studies on *D. impar* and, as noted for pit-fall trapping surveys for herpetofauna in general (Braithwaite, 1983; Friend, 1984), appear to be the result of institutional preference and survey histories at particular locations, rather than a quantitative or qualitative assessment of trap design.

In the ACT, trapping configurations have consistently been made up of a cross array with arms extending 25 metres in each direction and five pits spaced at five metre intervals along each arm (Williams & Kukolic, 1991). Elsewhere trapping configurations have included 50 metre drift lines with 10 pits spaced at five metre intervals (Coulson, 1990; Kutt, 1992), complex grid designs (Kutt, 1992; Whitby, 1995) and mini-grid formations (Larwill *et al.*, 1993; O’Shea, 1996; O’Shea & Hocking, 1997a). Studies on the efficiency of various pit-fall and drift-fence configurations for sampling herpetofaunal assemblages have produced conflicting results with some works suggesting that array designs, such as those used in the ACT, are more productive in that they are able to intercept animals from any direction (Bury & Corn, 1987; Corn, 1994). Others have found that these more complicated designs do not significantly increase trap success (Hobbs *et al.*, 1994). Additionally, although it is agreed that drift-fences act to increase capture success, an optimum drift-fence length has not been discerned for sampling herpetofaunal assemblages and it is thought that this may vary by habitat type, species and faunal groups and their characteristic behaviours and levels of activity (Braithwaite, 1983; Bury & Corn, 1987; Morton *et al.*, 1988; Friend *et al.*, 1989; Corn, 1994). It is suggested that longer traplines generally improve capture success (Hobbs *et al.*, 1994) and although Friend *et al.* (1989) found that the number of captures increased in direct proportion to drift-fence length, they also found that traps with independent drift-fences (i.e. one fence per pit) captured more animals.

Bury and Corn (1987) suggested that the total amount of fence is more important than individual lengths.

These factors need to be quantified to determine the best pit-fall trap and drift-fence configuration for sampling primary captures of *D. impar* and developing some level of standardisation, at least between populations forming individual management clusters and potentially, across all *D. impar* survey sites.

3.4.2 The effect of season

Delma impar can be successfully captured at Iramoo Wildlife Reserve using pit-fall traps and drift-fences in either Spring or Summer. However, both the number of individuals captured and the trap rate over a defined trapping duration were higher in Spring survey periods. This is consistent with previous observations (Kukolic *et al*., 1994; Osmond, 1994), although the reasons for such differences can only be presumed. It has been speculated that increased capture success in Spring is due to an increased level of activity, as this period coincides with the mating season. However, it is also plausible that animals are more surface active during this period due to thermoregulation requirements and that in Summer periods sub-surface temperatures are high enough to enable this cryptic species to adequately thermoregulate whilst maintaining a semi-fossorial habit. Such theories could be tested through tracking studies and also via studies of captive populations.

Although Spring pit-fall and drift-fence trapping yielded a higher *D. impar* capture and trap rate success compared with Summer trapping, there was no apparent difference in the level of recaptures. This lends further support to the idea that *D. impar* actively avoids recapture, as it would be expected that greater levels of surface activity leading to higher capture rates would also lead to a corresponding increase in recapture levels.

Generally, the *D. impar* population at Iramoo Wildlife Reserve appears to yield similar characteristics between Spring and Summer survey periods, in that the spatial distribution, sex ratio and population structure (excluding hatchlings) do not appear to differ. A lack of hatchlings in the Spring survey period is not surprising given the timing of hatching for this species. Yet, this also suggests that the rate of growth between hatching and the onset of brumation and/or between the cessation of brumation and commencement of Spring trapping in November is rapid, enabling them to outgrow the hatchling age class in a single year. This is consistent with growth rates observed for individuals hatched in captivity (Banks *et al.*, 1999).

The chi-square analyses for Spring and Summer survey periods did not suggest any significant difference from an even sex ratio. However, it is important to note that as with similar analyses in Chapter 2, the chi-square test has been violated due to low sample sizes, increasing the likelihood of a Type II error. Again a small proportion of the total plots sampled did not display ratios which were homogeneous with the remainder of the data set. For the three heterogeneous values removed from the data set in this study, there was no consistent pattern as to whether female or male individuals were more prevalent in those survey plots and therefore it is difficult to suggest any possible cause of this phenomenon. Presumably, these values are simply a result of random variation, with no underlying biological cause.

The mean ratio offers a quantified description of the sex ratio and even when data from survey plots displaying heterogeneity were removed, some mean ratio values were much less than 1.0 female/male. It is interesting to note that for all four survey periods over Spring and Summer, the mean ratio indicated that slightly fewer females than males were captured.

A lower condition factor for male *D. impar* in the Spring survey periods than in Summer survey periods suggests that the condition factor is a sensitive mechanism by which to assess the health of populations. A lower condition factor for males in Spring survey periods was expected, given that animals lose condition over the brumation period and possibly also through increased activity and stress in the Spring mating period. Future surveys and studies using this condition factor to assess population health should take the timing of the survey period into account.

3.4.3 Roof tiles as artificial shelters and as an alternative to pit-fall trapping

Roof tiles provide a viable alternative or addition to the use of pit-fall traps and drift-fences for monitoring *D. impar* populations. Although direct comparisons cannot be made, sighting rates of *D. impar* under roof tiles (2.5 *D. impar* per 100 tile turns) were comparable with Spring capture rates in pit-fall traps with drift-fences (2.8 *D. impar* per 100 pit-fall trap days) and this rate was even higher (4.3 *D. impar* per 100 tile turns) when survey data was limited to the months September to January. Recapture success was considerably higher when using roof tiles than when using pit-fall traps and drift-fences. The recapture success from roof tiles recorded in this study is a minimum and potentially would have been higher if large numbers of sighted individuals had not escaped capture.

Movement data gleaned from recaptured individuals under roof tiles concur with the limited movement observations of individuals recaptured in pit-fall traps. That is, both methods suggest that *D. impar* is a relatively sedentary species, generally only moving short distances within a small home range. This confirms previous findings and speculation for this species.

Roof tiles are useful for monitoring *D. impar* populations particularly in the months of September and October. This is most likely associated with the thermoregulation opportunities that the tiles offer during the Spring to early Summer period, in that tiles would be too cold in the Winter months and become too hot during the high Summer period. It is interesting to note that *D. impar* have been recorded in rock rolling surveys in the Winter months

(Whitby, 1995), suggesting that artificial shelters with greater insulation properties may provide a wider time frame over which such a monitoring method could be useful. The absence of *D. impar* under roof tiles during the late Summer to Autumn period is inexplicable, especially as animals were detected during these time frames at Iramoo Wildlife Reserve, using pit-fall traps and drift-fences. However, the absence of *D. impar* under roof tiles outside the Spring to early Summer period provides an explanation for the absence of animals under the 'Elabana' tiles in the O'Shea and Hocking (1997b) study, where monitoring was conducted over the months of February to July.

The use of roof tiles for monitoring *D. impar* populations in the Spring period creates some limitations to the type of data that can be collected, in that sex ratios appear to be skewed with a greater number of males observed under tiles until the swap-over period in November when females are more frequently observed. Furthermore, roof tiles do not sample for hatchlings due to the fact that hatching has not occurred at the time when tile monitoring is most successful, and juveniles are also poorly represented irrespective of the seasonal timing of surveys. Further work is required to validate the apparent alternate skew in sex ratios between August and December, as this study was only able to illustrate trends but was not able to statistically test them, given the small sample sizes.

Lowson and Avery (2000) suggested that one of the reasons that Slow-worms *Anguis fragilis* use artificial refuges is to increase their body temperatures by conduction, as a result of solar infra-red radiation heating the upper surface of the refuge. This is probably also the case for *D. impar* using roof tiles and suggests that individuals may be actively attracted to roof tile survey grids, an idea which is further supported by individuals displaying tile fidelity. As a result, it is possible that the density of *D. impar* within tile grids might be an overestimate of the density elsewhere within Iramoo Wildlife Reserve.

It is apparent from the broad confidence limits of the population size for roof tile grids on each monitoring occasion where individuals were captured that in this study at least, this method of estimating the overall population size at Iramoo Wildlife Reserve is not very accurate, although it suggests an order of magnitude. Improvements in this accuracy are likely to result from improved capture success, when close to 100 percent of individuals sighted are captured and marked. Monitoring which was conducted by two persons and with the use of the roof tile box, assisted with improving capture success, however 100 percent capture success was not achieved on all occasions.

Despite the suspected attractiveness of roof tiles to *D. impar*, the estimated population size at Iramoo Wildlife Reserve arising from roof tile monitoring is considered to be an underestimate in that hatchlings and juveniles are not well accounted for, males and females are not equally catchable all of the time and a large number of sighted individuals escaped capture and were not included in the calculations of the estimate. Furthermore, the Leslie, Chitty and Chitty test of equal catchability predicted a much larger number of new individuals entering the population throughout the monitoring periods than was actually observed. Future studies using the Jolly-Seber method (Krebs, 1989) might provide a more accurate estimate of the population size if all sighted animals were successfully captured and marked.

Although there are limitations in terms of the type of data that can be collected when using roof tiles to monitor *D. impar* populations, roof tiles are a viable alternative or addition to the more conventional pit-fall trap and drift-fence monitoring methods. The benefits of using the roof tile monitoring method include:

- cheap cost of materials and high availability. Second-hand roof tiles can be purchased for around $1 per tile, making them substantially cheaper than the cost of drift-fence netting and buckets;

- monitoring sites are quick and easy to establish, in comparison to the labour intensive and time consuming process of digging holes for pit-falls and burying drift-fences;
- the impact of installation on the habitat is substantially reduced, as no soil disturbance is required and vegetation disturbance is limited to the area covered by each tile;
- the impact of monitoring on the habitat is potentially reduced, as tiles require monitoring at a frequency no greater than once per fortnight, whereas pit-fall traps require daily monitoring;
- the reduced frequency of monitoring may potentially result in lower labour requirements. However, a less frequent monitoring regime may become more burdensome in terms of travel for remote sites. Furthermore, it is recommended that two persons participate in monitoring in an effort to reduce the number of escaped animals, and also for safety reasons;
- the presence of *D. impar* at a site may be confirmed by the detection of sloughs under roof tiles;
- roof tiles are considered a more humane method of monitoring, in that the animals are not constrained. This benefit was also noted by Schulz *et al.* (1995) who used sheets of galvanised iron to monitor the Alpine She-oak Skink *Cyclodomorphus praealtus* and Sutton *et al.* (1999) who used plywood coverboards to monitor the Sand Skink *Neoseps reynoldsi*. Interestingly, both these species, like *D. impar*, display limb reduction and cryptic semi-fossorial behaviours.

3.5 Conclusion

There is no evidence that permanent or temporary drift-fences influence the primary capture or recapture success for *D. impar*. Although it would therefore seem appropriate to leave drift-fences in place between survey periods, thus reducing labour requirements for trap installation and the amount of disturbance to the grassland habitat, several observations, during this study, of foxes foraging along permanently installed drift-fences between survey periods suggest that rates of predation on grassland fauna may be increased as a result of this. This should be taken into consideration when designing and implementing future pit-fall trap and drift-fence surveys.

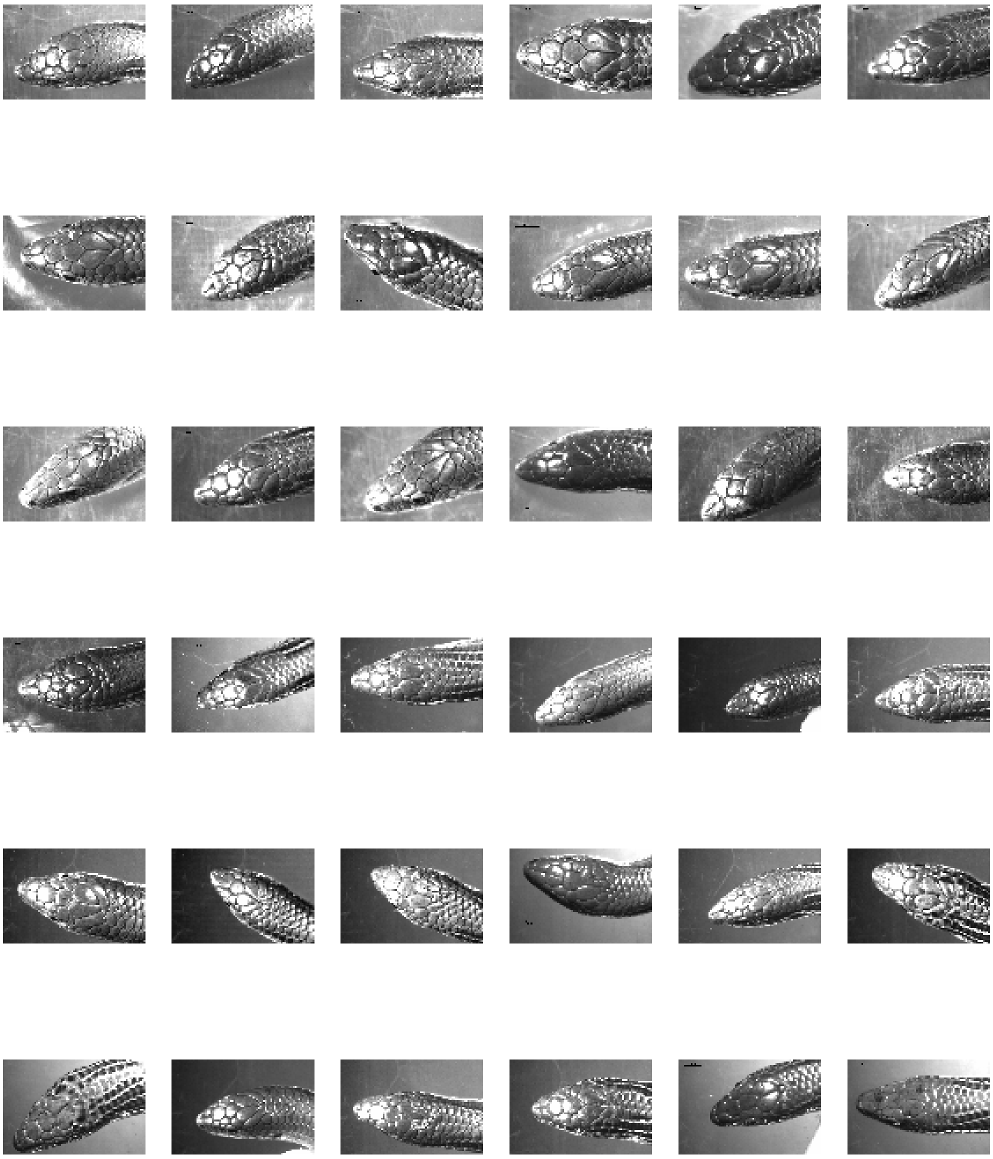
Pit-fall trapping with drift-fences in Spring (November to December) yields both a higher catch success and higher trapping rate than similar surveys conducted in Summer (January to March). Generally, there was very little difference in *D. impar* populations in terms of the spatial distribution of animals, age class distribution and sex ratio of individuals captured. However, it seems that no information on hatchlings can be obtained from Spring surveys using these methods, as would be expected given that eggs are known to hatch in the late Summer and early Autumn periods of the year. The condition factor of male *D. impar* captured in Spring is lower than the condition factor of males captured in Summer and for healthy *D. impar* populations within a similar geographic range to the Iramoo Wildlife Reserve population, it would be expected that the condition factor in Spring would be within the range 0.233 (±0.024) and in Summer this range would be 0.280 (±0.040).

Roof tile monitoring provides a viable alternative or addition to pit-fall trap and drift-fence monitoring for assessing *D. impar* populations. In this study both 'Elabana' and 'Standard' tiles were successfully used to monitor *D. impar*. Mowing under tiles did not influence the species' utilisation of tiles and recently laid out tiles were as successful as tiles which had been in place for longer periods of time. Tiles were most frequently utilised in September and October, however a bias in the sex ratio was observed with more males than females being recorded in these months. Roof tiles are not a useful tool for monitoring hatchlings and juveniles, with Summer pit-fall and drift-fence trapping being the better method for collecting information on these aspects of the population. The rate of sighting *D. impar* under roof tiles provides a reasonable estimate of basic population characteristics of *D. impar* when compared with the amount of effort required for pit-fall and drift-fence trapping.

The rate of recapture is notably higher with roof tiles, providing opportunities to easily and closely monitor the long-term survival and growth characteristics of individuals.

Further studies need to be conducted to determine the usefulness of the Jolly-Seber method for estimating population size when using roof tiles. In particular, efforts should be made to minimise the number of *D. impar* sighted under roof tiles from escaping capture and to maximise individual identification, thus potentially reducing such high variabilities in the Jolly-Seber population estimates . Further work should also investigate the most effective pattern of layout of roof tiles at field study sites. The low cost, ease of installation and monitoring of roof tiles provides opportunities for large areas of potential *D. impar* habitat to be monitored and provides opportunities for more accurate population estimates.

Chapter 4 Recognition of Individual Striped Legless Lizards



Chapter 4

Recognition of individual Striped Legless Lizards

4.1 Introduction

An increase in the use of animal marking for individual recognition has developed as descriptive natural history has evolved into the more analytical sciences of ecology and ethology, which require a more rigorous approach to field studies (Delaney, 1978). For reptiles and amphibians herpetologists documented the need for a simple system of recognising individuals at least as early as the 1930s (Blanchard & Finster, 1933). Since this time, the use of individual recognition methods has become popular in herpetological studies (Woodbury, 1956) and the range of techniques has broadened. The ability to identify individuals within a population can provide demographic information about a species, such as:

- movements and home range;
- behaviour;
- growth rates;
- sex ratios;
- population size;
- rates of fecundity and immigration;
- rates of mortality and emigration; and
- longevity.

For examples see: Blanchard & Finster, 1933; Carlström & Edelstam, 1946; Conant, 1948; Woodbury, 1956; Caughley, 1977; Fitch, 1987; Southwood & Henderson, 2000; Penney *et al.*, 2001.

When conducting studies of reptiles and amphibians that require the identification of individual animals, the method chosen should be humane, simple, accurate, durable and inexpensive. Lewke and Stroud (1974), Swingland (1978), Ferner (1979) and Southwood and Henderson (2000) provide lists of criteria for consideration when selecting a method for individual recognition. These are summarised as follows. The method:

- should not affect survivorship, growth or behaviour of the organism;
- should allow the animal to be as free as possible from stress and pain;
- should accurately identifiy the animal as a particular individual;
- should last indefinitely or as long as data collection is intended;
- should be easily read or observable;
- should be suitable for all sizes of the study species;
- can be used easily in both the laboratory and field;
- should use equipment that is easily obtainable at minimal cost; and
- should remain useful for identification after death.

Since Blanchard and Finster (1933) first suggested scale clipping as a method for individually marking snakes, this method has been adapted and refined (eg. Woodbury, 1956; Brown & Parker, 1976; Spellerberg, 1977) and alternative identification methods have been developed for a range of herpetofauna. Methods developed for identifying individual lizards include toe clipping (Medica *et al.*, 1971; Swingland, 1978; Ferner, 1979; Langston, 1996), pyrobranding (Clark, 1971), freeze-branding (Lewke & Stroud, 1974), external tagging (Swingland, 1978), passive implantable transponder (PIT) tagging (Camper & Dixon, 1988), painting (Jones & Ferguson, 1980; Boone & Larue, 1999; Quinn *et al.*, 2001), visible implant elastomer tagging (Penney *et al.*, 2001) and integument pattern mapping (Carlström & Edelstam, 1946; Stamps, 1973; Nelson *et al.*, 2000).

As Ferner (1979) pointed out, there are no techniques which satisfy all of the above criteria. Often the body size and anatomical features of the target species limit the choice of individual identification techniques (Penney *et al.*, 2001), as is the case for the Striped Legless Lizard *Delma impar*. The species has a small, slim body structure (average snout-vent length = 79 millimetres), and a tail approximately twice the snout-vent length which can be autotomised (Coulson, 1990). External evidence of the forelimbs are missing and the rear limbs have been reduced to small flap-like appendages (Greer, 1989). These morphological characteristics limit the range of individual identification techniques which can be applied to this species. To date, all work involving the individual identification of *D. impar* has involved the use of freeze-branding (Dorrough, 1995) or pyro-branding (Kutt, 1992; Nunan,1995; O'Shea, 1996; Hadden, 1998) techniques.

The practice of pyrobranding individual *D. impar* was continued throughout this study, using the methodology described by O'Shea (1996). This involved the use of a numbering system similar to that described by Brown and Parker (1976), whereby counting towards the anterior from the vent and excluding the first pair of pre-anal scales, the right-hand row of enlarged paired ventral scales was assigned as units and the left-hand row of enlarged paired ventral scales was assigned as tens. Values in the hundreds and thousands were counted from the tenth scale in the right-hand and left-hand rows of paired ventral scales, respectively (figure 4.1). This numbering system enabled the use of a microbranding technique similar to that described by Ehmann (2000), whereby ventral scales were individually marked using a hot, finely filed soldering iron, ensuring that the size of the brand was better suited to the small size of adult *D. impar*. However, this method was not considered suitable for animals smaller than 70 millimetres snout-vent length, which remained unmarked.

It became apparent throughout the study that the brands were becoming less visible over time (figures 4.2a & b) and in some cases this was noted as occurring in as little a period as eight months. As an example, one animal was re-branded with a different code after twelve months because there was no visible indication that it had previously been branded (this error was noted at a later stage through the use of the integument pattern mapping identification technique described later in this chapter).

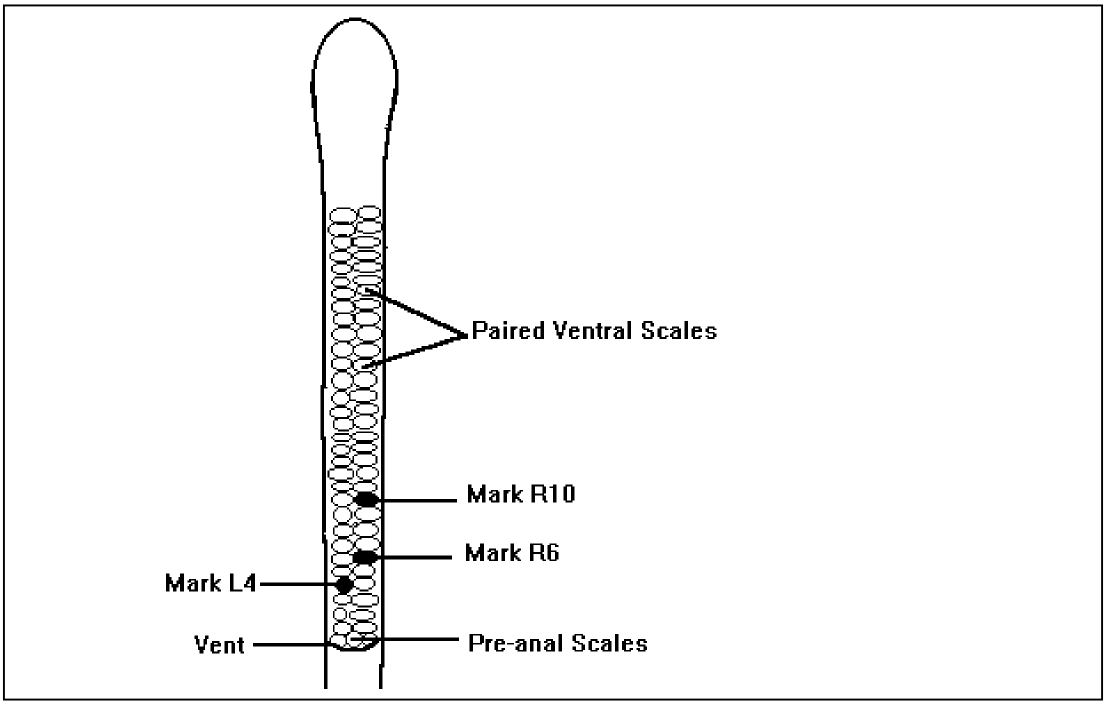


Figure 4.1: The pyrobrand marking scheme for *D. impar* 146 (source: O'Shea, 1996).

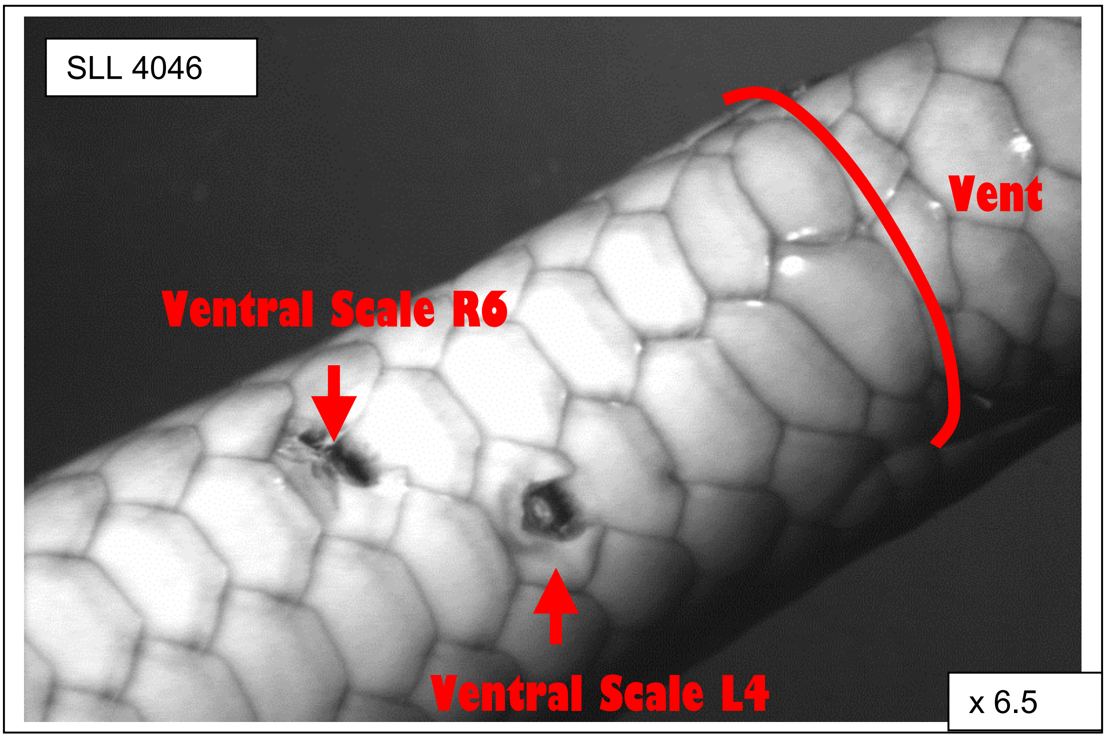


Figure 4.2a: Pyrobrands on Striped Legless Lizard 4046, thirteen days after marking. Note scales L14 (representing **4**000 in the ID 4046) and R10 (representing **0**hundreds in the ID 4046) are not displayed. First pair of pre-anal scales are not included in count.

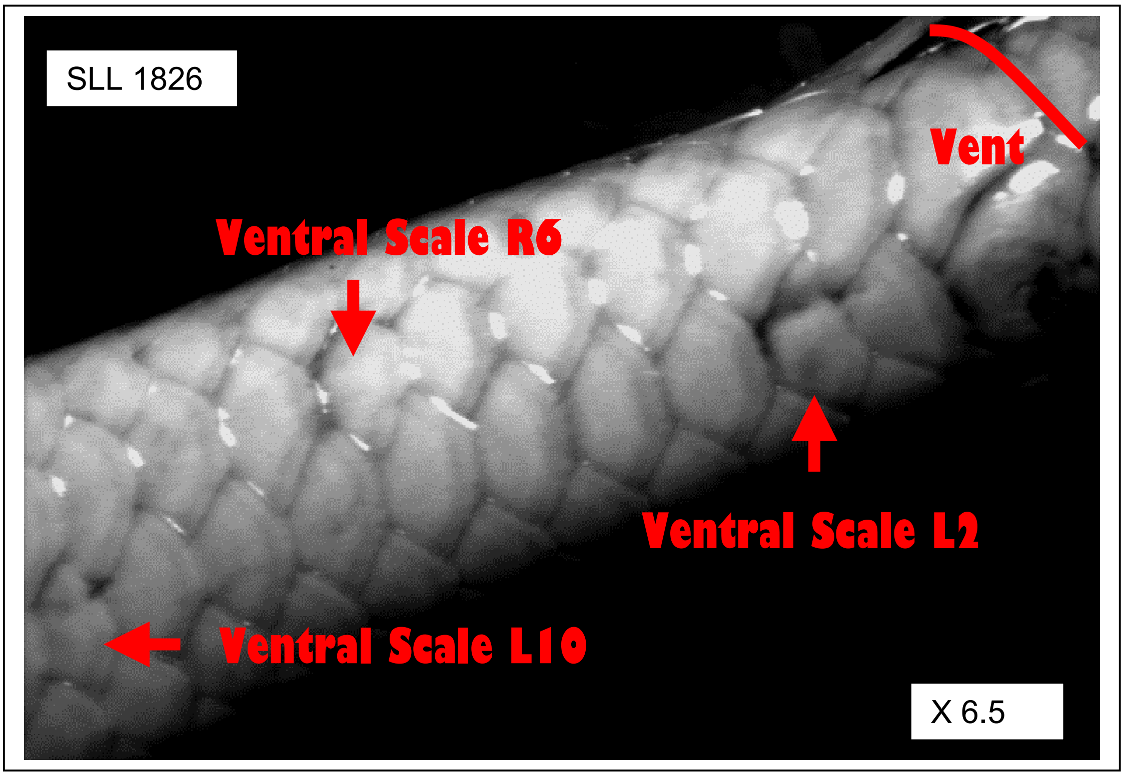


Figure 4.2b: Faintly distinguishable pyrobrands on Striped Legless Lizard 1826, five months after marking. Note scale R18 (representing **8**00 in the ID 1826) is not displayed. First pair of pre-anal scales are not included in count.

In order to assist with the identification of recaptured animals, a range of photographs were taken (in addition to pyrobranding) of each individual captured in January to March 1999 (refer to Chapter 2 for survey methods). Through this process it was noticed that the shape and pattern of the head scales appeared to be different for each individual.

Integument pattern mapping has been used for the individual identification of reptiles and amphibians and other wildlife previously. This technique has been applied to the identification of newts and other amphibians by taking photographs (Hagström, 1973; Jehle & Hödl, 1998) and photocopies (Andreone, 1986) of the pigmentation pattern on the ventral surface. Sweeney *et al*. (1995) extended the use of belly pattern characteristics for identification to develop the computerised automated newt identification system (ANIS). Carlström and Edelstam (1946) found that pigmentation patterns could also be used to identify individuals of a range of species of snakes and lizards. Nelson *et al.* (2000) photographed the pigmentation patterns of Grassland Earless Dragons *Tympanocryptis pinguicolla* and marked their ventral surface with ink to facilitate rapid identification of recaptures.

The shape and pattern of arrangement of *D. impar* head scales provided a possible alternative or additional technique for identifying individual animals within the Iramoo Wildlife Reserve population. Methods for recording individual head scale patterns were investigated and the suitability of these techniques for identification of individual *D. impar* within the population was assessed.

4.2 Methods

4.2.1 Developing a technique for recording head scale patterns

Initially, photographs of the head scale patterns of *D. impar* were recorded using a SLR camera (Nikon F401x) mounted on a Zeiss Stemi 2000-C dissecting stereomicroscope. However, this was found to produce varying quality photographs as a result of poor light levels and animal movements, which remained undetected until after photo processing by which time the animals had been released into the wild.

Following from this, a system was adopted whereby the SLR camera was replaced with a digital video camera (Dage-MTI CCD-100). Again, the video camera was mounted on a Zeiss Stemi 2000-C dissecting stereomicroscope. Lighting was provided by the microscope’s inbuilt light which illuminated the subject from above. A portable Olympus TL2 dissecting microscope lamp provided additional light from the side. The video camera was linked to the computer program Scion Image.

Use of this digitised system provided black and white images which could be instantly assessed for their clarity, providing the opportunity to take the image again, if the first image was not satisfactory. In addition, a calibrated scale could be easily applied to the image, if required. Images were saved, using the animal's individual identification number (pyrobrand code), onto computer disk and printed onto paper, saving on the costs and resources required if traditional photography had been used. Approximately 250 bitmap images of *D. impar* (each 385 kilobytes in size) could be saved onto a single 100 megabyte zip disk.

4.2.2 Accuracy of individual identification using the head scale pattern mapping technique

Individual *D. impar* forming a captive colony held at Victoria University (Victorian Department of Natural Resources and Environment, Wildlife Act 1975 Research Permit No.: 10000898 and Victoria University Animal Experimentation Ethics Committee application AEETH 12/99) were processed on arrival, using both the pyrobranding and digital head scale pattern mapping individual identification techniques. Length, weight and sex data were also recorded. These animals were salvaged from housing development areas adjacent to Iramoo Wildlife Reserve and were therefore representative of individual *D. impar* likely to be encountered in the Iramoo population.

The captive colony was scheduled to be monitored twice yearly, providing an opportunity to record a sequence of head scale images for each individual in the captive colony over time. In total, the head scale images of 52 individual *D. impar* from the Victoria University captive colony were recorded. The maximum number of images recorded for any individual was six. Table 4.1 provides a summary of the number of animals which had their images taken once or more times.

Number of times the image of an individual was taken	**Number of individuals**
1	33
2	0
3	0
4	3
5	13
6	3
Total	52 individuals

Table 4.1: A summary of the number of individual *D. impar* from the captive colony whose head scale image was taken once or more times.

To test the accuracy of the head scale identification method, all images were printed onto individual sheets of paper, with the identification number of the animal written on the back of each sheet, so that it could not be seen while studying the image. The collection of images was shuffled, to create a random order of all available images. Every image was compared with every other image. Images considered to be of the same individual were placed in discrete piles. Once all of the images had been compared to every other image and placed in piles, the images in each pile were compared with the identification numbers marked on the reverse of the sheet. Individual animals were considered to be positively identified if all of the available images of that animal were placed in a single pile. Variations in the number of images of each individual available for each identification trial served to prevent the observer from having expectations of the number of images that should be placed in each pile.

4.2.3 Assessing the influence of the head scale pattern mapping method on the detected rate of recaptures in pit-fall traps

In order to determine whether the new head scale pattern mapping individual identification technique had an influence on the detection of recaptures in comparison with the pyrobranding technique, the rate of recapture of *D. impar* in pit-fall traps for years when pyrobranding only was used was compared with the rate of recapture when each of the individual identification methods were used in conjunction (table 4.2). Such a comparison would provide an indication as to whether the apparently low recapture rates which have been recorded in studies using only the pyrobranding technique were due to failures in detecting branded individuals.

Pyrobranding only
Autumn 1996 (O'Shea, 1996)
Autumn 1997 (O'Shea & Hocking, 1997a)
Autumn 1998
Autumn 1999
Pyrobranding and head scale pattern mapping
Spring 1999
Autumn 2000
Spring 2000

Table 4.2: *D. impar* pit-fall trapping data available for survey periods when pyrobranding was the only individual identification technique used and for survey periods when both pyrobranding and head scale pattern mapping were used to identify individuals.

4.3 Results

4.3.1 Head scale patterns

The arrangement and pattern of head scales varied between individual *D. impar*. It was found that the shape of the enlarged parietal shields, the number, shape and arrangement of the nuchal scales immediately posterior to the parietals and variations in the frontal and prefrontal scales (figures 4.3a & b) displayed enough variation to enable animals to be differentiated from one another and also for individuals to be positively identified.

4.3.2 Accuracy of individual identification using the head scale pattern mapping technique

In total, 128 *D. impar* head images were assessed for the purposes of differentiating between animals and identifying individuals for cases where more than one image of an individual was available. Of the 128 images, 52 animals were differentiated from each other. Of the remaining 76 images, 75 were correctly identified as belonging to one of the 52 individuals of which only 19 had multiple images taken. Thus, the head scale pattern mapping technique had an accuracy of 99 percent.

Head scale images of individual *D. impar* in the captive colony were recorded over a duration of 30 months, indicating that this is a suitable identification technique for at least this length of time (figure 4.4). Slight changes in head scale patterns occurred in some individuals as a result of injury (figure 4.5). Although this sometimes made identification more difficult, a high level of accuracy could still be maintained. Injured scales generally appear to heal to the same scale formation as originally recorded.

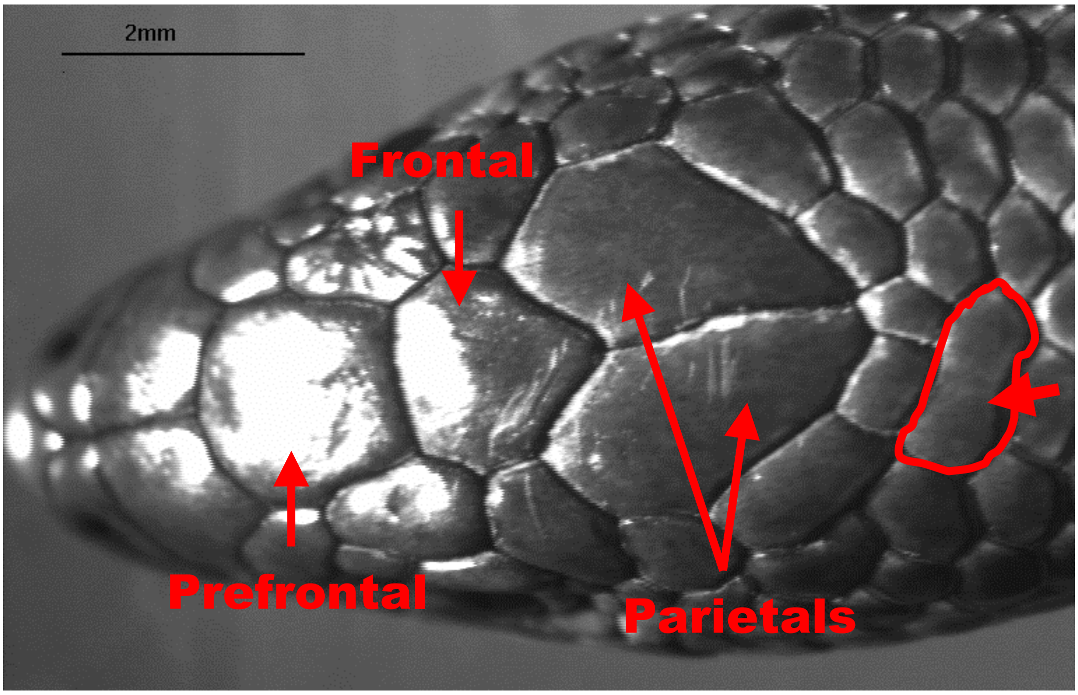
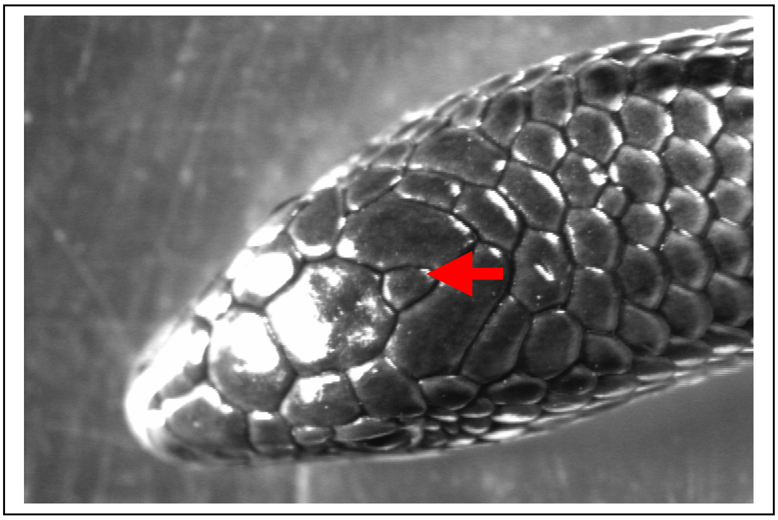
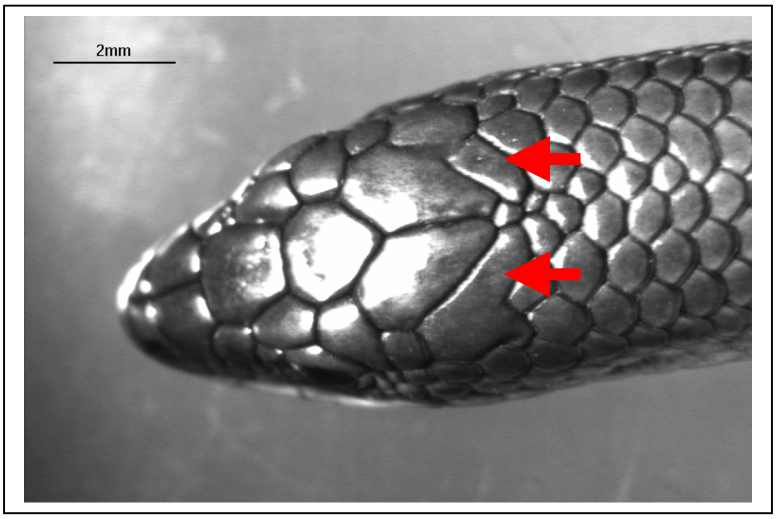


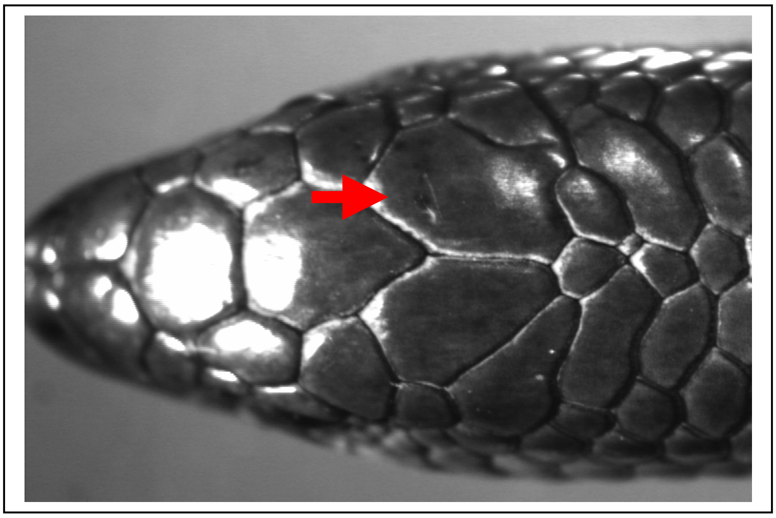
Figure 4.3a: SLL 4042. The shape and arrangement of scales on this head are common for *D. impar*. This individual can be distinguished by the enlarged nuchal scale (arrowed).



SLL 4000. The presence of an interparietal is uncommon for *D. impar*.



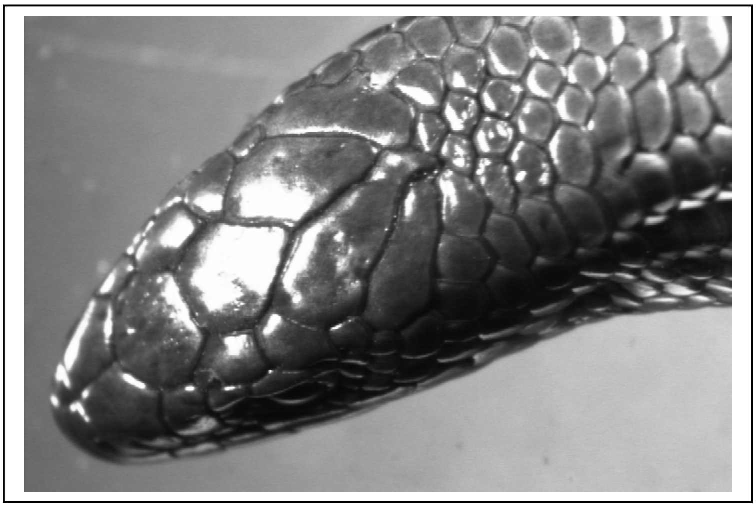
SLL 4004. The shape of these nuchal scales are unusual.



SLL 4028. This parietal shield has an uncommon shape.

Figure 4.3b: Variations in *D. impar* head scale patterns.

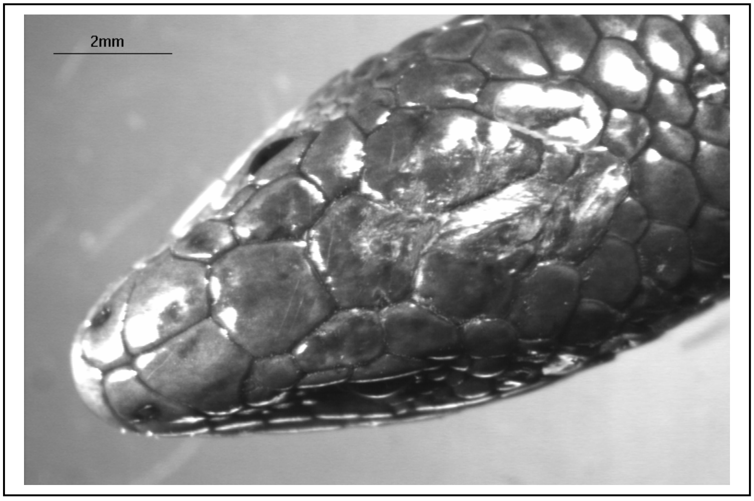
SLL 13. Image taken 19 April 2000



SLL 13. Image taken 25 September 2002

Figure 4.4: Using the head scale pattern mapping individual identification technique, this *D. impar* is easily identifiable after 29 months.

SLL 15. Image taken 19 April 2000



SLL 15. Image taken 3 May 2001

SLL 15. Image taken 6 November 2002

Figure 4.5: Individual *D. impar* can be identified using the head scale technique, even when scales have been injured and healed.

4.3.3 Assessing the influence of the head scale pattern mapping method on the detected rate of recaptures in pit-fall traps

The rate of *D. impar* recapture remained very low across all survey periods, regardless of whether the head scale pattern mapping technique was used for individual identification or not. Although the rate of recapture was not expected to alter within a single survey period, as brands are durable within this time frame, the rate of recapture of *D. impar* recorded in earlier survey periods remained very low when using both the pyrobranding and head scale pattern mapping identification techniques combined (table 4.3). Overall, the highest level of recapture of *D. impar* initially captured in an earlier survey period was 2.7 percent, in Autumn 1999 when pyrobranding was the only form of individual identification used.

Survey period	**Total captures**	**Total recaptures**	**Recaptures within survey period**	**Recaptures from an earlier survey period**
Pyrobranding only				
Autumn 1996	68	3 (4.4%)	3 (4.4%)	N/A
Autumn 1997	85	7 (8.2%)	7 (8.2%)	0 (0.0%)
Autumn 1998	18	0 (0.0%)	0 (0.0%)	0 (0.0%)
Spring 1998	62	4 (6.5%)	3 (4.8%)	1 (1.6%)
Autumn 1999	75	5 (6.7%)	3 (4.0%)	2 (2.7%)
Pyrobranding and head scale pattern mapping				
Spring 1999	141	4 (2.8%)	2 (1.4%)	2 (1.4%)
Autumn 2000	72	6 (8.3%)	5 (6.9%)	1 (1.4%)
Spring 2000	215	18 (8.4%)	14 (6.5%)	4 (1.9%)

Table 4.3: The rate of recapture of *D. impar* when individuals are identified by pyrobranding techniques only and when individuals are identified by pyrobranding and head scale pattern mapping combined.

4.4 Discussion

The head scale pattern mapping technique can provide a reliable, accurate and definitive means of identifying individual *D. impar*. If this technique is the sole means of identifying individual *D. impar*, it meets many of the criteria of a preferred identification method, as outlined in the introduction to this chapter.

In particular, because no physical mark or tag is applied to the animals, it is not likely that this method would affect the survivorship, growth or behaviour of individuals, especially in comparison to the possible and unevaluated negative effects of the traditional pyrobranding technique. In addition, the process of taking head scale images is free from pain, and stress to the individual would appear to be considerably lower.

In its current form this technique does not meet the criterion of being easily used in both the field and laboratory, in that the use of a dissecting stereomicroscope, video camera and computer require that animals are brought in to the laboratory for processing. However, recent developments in digital photography were found to be sufficient for recording the head scale image pattern using only a digital camera (Canon Powershot G2 Digital Camera) in the laboratory (figure 4.6). Although untested in this study, it is foreseeable that similar results could now be achieved in the field, thus replacing the head scale image recording equipment used in this study with a portable handheld digital camera. The use of this recently developed technology would enable animals to be processed in the field and released immediately following collection of morphometric data, thus reducing associated levels of stress even further. Images captured in the field with a digital camera could be downloaded upon return to the laboratory.

A significant drawback of the head scale pattern mapping method for individual identification is that manual comparison of images with previously captured individuals is tedious and time-consuming. Although being a relatively simple and straightforward method, it can at times require an experienced eye to distinguish subtle differences that were sometimes found between individuals. Furthermore, when identifying an individual on successive occasions, it is important that the examiner be mindful of the angle on which each image has been taken and also of the possibility that slight changes in the appearance of some scales might have occurred over time. Thus, the method may require the user to have a degree of experience before the high levels of accuracy reported in this study can be obtained.

Figure 4.6: Images of the head scale pattern can be recorded using a hand-held portable digital camera. Photo: Debbie Reynolds.

It is feasible that the development of a computerised system could be used to reduce the amount of time required to carefully scan and compare images. For example, Sweeney *et al.* (1995) was able to reduce the number of images of the individual belly patterns of Great Crested Newts *Tristurus cristatus* requiring manual comparison to 24 percent of those in the database by developing the Automated Newt Identification System (ANIS). This system used the techniques of image analysis to compare six belly pattern characteristics. If such a system were to be developed for *D. impar*, further refinement of the methods for capturing images may be required, so as to ensure that all images are digitally comparable.

Whether the head scale pattern mapping technique is performed using conventional photography or the preferred digital technology, the costs are moderate. In both cases a dissecting stereomicroscope capable of supporting a camera was required. Further start-up costs for the digital method require a moderate outlay for the video camera, computer and software. However, the ongoing cost of using the digital system are minimal. In comparison, conventional photography requires the start-up costs of an SLR camera, although these are generally available in most departments supporting natural history studies and projects. The costs of conventional photography for the purchase of film and processing are moderate but ongoing and the risk of a loss of images due to poor lighting and focussing are much higher than for the digital system used in this study. It is likely that the development and adoption of methods using a portable handheld digital camera in the field will make the head scale pattern mapping method of individual identification more cost efficient.

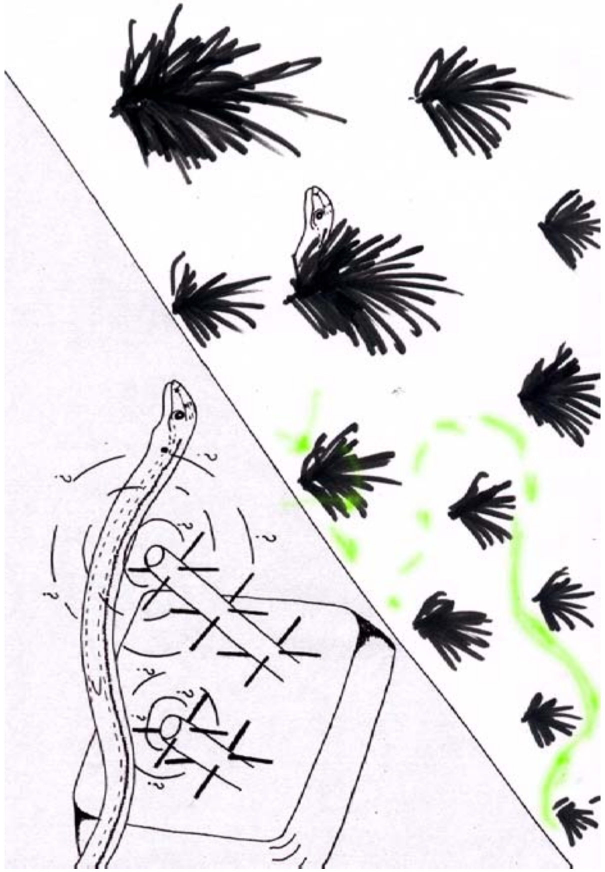
This study did not assess the accuracy of the head scale pattern mapping individual identification technique for individuals smaller than 70 millimetres snout-vent length, as animals of this size were not available in Victoria University's *D. impar* captive colony. Although 15 hatchlings and 70 juveniles captured during field studies at Iramoo Wildlife Reserve were identified using the head scale technique, only three of these individuals were recaptured and length, weight and location data further supported their identity. Two of these individuals were recaptured within two days of initial capture and thus no information regarding the potential change in head scale pattern with growth could be gleaned. The third animal (a juvenile measuring 6.7 millimetres SVL) was still recognisable when recaptured after approximately nine months. In the case of hatchling and juvenile *D. impar*, it may be that the lack of recaptures of these individuals is a true result of low numbers surviving or being recaptured. Alternatively, a low level of recaptures might indicate that the head scale patterns do alter with growth, precluding the use of this method in these smaller size classes. In consideration of the generally low recapture rate for all individuals in pit-fall traps, it is likely that the former is true. However, if the head scale method is to be adopted for future studies, it is advisable that potential changes in head scale patterns of hatchlings and juveniles be tracked over time in a captive population.

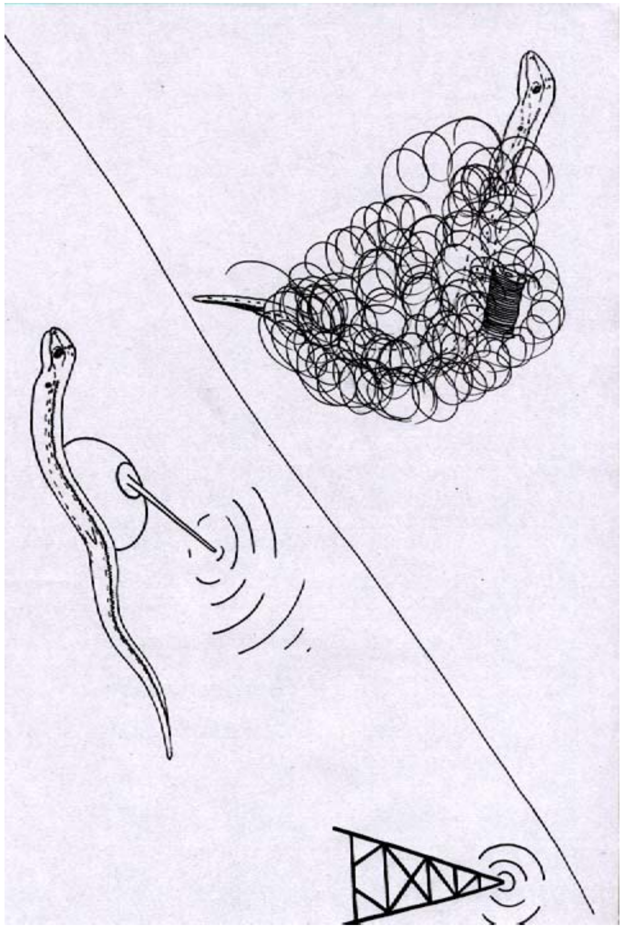
The head scale pattern mapping individual identification technique can be used on dead animals (providing the head scales have not been severely damaged or degraded) and on sloughs. Although not trialed in this study, it is foreseeable that the identification of sloughs with head scales intact could be a useful method in the case of artificial shelter monitoring, as described for roof tiles in Chapter 3, when a reasonable number of sloughs can be obtained. This would provide further information about the duration and periods over which an individual uses a given tile and may also provide information on an individual's movement habits, if sloughs are recorded under more than one tile.

When the use of pyrobranding in conjunction with the head scale pattern mapping technique was compared to the use of the pyrobranding technique only for identifying individual *D. impar*, no observed increase in the rate of recapture of *D. impar* in pit-fall traps was recorded for the combined techniques. This suggests that although the degradation and loss of brands has been observed to occur relatively quickly for this species, the low rate of recapture of individuals within a survey period and between survey periods cannot be attributed to the loss of pyrobrands. Although untested, it is possible that the pyrobranding individual identification technique has an impact on the behaviour or movement of individuals, thus precluding recapture. Future studies should compare rates of recapture of individuals identified using the pyrobranding technique only with the rates of recapture of individuals identified using the head scale pattern mapping method only.

4.5 Conclusion

The head scale pattern mapping individual identification technique is an accurate, durable, simple and humane method of identifying individual *D. impar*. It has the potential to meet most of the criteria for selecting a suitable identification method, especially with further development and adaptations of digital technology and computer programming, which can foreseeably make the method friendly for use in the field and reduce the amount of labour required. The major drawback for adopting this method of identification would appear to be the startup costs of acquiring the appropriate technology. Further work needs to be undertaken to make these methods useful in the field.





Chapter 5 The Harmonic Direction-finder – A Potential Tool for Monitoring the Movements of Striped Legless Lizards in the Field

Chapter 5

The harmonic direction-finder – a potential tool for monitoring the movements of Striped Legless Lizards in the field

5.1 Introduction

Animal tracking has been employed to investigate the microhabitat use of a range of species at a fine scale and to determine home range sizes, site fidelity, movement patterns, and daily and seasonal activities (Richards *et al.*, 1994; Langkilde & Alford, 2002). Tracking may assist in developing a better understanding of how conventional monitoring techniques can be more profitably utilised (Richards *et al.*, 1994), provide a basis for conservation planning for endangered fauna (size and location of reserves; vulnerability to habitat fragmentation, etc.) (Webb & Shine, 1997) and also in the development of appropriate translocation protocols.

For these reasons, the development of an effective method for tracking Striped Legless Lizards *Delma impar* has been identified as a high priority under the Striped Legless Lizard National Recovery Plan (Smith & Robertson, 1999). The development of improved and reliable detection techniques, including tracking, was also listed as a recommendation for achieving accurate data analysis and population assessments for *D. impar* at the Striped Legless Lizard Population and Habitat Viability Assessment Workshop (ARAZPA, 1996).

The choice of tracking method for a species is limited by its size, anatomical features, behavioural characteristics and the environment in which it occurs. In some instances, a simple mark-recapture study can provide adequate information about the movements of animals, however adequate empirically determined sample sizes (number of sightings) are required to avoid drawing erroneous conclusions (Stone & Baird, 2002). It is doubtful that this could be achieved for *D. impar*, given the poor levels of recapture success for the species.

Melville and Swain (1999) were able to describe the home-range characteristics of the small to medium-sized lygosomine Alpine Skink *Niveoscincus microlepidotus* in relatively open habitat by simply attaching squares of individually numbered yellow tape that could be identified from a distance, to the dorsal surface of the lizards. This is not a feasible option for *D. impar*, given the type of habitat in which it occurs and its cryptic, semi-fossorial behaviour.

Stott (1987) described the terrestrial movements of the freshwater tortoise *Chelodina longicollis* using a spool tracking device, a method which is precluded for *D. impar* by its size, anatomical features and probably also the often dense grassy habitat.

Slip and Shine (1988) were able to comment on the daily and seasonal movements, habitat use and home ranges of adult Diamond Pythons *Morelia spilota spilota* using miniature radio transmitters. Despite significant developments and further miniaturisation of this technology, radio tags are still too large for attachment or implantation into *D. impar*.

Fellers and Drost (1989) tracked Island Night Lizards *Xantusia riversiana* which deposited trails of fluorescent powder in which they had been dipped, for a minimum of five days. Kutt (1992; 1993a; 1993b) trialed this method for *D. impar* and was able to track one individual for a distance of 19.7 metres in a single day before the fluorescent powder wore off. In some instances, *D. impar* trails were as short as 1.25 metres and the average trail was 5.14 metres. Although these distances and time-frames are too short to draw any far reaching conclusions about *D. impar* movements and activity, further development of this tracking method may be warranted and should include an assessment of possible impacts on movement, behaviour and susceptibility to predation (Richards *et al.*, 1994).

Engelstoft *et al.* (1999) were able to monitor the movements of Sharp-tailed Snakes *Contia tenuis* in the field for 208 days, using a relatively new system, the harmonic direction-finder. This system offers a potential method for monitoring the movements of *D. impar* because the tag does not require a battery and is therefore small and lightweight.

The harmonic direction-finder functions by emitting a signal at a known microwave frequency from a hand-held transceiver which contains the power source. When the signal is received by a tag (consisting of a diode and antennae), which is attached to an animal, it is converted to an harmonic of the original signal and reflected back to the transceiver, which is tuned to detect the first harmonic. The received microwave harmonic is then converted to an audio signal which can be perceived by the researcher. Signals are highly directional, allowing the location of animals to be determined precisely.

Harmonic direction-finders have been used to monitor the movements of a range of terrestrial and aerial invertebrates (Mascanzoni & Wallin, 1986; Wallin & Ekbom, 1988; Kennedy & Young, 1993; Roland *et al.*, 1996; Lövei *et al.*, 1997). In each of these cases, the tag has been affixed to the elytra, thorax or abdomen of the invertebrate, with the antenna trailing behind. The harmonic radar is a similar system which has been used to provide omnidirectional and range information on bees (Riley *et al.*, 1996; Osborne *et al.*, 1997; Riley *et al.*, 1998; Carreck *et al.*, 1999; Capaldi *et al.*, 2000) but requires that the tag is vertically glued to the thorax.

Attempts have been made to use the harmonic direction-finder to track sparrows (Teamey, 1998), frogs (Langkilde & Alford, 2002) and agamid lizards (Langston, 1996). However, to date it appears that the only vertebrates to which this system has been successfully applied is in the tracking of snakes with snout-vent lengths less than 35 centimetres (Webb & Shine, 1997; Engelstoft *et al.*, 1999).

Generally the harmonic direction-finder transceivers used in these studies have emitted a continuous signal at about 917 megahertz and four to five Watts, thus receiving the signal at 1834 megahertz. Tags consist of a Schottky low-barrier diode and antennae and the signals of individual tags cannot be distinguished from each other. A range of diodes and antennae configurations have been used in the various studies and appear to be one of the major factors contributing to the success or failure of the harmonic direction-finder technique for monitoring the movements of animals. Detection ranges between 10 centimetres (Teamey, 1998) and 50 metres (Roland *et al.*, 1996) have been reported.

Diodes as large as three millimetres by 7.5 millimetres have been used to track beetles (Mascanzoni & Wallin, 1986) and antennae have been as long as 13 centimetres (Langkilde & Alford, 2002) and made of varying gauges of copper, aluminium and stainless steel wire. Although there have been some attempts to determine the best configuration of diodes, antennae and antennae length (Mascanzoni & Wallin, 1986; Webb & Shine, 1997; Teamey, 1998; Engelstoft *et al.*, 1999), no apparent consistency in the resultant read ranges was determined from the literature.

Given the small size of the snakes and implanted tags used by Engelstoft *et al.* (1999) and a reported read range of up to 12 metres, this system seemed most promising and applicable for use in studying the movements of *D. impar*. Thus, trials were undertaken to implant harmonic direction-finder tags, initially into a larger and non-threatened pygopodid species, the Olive Legless Lizard *Delma inornata* and eventually into *D. impar*.

5.2 Methods

The tags used for trial implantations into *D. inornata* and *D. impar* were constructed from Hewlett-Packard HP5316 Schottky diodes and 0.079 millimetre diameter enameled copper wire, as outlined in Engelstoft *et al.* (1999). The total length of the tag was eight centimetres, with a dipole antenna, and the weight was approximately four micrograms (figure 5.1).

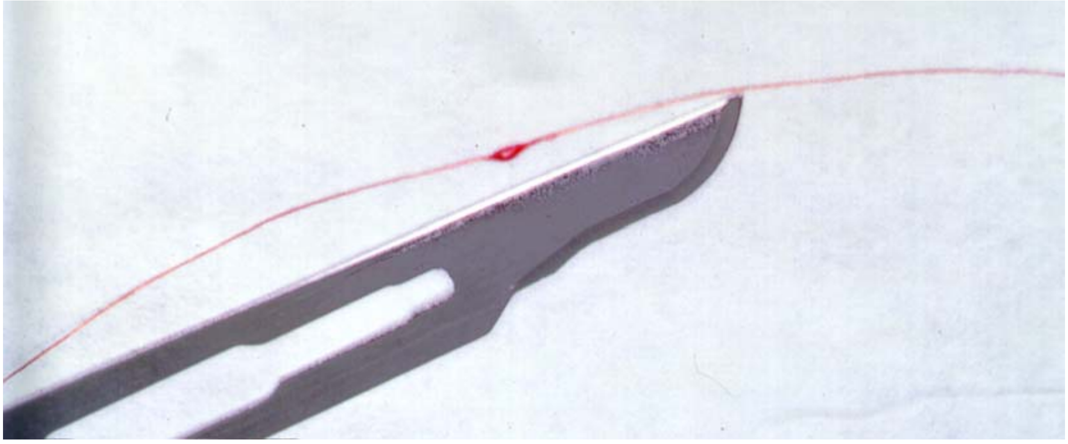


Figure 5.1: An harmonic direction-finder tag beside a scalpel blade. The enlarged section in the centre is the diode from which the antennae extend either side. Photo: Chris Banks.

Implantations were performed on two specimens of *D. inornata* and one specimen of *D. impar*, at the Animal Treatment Centre, Melbourne Zoo between 20 July and 19 October 1999. All three animals formed part of the Melbourne Zoo collection and had been kept under warm conditions indoors for at least two weeks prior to implantation. Animals were anaesthetised using two to five percent isofluorane in oxygen administered via an inhalation chamber throughout the procedure. As difficulties in maintaining continuous anaesthesia were experienced, an intra-muscular injection of the sedative Ketamine (40 milligrams per kilogram, otherwise expressed as 40 micrograms per gram) was also administered.

Two points of entry were marked on the dorso-lateral body surface of each lizard, at a distance of nine centimetres, keeping clear of the head. An exit point mid-way between the two entry points (i.e. 4.5 centimetres) was also marked. Two 25 gauge spinal needles were inserted subcutaneously at each entry point and threaded between the skin and muscle layer, to the central exit point. The stems of the spinal needles were removed, leaving a hollow tube through which the antennae of the tag could be back-threaded (figure 5.2). This resulted in the largest part of the tag (the diode) being centrally located and a small skin incision was required in order to implant this part of the tag (figure 5.3). The spinal needles were withdrawn, leaving the antennae threaded under the skin. Entry point incisions were closed with skin bond and the central exit point, where the diode was located, was sutured. The use of a binocular dissecting microscope assisted with such fine work. Tags were sterilized with ethylene-oxide prior to commencement of the implantation procedure.

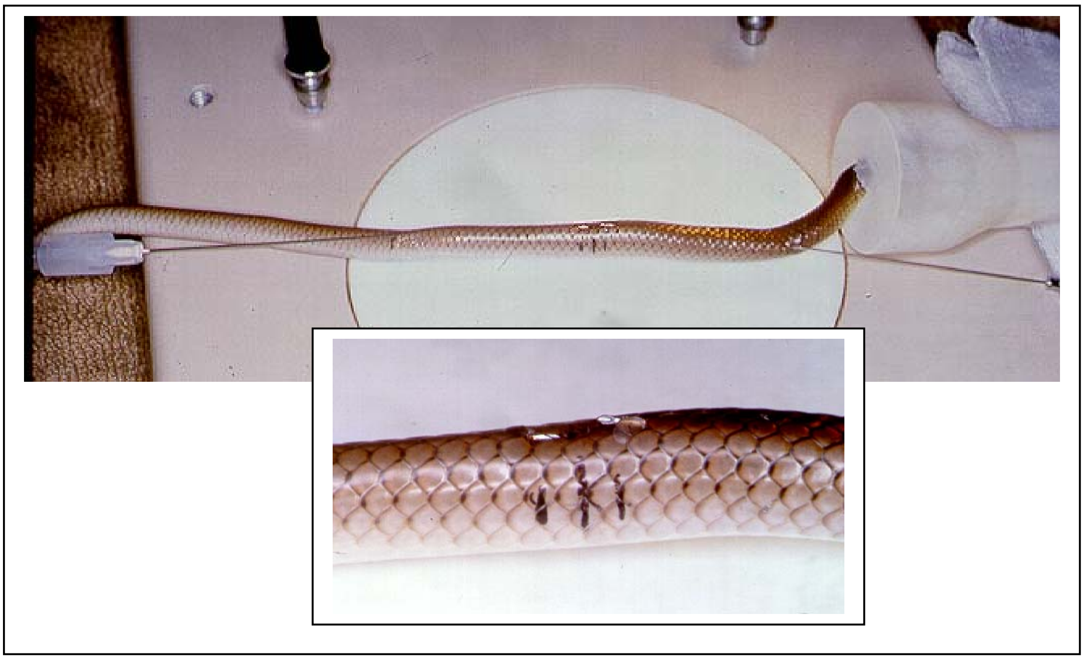


Figure 5.2: A *Delma inornata* with head in inhalation chamber and spinal needles threaded subcutaneously, ready for the tag to be back-threaded. Photo: Chris Banks.

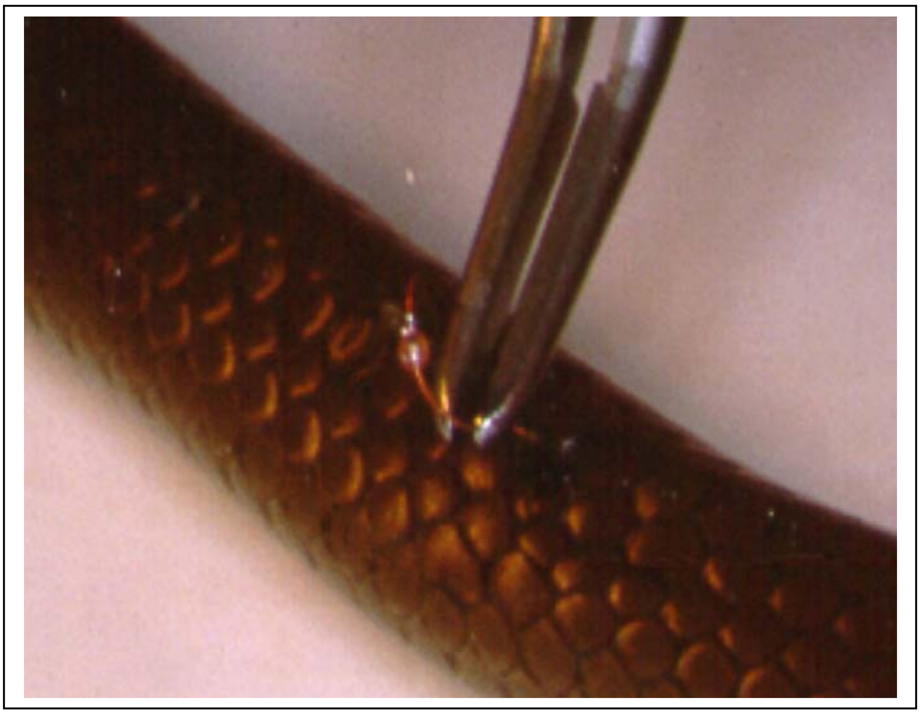


Figure 5.3: The tag has been back-threaded and the spinal needles removed. The diode component of the tag is being tucked under the skin, ready for closure. Photo: Chris Banks.

Each animal was checked for signal transmission and X-rays of the second *D. inornata* and the *D. impar* which were implanted with tags were taken immediately following the procedure and one week later. Animals were kept under observation at the Animal Treatment Centre for a minimum of 24 hours before being returned to housing arrangements similar to those in the weeks prior to implantation. Weights of each animal were regularly recorded until release into the normal holding environment.

5.3 Results

Immediately following implantation, signals could be obtained from each implantee, using a Recco harmonic direction-finder transceiver. However, in each case the signal became weak and disappeared after one to two weeks following implantation. In the case of the first *D. inornata* to be implanted, a slough was found in the enclosure a few days following implantation, with the diode component of the tag stuck to it. Apparently the tag had become stuck to the skin bond used to close the incision and was broken off during sloughing. For this reason, only suturing was used to close the central incision in subsequent implantation trials.

X-rays taken immediately after implantation into the second *D. inornata* and the *D. impar* showed an intact tag lying beneath the skin (figure 5.4). However, when X-rays were taken approximately one week later, the tags had fragmented into several pieces (figure 5.5), explaining why the signal was either weak or could not be received.

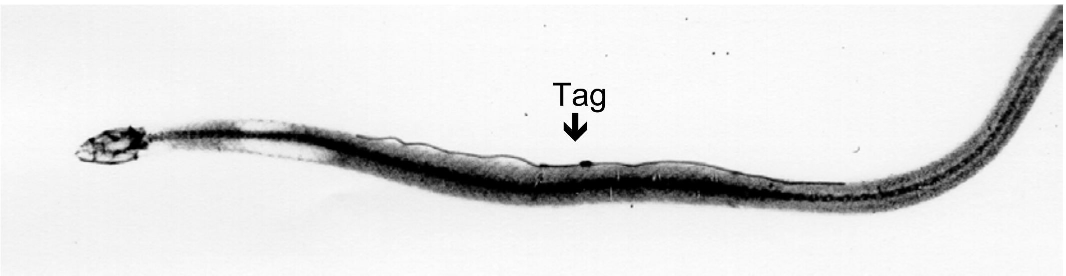


Figure 5.4: An X-ray of *D. impar* immediately following implantation, showing an intact tag beneath the skin.

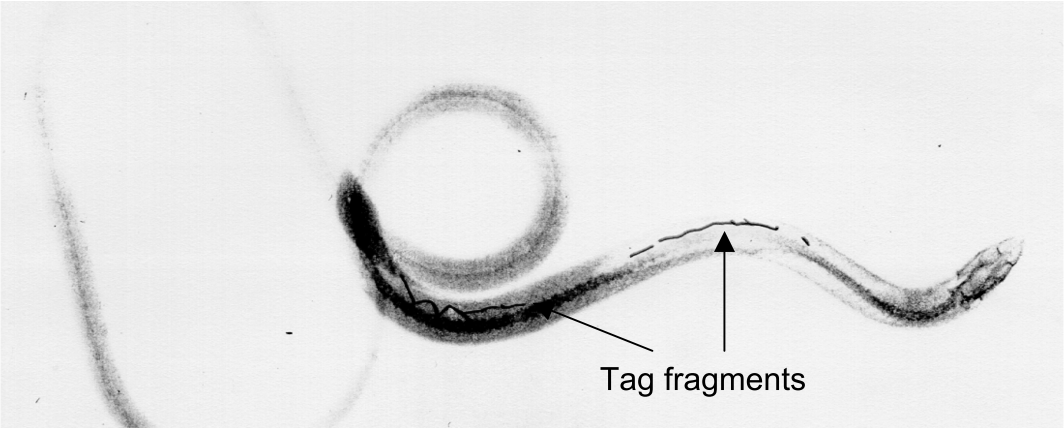


Figure 5.5: An X-ray of a *D. inornata* approximately one week after implantation, showing the tag broken into several fragments.

Despite the failure of the tags, the implantation procedure was considered partially successful, in that it was shown that the tags could be implanted and that both *D. inornata* and the smaller *D. impar* were robust enough to withstand such a procedure. Weights following implantation varied for each animal (table 5.1) and all three are still alive, approximately four years later.

	D. inornata 1	*D. inornata 2*	*D. impar*
Weight prior to implantation	11.60g	10.2g	-
Weight on day of implantation	12.0g	10.5g	9.0g
Weights following implantation	11.20g	10.86g	8.93g
	11.24g	10.6g	8.56g
	11.12g	-	8.25g
	-	-	7.85g
	-	-	7.5g
	-	-	8.91g

Table 5.1: The weights of *D. inornata* and *D. impar* before and after the implantation of an harmonic direction-finder tag.

Efforts to develop a more robust tag were undertaken. Initially, tags were encased in a protective sleeve of rubber silicon tubing, with an outer diameter of 0.94 millimetres. This meant that the tag could no longer be drawn through the hollow spinal needles, as performed in the previous implantation procedures. To solve this problem, a solid 10 centimetre long cutting type needle (20 gauge) was manufactured. The tag was attached to the blunt end of the needle by drawing the silicon tubing over a series of small barbs. The intention of this design (similar to a sewing needle and thread) was to draw the needle under the skin of the lizard for a distance of approximately eight centimetres before exit. The silicon encased tag would trail along behind the needle and when the tag was suitably positioned under the skin, the tubing would be cut from the needle, leaving the tag in position.

A single trial of this new procedure was performed on a live *D. inornata* at Melbourne Zoo. The trial was unsuccessful, as the needle could only be threaded subcutaneously for a distance of three centimetres before resistive forces lead to concern for the animal’s welfare and the procedure was aborted. The conclusion of this trial was that the needle and silicon tubing were too large in diameter to be drawn under the relatively tight skin of the animal.

Further efforts were undertaken to improve the robustness of the tag, whilst keeping the tag diameter small. A single tag was successfully manufactured in thin rubber silicon tubing (0.64 millimetres outer diameter). Each end of the silicon tubing was attached to small barbs at the blunt end of a solid cutting type needle (i.e. two individual needles) (figure 5.6). Each needle was 0.6 millimetres in diameter and 4.5 centimetres long. The surgical procedure in this instance was that both needles would enter the lizard at a central point and be drawn subcutaneously in opposite directions for a distance of approximately four centimetres each, before exiting. The silicon encased tag would follow the needles under the skin until suitably positioned and the tubing would be cut from each of the needles, thus leaving the tag in place under the skin.

Two attempts were made to implant the tag into two captive *D. impar* at Melbourne Zoo. However, on each occasion difficulties in maintaining adequate anaesthesia were experienced and the procedures were aborted.

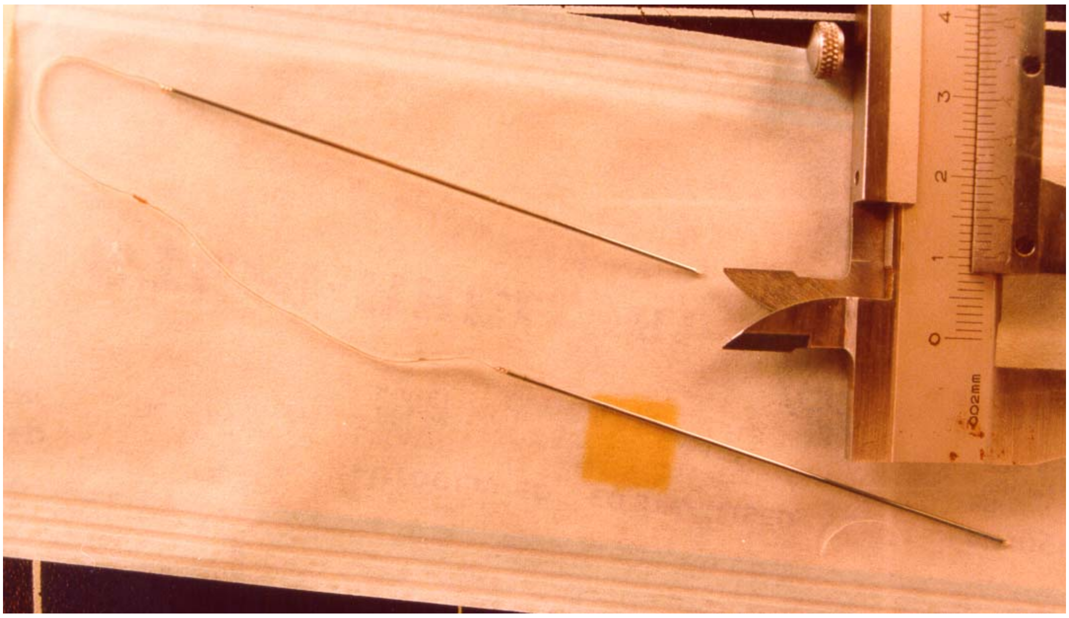


Figure 5.6: A silicon encased tag attached to two solid cutting type needles.

5.4 Discussion

Harmonic direction-finder technology offers potential for tracking the movements of *D. impar* and possibly other species of the family Pygopodidae, in that animals appear to be able to cope with having such devices subcutaneously implanted. In this study, the technology failed because the tags were not sufficiently robust to withstand the twisting movements of *D. impar*. In contrast, Engelstoft *et al.* (1999) reported being able to receive signals for at least 208 days, using the same tags, which were implanted into *C. tenuis*. However, these snakes, although capable of twisting movements similar to *D. impar*, have much looser skin which is not as likely to exert such great frictional forces on the tags. Thus, a more robust tag is required for this technology to be applicable to *D. impar*.

The smaller silicon encased tags and associated implantation procedures are potentially suitable for *D. impar*. However, problems with anaesthesia need to be overcome. It seems that individual *D. impar* have variable responses to the anaesthesia and if an alternative method is not devised, it may be feasible to commence implantation trials with several animals available and thus use the animal that responds best to anaesthesia.

During one of the trials to implant the smaller silicon encased tag, a spinal needle (gauge 23) was partially inserted under the skin and then removed, to create a tunnel between the skin and muscle layer through which the cutting type needle with the tag attached could travel. It was noted that the spinal needle was much easier to push under the skin, suggesting that it had a more highly polished surface which reduced the amount of friction experienced. If further implantation trials are to proceed, it would be advisable to have a similar finish on the cutting type needles. An alternative but less preferable option would be to simply use a spinal needle to create a tunnel between the skin and muscle layer, as described, prior to threading the needles with tubing attached.

Implantation of harmonic direction-finder tags into the body cavity of *D. impar* was seen as a potential option which could remove the need for implantation devices and may also lower the frictional forces exerted on the tags. Upon further consideration this option was rejected for several reasons: the length of the body cavity is not large enough to accommodate the long antennae of the tag; the surgical procedure would have a much greater physical impact, which would require animals to be removed from the field habitat for lengthy periods (i.e. weeks rather than days); and such a procedure would require lengthy anaesthesia, a method which is still being refined.

5.4.1 Animal welfare considerations

Weight and visual appearance were the only factors considered following the implantation of tracking devices into *D. inornata* and *D. impar* in this study. However, as Richards *et al.* (1994) and Langkilde and Alford (2002) point out, it is important to determine whether the implant or the implantation procedure has any effect on the subject animal's health, movement and behaviour before drawing conclusions relating to their normal patterns of movement in the field. Few studies involving the implantation of tracking or identification devices into reptiles have addressed this in any qualitative or quantitative manner. Generally, where assessments have been made, a large number of subjects has been compared against a control. Given the threatened status of *D. impar* and the labour-intensive nature of implantation procedures, it would not be feasible to conduct such studies for this species.

Table 5.2 identifies aspects of reptilian biology that have been suggested as being potentially affected by the implantation of tracking or identification devices. The results of studies designed to assess potential impacts have been summarised and methods for assessing these impacts on *D. impar* have been suggested for future implantation trials.

5.4.2 The harmonic direction-finder transceiver

Commercially manufactured transceiver units are available from RECCO AB (Sweden) and were priced at $US6,400 in 1999. These units do not comply with Australian Safety Standards and were only able to pick up signals from the tags used in this study at distances up to about five metres. Fourth year students from the Department of Electrical and Electronic Engineering, University of Melbourne, commenced construction of a new transceiver which would use pulsed power signals, instead of a continuous signal, to overcome issues of safety to field workers and animals. The use of a pulsed signal would also enable an increase in the power output, thereby enhancing the detection distance of the device. The students also attempted to reduce the level of false signals picked up by the receiver component. Although a new basic unit has been constructed, further work needs to be conducted to ensure that the system works safely and effectively. Teamey (1998) also undertook the construction of an harmonic direction-finder transceiver and provides substantial technical detail.

5.5 Conclusion

The harmonic direction-finder remains a potential option for monitoring the movements of *D. impar* in the field. Further efforts to develop suitable techniques for implanting tags or manufacturing more robust tags need to be undertaken. In addition, further development of the transceiver equipment is required to ensure that the system meets Australian Safety Standards and that the range of detection can be optimised. Commercially manufactured transceivers are expensive. Tags are not individually coded and therefore it would not be possible to distinguish between individual *D. impar*, without visually sighting the animals. The tags used in this study were not commercially available and were purchased from an electrical engineer in Canada at the considerable cost of $Ca40.00 per tag. Engelstoft *et al*. (1999)

provides details for the manufacture of tags, making it feasible for tags to be constructed in-house in the future.

Potential impact	Results of previous studies on reptiles	Proposed methods of assessment for *D. impar*
Weight loss	PIT chips implanted into peritoneal cavities of neonate garter snakes (n=12). Control (n=11). Snakes weighed at 3, 6 & 9 week intervals after surgery. No difference between groups (Keck, 1994).	Weigh animals 3 weeks, 1 week and immediately prior to surgery and once per week for one month following implantation and every month thereafter for 6 months.
Feeding	Radio tags implanted into peritoneal cavities of garter snakes, with subcutaneous antenna. Feeding rate prior to implantation compared with feeding rate after implantation (n=11). Snakes returned to normal feeding after 14 days (Lutterschmidt & Rayburn, 1993).	Compare feeding rate 3 weeks prior to implantation with feeding rate up to 4 weeks after implantation.
Growth	PIT chips intra-abdominally injected into rattlesnakes (n=15). Control (n=18). No difference in growth rates between groups (Jemison *et al.*, 1995).	Not testable due to slow growth rates of *D. impar*.
Defaecation	Observations only. No impact recorded.	Number of scats to be recorded 3 weeks prior and up to 4 weeks after surgery.

Table continued next page.

Potential impact	**Results of previous studies on reptiles**	**Proposed methods of assessment for *D. impar***
Movement	PIT chips injected intra-abdominally into rattlesnakes (n=15). Control (n=18). No difference in the net distance moved or rate of movement between groups (Jemison *et al.*, 1995). PIT chips implanted into peritoneal cavities of neonate garter snakes (n=12). Control (n=11). Speed measured. No difference between groups (Keck, 1994).	Measure the maximum height of saltation (jumping) (Bauer, 1986) at weekly intervals, 3 weeks before and up to 4 weeks after surgery.
Thermoregulation	PIT chips implanted into the peritoneal cavity of fence lizards (n=12). Sham (n=12), control (n=12). Selected body temperatures measured before and after surgery. Thermoregulation patterns not altered, allowing a few days recovery (Wang & Adolph, 1995).	Measure preferred body temperatures before and after surgery (see Osmond, 1994). Use sham implantation surgery to distinguish between effects of diode and effects of surgery.
Sloughing	Observations only. No impact recorded.	Visual observation to determine whether sloughing is impaired and incisions are healed.
Tag movement or loss	Several papers document the movement or loss of PIT chips in different reptile species.	X-ray *D. impar* immediately following implantation and 1 and 3 weeks thereafter.

Table 5.2: A summary of studies assessing the impact of implantation and surgery on reptiles and proposed methods of assessment for *D. impar.*

Chapter 6 The Use of Fire as a Management Tool in a Western Basalt Plains Grassland Remnant Supporting a Large Striped Legless Lizard Population

Chapter 6

The Use of Fire as a Management Tool in a Western Basalt Plains Grassland Remnant Supporting a Large *Delma impar* Population

6.1 Introduction

The Western Basalt Plains Grassland Community is listed as a Threatened Community under the Victorian *Flora and Fauna Guarantee Act* 1988 (Muir, 1994). Today less than 0.1 percent of its former extent remains (Stuwe, 1986; Craigie & Stuwe, 1992) in fragmented, isolated and degraded remnants, few of which are managed for nature conservation purposes (McDougall *et al.*, 1994). Prior to European colonisation, natural and Aboriginal-lit fires are thought to have played a pivotal role in influencing the ecological structure and functioning of the grassy plains of south-eastern Australian (Stuwe, 1994; Lunt & Morgan, 2002). Today, the floristic diversity of individual remnants is a result of fire and grazing regimes, mediated through competition from the dominant grass species, Kangaroo Grass *Themeda triandra* (Stuwe & Parsons, 1977; Morgan & Lunt, 1999; Lunt & Morgan, 2002).

Remnants with an open species-rich vegetation have a history of frequent burning and/or minimal stock grazing, whereas remnants with dense species-poor vegetation have been infrequently subjected to fire or grazing (Stuwe & Parsons, 1977; McDougall & Kirkpatrick, 1994). Continuous stock grazing has resulted in remnants with an open but species-poor vegetation (Stuwe and Parsons, 1977).

Often the dominant species in Western Basalt Plains Grassland Communities is *T. triandra* (Stuwe & Parsons, 1977; McDougall, 1989). At sites where the history has been one of infrequent sward disturbance *T. triandra* competitively excludes some forb species and the vegetation becomes species-poor (Stuwe & Parson, 1977; Kirkpatrick, 1986; Stuwe, 1986; McDougall, 1989; Stuwe, 1994; Morgan, 1999).

Although fire is considered to be a good management tool for maintaining native species diversity (Stuwe & Parson, 1977; Craigie & Stuwe, 1992; Lunt, 1991), the use of fire at long-undisturbed sites has been observed to lead to an increased abundance of annual exotics (Lunt, 1990). However, sward disturbance through the use of fire is considered an important aspect of management, as it has been demonstrated that when the inter-fire-interval exceeds six years, the number of tillers per *T. triandra* tussock and the total number of tussocks declines. By 10 to 11 years on productive sites without biomass removal (fire, grazing, mowing), the canopy collapses upon itself and forms a thick layer of dead thatch over the soil surface, allowing only minimal seedling recruitment and preventing new tiller initiation (McDougall, 1989; Morgan & Lunt, 1999). In these instances the above-ground biomass decline is mirrored below-ground, with the root biomass also being lower in long-undisturbed vegetation (Morgan & Lunt, 1999). Further to this, tussock senescence and decay is believed to lead to increased available soil nutrient (Wijesuriya & Hocking, 1999) and areas of dead grass are quickly occupied by rapid growth of exotic species (Lunt & Morgan, 1999).

This phenomenon of increased biomass following fire or grazing and eventual tussock senescence and decay in the absence of further biomass removal has been observed at several *T. triandra* grasslands in Victoria and is likely to have some generality in *T. triandra*-dominated grasslands in south-eastern Australia, although the rate of tussock decline will undoubtedly vary between sites, and depend chiefly on site productivity (Morgan & Lunt, 1999; Lunt & Morgan, 2002).

Although frequent biomass removal is considered necessary in maintaining the health and vigour of *T. triandra* dominated remnants, management of remnants can become problematic when significant fauna are also present (Morgan & Lunt, 1999; Lunt & Morgan, 2002). The Striped Legless Lizard *Delma impar* is a species where the perceived negative impacts of frequent burning have caused a shift in vegetation management regimes (Craigie & Stuwe, 1992; Webster *et al.*, 1992). However, the findings of Morgan and Lunt (1999) suggested that although changes in grassland structure, as a result of sward disturbance by fire, may affect the habitat suitability for *D. impar*, exclusion of fire for long periods is likely to cause sward deterioration and potential habitat loss or degradation for this species in the long-term. Thus, short inter-fire-intervals (less than five years) allowing *T. triandra* to maintain its competitiveness and habitat values in the long-term have been recommended (Morgan & Lunt, 1999).

Fire is considered to be one of the major long-term threats to *D. impar* population survival and habitat suitability (Working Party, 1994; ARAZPA, 1996; Smith & Robertson, 1999). Fire has been noted as a direct cause of *D. impar* mortality (Coulson, 1990; Moro, 1990; Walton, 1995) and it is thought that the degree of impact may vary with timing and amount of soil cracking (autumn/winter is low impact; spring/summer is high impact) (Coulson, 1990; ARAZPA, 1996), location (ARAZPA, 1996) and fire type (wildfire is low impact; control/convection burns are high impact) (Walton, 1995; ARAZPA, 1996).

Fire is also considered to be an indirect cause of *D. impar* mortality and population decline. However, evidence for this is limited and conflicting and the mechanisms of this impact have only been assumed. Based on low capture successes Moro (1990) recorded the greatest number of individual *D. impar* in vegetation dominated by *T. triandra* which had been burnt in the Spring prior to the January – February trapping programme. Kutt (1992) suggested that the influence of fire history did not appear to influence distribution and Hadden (1995) located sites where the species persisted under an annual burning regime. Alternatively, Coulson (1990) recorded the lowest rates of capture in areas with low vegetation cover as a result of fire and Walton (1995) only captured *D. impar* in areas that had not been burnt for more than two years. All studies were hampered by poor capture success and only Kutt (1992) presented data of replicated surveys over time, hence the statistical validity of assessing the impact of fire on most of these populations is under question.

Experts at the *D. impar* Population and Habitat Viability Assessment Workshop (ARAZPA, 1996) considered that indirect mortality and population decline as a result of fire was most likely to occur as a result of increased predation due to habitat loss (Coulson, 1990), decreased food availability due to habitat loss and lower recruitment as a result of habitat loss (ARAZPA, 1996). Burning was considered to have a greater impact on reproductive output than on adult survivorship, and models of these assumptions showed that when the fire frequency was once every year or greater, this resulted in negative population growth leading to extinction, especially in small populations (ARAZPA, 1996).

Generally, however, it has been considered that burning regimes with a frequency greater than once every five years are likely to negatively influence *D. impar* survivorship. In direct conflict with this is the recommendation by Morgan and Lunt (1999) that in order to maintain *T. triandra* health and competitiveness, an inter-fire-interval of less than or equal to five years is required.

In the study reported in this thesis, the impacts of biomass reduction burns on *D. impar* abundance, size class structure, sex ratio, condition, and predation levels over two years are assessed. However, this study was not able to assess the impacts of fire frequency, timing, size or fire type on *D. impar* population numbers or dynamics.

6.2 Methods

6.2.1 Project design

The effects of biomass reduction burns on *D. impar* were assessed over two years, 1999 and 2000, at Iramoo Wildlife Reserve. In 1999 the results of data collected from eight unburnt survey plots were compared to data collected from eight burnt survey plots. Unburnt survey plots were randomly selected from plots which had been used in previous *D. impar* survey periods. Burnt survey plots (see below for burn methods) were located in tussocking grassy vegetation in available spaces between pre-existing surveys plots across the reserve. Each individual burn survey plot was set at a minimum distance of 50 metres from any other survey plot.

In 2000 *D. impar* survey data was collected and compared from eight unburnt survey plots, seven survey plots 'burnt two-years-ago' and eight recently burnt survey plots. Unburnt and recently burnt survey plots were randomly selected from plots which had been used in previous *D. impar* survey periods. 'Burnt two-years-ago' survey plots consisted of the burnt plots from the 1999 burnt/unburnt experiment. Only seven 'burnt two years ago' replicates were used, as one of the original eight plots was re-burnt by an escaped fire.

Figures 6.1 and 6.2 indicate the distribution of survey plots and treatment types for the Spring 1999 and Spring 2000 survey periods, respectively.

6.2.2 Biomass reduction burns

Biomass reduction burns were conducted in mid-Summer (December/January) 1998-9 and mid-Autumn (April) 2000 by Parks Victoria and the Western Region Fire Brigade. All burns were approximately 0.5 hectares in size and constrained by a slashed fire break and water

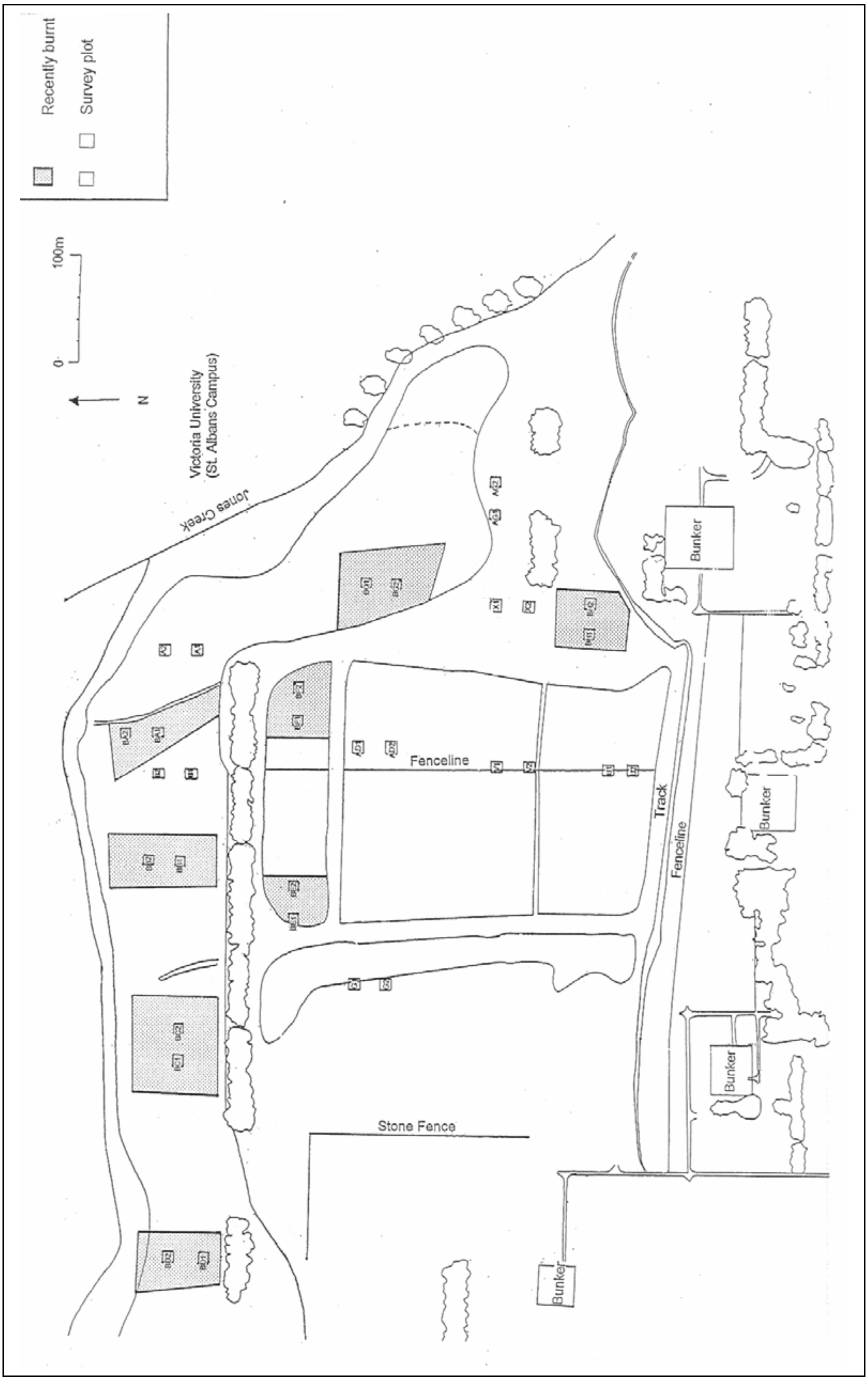


Figure 6.1: The location of unburnt and recently burnt survey plots, Spring 1999. A pair of boxes represent one survey plot.

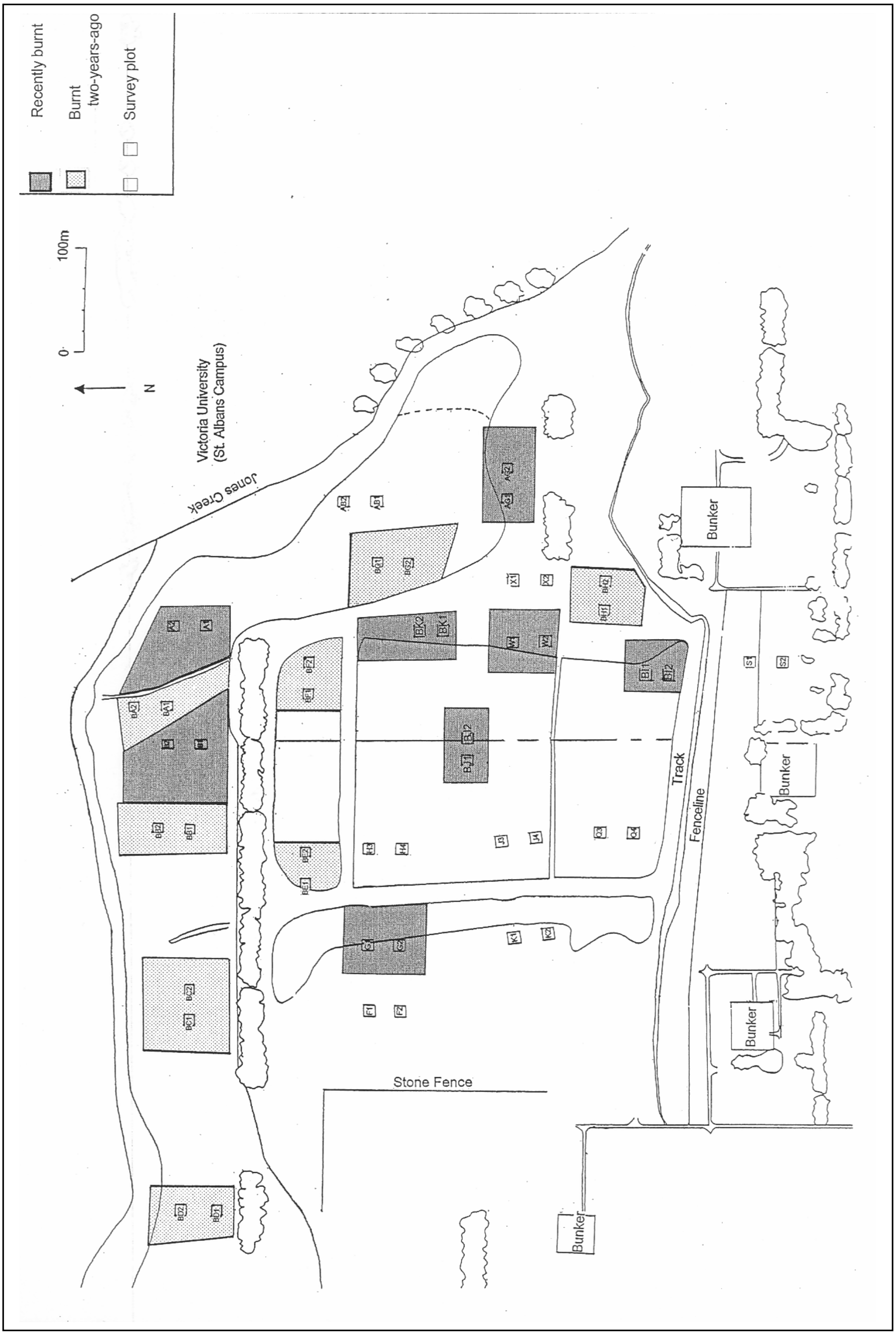


Figure 6.2: The location of unburnt, burnt two-years-ago and recently burnt survey plots, Spring 2000. A pair of boxes represent one survey plot.

containment lines. Medium to hot fires were achieved by burning into the wind and all above-ground vegetation was removed. The vegetation in the burnt areas and fire breaks rapidly resprouted in the intervening time between the burn events and the trapping surveys in the following Spring.

All burn sites were considered to be at a sufficient distance from other survey plots and artificial shelter monitoring sites, so as not to have an impact on any of the concurrent *D. impar* studies that have been described throughout this thesis.

6.2.3 Survey methods

Surveys for *D. impar* were conducted using a single survey plot of eight pit-fall traps and drift-fences (as described in Chapter 2), per treatment plot, for all treatments and replicates. Surveys were conducted from 12 November to 23 December 1999 (26 days total) and from 15 November to 19 December 2000 (25 days total), using the trapping and animal processing procedures outlined in Chapter 2.

6.2.4 Statistical analyses

Differences in the number of *D. impar* captured in the unburnt and recently burnt treatments were assessed using an independent samples t-test (Zar, 1996) for data collected in Spring 1999 and Spring 2000. A one-way ANOVA (Zar, 1996), incorporating the burnt two-years-ago data for Spring 2000 was also performed. Capture data for both survey periods met assumptions of normality and homoscedasticity using the Kolmogovor-Smirnov goodness-of-fit (Zar, 1996) and Levene's tests (SPSS Inc., 2000), respectively.

The spatial dispersion of *D. impar* was assessed for each treatment of each year, using the standardised Morisita index of dispersion (Krebs, 1989), as described in Chapter 2.

6.2.5 Population structure

To assess for possible differences in population structure between burn treatments, a Mann-Whitney U-test (Zar, 1996) was performed on *D. impar* snout-vent length data recorded from burnt and unburnt survey plots in Spring 1999. The Kruskal-Wallis independent samples test (Zar, 1996) was performed on *D. impar* snout-vent length data recorded from unburnt, burnt two-years-ago and recently burnt survey plots in Spring 2000.

6.2.6 Sex ratios

The homogeneity of sex ratios was individually assessed for each treatment of each survey period using a heterogeneity chi-square analysis (Zar, 1996). Departures from a 1:1 sex ratio were then assessed using chi-square analysis (Zar, 1996) and mean ratios (Krebs, 1989).

6.2.7 Condition

Given that surveys for *D. impar* in the burn treatments were only conducted in the Spring, when females were potentially gravid, the condition factor was calculated for males only. The condition factor was calculated using the criteria and formula described in Chapter 2, including the removal of hatchlings, juveniles and individuals displaying tail autotomy from the data set. The condition factor of male *D. impar* captured in unburnt survey plots was compared to the condition factor of male *D. impar* captured in recently burnt survey plots in the Spring 1999 survey period, using an independent samples t-test. The condition factor of male *D. impar* captured in unburnt, burnt two-years-ago and recently burnt treatments, was compared using a one-way ANOVA. Assumptions of normality and homoscedasticity were assessed using the Kolmogorov-Smirnov goodness-of-fit and Levene's test, respectively.

6.2.8 Predation

Differences in the level of predation on *D. impar* between burn treatments was assessed using log-likelihood ratio contingency tables (Sokal & Rohlf, 1995), for each survey period. Animals displaying tail breakage (including those with regenerated and recently broken tails) were categorised as having been subject to predation. Animals displaying no evidence of tail breakage were not considered to have been subject to predation. It was not possible to determine whether animals displaying evidence of tail breakage and subsequent regeneration had autotomised before or after the burn event. However, it was assumed that the proportion of animals displaying tail autotomy was the same across all survey plots prior to the burn events and therefore that any differences between recently burnt, burnt two-years-ago and unburnt survey plots were associated with the treatments.

6.3 Results

Immediately following all biomass reduction burns the survey plots were line-inspected for dead animals. Only three individual specimens were located within a single burn treatment area in the Summer 1998/9 burns. Cadavers were not located in any other burn plots.

In total, 140 individual *D. impar* were captured at an overall catch rate of 33.65 *D. impar* per 100 survey plot days in the 1999 survey period and 152 individual *D. impar* were captured at an overall catch rate of 26.44 *D. impar* per 100 survey plot days in the 2000 survey period.

Significantly more *D. impar* were captured in recently burnt survey plots than in unburnt survey plots for the Spring 1999 survey period (t=2.345; df=14; p=0.034) (figure 6.3) and for the Spring 2000 survey period (t=2.258; df=14; p=0.040) . However, no significant difference was observed for the Spring 2000 survey period when comparisons were made between all three treatments (unburnt, burnt two-years-ago and recently burnt) (F=1.505; df=2,21; p=0.245) (figure 6.4).

Generally, trap rates were higher in survey plots which had been burnt recently or burnt 2-years-ago. Overall, recaptures were low and the spatial dispersion of captures was aggregated (table 6.1a & b).

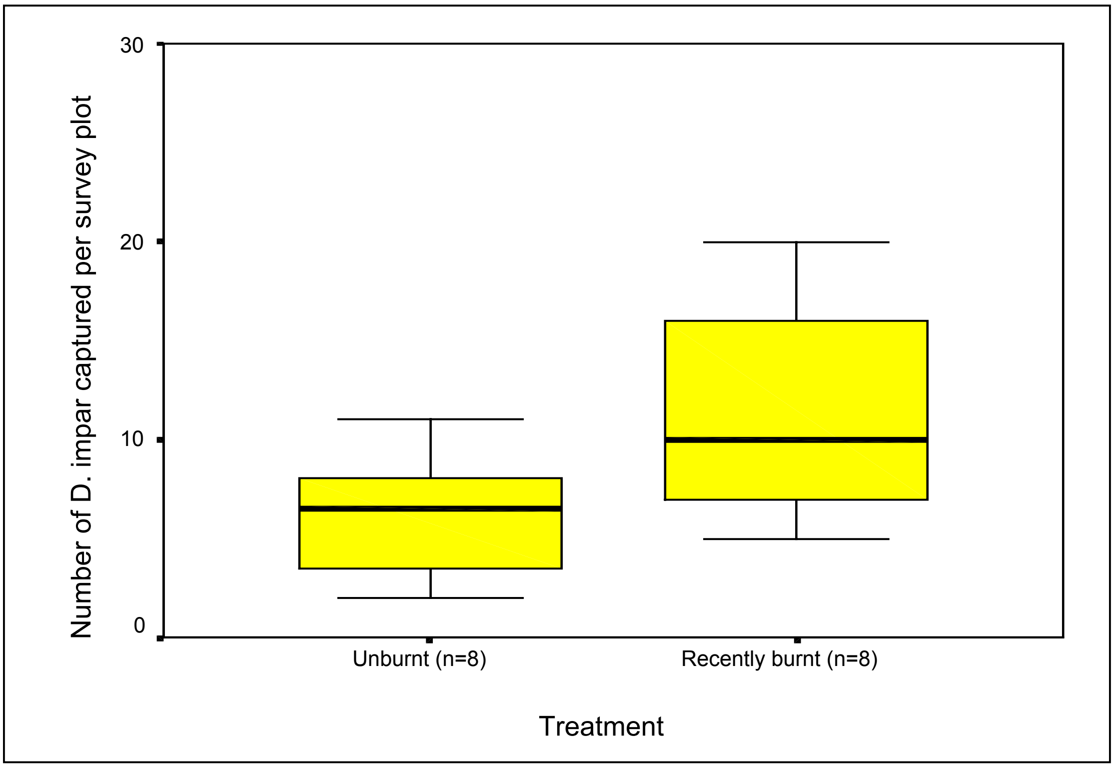


Figure 6.3: The number of *D. impar* captured in unburnt and recently burnt survey plots, Spring 1999. Horizontal bar = median; box = first quartile (lower edge) to third quartile (upper edge); whiskers = range.

	Spring 1999	
	Unburnt	**Recently burnt**
Total individuals captured	49	91
Trap rate (No. *D. impar* per 100 survey plot days)	23.55	43.75
Trap rate (No. *D. impar* per 1000 pit-fall trap days)	2.95	5.47
Recaptures (%total captures)	3 (6.1)	1 (1.1)
Dispersion index	0.200	0.524

Table 6.1a: Survey results for unburnt and recently burnt survey plots, Spring 1999.

	Spring 2000		
	Unburnt	**Burnt 2ya**	**Recently burnt**
Total individuals captured	32	56	63
Trap rate (No. *D. impar* per 100 survey plot days)	16.00	32.00	31.50
Trap rate (No. *D. impar* per 1000 pit-fall trap days)	2.00	4.00	3.94
Recaptures (% total captures)	3 (9.4)	5 (8.9)	4 (6.4)
Dispersion index	0.443	0.520	0.352

Table 6.1b: Survey results for unburnt, burnt 2-years-ago and recently burnt survey plots, Spring 2000.

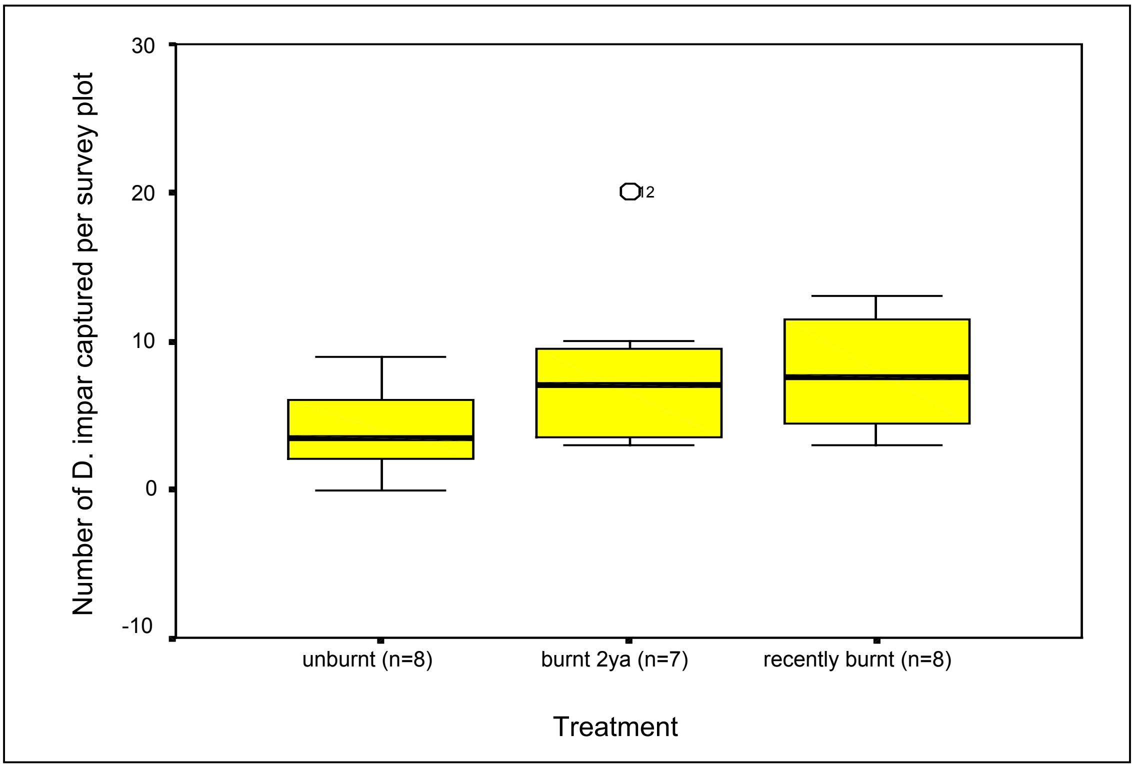


Figure 6.4: The number of *D. impar* captured in unburnt, burnt 2-years-ago and recently burnt survey plots, Spring 2000. Horizontal bar = median; box = first quartile (lower edge) to third quartile (upper edge); whiskers = range; circle = possible outlier.

6.3.1 Population structure

No indication of differences in the *D. impar* population structure between burn treatments was detected for the Spring 1999 survey period (U=1947.0; n=48,85; p=0.663) or the Spring 2000 survey period (*H*=1.88; n=30,54,63; p=0.910), based on comparisons of snout-vent lengths. Generally, 50 percent of individuals captured were distributed around the 85 millimetre size class, for all treatments except recently burnt survey plots for the Spring 1999 survey period, where the distribution of 50 percent of individuals captured was spread over a greater size class range (figures 6.5a & b; figures 6.6a & b).

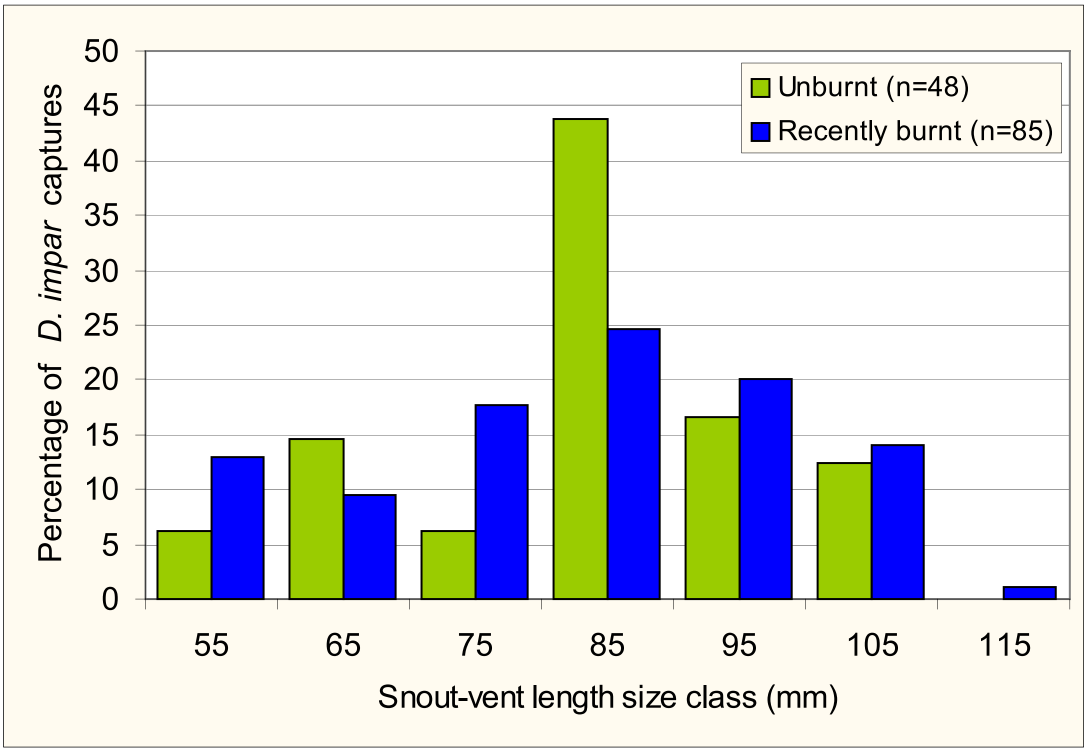


Figure 6.5a: The distribution of size classes of *D. impar* captured in unburnt (n=48) and recently burnt (n=85) treatments, Spring 1999.

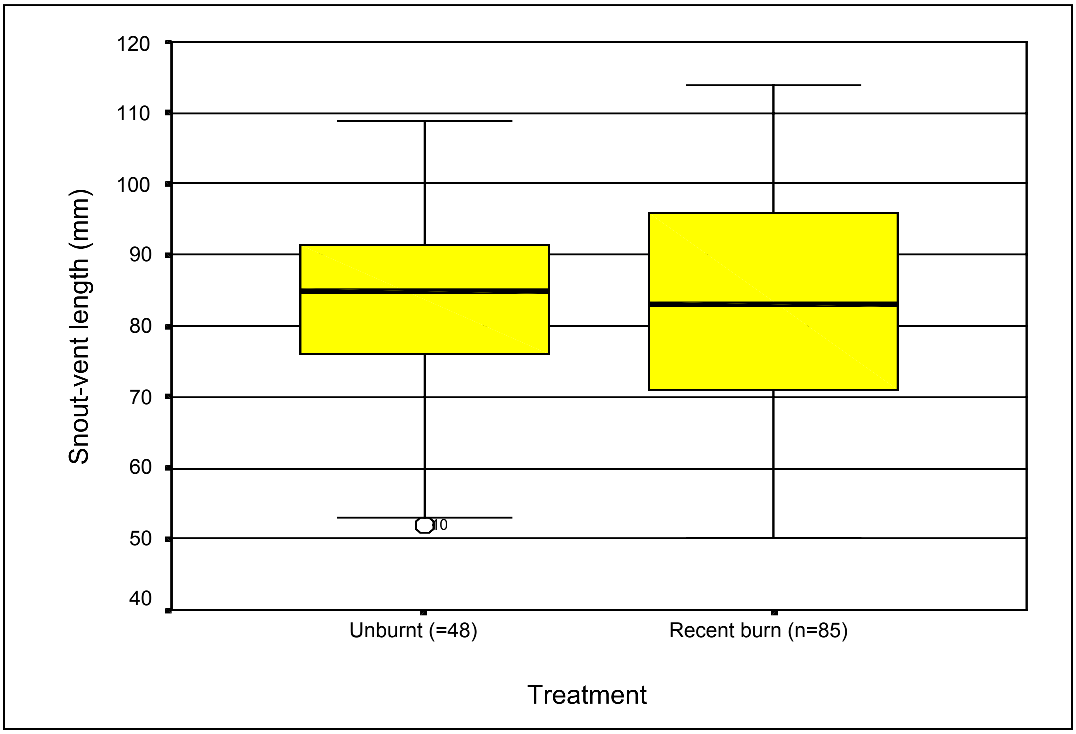


Figure 6.5b: The distribution of *D. impar* snout-vent lengths for unburnt and recently burnt treatments, Spring 1999. Horizontal bar = median; box = first quartile (lower edge) to third quartile (upper edge); whiskers = range; circle = possible outlier.

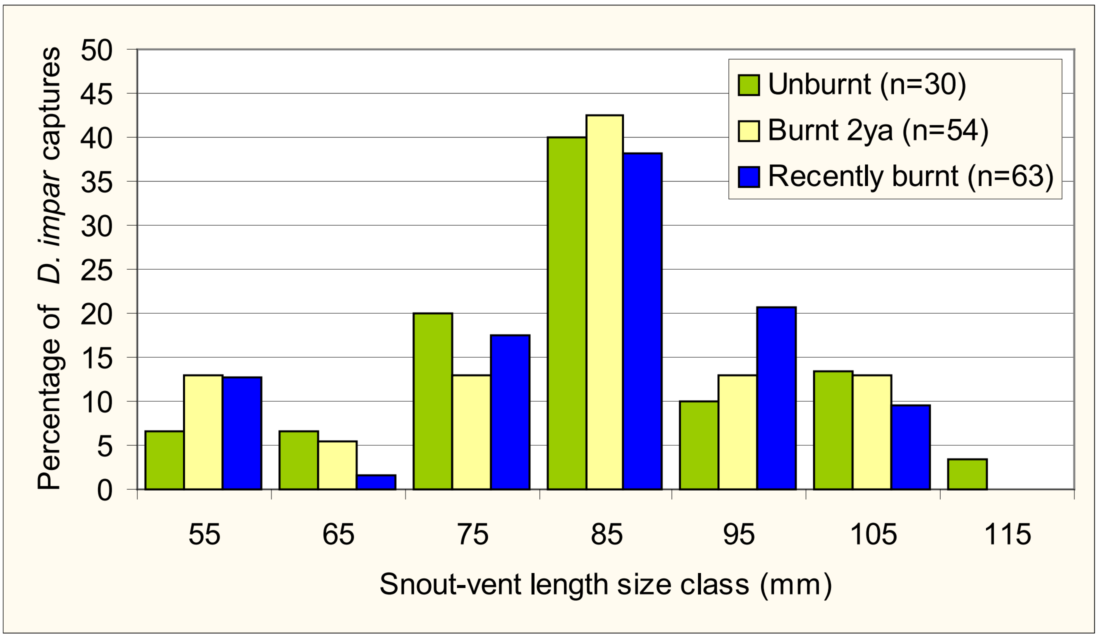


Figure 6.6a: The distribution of size classes of *D. impar* captured in unburnt (n=30), burnt 2-years-ago (n=54) and recently burnt (n=63) treatments, Spring 2000.

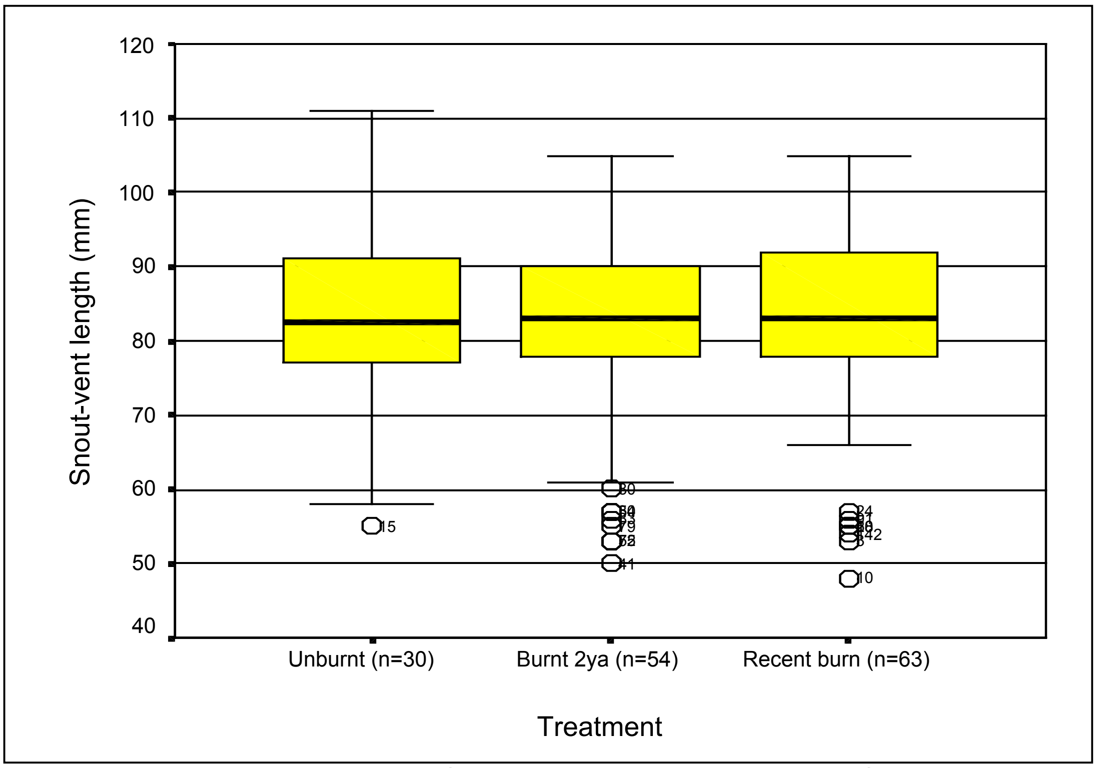


Figure 6.6b: The distribution of *D. impar* snout-vent lengths for unburnt, burnt 2-years-ago and recently burnt treatments, Spring 2000. Horizontal bar = median; box = first quartile (lower edge) to third quartile (upper edge); whiskers = range; circle = possible outlier.

6.3.2 Sex ratios

Sex ratios of *D. impar* were found to be homogeneous for all survey plots in Spring 1999 (χ^2=15.285; df=15; p=0.431) (81.3 percent of cells had expected values less than five) and Spring 2000 (χ^2=22.501; df=21; p=0.371) (84.1 percent of cells had expected values less than five). No significant departures from an even sex ratio were detected for either treatment in Spring 1999. However, all treatments in Spring 2000 displayed significant departures from an even sex ratio (table 6.2).

As in previous chapters where chi-square analysis has been used to assess sex ratios, the proportion of cells with expected frequency values less than five is greater than 20 percent, thus increasing the chances of making a Type II error. The mean ratio offers an alternative perspective on the sex ratios.

Although the mean ratio of female to male *D. impar* was only slightly greater than one in Spring 1999, the smaller mean ratio for Unburnt treatments and the much greater mean ratio for Recently burnt treatments (table 6.2; figure 6.7a) suggests that there is indeed a possibility that the null hypothesis was incorrectly accepted in the chi-square analyses for this survey period.

Overall, in Spring 2000, the mean ratio suggests that approximately only half the number of females were captured in comparison to the number of males captured (table 6.2; figure 6.7b). This trend was consistent for all three treatments in Spring 2000 and suggests that the null hypothesis was probably correctly rejected in the chi-square analyses for this survey period.

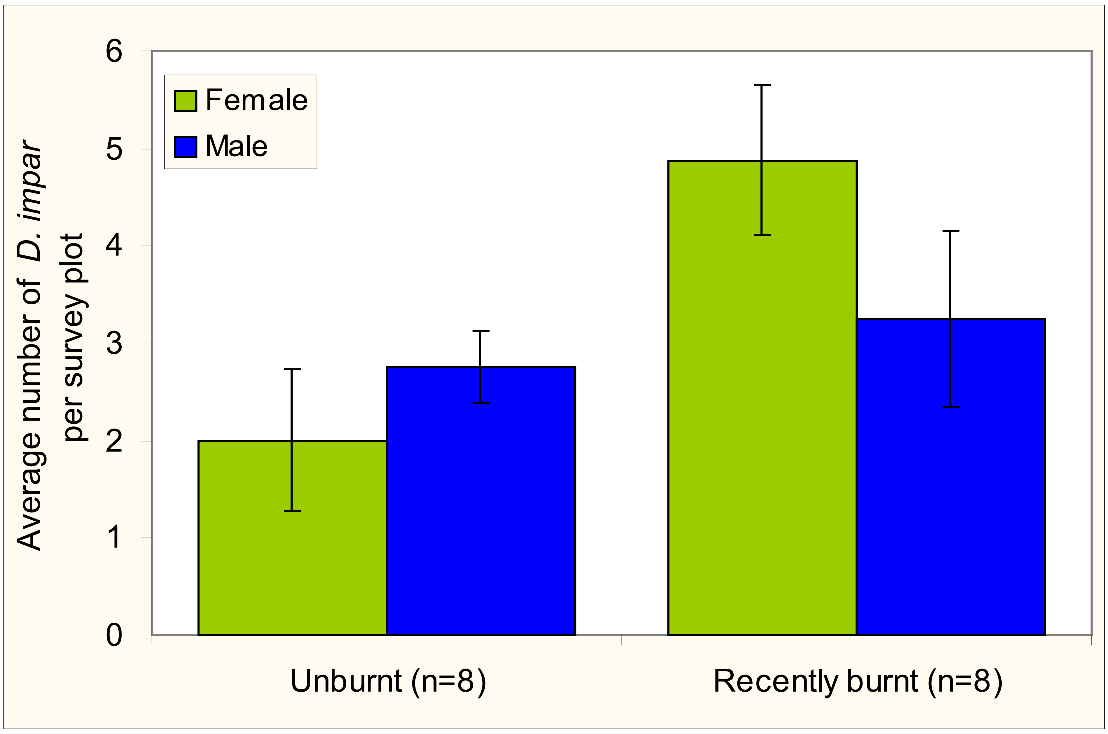


Figure 6.7a: The average number of female to male *D. impar* captured per survey plot for unburnt and recently burnt treatments, Spring 1999.

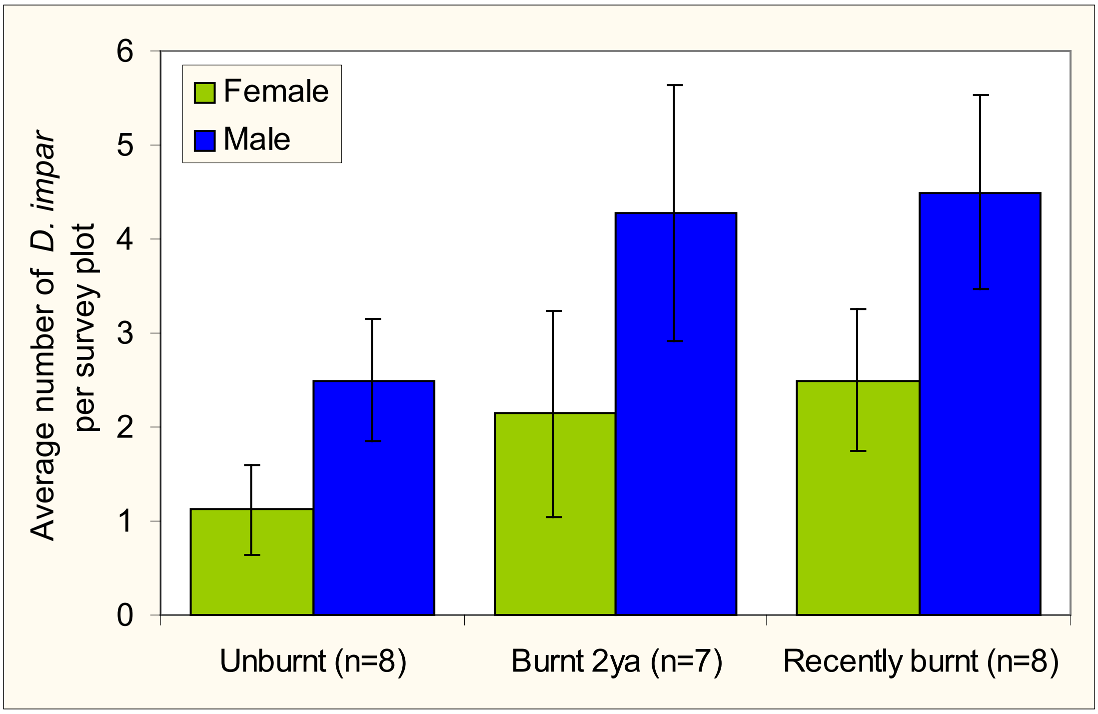


Figure 6.7b: The average number of female to male *D. impar* captured per survey plot for unburnt, burnt 2-years-ago and recently burnt treatments, Spring 2000.

Treatment	**Heterogeneity chi-square**	**1:1 chi-square**	**Mean ratio female/male**
Spring 1999			1.15
Unburnt 1999	χ^2=7.980; df=7; p=0.334	χ^2=0.947; df=1; p=0.330	0.73
Recently burnt 1999	χ^2=4.544; df=7; p=0.715	χ^2=2.600; df=1; p=0.107	1.50
Spring 2000			0.51
Unburnt 2000	χ^2=5.528; df=6; p=0.478	χ^2=4.172; df=1; p=0.041	0.45
Burnt 2ya 2000	χ^2=7.444; df=6; p=0.282	χ^2=5.000; df=1; p=0.025	0.50
Recently burnt 2000	χ^2=9.396; df=7; p=0.225	χ^2=4.571; df=1; p=0.033	0.56

Table 6.2: Results of *D. impar* sex ratio analyses for all treatments and survey periods.

6.3.3 Condition

Despite an apparent difference suggested by the averages and error bars (figure 6.8a), no significant difference was observed in the condition factor of male *D. impar* captured in the burn treatments for the Spring 1999 survey period (t=1.058; df=27; p=0.300) or in the burn treatments for the Spring 2000 survey period (F=2.075; df=2,51; p=0.136) (figure 6.8b).

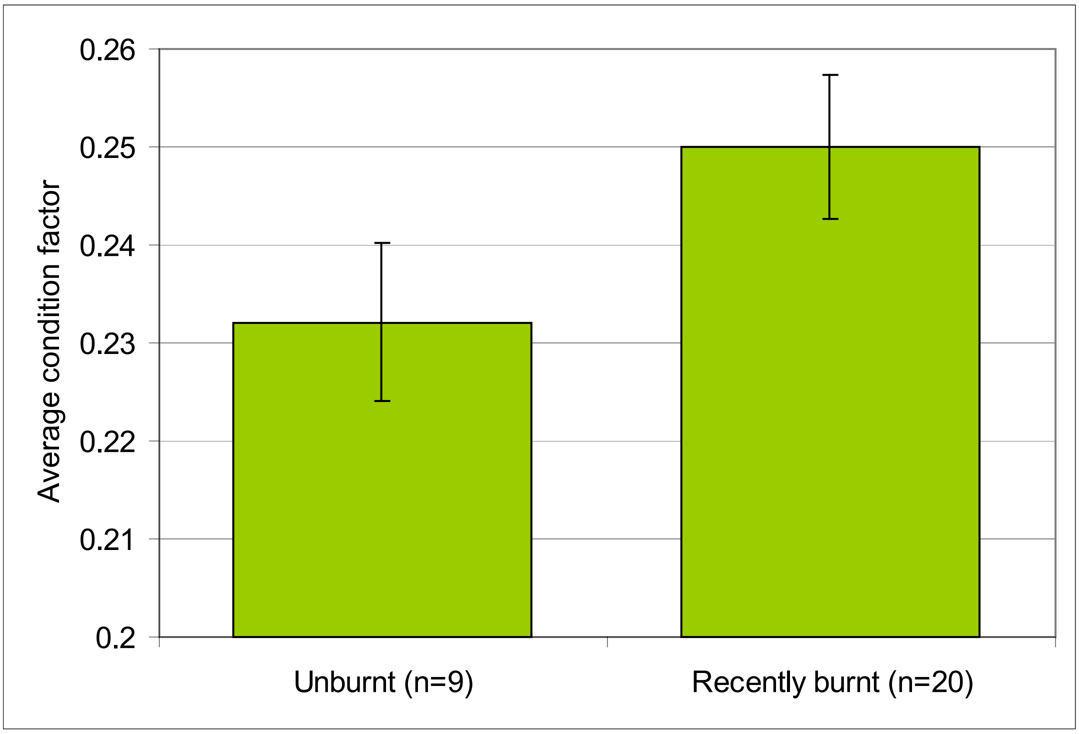


Figure 6.8a: The condition factor of male *D. impar* captured in unburnt and recently burnt survey plots, Spring 1999.

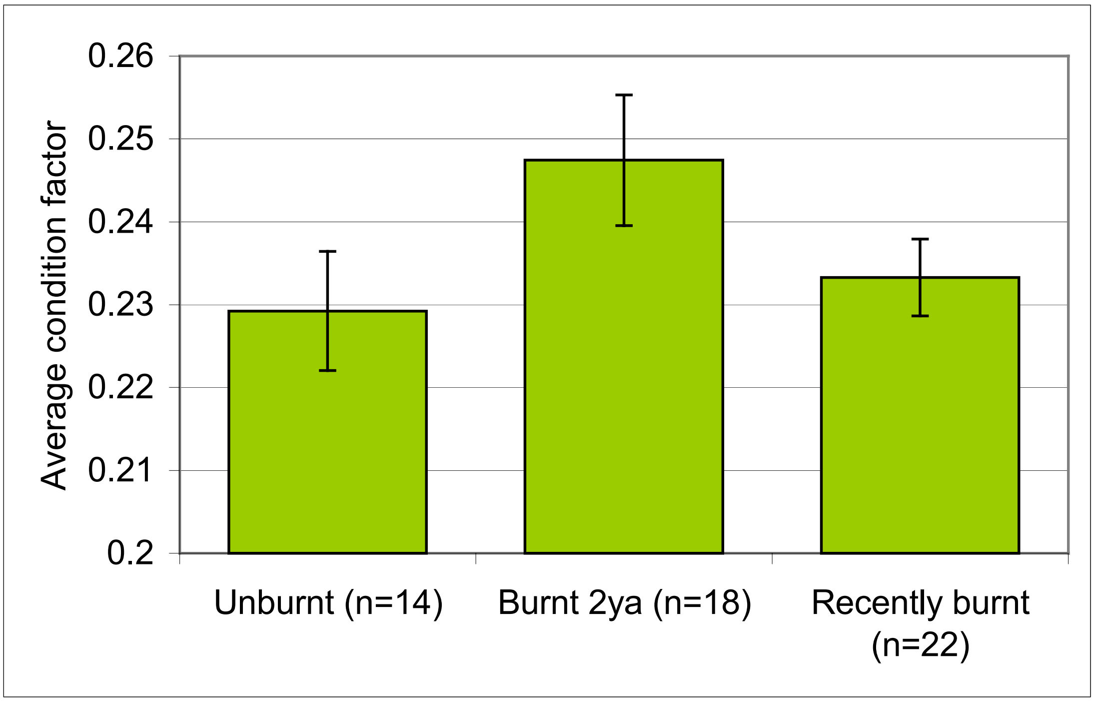


Figure 6.8b: The condition factor of male *D. impar* captured in unburnt, burnt 2-years-ago and recently burnt survey plots, Spring 2000.

6.3.4 Predation

The level of predation on *D. impar*, as determined by the proportion of tail breakages to unbroken tails, was significantly different between unburnt and recently burnt treatments in Spring 1999 (*G*=3.945; df=1; p=0.047) (figure 6.9a). In contrast, the level of predation on *D. impar* was relatively consistent between burn treatments for the Spring 2000 survey period (*G*=0.870; df=1; p=0.647), with approximately one third of animals in each treatment displaying tail autotomy (figure 6.9b).

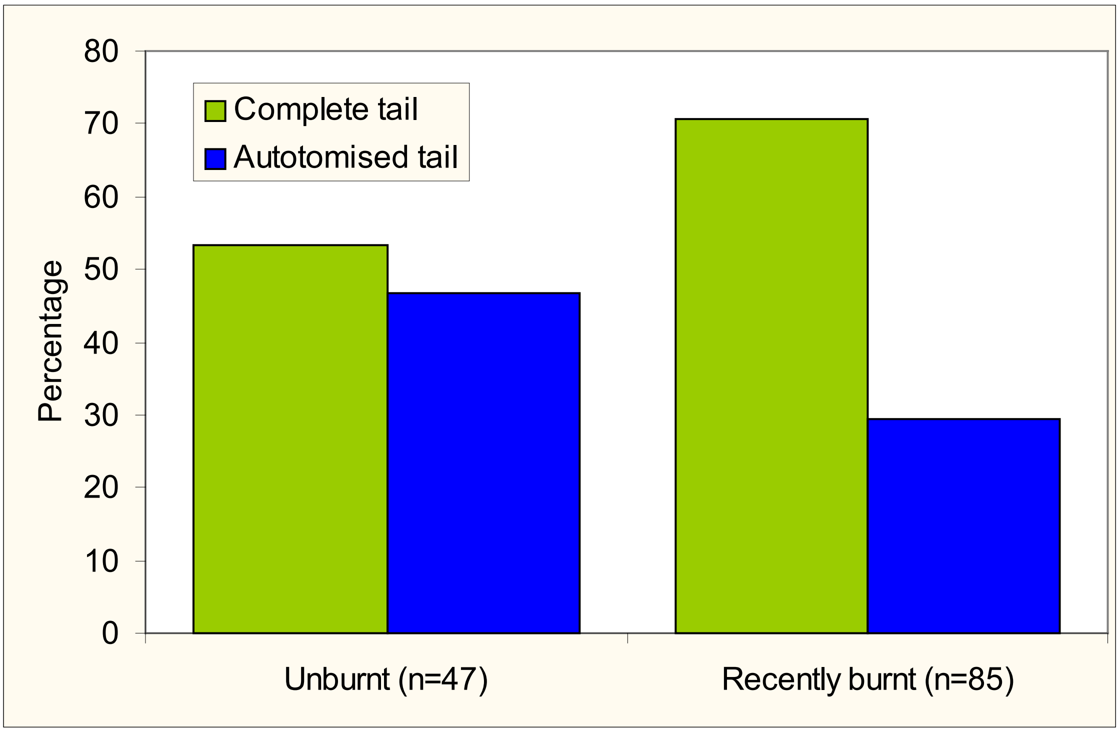


Figure 6.9a: The proportion of *D. impar* displaying unbroken and autotomised tails in unburnt and recently burnt treatments, Spring 1999.

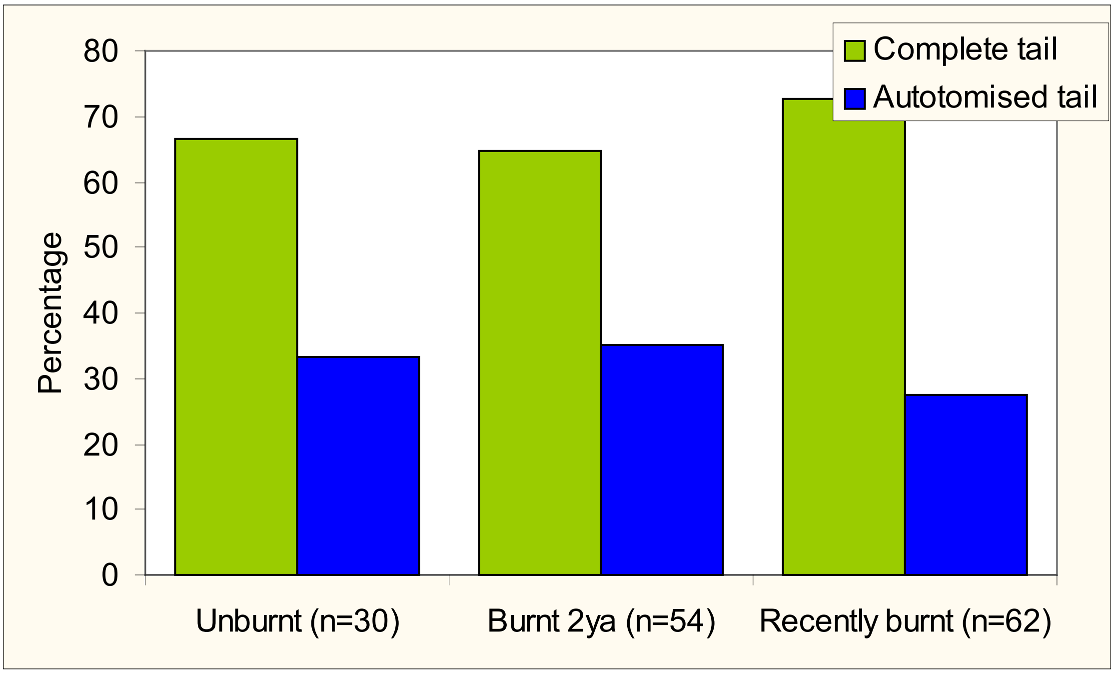


Figure 6.9b: The proportion of *D. impar* displaying complete and autotomised tails in unburnt, burnt 2-years-ago and recently burnt treatments, Spring 2000.

6.4 Discussion

Apart from the detection of three dead *D. impar* in a single burn replicate, this study makes no assessment of the acute impacts of fire on the species and its immediate post-fire survivability. However, assessments of the relative abundance of the species in the various burn treatments at periods greater than seven months following biomass reduction burns suggests that small scale Summer and Autumn burns do not render the habitat unsuitable for the species.

For both the Spring 1999 and Spring 2000 survey periods, the relative abundance of *D. impar* in recently burnt treatments was significantly higher than the relative abundance in unburnt habitat. This may either be a true reflection of the relative abundance of the species in each of the habitat treatment types or it may suggest that *D. impar* is more trappable in burnt habitats. In recently burnt areas *D. impar* may be more readily trapped for several reasons:

- lack of vertical structure forces animals to travel closer to the soil substrate and are therefore more likely to be intercepted by the drift-fence and thus trapped;
- lack of vertical structure around the entrance to the pit-fall trap provides less opportunity for animals to pull back into the grass once they have identified the trap. Thus they fall into the pit-falls more easily;
- due to the lack of vegetation cover animals may be moving across the terrain more rapidly and therefore are less likely to notice and avoid the pit-fall traps.

However, if small scale biomass reduction burns were to have a deleterious impact on the relative abundance of *D. impar*, then it would be expected that numbers would be significantly fewer in the burn treatment survey plots. In a study on the impacts of prescribed fires on reptile populations in a forest environment in Victoria, Humphries (1992) recorded an increase in the abundance of Coventry's Skink *Niveoscincus coventryi* and McCoy's Skink *Nannoscincus maccoyi* immediately following fire but found that the abundance of both species had declined five months later and remained low until the litter layer had recovered 28 months later, concluding that the initial high abundance may have been due to increased detectability.

If small scale Summer and Autumn biomass reduction burns were to have a major deleterious impact on *D. impar* abundance, it is probable that this would have been detected by comparing results across the time frames of seven months and 22 months following treatment. That is, sufficient time had elapsed to enable individuals which had survived the acute impact to subsequently either flee the burnt areas, be subject to high and deleterious levels of predation or undergo other forms of demise as a result of loss of habitat or food resources. Fox (1978) found that although large number of skinks survived the acute impacts of fire, many had died of starvation or predation in the weeks following. No evidence for any of these scenarios was observed for *D. impar* in the current study. However, as Whelan *et al.* (2002) pointed out, it is important to know whether individuals appearing after a fire are colonists from outside the burned area or previously occurring individuals simply emerging from refugia.

Although the timing of burn treatments was not a focus of this study, Friend (1993) suggested that outcomes for reptiles that survive the acute impacts of fire may be influenced by the season of the burn, in that Autumn fires would allow the Winter and early Spring period to ameliorate environmental conditions whilst animals were in a state of brumation and thus less subject to the forces of predation, reduced vegetation cover and potentially reduced food availabilty. Thus, *D. impar* may have survived the short-term impacts of small scale biomass reduction burns in such a manner, in the current study, by remaining relatively sedentary in the period between burn events and entry into brummation. Vegetation in burnt areas had resprouted by the time *D. impar* emerged from brummation in early Spring.

These results also add support to the suggestions of Dorrough (1995) and Hadden (1995; 1998) that habitat cover is not a critical factor for *D. impar* presence and abundance.

In addition to small scale biomass reduction burns having no apparent deleterious impact on *D. impar* abundance, there is no evidence that such treatments have deleterious impacts on any of the other population characteristics assessed. Based on the statistical analyses performed on population structure data, no significant difference was found in the proportion of size classes for each of the treatments, even though there appears to be a more even distribution of individuals across the size classes in the burnt treatment for the Spring 1999 survey period. The distribution of size classes in survey plots burnt prior to Spring 1999 more closely reflected the distributions for unburnt and recently burnt survey plots, when monitored (as burnt two-years-ago) in the Spring 2000 survey period, suggesting that even if burning had had an impact on some size classes of the population in 1999, the effects were not long-lasting and were not repeated in the Spring 2000 survey period.

Modeling at the Population and Habitat Viability Assessment Workshop (ARAZPA, 1996) predicted that burning would have a greater impact on reproductive output than on adult survivorship. Although reproductive output was not assessed in this study, it is encouraging that the 55 millimetre size class (ie. first year juveniles) was represented in all treatments. Indeed, the number of juveniles in this size class appears to be greater in burnt treatments than in unburnt treatments, although this was not tested statistically due to low sample sizes. Theoretically, this could mean that small scale biomass reduction burns do not have a negative impact on reproductive output or alternatively that juvenile animals may be forced into burnt areas. To date there is no evidence for differential use of habitat by different sizes classes of *D. impar*, adding weight to the idea that at Iramoo Wildlife Reserve small scale Summer and Autumn biomass reduction burns have not had a negative impact on reproductive output. It is even possible that recently burnt habitat may be preferable for *D. impar*.

No consistent trend in sex ratios was detected for burn treatments and it seems more likely that variations in observed ratios are a result of seasonal influences and timing of survey periods than habitat characteristics.

At the Population and Habitat Viability Assessment Workshop (ARAZPA, 1996), it was predicted that fire would lead to population decline as a result of indirect mortality due to decreased food availability as a result of habitat loss. In this study, no significant difference was observed in the condition of mature male *D. impar* for any of the habitat treatments, suggesting that food availability in burnt areas is not a deleterious factor for this component of the population at least. The availability of food resources following biomass reduction burns requires further investigation.

Reptiles display a diverse range of responses to fire and there is no consistent relationship between time-since-fire (successional stage) and reproductive and dietary patterns across reptile species (Friend, 1993). Although Bamford (1986) concluded that the effect of fire overall on reptiles was negligible, it seems that factors influencing a species success in environments after fire are dependent on their particular habitat requirements and foraging strategies. Fyfe (1980), Caughley (1985) and Lunney *et al.* (1991) found that fossorial refuge-seeking, burrowing and cryptic species and species which prefered open ground for foraging were more abundant in areas which had been most recently burnt. Perhaps *D. impar's* semi-fossorial habit also fits into this model, thus making it a fire tolerant species, at least under the conditions encountered in this study at Iramoo Wildlife Reserve. Whelan *et al.* (2002) suggested that a species plasticity in resource requirements could indicate a lower susceptibility to fire. *Delma impar* demonstrates some degree of plasticity in terms of grassland vegetation requirements (Dorrough, 1995; Hadden, 1995) and also in terms of food resource utilisation (Nunan, 1995; Kutt *et al.,* 1998; O'Shea & Hocking, 2000). Further investigation into the impacts on foraging and food resource utilisation following fire are required for this species.

6.5 Conclusion

On the basis of the parameters assessed in this study, it appears that small scale, mosaic-type Autumn burns, interspersed across large areas of unburnt habitat do not have an obvious acute impact on *D. impar* or a dramatic negative impact on the relative abundance, population characteristics or apparent habitat quality for the species.

On this basis and given the findings of Morgan and Lunt (1999) that the number and health of *T. triandra* tussocks in Western Basalt Plains Grassland remnants declines with time-since-fire, it is a reasonable and recommended action that remnants supporting dense senescing *T. triandra* vegetation be burnt at the recommended regime of at least once every fire years. Small scale burning in all remnants should be conducted in a mosaic-type style, allowing for the movement of *D. impar* and prey species between habitat types and larger areas of unburnt habitat. Small scale biomass reductions burns should be conducted in the drier months of the year, when opened soil cracks will provide refuge from the fire. This works suggests that burning relatively small patches of habitat (0.5 hectares) is a reasonable course of action.

Future research should use alternative methods of detection, such as the roof tile monitoring method described in Chapter 3, in an attempt to resolve whether biomass reduction burns lead to any change in the relative abundance of *D. impar* (either an increase or decrease) or whether trappability is altered as a result of structural changes in habitat.

In addition, Braithwaite (1987) found that the survival and reproduction of a range of lizard species varied between the different types of habitats created by different sorts of fires, as a result of seasonal timing and frequency, in the wet-dry tropics of Australia. The impact of factors such as frequency, timing, intensity and scale should be investigated as part of an adaptive management programme for *D. impar* habitat.

Chapter 7 Conclusion

Chapter 7

Conclusion

The Striped Legless Lizard *Delma impar* population at Iramoo Wildlife Reserve, St. Albans Victoria, is large and provides opportunity to conduct intensive, scientifically designed studies and experiments on the species' biology and ecology, with the intention of increasing our understanding and thus improving efforts at conserving the species.

In this study three methods of survey were utilised to assess the *D. impar* population at Iramoo Wildlife Reserve. These were Summer trapping with pit-fall traps and drift-fences, Spring trapping with pit-fall traps and drift-fences and roof tile monitoring. Each method has its advantages and disadvantages.

Pit-fall trapping with drift-fences, in general, is time consuming, expensive, labour intensive and potentially destructive to grassland habitat. Conventionally, results of various pit-fall trapping studies have been compared in terms of the resultant trap rate success. It is clear from this study and others (see Rauhala, 1997) that trap rate success is not a reliable indicator of fluctuations in populations and may vary in response to seasonal and habitat conditions, with very little relevance to real fluctuations in *D. impar* populations. Furthermore, trap rates are influenced by the duration of trapping, with shorter trapping programmes more likely to record higher trap rates.

Pit-fall trapping in general is a useful means for making comparisons on the effects of habitat treatments (eg. burnt and unburnt; native and exotic) at a given site. Basic data relating to relative densities, population structure, sex ratios, animal condition and variations in levels of predation can be obtained using this survey method. However, care should be taken with interpreting data, especially in relation to sex ratios where it is difficult to collect sufficient data to adequately meet the requirements of statistical procedures such as the chi-square test. Recapture rates are very low using this method of monitoring.

Pit-fall trapping with drift-fences in Spring survey periods yield a large number of *D. impar* over a shorter trapping duration, in comparison to Summer trapping. However, data collected in Spring provides no information on hatchlings and condition data should be collected for only males at this time of year, as it is difficult to interpret such data for gravid females.

Roof tiles provide a cheap, easy and less destructive option for monitoring *D. impar* populations, in comparison to pit-fall trapping with drift-fences. Recapture rates under roof tiles are high and it is foreseeable that this method of monitoring may provide more reliable estimates of the size of *D. impar* populations. Further work in this area is required. Roof tile monitoring should be conducted between the months of September and January, with best results obtained in the months September and October, when the temperatures under the tiles range from mid-20^{o}C to high-30^{o}C. As with Spring pit-fall trapping, no information on hatchlings can be obtained using roof tiles, and juvenile (i.e. individuals less than 70 millimetres snout-vent length) *D. impar* also seem poorly represented in data sets collected using this method. Care should be taken when interpreting sex ratio data for *D. impar* recorded under roof tiles, as males are more prevalent in the months of August, September and October, with females becoming predominant in November and December. Because roof tile monitoring is most successful in Spring, care should also be taken when making comparisons on the condition of animals. That is, it would not be appropriate to compare the condition of *D. impar* recorded under roof tiles, with the condition of *D. impar* recorded in pit-fall traps in Summer/Autumn.

Based on pit-fall trapping with drift-fences and roof tile monitoring, the calculations of the *D. impar* population size at Iramoo Widlife Reserve are considered to be underestimates but suggest that the order of magnitude is in the thousands of individuals, across the 37.5 hectare reserve. Densities were estimated at 16 *D. impar* per hectare based on pit-fall trapping with drift-fences in Summer survey periods and 91 *D. impar* per hectare based on roof tile monitoring. The structure of the *D. impar* population at Iramoo Wildlife Reserve appears to fluctuate from one active season to another but generally mature individuals account for over 50 percent of all animals recorded, with the median snout-vent length being greater than 75 millimetres. Methods for assessing the fate of hatchling cohorts over time need to be devised.

The data arising from the studies in this thesis suggest that the ratio of female to male *D. impar* might be slightly below parity, as indicated by mean ratio values. Sex ratios biased in favour of males have been reported for some Varanid lizards (Greer, 1989) and snakes (Shine, 1988). However, as both Greer (1989) and Shine (1988) point out, such biases may, in some instances, be attributed to differential levels of activity between the sexes, with more active individuals (usually males) being more frequently observed. This may also be the case for *D. impar* and requires further investigation.

The chi-square statistic provides an opportunity to hypothesis test a given sex ratio (for example 1:1), however this requires that a sufficient number of animals are captured in each sample so that assumptions of the test are not violated. This was not achieved in any of the experiments conducted in this study. Future investigations into *D. impar* sex ratios should attempt to increase the sample size, perhaps by increasing the survey area of each survey plot, which will enable the installation of a greater number of traps per survey plot.

The use of the condition factor for comparing the relative health of different *D. impar* populations needs to be further assessed. The condition factor is not useful for assessing hatchlings, juveniles and individuals which have autotomised their tail. It is also limited to the assessment of males in Spring survey periods but is useful for assessing females after the egg deposition period. The condition of individual *D. impar* meeting the assessment criteria at Iramoo Wildlife Reserve ranged from 0.162 to 0.489, with the average condition factor in Spring being 0.233 (±0.024) and the average condition factor in Summer being 0.280 (±0.040). These figures could form the baseline for assessing other *D. impar* populations.

The level or success of predation on *D. impar* at Iramoo Wildlife Reserve appears to vary between mature and immature individuals and also over time. Generally, greater than 30 percent of all *D. impar* captured displayed evidence of tail autotomy. It seems most likely that changes in the proportion of individuals with tail autotomy are a result of changes in levels of predation, rather than predator efficiency, if it is assumed that the types of predators has not changed throughout the study and that there has been no change in circumstances which would enable predators to improve their efficiency (ie. between habitat types such as native/exotic or burnt/unburnt). This seems a reasonable assumption, given that no difference in the proportion of tail autotomies was observed between native and exotic habitat and that no difference was found for the three habitat burn treatments in 2000. Thus, it appears that changes in proportions of tail autotomy are due to changes in the level of predation over time, however, it is not possible to determine in which direction these changes occur.

To date there has been little understanding of the impacts of habitat management on *D. impar* populations and managers have been reticent to implement programmes that may negatively impact on the species, despite current knowledge on ecosystem processes such as senescence in Kangaroo Grass *Themeda triandra* dominated temperate grasslands. This study shows that small (0.5 hectare) mosaic type Summer/Autumns biomass reduction burns interspersed between larger areas of unburnt habitat, do not have a deleterious impact on the species in terms of relative abundance, population structure and sex ratios or individual condition. It was not possible to determine the impact of potential changes in predation levels in burnt and unburnt areas, although given that there is no evidence for a decline in *D.*

impar densities or population structures in burnt habitat, it appears that any changes in levels of predation are not significant. Although no significant difference was recorded in the condition of *D. impar* in unburnt and burnt habitat, work should be conducted into the impacts of such habitat management procedures on the species' food requirements. Further work is also required in relation to particular fire regimes, in terms of seasonal timing frequency and intensity.

In order to further advance our understanding of *D. impar* biology and ecology and therefore improve our efforts at conserving the species, it is necessary to collect long-term data on individuals in wild populations. To date this has been very limited, due to a paucity of recapture data. The harmonic direction-finder provides an opportunity for collection of this information but requires further development in the areas of tag robustness, tag implantation techniques and transceiver design. The use of the head scale pattern mapping individual identification technique may also provide further opportunity to monitor individual *D. impar* over longer periods of time, if it is found that the traditional pyrobranding techniques contribute to the poor recapture rates for this species. The head scale pattern mapping technique is considered to be a more humane method for identifying individual *D. impar*, although initial set-up costs and labour requirements are higher than for pyrobranding. Further development could make the head scale pattern mapping individual identification technique user friendly.

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