The effects of introduced predators and the invasive weed *Tradescantia fluminensis* (Vell.) (Commelinaceae) on the land snail *Powelliphanta traversi traversi* (Powell) (Gastropoda: Pulmonata: Rhytididae).

A Thesis presented in partial fulfillment of the requirements for the degree of Master of Science in Ecology at Massey University.

Shaun James Bennett

2001

ERRATA

Abstract

p.ii, lines 3-4:

Rodent control was carried out continuously rather than using a single application of poison. Bait stations were topped up with Brodifacoum poison every 21-42 days over a period of 19 months.

p.ii, lines 12-13

Numbers of empty *Powelliphanta traversi traversi* shells found in each area decreased by approximately half when the interval between samples was reduced from 12 to 7 months (i.e. by approximately half). This suggested that the accumulation of shells in each area was related linearly to time rather than being strongly affected by any seasonal variation.

Chapter 3

p.23, lines 6-11

There were certain constraints in selecting suitable areas for this experiment, which lead to constraints in the experimental design. Poisoning had to be done over areas >200m across in order to successfully reduce rodent numbers and prevent re-invasion in the centres of the areas. Working on this assumption, only four large areas were available at Lake Papaitonga Scenic Reserve, and this resulted in only two replicates being possible.

p.25, lines 10-11

Permanent quadrats were used throughout the study, instead of a number of randomly chosen quadrats each time, to obtain mortality data by removing empty shells from these quadrats when they were searched. This was essential in determining the effect of reducing rat numbers in the areas that were poisoned. Consequently, a repeated measures design was used. The size of the quadrats ($100m^2$) ensured that sufficient numbers of snails and empty shells could be found, and were used as a compromise between a single larger quadrat and many smaller quadrats. It was also hoped that the quadrats established by this study would be used for further monitoring of *P. t. traversi*.

Chapter 4

p.51, lines 22-28

The main aim of the research was to determine what might happen to *P. t. traversi* in areas affected by *Tradescantia fluminensis* if this weed is removed. To answer this question, the use of *T. fluminensis* affected habitat by *P. t. traversi* was assessed, taking into account such factors as snail density in *T. fluminensis* affected habitat and native habitat, the size of the snails in each habitat type, and the effects of habitat type on snail movements. Ultimately, *T. fluminensis* removal experiments will need to be conducted to determine actual effects to *P. t. traversi*.

Chapter 5

p.82, lines 1-5

The effect of Grazon herbicide on *Cantareus aspersus* was tested further using a Chi-square test to compare mortality of the different treatments at 149 days after spraying. This confirmed a highly significant difference between treatments overall (χ^2 value = 76.48; df =2; P<0.001). It also showed a slightly non-significant interaction effect between treatment and age class (χ^2 value = 24.98; df =2; P=0.057).

Abstract

Powelliphanta traversi traversi (Powell) was studied at two forest remnants in the Horowhenua District. The effects of introduced predators, predator control, the invasive weed *Tradescantia fluminensis* (Vell.), and Grazon® herbicide on these snails were investigated. Brodifacoum poison was used in two areas of Lake Papaitonga Scenic Reserve to determine the effect of rodent control on *P. t. traversi*. Mouse abundance (inferred from tracking tunnel indices) was reduced in both poisoned areas below levels observed in two other areas that were not poisoned. Rat abundance was reduced below pre-poisoning levels but only to levels below one of the non-poison areas. In each poison and non-poison area, four 100m² quadrats were searched for *P. t. traversi* snails immediately before poisoning, and 12 and 19 months after poisoning commenced. After 19 months, only one poisoned area showed an overall increase in the number of snails, with significantly more live snails found (45) than at either of the two previous searches (22 before poisoning and 28 after 12 months of poisoning) ($P<0.05$). Numbers of empty *P. t. traversi* shells found in each area decreased at each search suggesting that shell accumulation is constant rather than seasonal. Rats were the greatest identified predator of *P. t. traversi* at Lake Papaitonga (17.87% of all empty shells), but the proportion of shells damaged by blackbirds and song thrushes was also high (11.91% of all empty shells) and increased from pre-poisoning numbers in three of the areas. Overall, there was no conclusive evidence to suggest that the numbers of live *P. t. traversi* increased as a result of rodent poisoning during the time period of this study. The effect of *T. fluminensis* on the movements of *P. t. traversi* at Prouse Bush was determined using harmonic radar. There was large variation in the movements and a highly significant difference between individual snails ($P<0.01$), with some snails regularly moving between areas of *T. fluminensis* and leaf litter. There was no significant difference in the mean daily displacement of movements by snails in leaf litter and *T. fluminensis*, but *T. fluminensis* did appear to affect home range size. Snails always found under *T. fluminensis* had significantly smaller mean 90% home range estimates (43.91 m²) than snails that were only ever found in leaf litter or those that moved between litter and *T. fluminensis* (171.35 m² and 610.14 m² respectively) ($P<0.05$). Snails in *T. fluminensis* had a significantly wider size-frequency distribution than those in leaf litter ($P<0.05$) and no live snails <35mm were found in leaf litter. There was no significant difference between the size-frequency distributions of empty shells found in both habitats, but their density was significantly greater in leaf litter ($P<0.05$). *Powelliphanta traversi traversi* regularly use *T. fluminensis* as a habitat and any control measures affecting this weed in native bush remnants need to be considered with regard to their possible effects on these snails. The toxicity of a 1.4% Grazon® solution (active ingredient triclopyr) to *P. t. traversi* was investigated by first using three life history stages of the brown garden snail (*Cantareus aspersus* Müller). After 149 days, there was significantly greater mean mortality of *C. aspersus* exposed to a direct spray and

to a sprayed environment (82.34% and 78.40% respectively) than in a control treatment (36.95%) ($P<0.05$). *Cantareus aspersus* egg mortality (86.00%) was significantly greater than adult and juvenile snail mortality (66.86 and 62.20% respectively) ($P<0.05$). Five *P. t. traversi* snails were also exposed to a single environmental spray of a 1.4% Grazon solution but no mortality or detrimental effects were observed after 149 days. A 1.4% Grazon solution does not appear to be toxic to *P. t. traversi* snails when sprayed on leaf litter where the snails live so Grazon appears to be a suitable herbicide for controlling *T. fluminensis* in forest remnants containing *P. t. traversi*.

Acknowledgements

As I am writing this, it must all be coming to an end. It has been a long journey getting here, but thanks to the help of so many people I have made it through. Because so many people helped me, in so many different ways, the following is a list of those I can remember to whom I am greatly indebted. I apologize to anyone I have failed to include and assure you that your help was greatly appreciated and did not go unnoticed.

I must begin by mentioning the special group of people that constitute the Ecology Group at Massey University. Of special note must be Cathy Lake, Paul Barrett, Hayden Hewitt, Hamish Mack, and Tracy Harris who helped me in so many ways that to mention them all would be impossible. Grant Blackwell helped with my predator control work greatly and always had answers to my questions. Ross Martin supplied tracking tunnel bases. Jens Jorgenson helped with equipment and tools. For getting dirty helping me look for snails, thanks to Melissa, Amy, Sharon, Hilary, Debbie, Evette. Jodi Matenga, Erica Reed, and Barabara Just made my life as a postgraduate bearable, as did all the other postgraduate students in the Ecology Group. Peter Van Essen and Drs Murray Potter and Gillian Rapson helped with plant identification and analysis of results.

I also received a great deal of assistance from many other people. David Mudge, Len Doel, Brian Bruce (Red Sky Film), Ian Townsend, and Paul Barrett (Auckland Zoo) each supplied me with their expertise and enthusiasm. Graham and Jan Taylor gave me a place to stay when I needed it. At Massey University, Duncan Hedderley helped with statistics, and Mark Osborne and Kerry Harrington supplied herbicide and spray equipment.

I would particularly like to thank Rachel Standish and my supervisors, Drs Ian Stringer and Alastair Robertson, for their knowledge, assistance in the field, and the development and revision of this thesis.

This investigation was made possible by funding from the Department of Conservation. Special thanks to Ian Cooksley and Kath Walker for their expertise, input, and answers to all those questions. Thanks also to David Agnew, Philippa Crisp, Raewyn Empson, Christine

Reed, and all the other Department of Conservation staff who helped with fieldwork, permits, and feedback on ideas.

I would especially like to thank Callum Kay and John Lepper for their friendship and help over the last few years, and also Matt Wong, Scott Bowie, and Susan Greenwood. Special thanks to my entire family for their support and interest, and to Miranda Olliff for all her help in the field, constructive criticism, and for getting me working and distracting me when I needed each most. This thesis is in part yours.

This is dedicated to my parents, Carol and Jim. Without their support and belief I would not have been able to do this.

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CHAPTER 1

Introduction

Powelliphanta traversi traversi (Powell) (Gastropoda: Pulmonata: Rhytididae) is a rare New Zealand endemic. It became fully protected in 1980 and is described as a Category B, second priority threatened species (Molloy *et al.*, 1994). Two current threats to it are introduced vertebrate predators and modification of their natural habitat (lowland native forest) associated with the invasion of the introduced weed *Tradescantia fluminensis* (Vell.). This study sought to assess the impacts of these two factors to provide the Department of Conservation with information that will aid management of *P. t. traversi* in lowland remnants in the Horowhenua District, New Zealand.

Taxonomy

Powelliphanta is an endemic New Zealand genus, but close relatives in the genera *Victaphanta* and *Melavitrina* exist in southern Australia (Climo, 1975), while representatives of the Family Rhytididae exist throughout the Indo-Pacific region (Climo, 1977). The taxonomic status of New Zealand *Powelliphanta* snails has had a dynamic history and consensus has been rare, some species having in the past been classified as belonging to the genus *Paryphanta* (Powell, 1936) and the subgenus *Powelliphanta* (O'Connor, 1945). *Powelliphanta* was eventually awarded full genus status in a revision of the Rhytididae by Climo (1977).

The Family Rhytididae is divided into two subfamilies based on differences in their respective reproductive structures (Climo, 1977). They are the Rhytidinae (which includes *Powelliphanta*, *Wainuia* and *Rhytida*), and the Paryphantinae. Members of the Subfamily Rhytidinae possess a spermatheca and have a normal penis structure, while the Paryphantinae have a greatly reduced penis and lack a spermatheca (Climo, 1977).

The comprehensive description of *Powelliphanta* by Powell (1930, 1932, 1936, 1938, 1949, 1979) formed the basis of our current knowledge of their taxonomy and distribution, and

lead to the subsequent evolutionary hypotheses (e.g. Climo, 1978). The magnitude of Powell's efforts are still unparalleled in New Zealand, and his distinction of ten species, thirty-four subspecies and four forms of *Powelliphanta* (Powell, 1979) has been followed in later ecological works (Meads *et al.*, 1984; Devine, 1997).

Powell's classification (Powell, 1979) remains the most substantial for the genus and provides a useful basis for ecological study because of the fine division of snails in separate populations. Analysis of genetic differences may result in some changes to this classification (Kath Walker, 2000, *pers. comm.*), but in the meantime, for comparative purposes, I follow Powell's classification here.

Powelliphanta traversi traversi (Powell, 1930) is recognised as a subspecies of *P. traversi*, together with one other subspecies, *P. t. otakia* (Powell, 1946). *Powelliphanta traversi traversi* is further divided to include five 'types', which are distinguishable but not taxonomically recognised (Powell, 1979). Each type predominantly occurs as one or more geographically isolated populations that differ from each other in shell pattern. These types are commonly referred to as forma (pleural formae) and are designated here by the abbreviation fa. They are: traversi (Powell, 1930) (the nominate subspecies), tararuaensis (Powell, 1938), koputaroa (Powell, 1946), florida (Powell, 1946), and latizona (Powell, 1948).

Distribution of Powelliphanta traversi

Snails in the genus *Powelliphanta* can be found from Lake Waikeremoana and Mount Taranaki in the North to Resolution Island, Lake Monowai and the Mataura Range in the South (Meads *et al.*, 1984). However, *P. traversi* is restricted to the lowland plains of the Horowhenua District and the foothills of the Tararua Ranges in the south of the North Island (Meads *et al.*, 1984) (Figure 1.1).

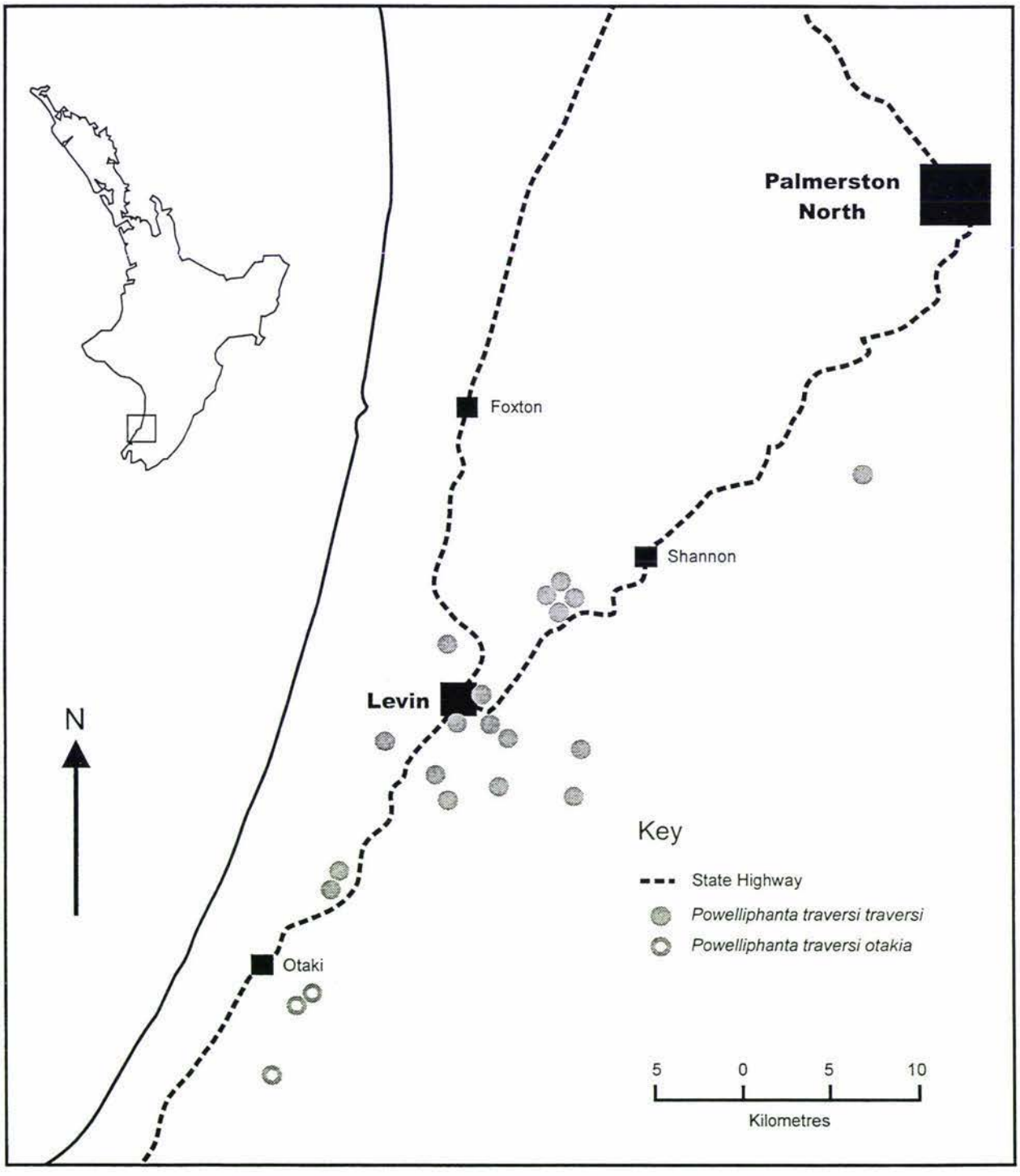

Figure 1.1 Distribution of *Powelliphanta traversi*.

Meads *et al.* (1984) confirmed the presence of *P. traversi* in fewer than half the 22 isolated localities where snails were previously known to occur up to 1965. More recently, *P. t. traversi* was only found at 9 isolated sites on the Horowhenua plains, and *P. t. otakia* is thought to exist at a further 3 sites (Rachel Standish & Shaun Bennett, *unpubl. data*). In addition, *P. t. traversi* has a limited distribution in the foothills of the Tararua Ranges.

Powell (1979) noted that each *P. t. traversi* forma had a restricted distribution with definite topographical boundaries. However, *P. t. traversi* fa. traversi and florida coexist at Lake Papaitonga Scenic Reserve together with a hybrid zone between them (Powell, 1946). This occurred because of course changes by the Ohau River. *P. t. traversi* fa. tararuaensis/traversi hybrids also occur in exotic pine plantations on the foothills of the Tararua Ranges.

Current knowledge of Powelliphanta traversi traversi

Irrespective of taxonomic organisation within the genus, it is apparent from previous investigations (e.g. Climo, 1975; Meads *et al.*, 1984; Devine, 1997) that many *Powelliphanta* populations face a host of common threats. In addition, due to their presence in lowland forest, *P. t. traversi* snails have likely suffered more than most from habitat destruction, predation by introduced species, and even from over-zealous collection by enthusiasts since European settlement (Meads *et al.*, 1984).

The small size of bush remnants containing *P. t. traversi*, together with their accessibility, the existence of rare or distinct formae, and the effects of introduced predators and weed species emphasise the need for effective strategies to protect these snails. Large-scale forest habitat removal has ceased, but many snail populations continue to be threatened with predation by introduced vertebrates and habitat modification caused by invasive weeds and surrounding land use.

Introduced vertebrates impose predation pressures never matched by native predators (Climo, 1975). Norway rats (*Rattus norvegicus* Berkenhout) were identified as a major predator of these snails in the Horowhenua (Meads *et al.*, 1984). However, ship rats (*Rattus rattus rattus* L.) also take snails, and have been trapped at Lake Papaitonga Scenic Reserve at rates far higher than in other comparable North Island sites (Devine, 1997).

Tradescantia fluminensis is an introduced weed now well established in many lowland forest areas of the North island and north of the South Island (Kelly & Skipworth, 1984a; Maule *et al.*, 1995). It is also present at sites in the Horowhenua where *P. t. traversi* occurs. It is not known what effect this weed may have on these snails directly although the snails are often found under it. *Tradescantia fluminensis* was shown to retard natural forest regeneration (Kelly & Skipworth, 1984a), so it may eventually affect the forest habitat of these snails.

There have been no published studies investigating the influence of *T. fluminensis* on *P. traversi*, or any other large New Zealand land snail species. This is despite the high potential for an interaction and the suggestion that this weed may provide benefits for some *P. traversi* populations (Anon., 1996).

Prior to my investigation, the only work to have investigated the distribution, movements, and morphometrics of *P. t. traversi* was by Devine (1997). Other works involving this species were limited to broad population estimates (e.g. Meads *et al.*, 1984; Walker, 1997a) and to taxonomic revisions (e.g. Climo, 1974, 1977, 1978). Consequently, there remains a paucity of life history information on this subspecies, as is the usual case with New Zealand's native land snails (Solem *et al.*, 1981).

Past attempts to gather basic information by maintaining snails of the genus *Powelliphanta* in captivity have met with mixed success. In one instance, eight captive *P. t. traversi* never mated or produced eggs, and most of the snails eventually died (Paul Barrett, 2000, *pers. comm.*). More success was achieved by Kath Walker who kept several species for several years. Many of them mated and produced eggs which hatched in captivity (Kath Walker, 2000, *pers. comm.*).

Thesis Focus

This study aimed to deal specifically with two main conservation issues concerning *P. t. traversi*. The first was to determine the effect on *P. t. traversi* of rodent control, and the second was to investigate the effect of *T. fluminensis* on their distribution and movements. The first objective aimed to determine the effectiveness of regularly poisoning rodents in a mainland reserve, and to quantify any effects this had on the local *P. t. traversi* population.

The second objective involved consideration of the effects *T. fluminensis* may have on snail movement and distribution, and the possible impact on *P. t. traversi* of weed control using herbicides. The toxicity of Grazon® herbicide to *P. t. traversi* was also tested because this is the best known herbicide for controlling *T. fluminensis* (e.g. Brown & Rees, 1995; McCluggage, 1998).

The broad aim of this investigation was to increase our knowledge of the ecology of *P. t. traversi* and to provide information to help manage these and other native land snails.

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CHAPTER 2

Study sites and General methods

2.1 STUDY SITES

Lake Papaitonga Scenic Reserve

Lake Papaitonga Scenic Reserve was chosen as the site for the experiment to determine the effects of a sustained poisoning operation on the recruitment of adult *Powelliphanta traversi traversi* (Powell) snails. The reserve is situated approximately 5 km Southwest of Levin (NZMS 260 S25 989 602). It comprises 52.20 ha of native bush surrounding a dune lake (Figure 2.1.1), and the total size of the reserve is 120.69 ha. The area was first set aside for the preservation of native bush in 1901 (Park, 1995; Anon., 1996) and has been under formal protection since the land was purchased in 1980 and 1981 (Devine, 1997).

Lake Papaitonga Scenic Reserve has an intact habitat sequence from wetland to mature dry terrace forest, and this forest habitat contains one of the largest extant populations of *P. t. traversi* (Anon., 1996). Another carnivorous snail, *Rhytida greenwoodi greenwoodi* (Gray), is also present in the reserve. The forest is dominated by tawa and titoki on the upper terrace, with low open swampland adjoining the shallow dune lake. The lake includes two islands which total 1.3 ha. Approximately 15.66 ha. of land is swamp, flax-koromiko shrubland or sedgeland (Devine, 1997). These latter areas are difficult to search and contained few snails (Devine, 1997). Thus, there is an effective land area of approximately 35.9 ha. of suitable habitat for *P. t. traversi*.

Powelliphanta traversi traversi formae traversi and florida coexist at Lake Papaitonga and have formed a hybrid zone where the two overlap (Powell, 1946). These snails were recently studied by Devine (1997), who calculated snail density to be 282 ± 17 snails per hectare, giving an estimated population in the reserve of between 13488 and 15219 snails.

Figure 2.1.1 Aerial photograph of Lake Papaitonga Scenic Reserve.

Snails of the genus *Powelliphanta* suffer depredations by many native and introduced vertebrates (Climo, 1975; Meads *et al.*, 1984), but ship rats (*Rattus rattus rattus* L.) and Norway rats (*Rattus norvegicus* Berkenhout) were identified as the primary predators of *P. t. traversi* (Meads *et al.*, 1984; Devine, 1997). Shell damage believed to be diagnostic of both rat species can be found at Lake Papaitonga (Meads *et al.*, 1984; Devine, 1997), but only ship rats were trapped there recently (Devine, 1997).

Prouse Bush

Prouse Bush was chosen as the site to investigate the movement and distribution of *P. t. traversi* snails in relation to areas of *Tradescantia fluminensis* (Vell.) and areas lacking the weed. This was done to evaluate the possible effects on the snails of *T. fluminensis* removal. *Powelliphanta traversi traversi* fa. traversi are present at Prouse Bush but there has been no previous work with the snails at this site other than brief searches for estimating snail abundance and predation rates (Meads *et al.*, 1984).

Prouse Bush is a small native remnant (approximately 5.3 ha.) within the urban limits of Levin (NZMS 260 S25 028 611). Residential properties, farmland, and industrial businesses bound the bush (Figure 2.1.2). Prouse Bush was first classified as a public reserve in 1928 and was gifted to the people of Levin by Christina Prouse in 1951 (Strong, 2000). It is currently under the administration of the Horowhenua District Council.

The canopy is tawa-titoki dominant with some rewarewa, mahoe and kawakawa (Ravine, 1995) but is greatly degraded in places. Little native understory exists apart from karaka and kawakawa saplings. The invasive weed *T. fluminensis* is well established throughout the bush, but is most prevalent where there is more light near the edges of the bush and where canopy cover is decreased. *Tradescantia fluminensis* is the most dominant ground vegetation at the site, covering approximately 55.35% (2.77 ha.) of the area. The remainder of the ground is covered predominantly by native leaf litter.

A Horowhenua District Council management plan for the area was prepared in 1992, but no action was taken to restore the native vegetation (Strong, 2000). The current management plan recognises that cover provided by *T. fluminensis* may protect the *P. t. traversi* there

(Strong, 2000). Consequently, a cautious approach was adopted by the district council and no *T. fluminensis* control was undertaken.

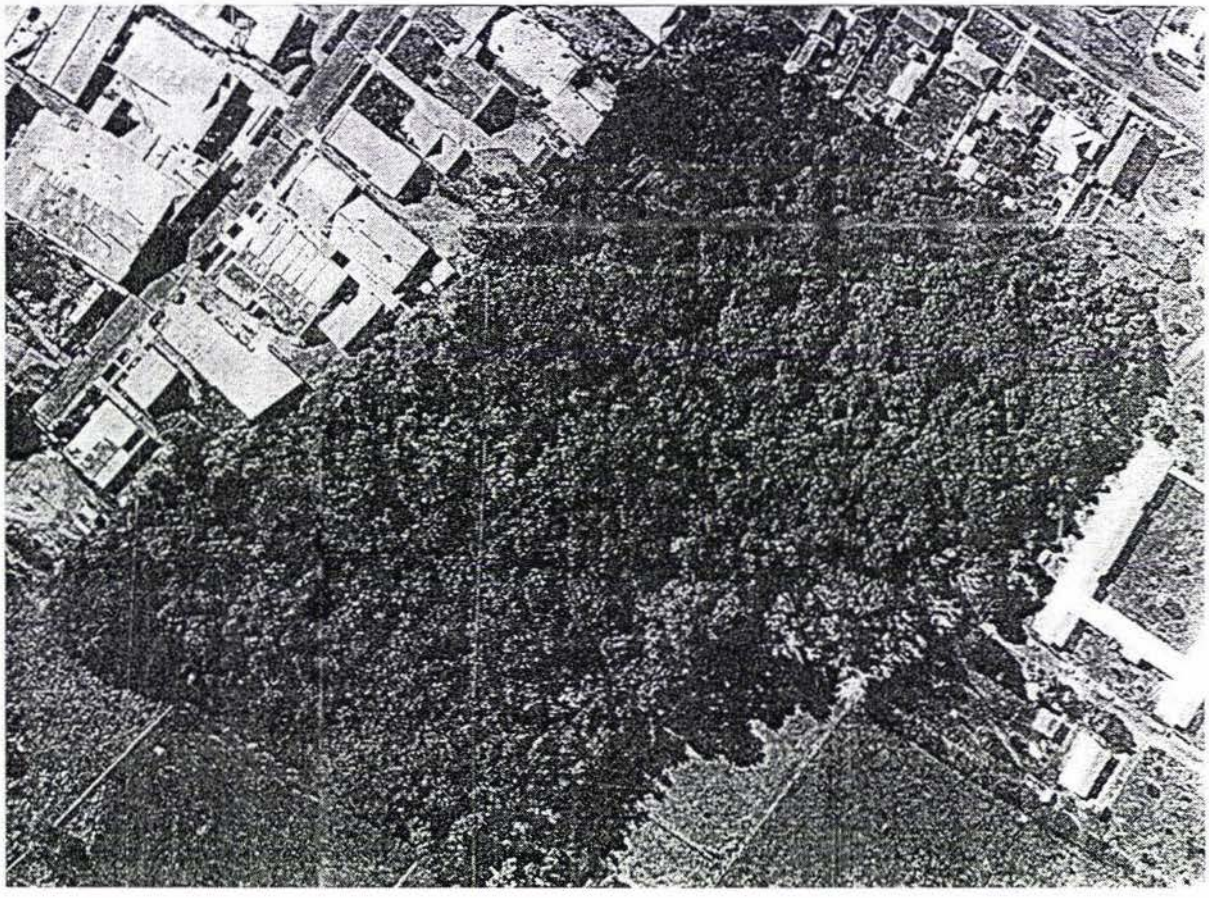


Figure 2.1.2 Aerial photograph of Prouse Bush.

2.2 GENERAL METHODS

Measuring and describing shells

Maximum shell length (ML), maximum shell width (MW), and maximum aperture width (MA) (Figure 2.2.1) were recorded using vernier calipers (±0.01 mm) for live snails and empty shells found at Lake Papaitonga and Prouse Bush. When shell damage or degradation prevented accurate measurement, shell measures were estimated by comparison with similar sized intact shells. Weight was recorded for all live snails using portable electronic scales (Sartorius (Germany) (±0.1 g) and Jadever (Taiwan) (±0.01 g)).

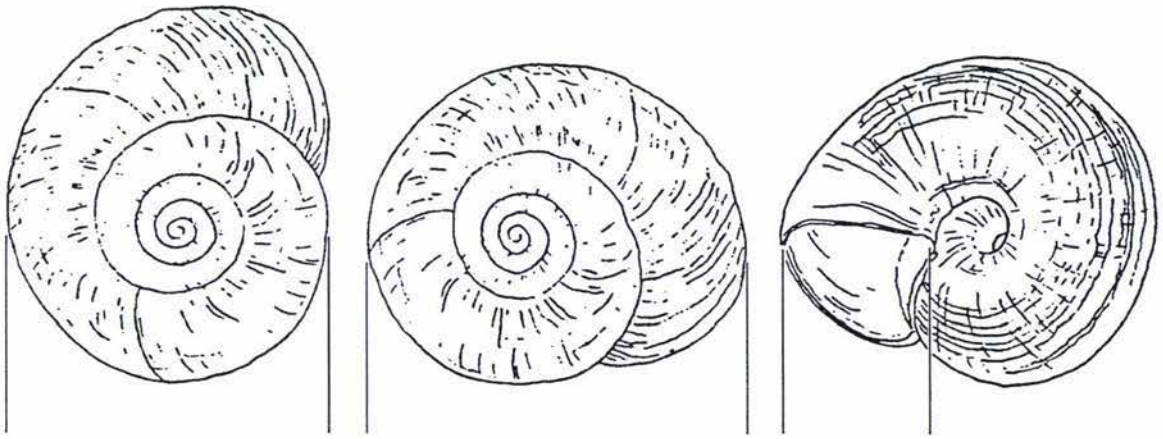
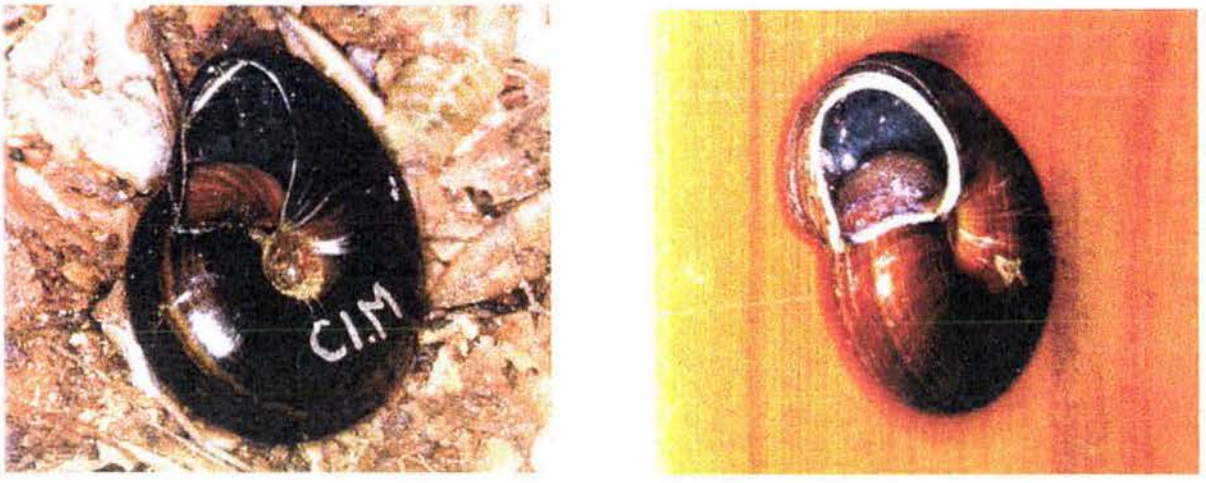


Figure 2.2.1 Shell measurements used in this study. Maximum width and length follow Meads *et al.* (1984).

Common shell characteristics or unusual features were also noted. These included shape, appearance and flexibility of the peristome (aperture lip) for all live snails, and the presence of an adult lip. The latter was identified for some other Rhytididae (Powell, 1946), but Devine (1997) was not able to differentiate between juveniles and adult *P. t. traversi* except on the basis of size. However, *P. t. traversi* snails are known to develop distinctive lip characteristics at the two study sites. Larger snails at Lake Papaitonga typically have a dropped or 'drooped' aperture lip (Devine, 1997) formed by downward growth of the aperture lip (Figure 2.2.2(a)). By comparison, many snails at Prouse Bush posses a

thickened lip (Figure 2.2.2(b)), thought to be the result of growth during dry or stressful environmental conditions (Kath Walker, 2000, *pers. comm.*).



(a) (b)

Figure 2.2.2 Ventral views of *Powelliphanta traversi traversi* found at (a) Lake Papaitonga and (b) Prouse Bush. Note the peristome and the method of marking using numbers engraved through the periostracum.

Marking snails

Live snails >23 mm (ML) at Lake Papaitonga were marked for subsequent identification with a letter-number combination on the ventral periostracum using an electric engraver (Arlec Engraver, Dick Smith Electronics) (Figure 2.2.2). Smaller individuals were marked with a small amount of Liquid Nails adhesive (Selleys) placed on the centre of the shell (over the protoconch) after it was lightly scoured with carborundum paper. This was done so as not to risk damaging small snails with the engraver but it did not provide individual identification. Most snails followed at Prouse Bush were engraved with a number but some small snails were marked with Liquid Nails (as for snails at Papaitonga), and others had a 'Queen-bee' label imbedded in 5-minute Araldite (Selleys).

Attaching Harmonic Transponders

Movements made by *P. t. traversi* snails at Prouse Bush over periods of a month or more were recorded by regularly finding snails using harmonic radar. Harmonic radar tracking was first developed to follow the movements of carabid beetles (Mascanzoni & Wallin, 1986) and it allowed safe and time efficient capture-recapture information to be gathered from other invertebrates (e.g. Riley *et al.*, 1996; Lövei *et al.*, 1997; Riley *et al.*, 1999). The use of harmonic radar was specifically adapted in New Zealand for tracking *Paryphanta busbyi watti* (Lövei *et al.*, 1997), but it was also used to follow the movements of *P. t. traversi* (Devine, 1997).

Harmonic radar tracking uses a hand held, portable transmitter-receiver unit (Recco Rescue Systems, Lidingo, Sweden), and a passive reflector or 'transponder' attached to the snail. A harmonic signal is reflected by the transponder, received by the radar unit and transformed into an audible signal that changes with proximity to the transponder (Lovei *et al.*, 1997). Because the hand held radar unit supplies the power for the transponder, transponder cost and weight are low, and the effective life of the transponder is increased beyond that of conventional transmitter-receiver devices which depend on batteries. One shortcoming of this technique is that 'displacement' (the point-to-point distance between successive positions) may not reveal the actual route (and therefore distance) traveled by a snail between relocations.

Transponders used for this study followed the design of Devine (1997), using a Z3040 diode (Dick Smith Electronics) soldered to a shaped copper aerial. Three sizes of transponders were produced to allow the use of the largest possible transponder for a range of snail sizes (Figure 2.2.3). They varied in mass from 0.70 - 1.20 g.

Transponders were attached to snails using Liquid Nails (Selleys) or 5-minute Araldite (Selleys) after lightly scouring the periostracum with carborundum paper. The transponders were then lightly covered with adhesive to ensure that they were secure and that there was a smooth surface over the shell. Attaching a transponder added on average 2.21 g (after the adhesive had cured) to the weight of a snail.

Figure 2.2.3 Transponder sizes used for harmonic radar tracking of different sized snails.

Spool and thread tracking

A spool and thread was used to track the nightly movements of *P. t. traversi* snails at Prouse Bush. A plastic sewing machine spool was attached to the snail shell using a loop of thread and 5-minute Araldite (Selleys) (Figure 2.2.4). This method gives information about the actual movement of a snail and was used to check where snail went between harmonic radar relocations.

The free end of the thread was securely attached to a peg or stick and when a snail moved the thread pulled out and snagged on small features of the environment. This produced an approximate record of the snails movements. Spools require <0.005 N to unravel (Devine, 1997) and could contain up to 30 m of thread. Attaching a spool added between 1.37 and 1.95 g to the weight of each snail.

None the less, spool and thread tracking has limitations. It is unable to record movements greater than the amount of thread held by the spool, and the thread can become easily tangled which may prevent a snail from moving (Devine, 1997; Coad, 1998). For this reason, spool and thread tracking was only used for intervals of up to 72 hours.

Figure 2.2.4 *Powelliphanta traversi traversi* snail with transponder and spool attached.

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CHAPTER 3

The effects of introduced predators and predator control on *Powelliphanta traversi traversi* (Powell).

ABSTRACT

Brodifacoum poison was used in two areas of Lake Papaitonga Scenic Reserve to control rats and mice. Mice abundance (inferred from tracking tunnel indices) was reduced in both poisoned areas below levels observed in two other areas that were not poisoned. Rat abundance was reduced below pre-poisoning levels but only to levels below one of the non-poison areas. In each poison and non-poison area, four 100m² quadrats were searched for *Powelliphanta traversi traversi* (Powell) snails immediately before poisoning, and 12 and 19 months after poisoning commenced. After 12 months of poisoning there was an increase in the numbers of live snails in all of the four areas. The increase in one non-poisoned area (from 24 to 40 snails) was significant ($P<0.05$). After 19 months of poisoning, only one poison area showed another increase in the number of snails, with significantly more snails (45) being found than at either of the two previous searches (22 before poisoning, 28 after 12 months) ($P<0.05$). Overall, there were highly significant differences in the number of live snails found in each area ($P<0.01$). The mean (± std. error) number of live snails per 100 m² quadrat was 5.13 (± 0.652), giving a 95% confidence interval for the *P. t. traversi* population at Lake Papaitonga of between 13678 and 23120 snails in 35.9 ha. The numbers of empty shells found decreased at each search (empty shells were removed from quadrats when first found) suggesting that shell accumulation is constant rather than seasonal. Rodents were responsible for more predated shells than any other identifiable predator, but the numbers of bird damaged shells increased in three of the areas. Overall, there was no conclusive evidence to suggest that the numbers of live *P. t. traversi* increased as a result of rodent poisoning. Possible reasons for the changes in live snail and empty shell numbers, and the effects of rodent poisoning are discussed.

3.1 INTRODUCTION

New Zealand's native biota is characterised by a high degree of endemism presumably due to its long isolation from other land masses, its Gondwanan ancestry, and the absence of many mammals (Cooper & Millener, 1993; Daugherty *et al.*, 1993). New Zealand lacked land mammals other than bats and seals until around 1000-1200 years ago (King, 1990), so many of the traits developed in the absence of mammalian predators (e.g. gigantism and low reproductive rates) resulted in rapid declines of many endemic species after the arrival of such mammals (Daugherty *et al.*, 1993).

Polynesians arrived in New Zealand around 1000 years ago and successfully introduced dogs and kiore (*Rattus exulans* Peale), and reduced the native forest cover. The arrival and

settlement of Europeans since around 1840 resulted in much greater forest reduction and the establishment of 82 species of exotic mammals, birds and fishes (Atkinson & Cameron, 1993). Their introduction has had significant and often obvious effects on New Zealand's vegetation (Campbell, 1978), its vertebrate biota (Atkinson, 1978), and its endemic invertebrate fauna (Ramsay, 1978).

Endemic land snails of the genus *Powelliphanta* have probably always had some degree of 'natural' predation from endemic bird species such as kaka, kea, and weka, but these predators alone are not thought to threaten these snails (Meads *et al.*, 1984). In contrast, the introduction of vertebrate predators was thought to have caused a widespread decline in both the numbers of species of snails and the distributions of the three largest snail genera, *Powelliphanta*, *Paryphanta* and *Placostylus* (Meads *et al.*, 1984). For example, *Powelliphanta traversi traversi* (Powell) has suffered greatly from both habitat destruction and predation by introduced vertebrates. These snails may have once occurred throughout the Horowhenua and Manawatu districts and Tararua ranges in the lower North Island. The majority of extant populations are now reduced to small isolated bush remnants on the plains of the Horowhenua District (Meads *et al.*, 1984).

Rats, mice, brushtailed possums, pigs, hedgehogs, mustelids, blackbirds, song thrushes, and cats have all been cited as possible predators of *Powelliphanta* and other native land snails (Powell, 1946; Meads *et al.*, 1984; Walker, 1997a; Coad, 1998; Efford, 1998). All occur in the Horowhenua District.

Brushtailed possums (*Trichosurus vulpecula* Kerr) have eaten live *Powelliphanta* snails presented to them during predator trials (Kath Walker, 2000 *pers. comm.*) and are prolific predators of *Powelliphanta* snails in localised areas. However, there is no evidence of possum predation on *P. t. traversi* in bush remnants of the Horowhenua District (Meads *et al.*, 1984; Devine, 1997).

Stoats (*Mustela erminea* L.) were implicated as predators of *Powelliphanta marchanti* (Powell) from damage to their shells, but they are not believed to be significant predators of *Powelliphanta* (Meads *et al.*, 1984). They have successfully 'attacked' baited shells of *Paryphanta busbyi busbyi* (Gray) (Coad, 1998) and *P. t. traversi* (see Appendix 1), but they

failed to attack live *Powelliphanta* when presented to them (Kath Walker, 2000, *pers. comm.*). It is likely that ferrets and weasels could also prey on *P. t. traversi* snails in the Horowhenua.

Hedgehogs (*Erinaceus europaeus occidentalis* (Barrett-Hamilton)) are abundant throughout lowland coastal areas of the North Island (Brockie, 1990) and have been specifically blamed for reductions of *P. t. traversi* at Waiopehu Reserve (Powell, 1946). They are also suspected to prey on snails of the genera *Placostylus* and *Paryphanta* (Parrish *et al.*, 1995; Sherley *et al.*, 1998). Hedgehogs are known to feed on the introduced brown garden snail (*Cantareus aspersus* (Müller)) (Brockie, 1990), and they are also known to eat the medium sized native snails *Wainuia urnula* (Pfeiffer) and *Rhytida greenwoodi greenwoodi* (Gray) (Brockie, 1990; Efford, 1998). They will eat small *Powelliphanta* (Kath Walker, 2000, *pers. comm.*) and *Paryphanta busbyi busbyi* in feeding trials (Coad, 1998), but appear to be restricted to smaller sized *Powelliphanta* (Kath Walker, 2000, *pers. comm.*).

All four rodent species known in New Zealand have had a considerable impact on invertebrates throughout New Zealand (Ramsay, 1978). All belong to the Family Muridae, and in order of their arrival are: the polynesian rat or kiore (*R. exulans*), the Norway or brown rat (*Rattus norvegicus* Berkenhout), the house mouse (*Mus musculus* L.), and the ship or black rat (*Rattus rattus rattus* L.) (Innes, 1992). Rats are believed to be the main predators of *P. t. traversi* (Meads *et al.*, 1984; Devine, 1997).

Kiore were implicated in the extinction of two species of large land snails (*Placostylus hongii* (Lesson), and *Amborhytida tarangensis* (Powell)) on Lady Alice Island (Brook, 1999), as well as extinctions and reductions on mainland New Zealand of an unknown number of larger invertebrates, including land snails (Atkinson & Moller, 1990). Meads *et al.* (1984) reported evidence of kiore predation on *Powelliphanta* snails, but their full impact on small animals on mainland New Zealand can only be inferred because Kiore disappeared from the mainland except the lower South Island by the late 1800's (Atkinson, 1973).

Mice were considered to be predators of New Zealand's large land snails by Climo (1975) and are possible predators of *Powelliphanta* (Meads *et al.*, 1984). Invertebrates are an important food source for mice (Murphy & Pickard, 1990), which are capable of successfully

'attacking' baited *P. t. traversi* shells (see Appendix 1), so it is likely that they would be capable of attacking at least small *Powelliphanta* snails.

Ship and Norway rats are able to coexist, with ship rats being mainly arboreal and Norway rats being predominantly ground dwelling and typically associated with water (Moors, 1990). Evidence of snail predation by both ship and Norway rats is substantial (e.g. Meads *et al.*, 1984; Devine, 1997; Walker, 1997b; Sherley *et al.*, 1998). Rat depredations (either ship or Norway) have accounted for a major proportion of empty *P. t. traversi* shells found at many sites in the Horowhenua (Meads *et al.*, 1984; Devine, 1997) and appear to directly threaten several snail populations. Meads *et al.* (1984) thought Norway rats were most likely responsible for the heavy rat depredations of *P. t. traversi*.

Much of New Zealand's modern conservation effort has gone toward the control or eradication of rodents because of their wide distribution. Poison was widely used to successfully eradicate them, especially in offshore island operations (see Veitch (1992) for summary of operations). An important step in the success of these poisoning operations was the development of second-generation anticoagulant poisons (such as brodifacoum). Such poisons can kill after a single feed but they allow a rodent to consume a lethal dose long before they suffer toxic effects, so they are not known to cause bait shyness following a sub-lethal dose (Taylor, 1992). Brodifacoum was used in 28 of 33 mammal eradication programs undertaken by the Department of Conservation (DoC) on offshore islands in recent times (I. McFadden, *pers. comm.*, cited in Innes, 1999), and it has also been used to control mice in crop fields (Brown & Singleton, 1998).

Studies of the effects of rodents on land snails have shown variable benefits of rodent control. Sherley *et al.* (1998) found that all juvenile snails of a subspecies of *Placostylus ambagiosus* (Suter) with shells larger than 10 mm increased significantly following long term (8 years) 'pulse' poisoning of rodents. In contrast, surveys of *Powelliphanta traversi otakia* (Powell) during 18 years of rodent control have not shown any clear long term benefits (Walker, 1997b). However, neither study assessed the number of rodents present, so the effectiveness of poisoning was not known.

The research in this chapter was aimed at assessing what benefit rodent control has on the numbers of *P. t. traversi*. This was done by comparing the numbers of rodents and the numbers of *P. t. traversi* snails in two experimental control areas with those in two areas where brodifacoum baits were applied over 22 months in Lake Papaitonga Scenic Reserve (NZMS 260 S25 989 602), near Levin (see Chapter 2 for site description).

3.2 **METHODS**

Effect of rodent control on snails

Two paired areas were set up at Lake Papaitonga Scenic Reserve to test the effects of rodent control on recruitment of *P. t. traversi*. Both areas were situated in similar habitat but separated from each other by an area of wetland (Figure 3.2.1). One area of each pair was poisoned and one was not poisoned. The four areas were designated P1, P2, and N1, N2, for poisoned and non-poisoned areas respectively. They were estimated to occupy approx. 2.99, 2.94, 2.92, and 2.60 ha. respectively using an aerial photograph.

Seventeen Philproof® bait stations were placed on a 50 x 50 m grid in each poison area (Figure 3.2.1). Bait stations were affixed low on trees. Each had sticks from the opening to the ground to aid rodent access. Orange-flavoured PestOff® possum bait (Animal Control Products Limited, Wanganui), a wax-coated cereal pellet (active ingredient 0.02g/kg brodifacoum), was used. Six pellets were placed in each bait station to assess bait take on 20/10/1998. All but four bait stations were empty five days later so they were all restocked with 100-250 g of poison (approximately 38-87 pellets) depending on bait take. Bait stations were checked and re-stocked as necessary every 21-42 days until 14 April 2000, and again on 10 July 2000.

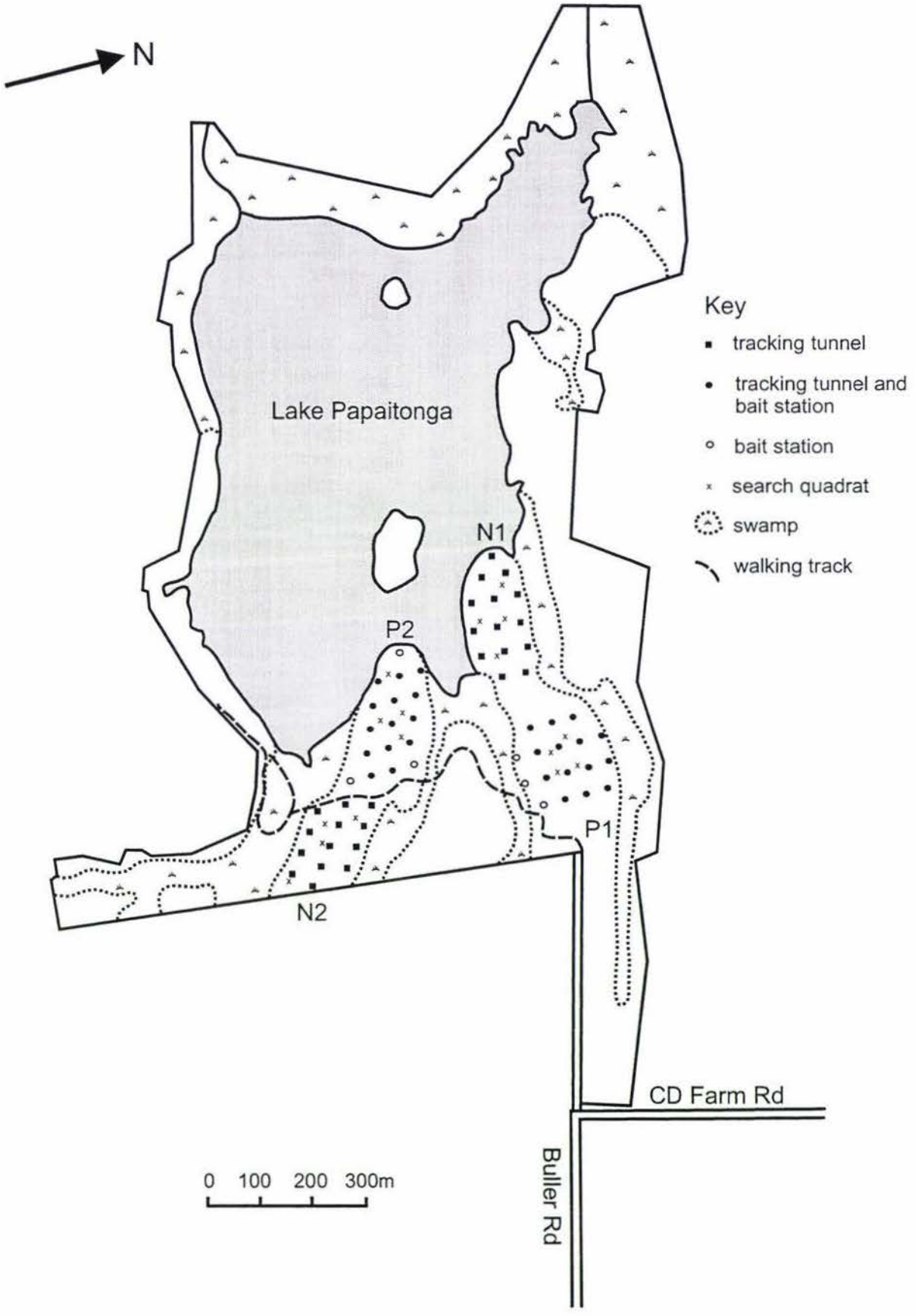

Figure 3.2.1 Map of Lake Papaitonga Scenic Reserve showing experimental areas.

Rodent abundance was monitored in all areas on two occasions before poison was applied and on 17 occasions during the poisoning program using tracking tunnels with ferric chloride ink and tannic acid-sprayed tracking papers (King *et al.*, 1994) (Figure 3.2.2). An index of rodent numbers was obtained each time the site was visited using tracking tunnels placed next to the bait stations in P1 and P2, and on a 50 x 50 m grid in N1 and N2. On each occasion, tracking papers were replaced for 3 consecutive nights and a small amount of crunchy peanut butter (Sanitarium Co.) was added to each end of the tracking tunnels. The average percentage of tunnels with tracks during the three nights was used as an index of rodent abundance for each area.

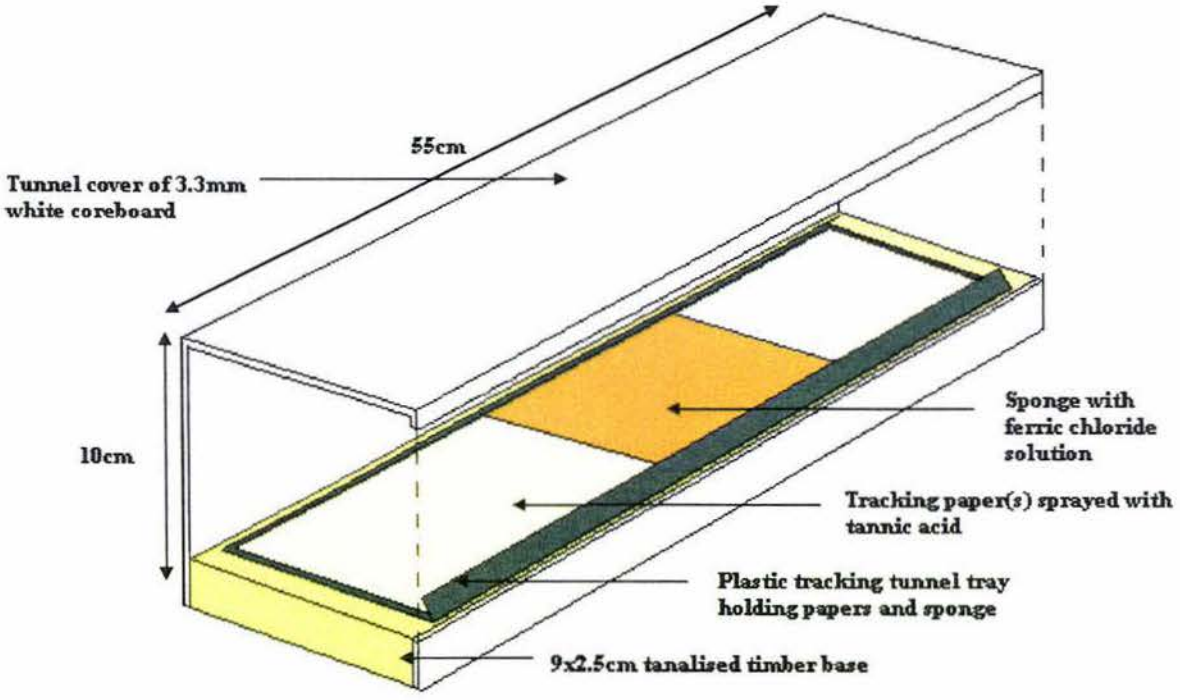

Figure 3.2.2 Cut-away view of tracking tunnel showing construction of tunnels used to monitor rodent levels at Lake Papaitonga Scenic Reserve. Figure not drawn to scale.

Monitoring P. t. traversi snails

Four 20 x 5 m quadrats were established in each experimental area at Lake Papaitonga (total of sixteen quadrats) for monitoring *P. t. traversi* snails (Figure 3.2.1). Each was marked with painted wooden stakes at each end of a 20 m-long centre line. All quadrats were searched in October 1998 before the first poison was applied, and then 12 and 19 months after rodent control commenced (October 1999 and May 2000).

The size and weight of each live snail found was measured each time it was found (see Chapter 2 for dimensions). They were then marked for identification (see Chapter 2 for details of marking) and replaced where they were found. Empty shells were measured and the probable cause of death noted. They were then discarded well outside the quadrats in which they were found, so they would not be collected in future searches.

3.3 **RESULTS**

Changes in numbers of rodents

Both rat and mouse track indices increased in all areas between the first (August 1998) and second (October 1998) pre-poison sampling occasions. On the second occasion, immediately before poisoning, P1 and N2, and P2 and N1 had similar rat track indices (Figure 3.3.1), but the indices for P2 and N1 were considerably higher (45% and 38% respectively) that those at P1 and N2 (14% and 12.33% respectively). The mouse track indices for all areas were greater than the corresponding rat track indices (Figure 3.3.2). Also, the mouse track indices at P2 and N1 (52.33% and 47.67% respectively) were more similar than those at P1 and N2 (69% and 89.5% respectively).

No rats were recorded in P2 during November and December 1998 after poisoning began (Figure 3.3.1) and the rat track indices for this area remained mostly below 5% and always below 12% thereafter (Table 3.3.1). However, there was a slower decline in the rat tracking index at P1, until no rat tracks were observed there by March 1999. Thereafter, the rat track index ranged between 0 and 5%, except for one occasion when it reached 14.33% (January 2000) (Table 3.3.1). The rat track index at N1 was consistently higher than any other area when poison was applied (Figure 3.3.1) and this area also had the highest average rat track index (46.04%) (Table 3.3.1). In contrast, the rat track indices for N2 was similar to the two poison areas both in range and overall average (Table 3.3.1).

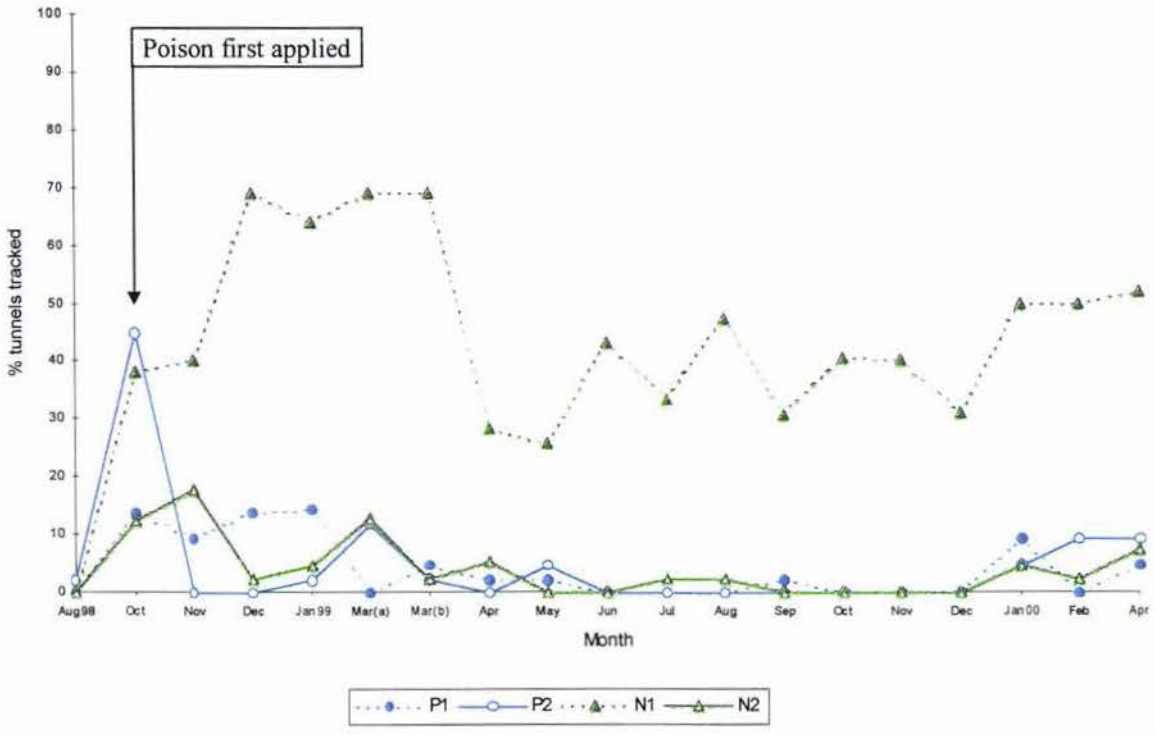

Figure 3.3.1 Percentage of tunnels with rat tracks in each of the four areas at Lake Papaitonga Scenic Reserve. Average percentages of three consecutive tracking nights are shown. Poison baits were first put out immediately after the tracking index was taken for October 1998 and ceased in July 2000.

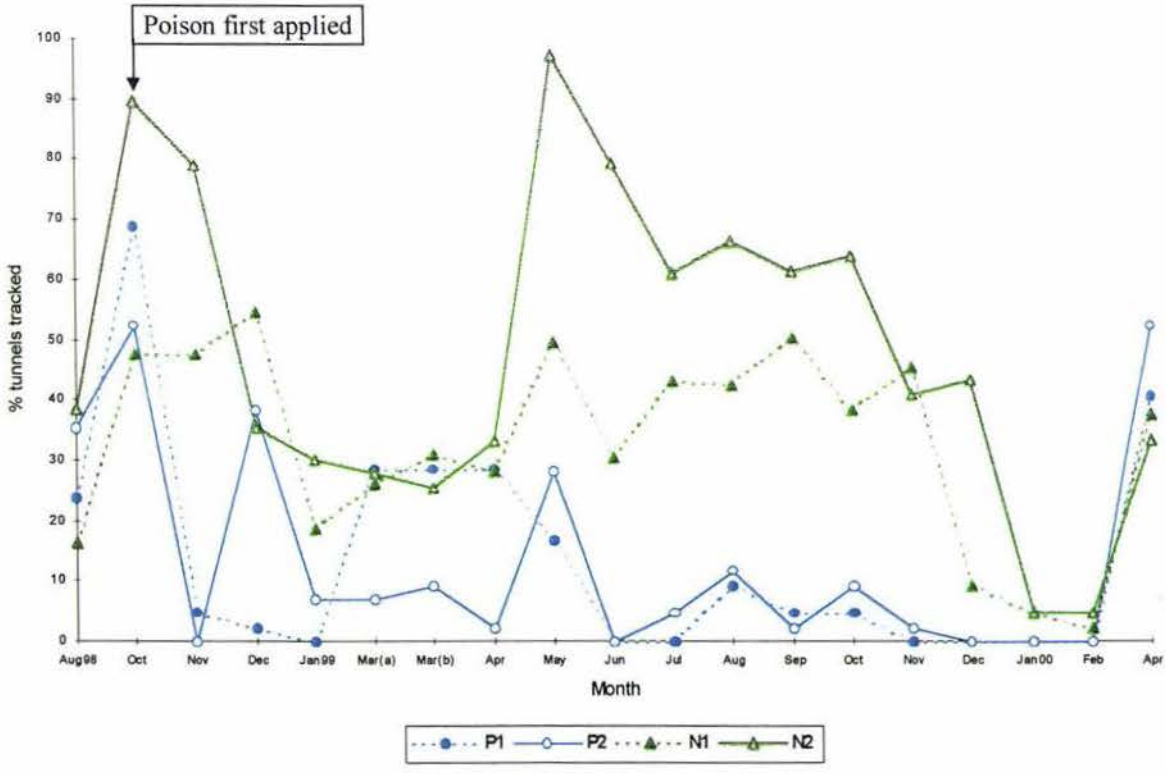

Figure 3.3.2 Percentage of tunnels with mouse tracks in each of the four areas at Lake Papaitonga Scenic Reserve. Average percentages of three consecutive tracking nights are shown. Poison baits were first put out immediately after the tracking index was taken for October 1998 and ceased in July 2000.

There was an immediate reduction of 93% - 100% in the mouse track index at P1 and P2 following the first application of poison (Figure 3.3.2). The average mouse track index in both poison areas was much lower than in the non-poison areas (Table 3.3.2), but the mouse track index for one poison area was at least as high as a non-poison area in December 1998 and March - April 1999. The mouse track indices for P1 and P2 even exceeded both non-poison areas on one occasion (April 2000). Overall, the mouse track index was generally highest in N2 (Figure 3.3.2 and Table 3.3.2) and contrasted with the rat track indices which were highest in N1 (Figure 3.3.1 and Table 3.3.1). In general, the mouse track indices during poisoning were generally higher than the corresponding rat track indices, except in N1.

The mouse track index in P2 was generally lower than the index in N2 but the two indices showed similar patterns of increases and declines in the percentage of tunnels with tracks (Figure 3.3.2). There was also a similar increase followed by a decline in the rat track indices in P2 and N2 between December 1998 - March(b) 1999 (Figure 3.3.1). There were no apparent similarities between mouse and rat track indices in P1 and N1. Such results suggest that some unknown factor(s) has a similar influence on rodents in P2 and N2 during poisoning, and that they either did not operate in P1 and N1 or that they were so low as to not be detected. However, mouse track indices in all four areas increased similarly at the time this was taken in April 2000.

Table 3.3.1 Summary of rat tracking indices at Lake Papaitonga. Total range includes tracking index values observed before poison was applied.

Area	Pre-poisoning maximum	Total range	Average during poisoning
P1	14.00	0.00 – 14.33	3.72
P2	45.00	0.00 – 11.67	2.61
N1	38.00	0.00 – 69.00	46.04
N2	12.33	0.00 – 17.67	3.77

Table 3.3.2 Summary of mice tracking indices at Lake Papaitonga. Total range includes tracking index values observed before poison was applied.

Area	Pre-poisoning maximum	Total range	Average during poisoning
P1	69.00	0.00 – 48.67	9.94
P2	52.33	0.00 – 52.33	10.29
N1	47.67	2.33 – 54.67	32.96
N2	89.50	4.67 – 89.50	46.36

Live snails

Numbers before poisoning

There were large differences in the number of live snails in the four areas when these were sampled before poisoning commenced in 1998. Numbers ranged from 6 to 23 (Figure 3.3.3). There were significantly fewer live snails found at N2 (6 snails) compared to P2 ($P<0.05$, Chi-square), P1 ($P<0.01$), and N1 ($P<0.01$).

There was also a large variation (mean ± std. deviation) in the numbers of live snails found in quadrats (100 m²) in the areas P1 (5.5 ± 4.12), P2 (4.0 ± 3.56), and N2 (1.5 ± 1.29). The least amount of variation occurred in N1 (6.0 ± 2.16).

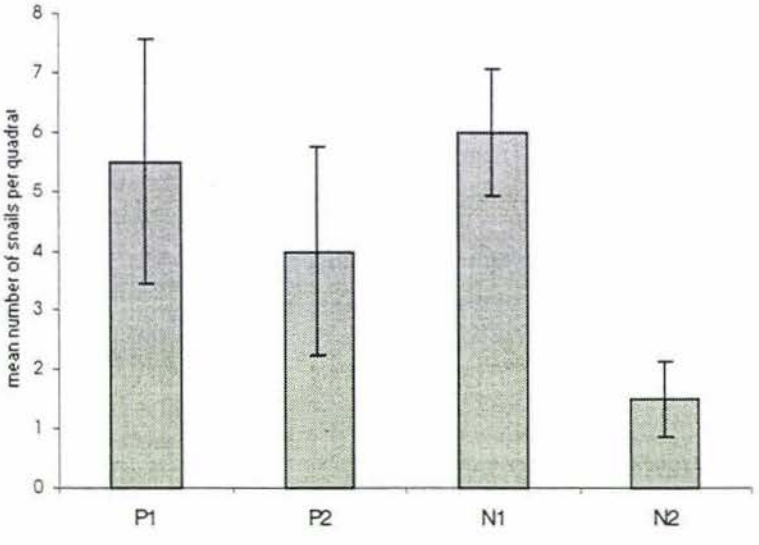

Figure 3.3.3 Mean (± S.E.) number of live *P. t. traversi* snails per 100 m² quadrat in each area before poisoning began in P1 and P2.

Changes in numbers of snails during poisoning

The numbers of live snails found in each area increased after 12 months (1999) of poisoning in P1 and P2 (Figure 3.3.4). N1 remained the area with the greatest number of live snails (40 snails) and N2 the area with the least (8 snails).

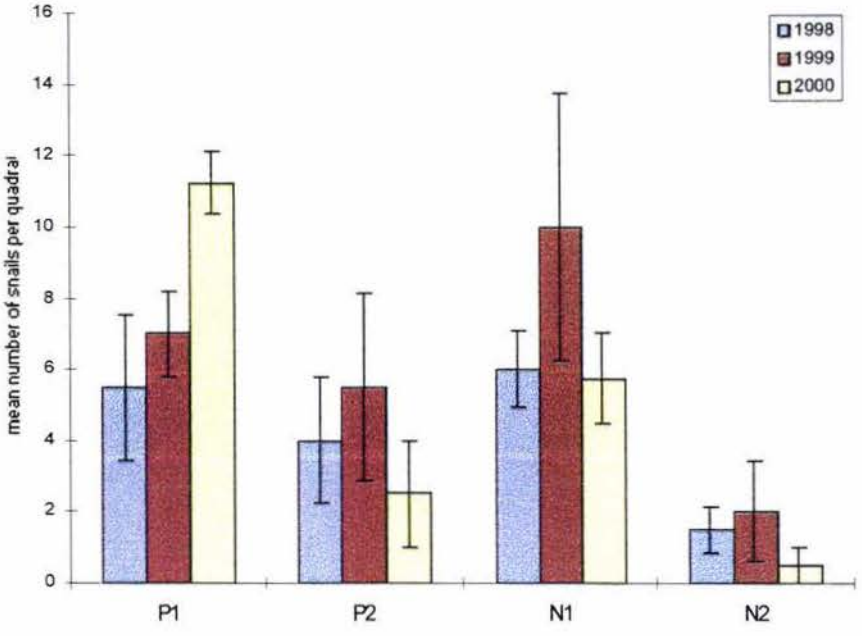

Figure 3.3.4 Mean (± S.E.) numbers of live *P. t. traversi* snails found per 100 m² quadrat at each search in each area of Lake Papaitonga Scenic Reserve.

The greatest number of live snails found in 2000 was at P1 (45 snails). This was the largest number of live snails found at one area in any year. The numbers of live snails at P2, N1, and N2 were lowest in 2000. Over the three searches, only P1 showed a consistent increase in the number of live snails (22, 28, and 45 snails, see Figure 3.3.4).

The difference in the number of live snails found in each area (between the three searches) was highly significant ($P<0.01$, Fisher's exact test). Of the changes, the increase in the number of snails found at P1 in 1999 and 2000, and at N1 in 1999 made the greatest contribution to this result. The number of live snails found at P1 in 2000 (45 snails) was a highly significant increase ($P<0.01$, Chi-square test) from the numbers of snails found in 1998 (22 snails), and a slightly significant increase ($P<0.05$, Chi-square test) from the number found in 1999 (28 snails). The number of live snails found in area N1 in 1999 (40) was only slightly significantly different ($P<0.05$) from the number of snails found in N1 in 1998 (24 snails).

The decline in the numbers of live snails between 1999 and 2000 in N1 (40 and 23 snail respectively) and P2 (22 and 10 respectively) were also significant ($P<0.05$, Chi-square test). No other changes in the number of live snails (increases or declines) from 1998 levels were significant for any area.

Further comparisons of changes in the number of live snails (Fisher's exact test) were made between adjacent poison and non-poison areas (that is, P1 was compared with N1, and P2 was compared with N2). The increase in the number of live snails in P1 was highly significant ($P=0.010$) compared with the changes in snail numbers in N1. However, compared with N2, the change in number of live snails found in P2 was not significant ($P>0.05$).

The mean (± std. error) number of live snails per 100 m² quadrat for all 48 searches was 5.13 (± 0.652). This gives a estimate for the *P. t. traversi* population at Lake Papaitonga Scenic Reserve of between 13678 and 23120 snails (95% confidence interval) for 35.9 ha.

Snail relocations

Ten snails were found again during subsequent searches. One snail, first found in 1998, was found in 1999 and 2000 at P1 and one other snail found in 1998 was found again in 1999. Eight snails found for the first time during the 1999 searches were found again in 2000. Seven of these were found in the same quadrat they occupied previously. The increase in the number of snails found again in 2000 suggests that the chance of finding a snail again in the same area increases when the time between searches decreases (from 12 to 7 months).

Changes in snail size

There was a significant change ($P<0.05$, Fisher's exact test) in the size frequency distribution of live snails between 1998 and 1999 in area P2 (Figure 3.3.5) and N1 (Figure 3.3.6). In area P2 there was an increase in the number of snails between 30-44mm maximum shell length in 1999 (Figure 3.3.5), whereas in N1 there was a large increase in the number of live snails with a maximum shell length of 40-44mm in 1999 (Figure 3.3.6). No other comparisons within each of the four areas showed any significant change in the size distribution of live snails between 1998 and 1999 or 2000.

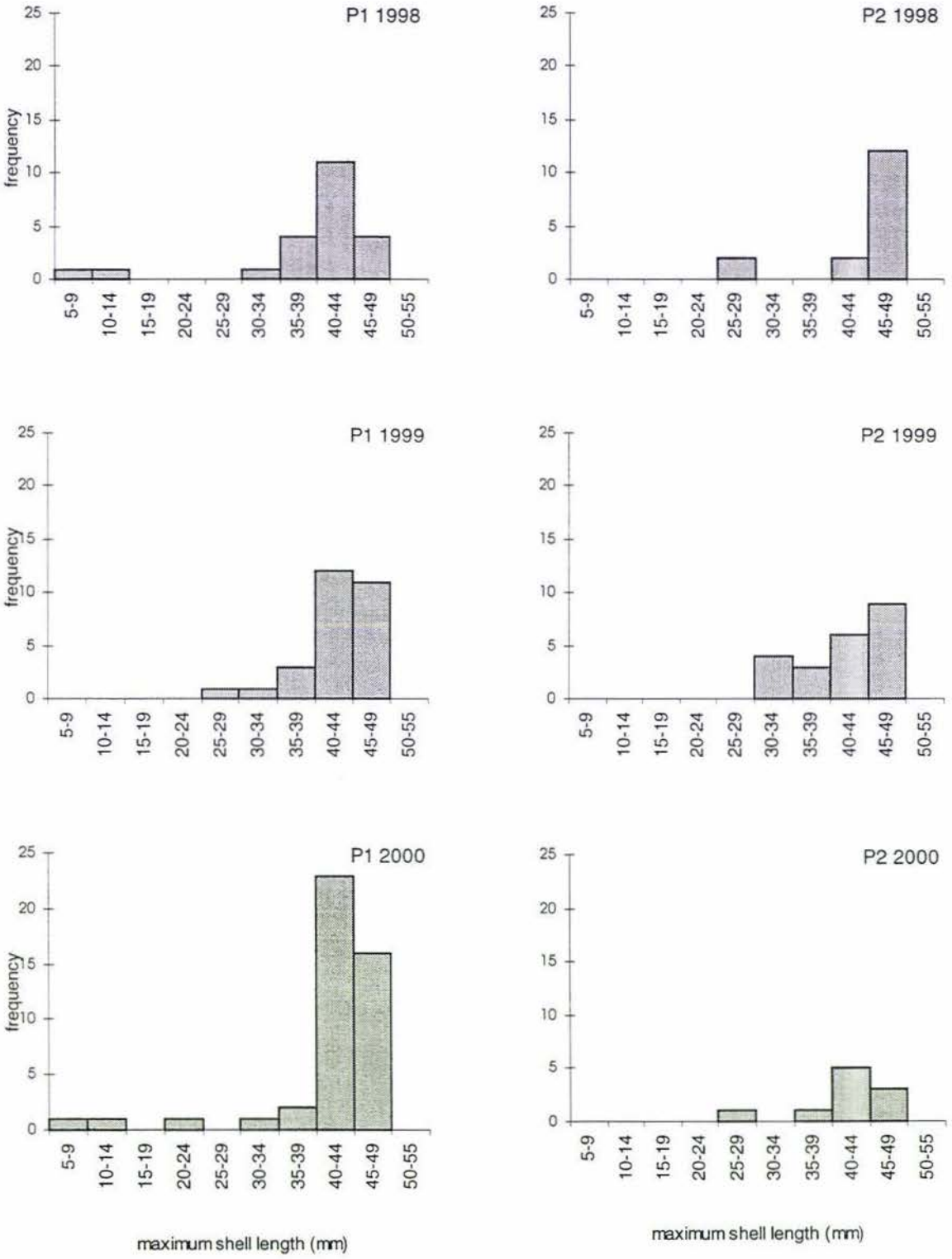

Figure 3.3.5 Size-frequency distributions (maximum shell length) of live *P. t. traversi* found during searches in 1998, 1999, and 2000, in areas P1 and P2.

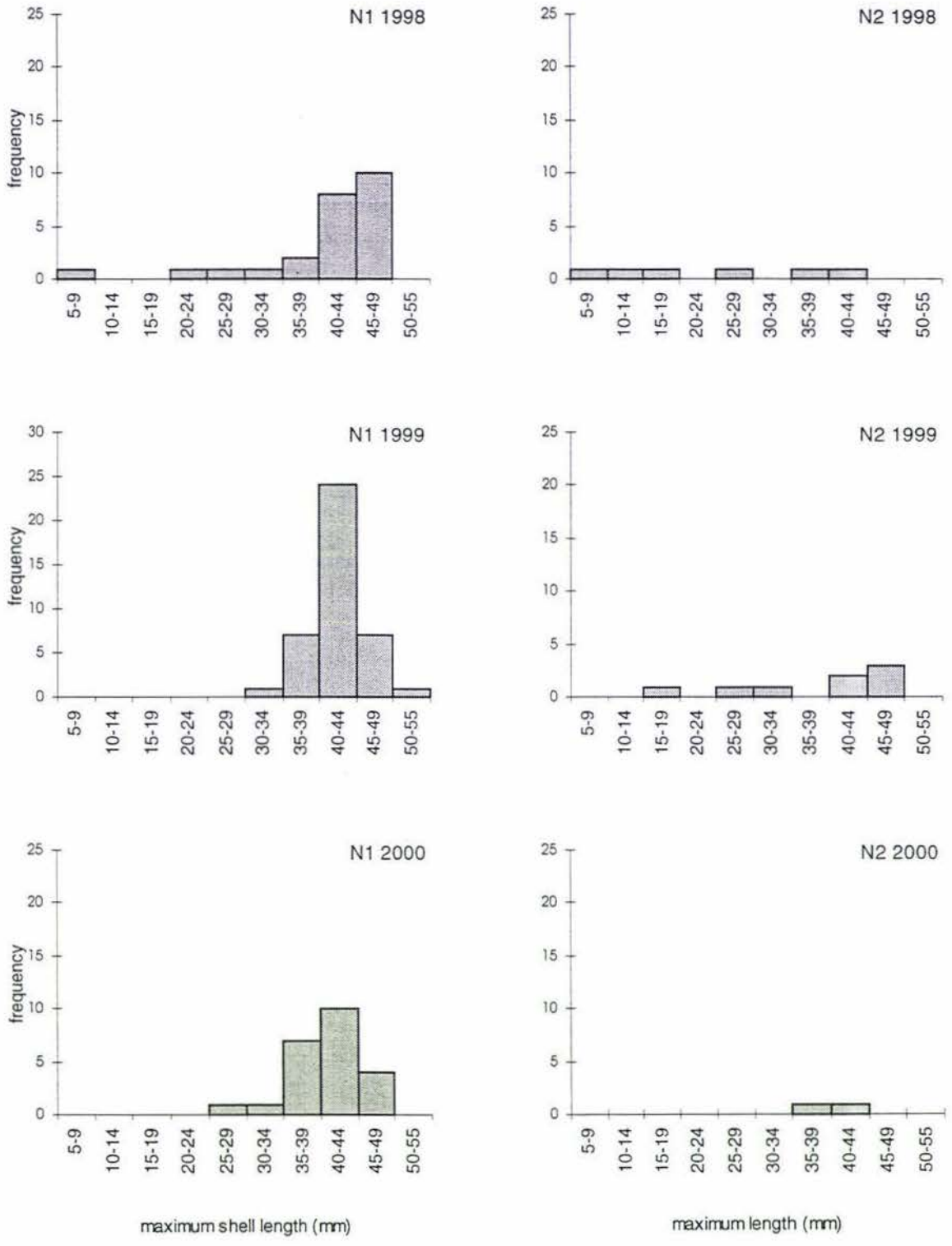

Figure 5.3.6 Size-frequency distributions (maximum shell length) of live *P. t. traversi* found during searches in 1998, 1999, and 2000, in areas N1 and N2.

Numbers of empty shells

The total number of *P. t. traversi* shells found in each of the four areas decreased at each search by approximately half. This result is expected considering the sampling period each search represented. The 1998 search effectively sampled a period proceeding the search that equaled the life span of the shells in this environment. The decrease in shell numbers between the 1999 and 2000 searches corresponded with the decrease between searches from 12 months (1998-1999) to 7 months (1999-2000). This suggests that the accumulation of shells in each area was related linearly to time rather than seasonal variation.

Numbers of whole shells, rat damaged shells, and other partial shells (shells with natural or indeterminate damage) decreased or remained constant in all areas (Figure 3.3.7). No rat damaged shells were ever found at P2 or N2. There were significantly fewer rat damaged shells ($P<0.05$) in P1 in 1999 (6) and 2000 (2) compared with 1998 (23). By comparison, there was no significant change in the number of rat damaged shells in N1.

No bird damaged shells were ever found in N2, but the number of bird damaged shells found at P1 increased in both 1999 and 2000, and at P2 and N1 in 1999 (Figure 3.3.7). Only the increases in P1 were significant ($P<0.05$).

Empty shells were found with damage consistent with both Norway and ship rat predation during this study. Several empty shells (and two live snails) were found that had suffered rat damage to the aperture lip (peristome) but had continued to grow so it is apparent that snails are able to survive some degree of rodent attack. Both types of rat damage, and snails that had survived rodent attack were also observed by Devine (1997).

In total (all searches), more whole shells were found than any other (51.49%), followed by other partial shells (18.72%), rat damaged shells (17.87%) and bird damaged shells (11.91%).

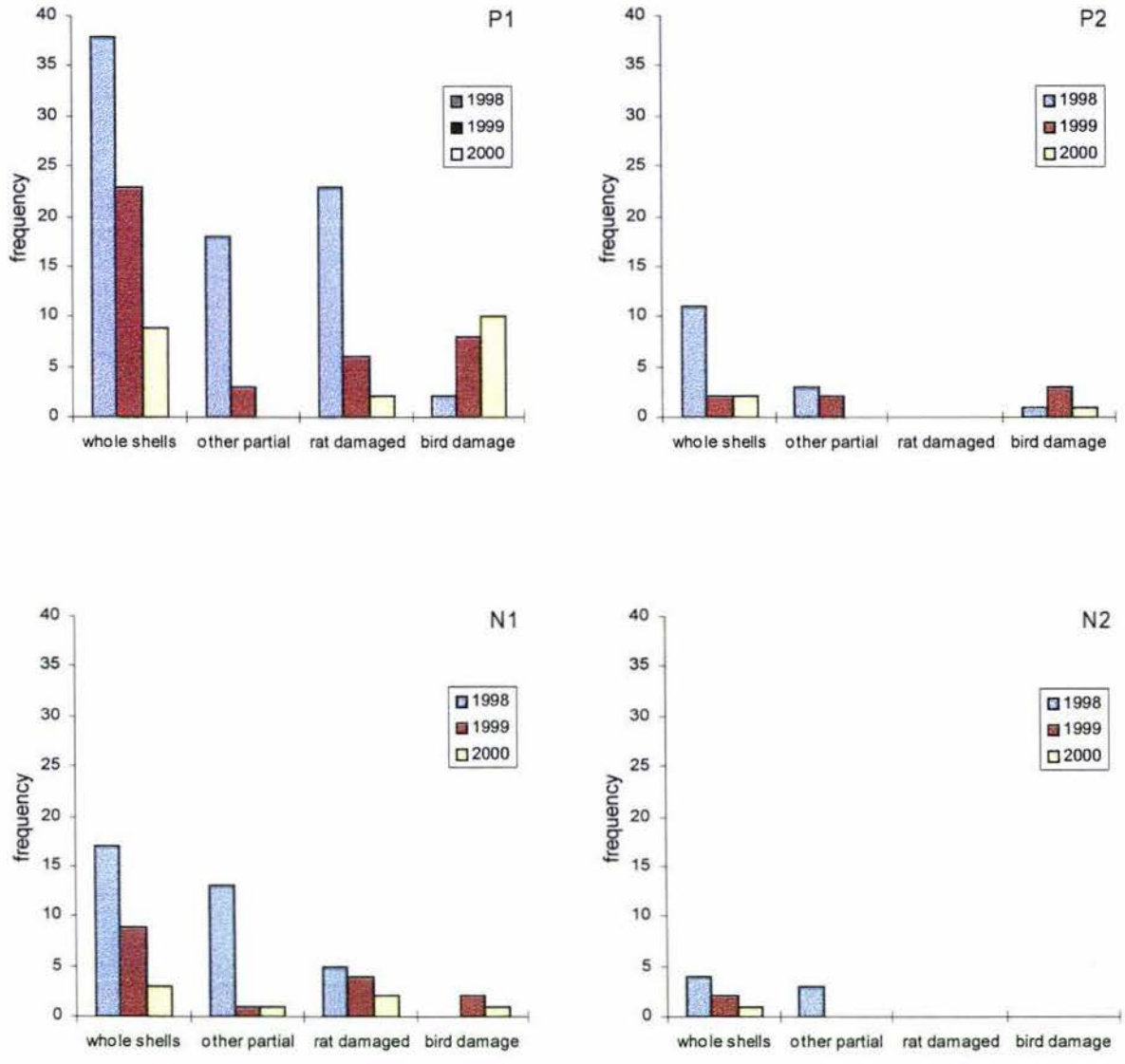

Figure 3.3.7 Numbers of empty *P. t. traversi* shells found during searches in each area.

Empty shell size

When the total numbers of whole, rat damaged, bird damaged, and other partial shells were examined, the majority of whole and rat damaged shells were in the 3 largest size classes (40.00 -54.99 mm) (Figure 3.3.8). By comparison, partial shells were distributed across all size classes and bird damaged shells found were less than 25 mm (with the majority less than 20 mm).

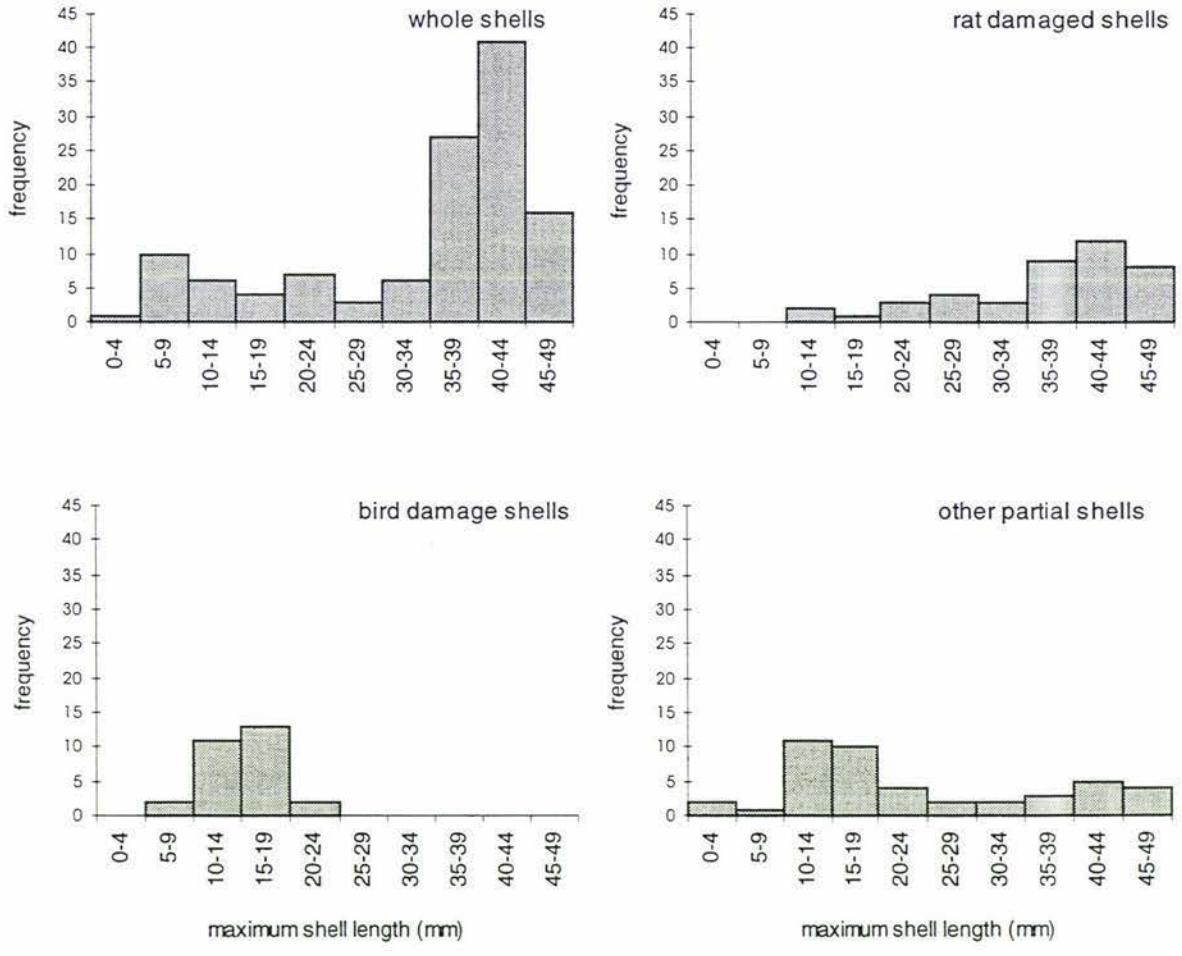

Figure 3.3.8 Size-frequency distributions (maximum shell length) of whole, rat damaged, bird damaged, and other partial shells.

There was a highly significant difference ($P<0.01$, Chi-square test) between the four types of empty shell, when the size-frequency distributions were compared. The high number of empty whole shells between 40.00 - 49.99 mm and bird damaged shells between 10.00 - 19.99 mm contributed most to the significant result.

3.4 **DISCUSSION**

The primary purpose for poisoning at Lake Papaitonga Scenic Reserve was to control rats as they are thought to be the primary predator of *P. t. traversi* snails in the Horowhenua (Meads *et al.*, 1984; Devine, 1997). This appeared to be reasonable considering that rat damage accounted for the majority of empty shells with determinable damage found before poisoning began. In addition, the hope was that mice numbers could also be controlled at Lake Papaitonga because brodifacoum has been used to control mice elsewhere (Brown & Singleton, 1998; Dowding *et al.*, 1999). If the observed rat and mice track indices are assumed to directly reflect actual population densities, then poisoning successfully achieved both objectives.

However, control was not immediate in some instances and poisoning did not appear to control mice as strongly as rats in either of the poisoned areas. If mice are predators of *P. t. traversi* snails, then long term poisoning may not be as effective at reducing the threat of mice depredations as it appears to be in reducing the threat of rat depredations. These observations may be related to the possibility that once rodent numbers were reduced by poisoning the removal of the remaining individuals became increasingly harder because the relative abundance of natural food increased (Thomson, 1992). In addition, the presence of bait may not induce rats to leave their home ranges (Innes, 1977). This makes the positioning of bait stations critical if effective control is to be achieved.

The increase in the mouse track index in area P2 after a 100% decline following the first application of poison is similar to that reported in other studies where mice numbers were observed to increase following initial declines after poisoning (Innes *et al.*, 1995; Miller & Miller, 1995; Hunt *et al.*, 1998). Innes *et al.* (1995) showed that increases in mice were highest in areas where rats had been most reduced. This was also the case at Lake Papaitonga. Of the two poisoned areas, P2 had the highest rat tracking index before poisoning and the greatest decline in rat tracking once poisoning began, and it was here that greatest increase in the mouse tracking index was observed during poisoning.

Rat populations can recover within 2-5 months of a single poison drop and mice and rat tracking may increase at the same rate (Innes *et al.*, 1995). The continuous monthly baiting

at Lake Papaitonga shows that rat abundance can be kept at low levels in a small area, but that mice abundance may still increase during poisoning. Similar results were reported in other studies where mice increased when poison was still available (Innes *et al.*, 1995; Gillies & Pierce, 1999). Furthermore, when baiting was not done for 2 months (from February to April 2000), the rat track indices remained low while mice abundance in all four areas increased markedly. This suggests that mice numbers increase much faster than rats. These observed differences between mice and rats, and between areas, highlight the importance of monitoring targeted pest species during control operations. Surprisingly, this consideration has been overlooked occasionally in poisoning operations when investigating the impact of predators on land snails (e.g. Walker, 1997b; Sherley *et al.*, 1998).

The major outcome of this study is that a significant increase in live *P. t. traversi* occurred (in P1) when the abundance of rodents (inferred from tracking indices) was reduced by poisoning and the relative proportion of *P. t. traversi* depredations by rodents decreased. Such a result, is similar to that reported by Churchfield *et al.* (1991) who found significantly more large gastropods when small mammal predators (namely shrews) were excluded from grassland plots in England over a 2 year period. There are clearly differences between that study and my own, in both the predators involved and the habitat, but the results of poisoning at Lake Papaitonga supports the suggestion that small mammal predators can have a major impact on terrestrial invertebrates because of their capacity for high population densities and consumption rates (Churchfield *et al.*, 1991).

Sherley *et al.* (1998) recorded significant changes in the demography of *Placostylus ambagiosus paraspritítus* (Powell) with rodent control, mostly from relative increases in numbers of juveniles, but adult recruitment required at least 8 years of rodent control. The increase in *P. t. traversi* numbers at P1 after 19 months was not the result of a significant change in demography, but the increase in numbers of live snails was greatest in size classes >40 mm. Similarly, there was an increase in the numbers of live snails >30 mm at P2 in 1999 which resulted in a significant change in demography. These increases correspond to the suggestion that rat predation was reduced, as more shells between 35.00 - 49.99 mm were found with rat damage than shells <35 mm.

The most similar study to my work, on the basis of predator control, habitat, and location, involved 18 years of rodent poisoning at Hillas covenant, a protected native bush remnant in the Wellington region containing *P. t. otakia* snails. The number of live snails increased from 0 to 10 after 3 years of poisoning, and to 42 after 8 years of poisoning. Since then, however, the number of live snails has fluctuated and the reason for these changes is not known because monitoring has not been regular enough (Kath Walker, 2000, *pers. comm.*). An obvious consideration is that comparatively low rat densities were maintained at Lake Papaitonga because baiting was done monthly. No rodent monitoring was done at Hillas covenant, baiting was less regular (every 2 - 3 months), and the area poisoned was only about 300 m^2 (Kath Walker, 2000, *pers. comm.*). Consequently, rodent control may not have been as effective as in my study because of reinvasion of rats.

Innes and Barker (1999) pointed out that a 'pulse' regime to control vertebrate pests is likely to be ineffective in benefiting invertebrates whose adults would be vulnerable to predation at inter-pulse periods. Growth of *P. t. traversi* is indeterminate and they do not develop any morphological attributes as adults that afford them protection from rodent predation. In contrast, *Placostylus* snails have determinate growth, developing a thickened varix as adults, and this prevents rats from preying on them (Sherley *et al.*, 1998). In the latter case, pulse control may be suitable to establish adult numbers. However, if predation of *P. t. traversi* by rodents is to be controlled, continual efforts must be made so that rodent densities remain at reduced levels for adult snail numbers to increase sufficiently to allow increased reproduction.

Poisoning did not appear to be as effective at P2 as at P1 in eliciting a significant increase in *P. t. traversi* numbers. There was an increase in the numbers of live snails in 1999, but there was an overall decrease in 2000. Furthermore, no live snails <25 mm were ever found at P2 and only a small number of empty shells this size were found. Considering that no rat damaged shells were ever found in P2, factors other than rat predation may have caused the absence of small *P. t. traversi* and the decline in snail density at this area. One explanation could be that there were insufficient numbers of reproductive adults to produce young. Numbers of small land snails can also be reduced in areas where carnivorous *Rhytida* (Albers) snails are in high concentrations (Climo, 1975). *R. greenwoodi greenwoodi* (Gray) was present at Lake Papaitonga, and the greatest numbers were in P2. It is possible then that

predation by invertebrates could be one reason for the absence of small snails. Furthermore, predation of *P. t. traversi* by invertebrates would be underestimated as they may leave little or no obvious shell damage.

Alternatively, depredations by mice may be a greater threat to small snails than rats are at P2. This is because mice are able to attack *P. t. traversi* shells up to approximately 27mm (see Appendix 1) and on the basis of tracking indices, mice were not controlled as effectively as rats at P2 (no mice tracks were observed on only 5 occasions (out of 17) compared with 10 occasions for rats). However, shell damage caused by mice is similar to rat damage (*pers. obs.*) so any small shells predated by mice were diagnosed as rat damaged.

Another possible reason for the general lack of small live snails at Lake Papaitonga may lie in the increase in proportion (and number) of bird damaged shells observed in both poisoned areas. Blackbirds (*Turdus merula* L.) and song thrushes (*Turdus philomelos* Brehm) occur throughout the reserve and both are recognised predators of *Powelliphanta* snails (Meads *et al.*, 1984). Both species are considered equally responsible for snail depredations because they probably cause the same type of shell damage even though each typically uses a different method of snail extraction. Song thrushes tend to use anvils to smash open snail shells (Heather & Robertson, 1996), while blackbirds hammer and prise open the shells (Meads *et al.*, 1984). Predation of *P. t. traversi* by these two species was limited to small snails (<25 mm maximum shell length) so their depredations may be a primary cause of the low numbers of these small snails. Meads *et al.* (1984) warned that shells at anvils and the inability of searchers to find small-predated shells could influence an accurate measure of bird predation, but the smallest shell found each year during this study was never a bird-damaged shell and no obvious shell deposits were found as have been seen elsewhere in the Horowhenua (*pers. obs.*). Therefore, the observed changes in bird damaged shells are believed to accurately reflect actual changes in predation by these birds.

Predation by rats on New Zealand's native avifauna is well documented (Atkinson, 1978; Brown *et al.*, 1998). Furthermore, ship rats and mice also kill blackbirds and song thrushes in New Zealand (Kikkawa, 1966; Moors, 1983). Therefore, rats may reduce snail predation by birds by influencing bird numbers. Alternatively, predation by rats and mice could have reduced the availability of the smaller snails that these birds typically attack. Of course,

birds visiting the bush from outside the immediate area may have been just as responsible for the predation at Papaitonga. If this is the case, the ratio between reserve area and boundary length might affect the level of this type of predation (Devine, 1997). However, Devine (1997) did not report any bird damaged shells from seventeen 100 m² quadrats he searched at Lake Papaitonga. This is surprising considering that bird damaged shells were found in 9 of the 16 quadrats searched during my study. This difference suggests that bird predation may only be recent at Lake Papaitonga, and that the increase in bird damaged shells was the result of natural variation in bird foraging (e.g. site preference and visit-frequency). Such reasons could also account for the small increase in the number of bird damaged shells in N1 where the rat tracking index remained high.

Hunt *et al.* (1998) suggested that studies investigating the benefits to invertebrates of sustained poisoning for rodents require around 6 years of field work, including 2 years before pest control. Such efforts were far beyond the scope of this study, but some of the observations support this suggestion. Determining the true effectiveness of poisoning would have been easier if pre-poisoning monitoring had been done to determine natural fluctuations in both snail and rodent densities. Dowding and Murphy (1994) showed that ship rat home ranges may change during the year and vary greatly between different areas, and urged caution when interpreting data from tracking tunnels and comparing different areas. This is especially important when no historical patterns are known. The inability to carry out snap-trapping to calibrate tracking indices with actual rat and mice numbers (e.g. Brown *et al.*, 1996) also meant the true effectiveness of poisoning in reducing rat densities and the accuracy of tracking tunnels at Lake Papaitonga was not known.

Despite these complications, there were positive outcomes from my study. Because snail distribution varied greatly between the areas established in this study, searching four 100 m² quadrats in each area, rather than a single larger quadrat (e.g.Walker, 1997a), was worthwhile because it showed that the variation between areas was generally consistent within each area. Terrestrial molluscs frequently have an aggregated or non-random distribution (Peake, 1978), and this appears to be the case for *P. t. traversi* at Lake Papaitonga where there are greater snail densities in P1 and N1 compared with P2 and N2. Devine (1997) also found that 100 m² quadrats were of sufficient size to effectively sample *P. t. traversi* at Lake Papaitonga. In future studies, searching sufficient numbers of smaller

quadrats could be more productive in terms of sampling variation in the snail population, and it is more suitable than searching very large quadrats if there are only a small number of searchers.

The *P. t. traversi* population estimate from my search data (between 13678 and 23120 snails) is greater than, but includes most of the 95% confidence interval calculated by Devine (1997) (between 13488 and 15219 snails) for the same area (35.9 ha.). This suggests that the *P. t. traversi* population at Lake Papaitonga has not declined since Devine (1997) sampled the population, and has possibly increased slightly.

Poisoning not only resulted in an increase in *P. t. traversi* numbers in P1 but also appeared to have additional benefits for Fantails (*Rhipidura fuliginosa placabilis* Bangs) in the area. Fantails nests are known to be plundered by rats (Moors, 1983) and there was an increase in breeding activity and number of fledged chicks (from 2 to 10) following poisoning at P1 (David Mudge, 1999, *pers. comm.*). An increase in Fantail numbers was observed at P1 during the experiment and fledging rates were well above average for the area following the commencement of poisoning (David Mudge, 1999, *pers. comm.*). By comparison, no chicks were seen in either of the non-poisoned areas. No observations were made in P2.

Meads *et al.* (1984) believed that Norway rats were the primary predator of *P. t. traversi* in the Horowhenua but recognised that both Norway and ship rats may exhibit similar predator sign. I found one dead ship rat near area P2, and the most recent trapping efforts only caught ship rats (Devine, 1997). Ship rats therefore appear to be the most likely predator of *P. t. traversi* at Lake Papaitonga, yet despite their depredations these snails are still able to reach maximum sizes there. The largest live snail found as part of my study was 55.54 mm (ML), and this was larger than the maximum sizes reported for this subspecies by either Powell (1979) (53.5 mm) or Devine (1997) (54 mm). In addition, a 58.19 mm snail was collected in a pitfall trap at N2 (Melissa Hutchinson, *unpubl. data*). This suggests that even with the current threats to these snails, *P. t. traversi* populations with similar snail numbers should contain reproductively mature individuals.

Brodifacoum was widely used to control both rats and mice, both on the mainland and offshore islands (e.g. Innes *et al.*, 1995; Ogilvie *et al.*, 1997; Brown & Singelton, 1998;

Dowding *et al.*, 1999; Empson & Miskelly, 1999; Gillies & Pierce, 1999). The increase in the numbers of live *P. t. traversi* when rodent poisoning was done supports continued rodent control (poisoning and/or other methods) to benefit populations of these endangered snails in small native remnants. However, comparison with other rodent control efforts suggests that any control efforts must be intensive and constant in order to kill as many rodents as possible to maintain reduced rodent densities. In addition, mice may increase even when poison is present. If mice are regular predators of *Powelliphanta*, then their effects may not be significantly reduced by poisoning alone. Predation of small *Powelliphanta* by blackbirds and song thrushes is of concern because they may prevent the recruitment of young snails and their effects would be difficult to control.

No conclusive evidence was produced that rodent poisoning uniformly benefited *P. t. traversi* at Lake Papaitonga, because increases in the numbers of live snails were observed in all four areas after 12 months of rodent control in the two poisoned areas. However, the significant increase in the numbers of live snails at P1 after 19 months of poisoning is encouraging. Rodents remain the primary identifiable predator of *P. t. traversi* snails at Lake Papaitonga, in which case rodent control should be looked at as a necessary management strategy for the conservation of these snails at this site, and should be considered for other populations in bush remnants in the Horowhenua District. In addition, rodent snap-trapping should be done to confirm the incidence of *P. t. traversi* in the diet of rodents, and the effectiveness of rodent control. Regular (presumably annual) monitoring of all *P. t. traversi* populations would be worthwhile in providing valuable information about the state of the different formae of this subspecies, and would give an insight into variations in snail numbers caused by various predators at different sites.

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CHAPTER 4

The effects of *Tradescantia fluminensis* (Vell.) on movement and demographics of *Powelliphanta traversi traversi* (Powell).

ABSTRACT

The movements of 40 *Powelliphanta traversi traversi* (Powell) snails in leaf litter and *Tradescantia fluminensis* (Vell.) at Prouse Bush, Levin, were observed using harmonic radar. The mean daily displacement for snails was greatest in leaf litter (0.201 m/day) but did not differ significantly from that in *T. fluminensis* (0.064 m/day). There was large variation in the movements observed and a highly significant difference between individual snails ($P<0.01$), and some snails regularly moved between the different habitats. Calculated home ranges suggested *T. fluminensis* affected home range size, snails always found under *T. fluminensis* had significantly smaller 90% home ranges (43.91 m^2) ($P<0.05$) than those snails found only in leaf litter or those that changed habitat (171.35 m^2 and 610.14 m^2 respectively). There was a highly significant difference between the size-frequency distributions of live snails found in litter and *T. fluminensis* ($P<0.01$) and the only snails >35mm were found in leaf litter. There was no significant difference in live snail densities between these habitats. In contrast, there was no significant difference between the size-frequency distributions of empty shells found in both habitats, but their density was significantly greater in leaf litter ($P<0.05$). *P. t. traversi* effectively use *T. fluminensis* as a habitat and this seems to be especially favored by small individuals. Possible reasons for the differences in movement and distribution between the habitats are discussed.

4.1 INTRODUCTION

Extant populations of *Powelliphanta traversi traversi* (Powell) now only survive on the lowland plains of the Horowhenua District (lower North Island), in predominantly small (<5 ha.) isolated native bush remnants. Here they are found under leaf litter during the daytime, moving mainly at night when moisture is high (Devine, 1997). The original distribution of this endemic land snail is not known because much of the native bush in the Horowhenua and Manawatu regions was rapidly cleared in the late 19th century.

Several of these remnants are currently threatened by the invasive weed *Tradescantia fluminensis* (Vell.) (Commelinaceae). This weed has the potential to cover the forest floor once it has established (Kelly & Skipworth, 1984a), and hence it can potentially destroy the

original habitat of *Powelliphanta* snails. *Tradescantia fluminensis* is a perennial, herbaceous species native to tropical regions (Maule *et al.*, 1995). It is commonly known as wandering Jew (Healy, 1984) and is typically found in areas of New Zealand with medium to high soil fertility (Ogle & Lovelock, 1989; Atkinson, 1997). It has rapidly invaded and become established in native lowland forest in the North and South Island of New Zealand (Maule *et al.*, 1995), as well as on at least five offshore islands (Atkinson, 1997).

Tradescantia fluminensis was first introduced into the Manawatu District in 1910 to stabilise banks (Kelly & Skipworth, 1984a). It has since dispersed via streams (Esler, 1978), and has been spread in garden refuse dumped near forest remnants (Timmins & Williams, 1991) and by livestock (Ogle & Lovelock, 1989). *Tradescantia fluminensis* is a threat to New Zealand native forests because it can form a dense mat that may limit seedling survival and growth, thus affecting forest regeneration in the long term (Kelly & Skipworth, 1984a, 1984b).

There is a high positive correlation between light intensity and the standing crop of *T. fluminensis* (Kelly & Skipworth, 1984a). While the deleterious effects of the weed may be minimised (less standing crop) by maintaining low light levels in native forest, *T. fluminensis* is shade tolerant (Timmins & Mackenzie, 1995) and it is able to persist for an indeterminate period of time at low light levels once it is established. It can still form a dense cover in shade (Atkinson & Cameron, 1993) and growth is possible at irradiance levels from 1% to 90% normal daylight (Maule *et al.*, 1995).

Manual methods have been used in the past to control *T. fluminensis*, and removal by hand can successfully clear small areas (Ogle & Lovelock, 1989). However, hand weeding is not a feasible control method for large areas and may be ineffective in some situations (Brown & Rees, 1995). The ability of *T. fluminensis* to establish from small stem fragments represents a major drawback. Alternatively, various herbicides are effective in controlling *T. fluminensis* in native bush (Kelly & Skipworth, 1984b; Ogle & Lovelock, 1989; Timmins & Mackenzie, 1995; McCluggage, 1998) and the method may be more cost effective and less labour intensive than manual methods. Hence, herbicide use currently represents the best option for *T. fluminensis* control.

Tradescantia fluminensis infestation of natural areas and the potential for further infestation is high in the Wellington region (Timmins & Mackenzie, 1995). If *T. fluminensis* continues to spread through native areas in the Horowhenua, the application of herbicide may be one management decision necessary for the long-term conservation of natural areas and their constituents.

Powelliphanta traversi traversi has been found in high densities underneath *T. fluminensis* (Ian Cooksley, 1994, *pers. comm.*, cited in Devine, 1997) and it has been suggested that these snails have only survived at some sites because of the presence of this weed (Anon., 1996). Consequently, these snails are not only at risk from any detrimental effects *T. fluminensis* may have on them or their habitat but also from removal of the weed and the methods used to do so. The distribution of *T. fluminensis* at sites which contain these snails is usually patchy (Ogle & Lovelock, 1989). Thus, the spatial distribution of *P. t. traversi* snails within affected remnants needs to be quantified to determine the likelihood of any detrimental effects of herbicide use. If this can be achieved, we may be able to predict with greater certainty the likely impacts of *T. fluminensis* removal on *Powelliphanta* snails, including both direct and indirect effects.

The only known intensive study of distribution and movement of *P. t. traversi* is that by Devine (1997) at Lake Papaitonga Scenic Reserve. However, he did not study the relationship between *T. fluminensis* and this snail although the weed was present. As such, this is the first known investigation into the use of *T. fluminensis* by any *Powelliphanta* snail and may be the first detailing its use by a New Zealand land snail species.

The main aim of the research in this chapter was to determine the extent of true use of *T. fluminensis* affected habitat by *P. t. traversi*. This involved estimating snail density, demographic composition and snail movements within and between the two habitat types at Prouse Bush, Levin (see Chapter 2 for site details). Preliminary observations showed that *T. fluminensis* comprised up to 55% of the ground cover in Prouse Bush. To my knowledge, there had been no previous work with *P. t. traversi* at this site other than brief population estimates, but snails were known to have been found under *T. fluminensis* there.

4.2 METHODS

The movements of *P .t. traversi* fa. traversi were examined in Prouse Bush, Levin (NZMS 260 S25 028 611), between July 1998 and July 2000. Long-term movements (with intervals >1 month) were followed by finding the snails with harmonic radar (see Chapter 2 for description of method). Short-term movements (24-72 hours) were followed by attaching a spool and thread to the shells of the snails (see Chapter 2 for description of method). Individual snails were followed for different lengths of time depending on when they were first found and when they died or disappeared.

Harmonic Radar

Searches were made by hand of leaf litter and *T. fluminensis* for snails that were large enough (maximum shell length >34 mm) to have an harmonic transponder attached to their shell (see Chapter 2 for method).

Snails with transponders were relocated every month between July 1998 and July 2000 with the exceptions of October 1999, and May and June 2000. Harmonic radar allowed the area where the snails were situated to be narrowed down to about 0.25 m^2, minimising the amount of habitat disturbed to find the snails.

The maximum shell length, maximum shell width, maximum aperture width (see Chapter 2 for description of measures) and weight were measured each time a snail was found, and its position relative to a marked reference point and habitat was recorded. Marked trees were used as reference points. These were selected for their proximity to several snails. The depth of leaf litter was measured to the nearest centimetre, and the maximum stem height was taken for *T. fluminensis* in the ranges 0-30 cm, 30-50 cm, and >50 cm. The relative percentages of litter and *T. fluminensis* together with their depth/height were measured in areas that included both habitat types (referred to here as 'combination' habitat).

Movements were converted to average daily displacements when comparing snails followed for different lengths of time. When a snail with a transponder showed no change in position prior to its death, this was included in the movement analysis. Successive positions of snails

were mapped using SigmaPlot (v. 1.02, Jandel Corporation, 1994), and home range estimates were made (adaptive kernel method) using CALHOME (v. 1.0, Californian Forest Service, 1994).

A logarithmic transformation was used to normalise the data when daily movements and home ranges were compared for snails within leaf litter, *T. fluminensis* and combination habitat. Snail movements and home ranges were compared using a generalised linear model procedure with SAS (SAS Inc. 1996). Because movement data was repeatedly collected from individual snails, there was the potential for some correlation between the movements of an individual snails. This was reduced by fitting a term for each individual to the general linear model used for movement analysis.

Short term movements

Movements using spools and thread as a method (see Chapter 2 for description of method) were followed in leaf litter, *T. fluminensis*, and combination habitat on a total of 29 nights between March 1999 and February 2000. Twenty snails with transponders and five without transponders had spools attached to their shells. When a snail moved, the amount of thread pulled out and the distance between the snails initial and final position was measured to the nearest millimetre.

Snail distribution

Searches for snails were carried out in leaf litter and in *T. fluminensis* to compare the density and distribution of snails in these habitats. Additional data was collected from a further search of litter and *T. fluminensis* in July 2000, which included large areas within *T. fluminensis* patches (up to 25 m^2). These latter searches were left until the end of this study because it was destructive and damaged the habitat. Previous searches to find snails in *T. fluminensis* for attaching transponders to had only been in small areas at the edges of *T. fluminensis* patches (approximately 4 m^2). Different sized areas were searched in each habitat so live snail and empty shell densities were calculated for the area of each search rather than for the total area of all searches. Size-frequency distributions of live snails and

empty shells found in the two habitats were compared using Fisher's exact test and Chi-square using SAS (SAS Inc. 1996).

4.3 **RESULTS**

Harmonic Radar

Snail movements

Forty (40) snails had transponders attached to their shells. These ranged in maximum shell length between 34.68 - 55.53 mm. Of these, 23 were found in leaf litter, 16 in *T. fluminensis*, and 1 in combination habitat (Figure 4.3.1).

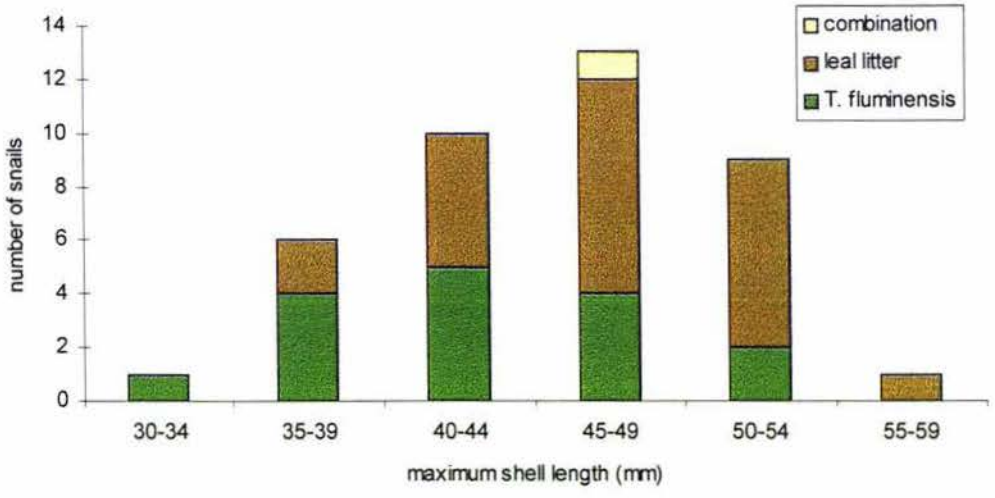

Figure 4.3.1 Size-frequency distribution (maximum shell length) of snails with harmonic radar transponders in Prouse Bush (n=40). The maximum length of one snail in *T. fluminensis* (48.72mm) was estimated because of damage to the aperture lip.

All 40 snails with transponders attached were each relocated at least once, and 30 were relocated 5 or more times (see Appendix 2 for summary of relocations). The largest number of times that one snail was relocated was 19, and this occurred over 662 days (snail 12). However, the longest time one snail was followed was 692 days during which it was relocated 17 times (snail 6).

In total, 371 movements were recorded using harmonic radar. Of these, 199 were by snails in leaf litter, 121 in *T. fluminensis*, 10 in combination habitat, and 39 were movements from one habitat to another (Figure 4.3.2). The remaining 2 movements were by 2 different snails that moved from one patch of *T. fluminensis* to another, and could have only done so by traveling over leaf litter. Because these changes in habitat were not measured, the movements were not used in the movement analysis. The fact that only 10 consecutive relocations were made in combination habitat was likely due to the small amount of combination habitat at Prouse Bush (compared with leaf litter and *T. fluminensis*) and because snails had to only travel a small distance to move into either of the other habitats.

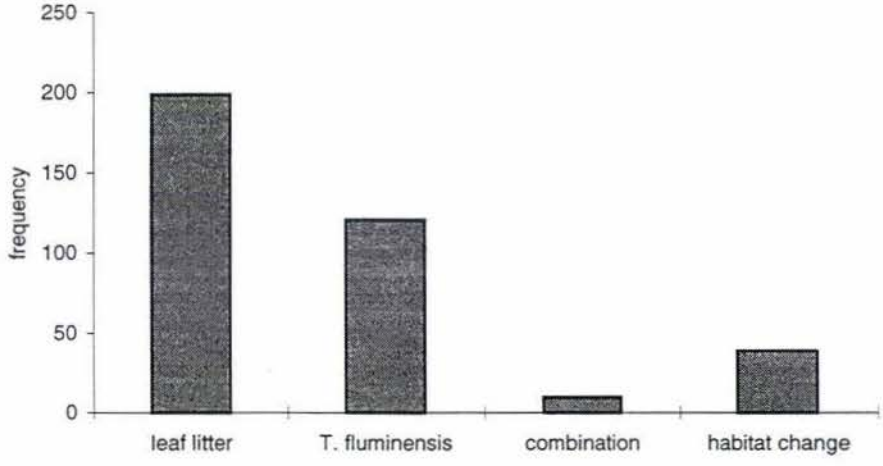

Figure 4.3.2 Numbers of movements by snails with transponders in three different habitats and between these habitats in Prouse Bush, observed using harmonic radar relocations of snails (n=371).

Mean displacement (the average distance between relocations) was greatest for movements when a snail changed habitat (7.21 m), followed by movements in leaf litter (5.48 m), *T. fluminensis* (2.71 m), and combination habitat (0.35 m) (Table 4.3.1). However, when a snail changed habitat it was relocated on average less frequently (38.64 days) than snails that remained in leaf litter or *T. fluminensis* (36.10 and 35.68 days respectively). Consequently, mean daily displacement was actually greatest for movements in leaf litter (0.201 m/day), followed by movements between habitats (0.169 m/day), in *T. fluminensis* (0.064 m/day), and in combination habitat (0.031 m/day) (Table 4.3.1).

Table 4.3.1 Comparison of movements by snails in three habitats and when a snail changed habitat. All displacement measures refer to distance in metres. Mean no. days shows the average time between relocations.

	Leaf litter	*T. fluminensis*	Combination	Change	Total
No. movements	199	121	10	39	371
Range of displacements	0.05-28.13	0.05-18.71	0.01-1.08	0.11-44.55	0.01-44.55
Mean no. days	36.10	35.68	39.90	38.64	36.18
Mean displacement	5.48	2.71	0.35	7.21	4.42
Mean daily displacement	0.201	0.064	0.031	0.169	0.150

A general linear model fitted to the log transformed daily displacement data showed a highly significant difference in daily displacement between the snails followed ($P<0.01$) and between the different habitats ($P<0.01$). Thus, even when the variation between individual snails was controlled, there were still highly significant differences in daily displacement between the four types of habitat use. However, the model only accounted for 34% of the variation ($r^2 = 0.3439$). Hence, while differences between individual snails and habitats were significant, these factors alone were not sufficient to explain most of the variation in daily displacement.

Mean daily displacements for the different movements were compared with Tukey's, Duncan's and Least squares means procedures (SAS) to establish significant differences ($P<0.05$). Tukey's test gave no significant difference in mean daily displacement between movements in leaf litter, when a snail changed habitat, or in *T. fluminensis*. Mean daily displacement of all three were significantly greater than for movements by snails in combination habitat, but again the area of combination habitat was small and few observations were made ($n=10$). In contrast, Duncan's and Least squares means procedures showed that the difference in daily displacement between leaf litter and *T. fluminensis* was significant, but both procedures are less conservative than Tukey's test.

Of the 39 movements observed where snails moved between habitats, movement from *T. fluminensis* into leaf litter was undertaken by more *P. t. traversi* (8 snails) and more often (9 movements) than any other change in habitat (Table 4.3.2). Only 15 different snails changed habitat. Five snails changed habitat just once, but six changed habitat three or more times. Snail 19 was the most active with regard to this, and changed habitat 6 times in 483 days (including all three habitats).

Table 4.3.2 Habitat changes made by 15 snails with transponders attached to their shells. A change in habitat was shown by successive relocations in two different habitat types.

Change in habitat	no. observations	no. snails
T. fluminensis → Combination	7	6
T. fluminensis → Leaf litter	9	8
Leaf litter → Combination	6	6
Leaf litter → *T. fluminensis*	8	6
Combination → *T. fluminensis*	5	4
Combination → Leaf litter	4	4

The greatest total distance moved by a snail with a transponder was 214.42m (over 627 days) (snail A4, Appendix 2) and included three movements between leaf litter and *T. fluminensis*. The smallest total distance was 0.05 m (over 66 days) (snail 18, Appendix 2).

For those snails relocated more than once (n=37), the distance from the position it was last found to its point of origin, on average accounted for 37.85% of the distance it actually moved (inferred from displacement measures) (range 3.28 - 100%) (Appendix 2).

Home ranges

Home ranges were calculated to give the area of 50% and 90% of a snail's distribution. This could only be done for 36 and 27 of the snails respectively because of the small number of relocations for the other snails. For comparative analysis, home ranges were categorised as either leaf litter, *T. fluminensis*, or combination (both litter and *T. fluminensis*) based on where a snail had been found in the field. This did not take into account whether a home range calculated for snails only ever found in one type of habitat included areas of alternative habitat. The effect of snail size on home range size was tested by dividing the maximum shell length of the snails into size-classes.

Mean 50% and 90% home range sizes were greatest for snails that were observed to have changed habitat (130.25 and 610.14 m^2 respectively), followed by snails in leaf litter (24.65 and 171.35 m^2), and in *T. fluminensis* (6.18 and 43.91 m^2) (Table 4.3.3).

Table 4.3.3 Mean home range sizes (m^2) and sample sizes for snails in leaf litter, *T. fluminensis*, and in both habitats (combination). The values shown were calculated using the adaptive kernel method (CALHOME).

Home range	Leaf litter	*T.fluminensis*	combination
50%	24.65 (n = 16)	6.18 (n = 6)	130.25 (n = 14)
90%	171.35 (n = 13)	43.91 (n = 3)	610.14 (n = 11)

A general linear model fitted to the data showed no significant effect of habitat, snail size or habitat-size interaction on 50% home range size ($P>0.05$). In contrast, a general linear model for the 90% home range data showed a significant effect of both habitat and snail size ($P<0.05$), but no significant interaction effect ($P>0.05$). The mean 90% home range for snails that changed habitat (610.14 m^2) and for those only found in leaf litter (171.35 m^2) were significantly greater than for snails always found in *T. fluminensis* (43.91m^2) ($P<0.05$, Tukey's test).

Short term movements

A total of 97 paired observations of thread length and displacement were made of snails with spools attached: 10 in combination habitat, 31 in *T. fluminensis*, and 56 in leaf litter. However, movement was observed on only 39 (38.25%) instances: 2 in combination habitat, 9 in *T. fluminensis*, and 28 in leaf litter. For these movements, the mean amount of thread pulled out was 0.283m compared with a mean displacement of 0.207 m (Table 4.3.4), and for 19 movements (48.72%) displacement was equal to the amount of thread pulled out.

Table 4.3.4 Summary of movements in leaf litter and *T. fluminensis* by *P. t. traversi* snails with spools attached to shells. Thread length and displacement measures are shown in metres.

		Leaf litter	*T. fluminensis*	combination	Total
No. of movements		28	9	2	39
Range -	thread length	0.034-1.302	0.048-0.410	0.112-0.119	0.034-1.302
	displacement	0.000-0.722	0.000-0.326	0.112-0.119	0.000-0.722
Mean -	thread length	0.324	0.173	0.151	0.283
	displacement	0.230	0.131	0.151	0.207

Of the observations made of snails in each habitat, snails in leaf litter moved proportionally more often (50%) and further (0.324 m mean thread length) than snails in *T. fluminensis* (29%, 0.173 m) or combination (20%, 0.151 m). When a snail did move, movement was regularly in a single direction with little deviation in each habitat. This likely accounted for the high percentage of movements explained totally by displacement (48.72%). However, there were also instances when a snail made several changes in direction or even backtracked and no displacement was observed.

Linear regression of the 39 paired observations (SAS) produced Equation 4.3.1. and accounted for r^2 (adjusted) = 80.96% of the variation between thread length and displacement. The positive y intercept (0.040747m) shows that the spool and thread method detected movement that displacement did not. However, this difference was small and was likely the outcome of only relatively short distances ever being observed using this method (1.302 m maximum thread length) and because many of the movements were in a single direction meaning that they were completely explained by a displacement measure (48.72%).

Equation 4.3.1. Displacement = 0.040747 + 0.58766(thread length)

Snail distribution

Snail density in litter and T. fluminensis

Many more live snails were found in *T. fluminensis* (54) than in leaf litter (11) during searches (Table 4.3.5). Another 18 live snails were found in leaf litter but could not be included in the density analysis because they were not found while searching a measured area. Only slightly more empty shells were found in leaf litter (48) than in *T. fluminensis* (43).

Table 4.3.5 Results of searches to estimate *P. t. traversi* density in leaf litter and *T. fluminensis* at Prouse Bush.

	Leaf litter	*T. fluminensis*
Area searched	285 m²	293 m²
No. live snails	11	54
No. empty shells	48	43
No. searches with no live snails	5	11
No. searches with no shells	2	17
Average live snail density (*std. deviation*)	0.0479 m² (*0.0675*)	0.2519 m² (*0.4735*)
Average empty shell density (*std. deviation*)	0.1706 m² (*0.1759*)	0.1242 m² (*0.2065*)

A general linear model fitted to the log transformed density data showed that there was no significant difference in live snail density between leaf litter and *T. fluminensis* ($P>0.05$). The model did find a slight significant difference in the density of empty shells between the two habitats ($P<0.05$). However, both models were not able to provide a very good fit to the data ($r^2 = 0.1023$ and 0.1154 respectively). This was likely due to the proportionally large number of searches where no live snails or empty shells were found in both habitats, and suggests there was substantial natural variation in density between searches.

Snail size in litter and T. fluminensis

There was a smaller mean maximum shell length for live snails and empty shells found in *T. fluminensis* (29.82 mm and 31.16 mm respectively) than in leaf litter (47.16 mm and 49.53 mm respectively). Furthermore, live snails in leaf litter were all greater than 35 mm maximum length (Figure 4.3.4). By comparison, live snails found in *T. fluminensis* had a greater size range, and this included the smallest found (4.86 mm).

The size-frequency distributions for empty shells found in leaf litter and *T. fluminensis* were similar (Figure 4.3.5). There appeared to be two modes for empty shell size in liter and *T. fluminensis* separated by an absence of empty shells measuring between 30.00-34.99 mm. Of the empty shells found in leaf litter more were in the larger size classes, as for live snail sizes (Figure 4.3.5).

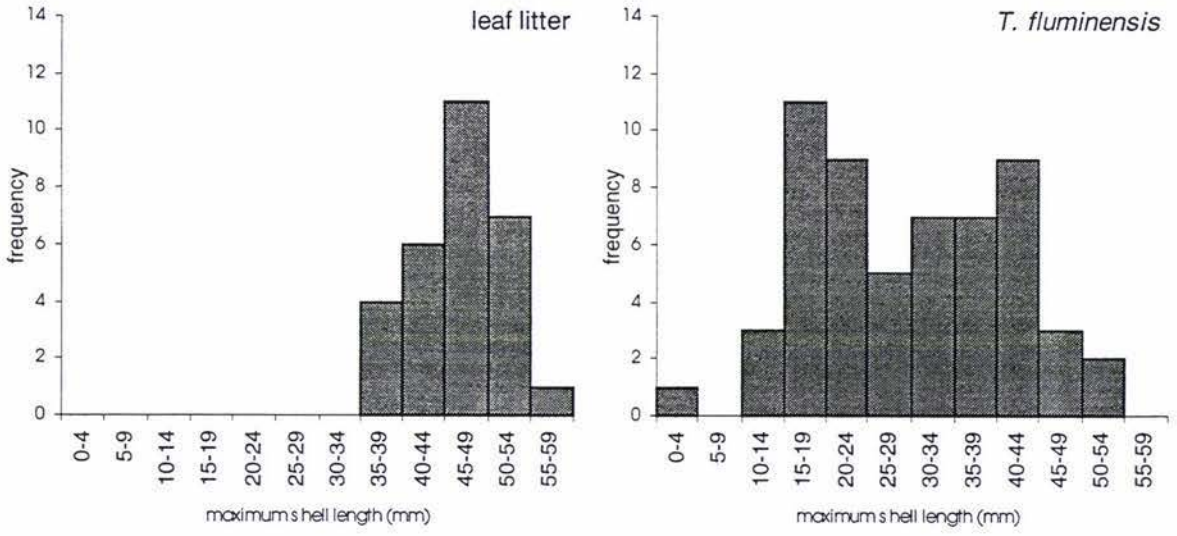

Figure 4.3.3 Frequency distributions of maximum shell length for live snails found in leaf litter (n=29) and *T. fluminensis* (n=57).

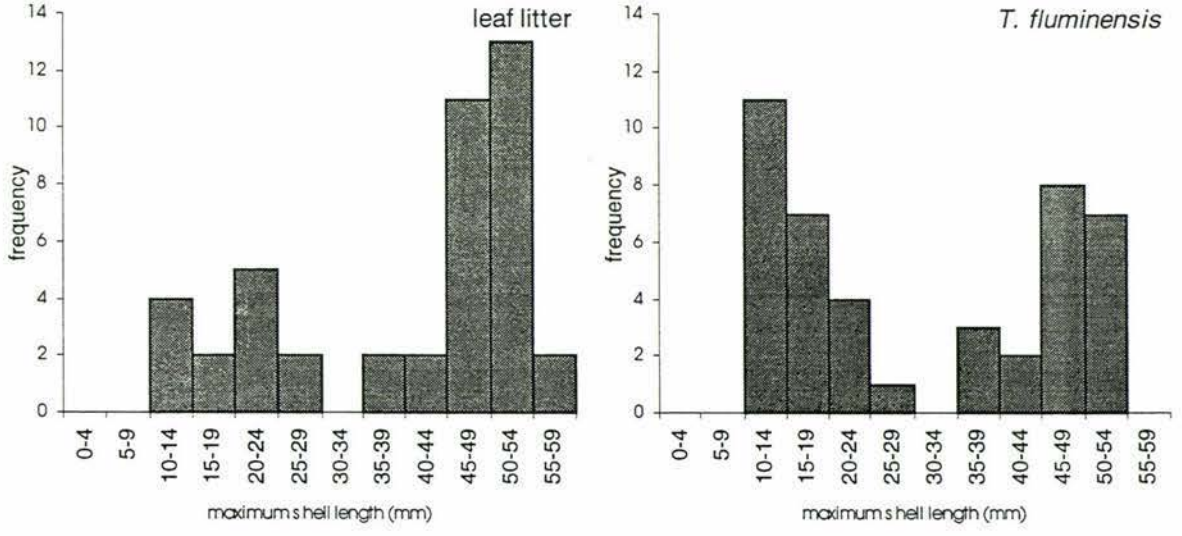

Figure 4.3.4 Frequency distributions of maximum shell length for empty shells found in leaf litter (n=43) and *T. fluminensis* (n=43).

The size-frequency distributions for all live snails and empty shells found in leaf litter (29 and 43 respectively) and *T. fluminensis* (58 and 43 respectively) were compared using Fisher's exact test (SAS). Analysis showed a highly significant difference in the size-frequency distributions for live snails between the two habitats ($P<0.01$), but there was no significant difference between the size-frequency distributions for empty shells ($P>0.05$).

Snail deaths

Twenty-four snails with transponders attached (58.54%) died between August 1998 and July 2000. Death was only suspected at first for 6 snails and was confirmed by subsequent relocations.

For 23 known deaths, the cause of death could not be determined (i.e. no shell damage). Thirteen (54.17%) of these were in leaf litter, 7 (29.17%) in *T. fluminensis*, and 3 (12.50%) in combination habitat. The remaining known death (4.17%) was the result of rat predation on a snail in *T. fluminensis*. There was rat damage to the aperture lip of one snail found in *T. fluminensis* and the aperture and periphery of one snail found in leaf litter prior to attaching transponders. Both snails later died but their deaths could not be attributed to the damage.

Snails with transponders attached were found dead each month in leaf litter over a ten month period during the study and there was no apparent pattern to this (Figure 4.3.6). The three snails found dead in combination habitat were found in March (33.33%) and April (66.66%) but so few snails were ever observed in this habitat that little can be taken form this small variation in time. Five of the seven dead snails (71.43%) found dead in *T. fluminensis*, were in the summer months (December-February).

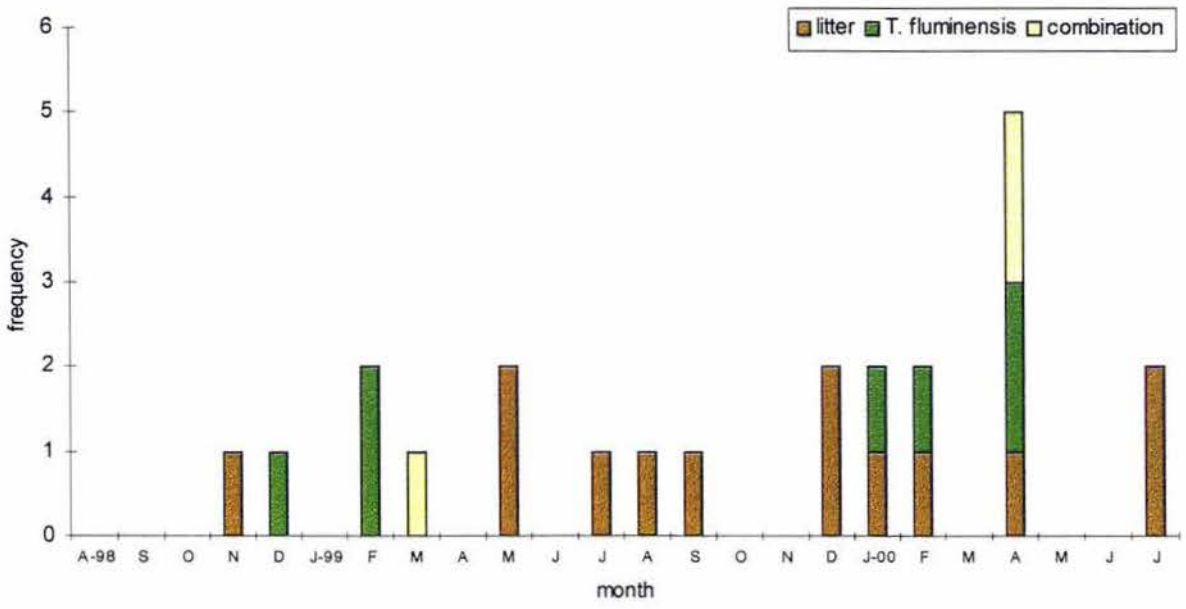

Figure 4.3.5 Time of year when dead snails with transponders were found in the three different habitats at Prouse Bush (n=23). One snail lost to rat predation is not shown. The distribution shows when each snail was confirmed dead and not when they necessarily died.

4.4 **DISCUSSION**

There is substantial overlap of habitat use by *P. t. traversi* and *T. fluminensis* in Prouse Bush. More live snails were found in this weed than in leaf litter and many snails were only ever repeatedly found in *T. fluminensis*. Furthermore, some snails regularly moved between the two habitats and occasionally moved relatively large distances often over short periods. Mean daily displacement by *P. t. traversi* in leaf litter (their natural habitat) and in *T. fluminensis*, was not significantly different suggesting that *T. fluminensis* has little influence on the distances the snails moved. The greatest mean daily displacement was for *P. t. traversi* that changed habitats, but fewer than half of the snails followed moved between habitats. The most important factor influencing the movements of *P. t. traversi* appeared to be individual variation between snails.

Devine (1997) provides the best comparison for this study. The short term movements by *P. t. traversi* at Prouse Bush differ from results gathered by him at nearby Lake Papaitonga Scenic Reserve. The displacement explained a large proportion of the actual overnight movement by *P. t. traversi* snails (80.96%) at Prouse Bush as ascertained by spool and thread tracking, and this was greater than reported by Devine (1997) (67.3%). The nightly

movement by *P. t. traversi* snails (80.96%) at Prouse Bush as ascertained by spool and thread tracking, and this was greater than reported by Devine (1997) (67.3%). The nightly movements of many snails at Prouse Bush appeared to be in one predominant direction, a phenomenon noted also at Lake Papaitonga Scenic Reserve (Devine, 1997), as well as for *Paryphanta busbyi busbyi* (Gray) (Coad, 1998), and *Oxychilus helveticum* (Blum) (Verdcourt, 1947). However, movement in one direction would rarely occur over long distances (Verdcourt, 1947). Thus, because the largest distance travelled during a single night at Prouse Bush (1.302 m) was much smaller than that reported by Devine (1997) (5.58 m) and far less than that reported in *P. b. busbyi* (9.5m (Coad, 1998)), this may have influenced the improvement in displacement accuracy. Spool tracking may have produced errors when the thread became tangled. This study reinforces Verdcourt's (1947) conclusion and shows that displacement can only be expected to explain a certain percentage of the actual distance a snail has travelled. Caution is therefore needed over longer periods of time when the actual distance not explained by displacement may become large.

Tradescantia fluminensis did appear to affect the home range of *P. t. traversi* because snails that were only found in *T. fluminensis* had significantly smaller home ranges. This suggests that although the average distances moved daily were not significantly different between snails in litter or *T. fluminensis*, overall dispersal by snails in *T. fluminensis* was smaller than in leaf litter or for snails that moved between habitats. The reason for this difference may be related to the differences between the sizes of snails in different habitats at Prouse Bush. Live snails in leaf litter were significantly larger so they may have had greater tendency to move further than the overall smaller snails in *T. fluminensis*. Pomeroy (1969) showed that juvenile *Helicella virgata* were more active and moved further than adults, whereas Verdcourt (1947), did not consider size to be an influence (within a species) the speed of a snail because resistance to motion would increase with size and muscular power. He did consider the rate of snail dispersal to be influenced by the actual speed of the snail, but noted that this factor was in turn dependent on atmospheric conditions, the nature of the surface, the internal state of the animal, and its food supply. Devine (1997) showed that *P. t. traversi* is mostly nocturnal and activity is determined primarily by moisture related factors.

My own casual observations suggested that *T. fluminensis* supported a greater abundance of larger soil invertebrates than leaf litter at Prouse Bush so more food may be available under

T. fluminensis. Soil moisture beneath *T. fluminensis* also appeared to be higher than under leaf litter. I was able to collect a small amount of temperature data (11 days) that suggested temperature was only slightly lower on average under *T. fluminensis* (6.70 °C) than under leaf litter (7.09 °C) at Prouse Bush. Despite little apparent difference in temperature, if there was more food and/or moisture under *T. fluminensis* than in leaf litter, then there may be less need for snails in *T. fluminensis* to disperse as far as those in leaf litter. Coad (1998) found correlations between the density of *P. b. busbyi* and the relative abundance of earthworms, and Verdcourt (1947) considered that food abundance would play a considerable role in snail dispersal because snails do not need to leave habitats with ample food. High prey density was shown to affect behavioural patterns and the speed of movement of some carabid beetles including increased turning frequency and patch residence times (Wallin, 1991; Wallin & Ekbom, 1994). Lower dispersal by *P. t. traversi* in *T. fluminensis* may therefore reflect similar responses to prey densities.

The possibility that *P. t. traversi* adhere to a home range or territory was suggested by Devine (1997), and homing behaviour has been proposed or demonstrated for other mollusca (e.g. Cook *et al.*, 1969; Gelperin, 1974; Pollard, 1975; Cook, 1977; Peake, 1978; Lorvelec, 1990; Coad, 1998). Never the less, whether *P. t. traversi* do display homing tendencies is still not clear. If they do have home ranges then it is clear that those of several snails may overlap greatly. This was also apparently the case for *P. b. busbyi* (Coad, 1998). Although Devive (1997) suggested that *P. t. traversi* had home ranges, he admitted that the existence of a home range still "does not preclude the possibility that movement may be random within the home range". Verdcourt (1947) also recognised the existence of random movements by snails "beyond human perception". Personally, I have reservations over placing much importance on consideration of home ranges because of unknown differences between individual snails and the difficulty in determining the true purpose of certain movements (e.g. homing; searching for food, mates, egg laying sites, or a more suitable micro-environment). Evans (1972, cited in Peake, 1978) summarised these problems when he stated "the ecology of a snail species may enable it to occur in a number of, what to a man's eye are different, environments, but which to the snail are identical".

Many *P. t. traversi* snails found in leaf litter at Prouse Bush had a thickened peristome but snails found under *T. fluminensis* typically did not. Thickening of the aperture lip is believed

to represent shell growth under less favorable conditions (Kath Walker, 2000, *pers. comm.*) that results in a reduction in aperture size. This may be a response to help retard water loss under increased evaporative conditions brought about because of the degraded state of Prouse Bush. Activity of terrestrial snails is generally governed by moisture related factors (e.g. Pomeroy, 1969; Machin, 1975; Pollard, 1975) so the state of the bush may also explain the difference between the mean daily average for *P. t. traversi* movements in leaf litter at Prouse Bush (0.201 m/day) with those at Lake Papaitonga Scenic Reserve (0.293 m/day (Devine, 1997)) where evaporative conditions may be less because the vegetation is less degraded.

The reason why there is a significant difference in live snail sizes between the two habitats is unclear but this may also be related to the state of the bush. Water is lost by evaporation from all exposed surfaces (including slowly from the shell) of a terrestrial pulmonate's body and this is affected by environmental parameters such as humidity, temperature, and wind speed (Machin, 1975). Such factors are in turn affected by fragmentation of bush, with smaller remnants being more affected because of proportionally greater edge effects (Saunders *et al.*, 1991). At Prouse Bush, no live snails <35 mm were ever found in leaf litter while the majority of snails found under *T. fluminensis* were <35 mm. Furthermore, the largest empty shell and live snail were found in leaf litter. Smaller snails are more at risk from desiccation than larger snails because of their proportionally larger surface to volume ratio. The fact that nearly twice as many snails followed with harmonic radar died in leaf litter than in *T. fluminensis* suggests environmental conditions under *T. fluminensis* may be more suitable for smaller snails. There is also the chance that sampling error influenced the significant difference because there was substantial variation in numbers of empty shells and live snails between different areas and many snail populations show a marked degree of clumping or non-random distribution (Pomeroy, 1969; Peake, 1978).

The Department of Conservation (DoC) has been cautious concerning the removal of *T. fluminensis* from some sites because of a perceived beneficial effect of *T. fluminensis* to *P. traversi* snails (Anon., 1996). This was also recognised by the Horowhenua District Council (Strong, 2000) who administer Prouse Bush, so they have not undertaken any steps to control this weed. The abundance of *P. t. traversi*, the apparent maintenance of home ranges in areas of *T. fluminensis*, and the presence of small snails in this weed at Prouse Bush (when

none were found in leaf litter) all support the suggestion that these snails may benefit from some environmental factor(s) in the presence of *T. fluminensis*. Devine (1997) found that karaka (*Corynocarpus laevitagus*) was associated with areas of high *P. t. traversi* snail numbers at Lake Papaitonga Scenic Reserve, leading him to suggest that karaka may reflect good snail habitat. Similarly, vigorous growth of *T. fluminensis* is considered to indicate high fertility soils (Ogle & Lovelock, 1989; Atkinson, 1997), which may reflect sites preferred by *P. t. traversi* snails.

Predation by introduced species is another serious threat to *P. traversi* snails (Climo, 1975; Meads *et al.*, 1984) but it is unclear how *T. fluminensis* may influence the predation rate on *P. t. traversi*. Blackbirds (*Turdus merula* L.) and Song thrushes (*Turdus philomelos* Brehm) are known predators of smaller *P. t. traversi* (Meads *et al.*, 1984) and I found bird damaged shells in litter but none under *T. fluminensis*. It seems likely that these birds would find it difficult to locate snails under areas where this weed forms a dense mat. The same cannot be said for rat predation. The only snail with a transponder known to be killed by a rat was in *T. fluminensis*. The aperture and outer whorl were removed, damage characteristic of predation by a Norway rat (*Rattus norvegicus* Berkenhout) (Meads *et al.*, 1984). A second live snail found in *T. fluminensis* had survived an attack to its aperture and the only empty shells damaged by rats were found in *T. fluminensis*. All the latter had damage consistent with Norway rat predation. Norway rats are found in most towns and cities (Moors, 1990). The only dead rat found at Prouse Bush was, however, a Ship rat (*Rattus rattus* L.).

Despite this, rats and birds appear to have only a minor effect on *P. t. traversi* at Prouse Bush compared with snails at Lake Papaitonga Scenic Reserve (see Chapter 3 for comparison). Businesses near Prouse Bush may have the potential to support high rodent numbers, but domestic cats (*Felis catus* L.) at Prouse Bush may restrict rodent and bird numbers. In addition, rodent numbers may have been affected by poison in bait stations present adjacent to one side of Prouse Bush. No rat tracks were recorded in Prouse Bush using tracking tunnels in leaf litter and *T. fluminensis*, but mouse tracks were recorded from one tunnel in *T. fluminensis* and another next to an area of *T. fluminensis*.

In summary, this study shows that *P. t. traversi* snails commonly occur under *T. fluminensis* throughout Prouse Bush and that there are regular movements of snails between leaf litter

and this weed. Furthermore, some snails were only ever found under *T. fluminensis* over an entire year. Hence, the control of *T. fluminensis* does have the potential to affect many snails. *Tradescantia fluminensis* apparently provides a refuge for smaller snails. Hence, mortality of these snails in particular (either by increased environmental or predation impacts) may increase if *T. fluminensis* is removed and alternative suitable habitat is not available.

Recent trials have shown Grazon® herbicide to be effective in controlling *T. fluminensis* (e.g. Brown & Ress, 1995; McCluggage, 1998), so investigation of its toxicity to *P. t. traversi* is warranted. If this herbicide is toxic, further trials (spray concentrations, spray regimes, other herbicides) would still be valuable as herbicides appear to provide the quickest and most cost effective method of weed control (McCluggage, 1998). Alternatively, because the area affected by *T. fluminensis* is relatively small it could be cleared using manual methods. This would likely require continued monitoring and weeding, but any snails under the weed could then be moved to suitable areas of leaf litter.

Because the common aim of *T. fluminensis* control is to increase regeneration of native bush, restoration work prior to weed control by planting seedlings in areas of *T. fluminensis* should be considered. This is being investigated by Rachel Standish (*in prep.*). This may prevent or lessen the effects of removing *T. fluminensis* on *P. t. traversi* snails in areas where leaf litter is sparse in degraded bush remnants.

My results cannot be applied to other sites and snail species without further study. However, *P. t. traversi* in other sites affected by *T. fluminensis* in the Horowhenua are most likely to display similar dynamics to those at Prouse Bush. Investigations by students (e.g. Penniket, 1981; Devine, 1997; Coad, 1998) have extended our knowledge of the ecology of New Zealand native snails greatly and should continue to be supported. My work provides valuable information about the potential remnant-scale distribution of *P. t. traversi* and the possibility that control of *T. fluminensis* may have negative effects on these snails. In light of these results caution is recommended when contemplating the control of weeds in native bush.

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CHAPTER 5

Effects of Grazon® herbicide on the Brown Garden snail (*Cantareus aspersus* (Müller)) and *Powelliphanta traversi traversi* (Powell).

ABSTRACT

The toxicity of a 1.4% Grazon® solution (active ingredient triclopyr), known to control the invasive weed *Tradescantia fluminensis* (Vell.), was investigated using three life history stages of the brown garden snail (*Cantareus aspersus* (Müller)) and 5 individuals of the protected snail *Powelliphanta traversi traversi* (Powell). The mean mortalities of *C. aspersus* exposed to a direct spray and to a sprayed environment (82.34% and 78.40% respectively) were significantly greater than the mortality in a control treatment (36.95%) ($P<0.05$) after 149 days, suggesting that Grazon was toxic to *C. aspersus*. However, there were apparent differences in the toxicity between the three life stages. Overall egg mortality (86.00%) was significantly greater than adult and juvenile snail mortality (66.86 and 62.20% respectively) ($P<0.05$). Five *P. t. traversi* snails were exposed to a single environmental spray of Grazon but none died after 149 days, and all increased in weight and all but one in maximum length. Two *P. t. traversi* in a control group produced 10 eggs, and three snails exposed to Grazon produced 11 eggs, but none had hatched by the conclusion of the experiment. A 1.4% Grazon solution does not appear to be toxic to *P. t. traversi* snails when sprayed on leaf litter where the snails live. The most likely reason for these results is that *C. aspersus* is herbivorous while *P. t. traversi* is carnivorous. Thus, *C. aspersus* ingested the Grazon but *P. t. traversi* did not. The results suggest that Grazon is a suitable herbicide to control *T. fluminensis* containing *P. t. traversi*.

5.1 INTRODUCTION

Certain site characteristics such as proportionally large borders, degraded open canopies, close proximity to towns and high human use are important factors that influence the number of weeds present in New Zealand natural reserves (Timmins & Williams, 1991). Many native forest remnants in the Horowhenua District contain the endemic land snail *Powelliphanta traversi traversi* (Powell) and have these characteristics so they are susceptible to weed invasion. One weed that has successfully invaded many of these lowland remnants, including sites containing *P. t. traversi*, is *Tradescantia fluminensis* (Vell.) (Commelinaceae).

Tradescantia fluminensis (or wandering Jew) was first introduced to the neighbouring Manawatu District for stabilising steep banks (Kelly & Skipworth, 1984a) and it was also a

popular ornamental garden plant. The weed has spread on its own accord via streams (Esler, 1978), but it has also been spread by livestock (Ogle & Lovelock, 1989) and through garden refuse being dumped by forest remnants. New plants can successfully establish from fragments > 1 cm long (Kelly & Skipworth, 1984a), and once introduced to a bush remnant, the high soil fertility and moderate rainfall (Esler, 1978; Ravine, 1995) that generally occurs in the Horowhenua district favors its establishment.

The extent of *T. fluminensis* infestation appears to be determined by light levels in forest, so that open areas at forest edges, by stream banks and beneath canopy gaps are the most heavily infested (Kelly & Skipworth, 1984a; Brown & Rees, 1995; Maule *et al.*, 1995; Standish *et al.* (*submitted*)). However, *T. fluminensis* can occupy a site, even at low light levels, for a very long period because it is very efficient at recycling nutrients (Maule *et al.*, 1995). At high levels of infestation, *T. fluminensis* forms a carpet up to 0.6 m tall which can out-compete native ground plants and seedlings (Kelly & Skipworth, 1984a; Brown & Rees, 1995; Standish *et al.*, *submitted*) so it will eventually affect forest regeneration (Timmins, 1995). Clearly, invasion by *T. fluminensis* is a threat to the natural habitat of *P. t. traversi*.

The development of an effective and cost effective method to control *T. fluminensis* has been investigated for some time, and until recently, most control techniques had mixed success. Hand weeding and physical removal may rid small areas of the weed, but results can vary because of simple differences in weeding technique (Ogle & Lovelock, 1989). Furthermore, manual removal is not feasible for large areas, and it has also proven ineffective in some situations (Brown & Rees, 1995). It is labour intensive and the ability of *T. fluminensis* to grow from small stem fragments represents a major drawback.

The effectiveness of various herbicide sprays and concentrations for controlling *T. fluminensis* were tested in response to these problems (e.g. Kelly & Skipworth, 1984b; Brown & Rees, 1995; Timmins & Mackenzie, 1995; McCluggage, 1998). They showed that effective control using herbicide is possible and may be cheaper than mechanical control (e.g. Ogle & Lovelock, 1989). Despite these advantages, continued annual inspection and further control action is likely to be required for effective long-term control. This was estimated to require up to four years at one site in the Manawatu (Ogle & Lovelock, 1989).

Paraquat, glyphosate, and triclopyr have each received attention as possible control agents for *T. fluminensis*. Paraquat was effective in reducing the standing crop of *T. fluminensis* by over 50%, but it also damaged native plants (Kelly & Skipworth, 1984b) and it is toxic to both animals and to the operator (Ogle & Lovelock, 1989). Glyphosate (the active ingredient in Roundup®) was suggested as a herbicide for *T. fluminensis* in the Manawatu, by using an initial spraying with Roundup followed by hand weeding or spraying of re-growth in following years (Ogle & Lovelock, 1989). Kelly and Skipworth (1984b) found it was not as effective as paraquat in reducing the standing crop of *T. fluminensis*, but they used a concentration lower than necessary to control *T. fluminensis* (Brown & Rees, 1995; McCluggage, 1998).

Recently, herbicides containing the active ingredient triclopyr (or 3,5,6-trichloro-2-pyridyloxyacetic acid) have proven highly effective in controlling woody plants and broadleaf weeds (e.g. Whisenant & McArthur, 1989; Mislevy *et al.*, 1999). One registered triclopyr herbicide in New Zealand is Grazon® (DowElanco) which has already been used in New Zealand to control *T. fluminensis* in field trials (Brown & Rees, 1995; McCluggage, 1998). It proved to be cost-effective and it killed all treated *T. fluminensis* (McCluggage, 1998).

Triclopyr is present in Grazon as a butoxyethyl ester which was shown to be toxic to fish and daphnids (Wan *et al.*, 1987; Kreutzweiser *et al.*, 1994). However, the ester is only present in the environment for a short time before it is converted by contact with soil or water to the triclopyr acid, which has low toxicity to fish, daphnia, birds, mammals, and bees, (Gersich *et al.*, 1984; Mayes *et al.*, 1984; Anon., 1991; Holmes *et al.*, 1994) and is non-toxic to cattle (Eckerlin *et al.*, 1987). The primary route of breakdown of triclopyr is through aerobic microbial action in soil and by sunlight in water. There is no evidence that triclopyr accumulates in aquatic or terrestrial food chains.

By comparison to their breakdown in water, triclopyr residues remain in soils and plants for long periods and are known to increase with an increase in herbicide dose (Siltanen *et al.*, 1981). Triclopyr residues in plants are highest immediately after application but have been detected 365 days (Norris *et al.*, 1987; Whisenant & McArthur, 1989) and 13 months (Siltanen *et al.*, 1981) after application. Triclopyr has also been shown to remain in soils for

up to 365 days (Norris *et al.*, 1987), with peak concentrations tending to occur some time after spraying, due to the delayed transference of residues to the soil with wash-off from sprayed vegetation (Norris *et al.*, 1987). In a study by Rachel Standish (*unpubl. data*), triclopyr residues in soil beneath *T. fluminensis* sprayed with a 0.66% Grazon solution were maximal 61 days later, and were still detectable 141 days after the application.

The toxicity of triclopyr to invertebrates has received much attention (e.g. Gersich *et al.*, 1984; Kreutzweiser *et al.*, 1992; Kreutzweiser *et al.*, 1994; Maloney, 1995), but snails have not been routinely tested (Melo *et al.*, 2000). The effects of other herbicides on snails was investigated using aquatic species (e.g. Rajyalakshmi *et al.*, 1996; Tate *et al.*, 1997; Melo *et al.*, 2000), and *Cantareus aspersus* (Müller) was used by Salama and Radwan (1995) to test the metabolism of the insecticide phorate by adult snails. However, no studies are known that involve triclopyr and terrestrial pulmonates. One study of the effects of triclopyr on invertebrates in New Zealand indicated that there was no change in benthic aquatic macroinvertebrate abundance or species composition attributable to its use (Maloney, 1995). Despite this, the potential toxic effect of Grazon still needs to be evaluated with respect to other New Zealand environments and biota.

Because of its success in recent trials (Brown & Rees, 1995; McCluggage, 1998) Grazon appears to be the best option for controlling *T. fluminensis* amongst the range of herbicides presently available in New Zealand. Consequently, there is a need to evaluate the effect of Grazon itself on native species that may be at risk. In the Horowhenua, *P. t. traversi* snails fall into this category as they are endangered and have been found under *T. fluminensis* in several native bush remnants.

In this study, the toxic effects of Grazon were first tested on the introduced brown garden snail (*C. aspersus*) (Gastropoda: Stylommatophora). This snail is common in coastal areas of the North Island including disturbed native forest (Barker, 1999). It has also been found in *T. fluminensis* together with *P. t. traversi* in bush remnants in the Horowhenua. My intention was to investigate the toxicity of Grazon on *P. t. traversi* snails following the outcome of the experiment with *C. aspersus*.

This chapter outlines experiments to investigate the toxic effect of a single application of a 1.4% Grazon solution to both snail species. This concentration of Grazon solution is the highest known to have been used to control *T. fluminensis* (McCluggage, 1998) so it was used for these experiments in order to be sure as possible of its potential toxicity to *P. t. traversi*.

5.2 METHODS

Preliminary trials with the brown garden snail

Three age classes of *C. aspersus* (referred to as adult snails, juvenile snails, and eggs) were each given a single spray application of 1.4% Grazon solution. *C. aspersus* snails develop a reflected peristome (aperture lip) upon reaching adult size, but snails lacking a deflected lip may successfully reproduce (Barker, 1999). Following Johnson and Black (1991), the presence of a reflected lip was used as the operational classification of an adult snail, and juvenile snails lacked a reflected lip.

The toxicity of Grazon to each age class was tested using 3 treatments (Table 5.2.1). Treatment 1 (Direct spray) and treatment 2 (Environmental spray) were different methods of exposure to Grazon and treatment 3 (Control) was an experimental control. There were also two replicates of the direct and environmental spray treatments, and one of the control treatment. Fifteen adult snails, 40 juvenile snails, and 50 eggs were used for each replicate.

Table 5.2.1 Experimental procedure of the three treatments used for each age class of the brown garden snail.

Treatment	Action
1 – Direct Spray	Grazon solution sprayed on snails/eggs, their food and environment
2 - Environmental Spray	Grazon solution sprayed on food and environment only (snails/eggs reintroduced)
3 – Control	Tap water sprayed on snails/eggs, their food and environment

Spraying was done using a Solo 425 (Germany) knapsack sprayer (15L capacity). This was thoroughly rinsed with clean tap water before use, and it was used to spray water for the

control treatment first. Spraying was done until the environment was thoroughly wet following the manufacturers instructions (DowElanco).

Adult and juvenile snails were kept in glass aquaria of varying sizes (between 37x20x19 cm and 54x50x34 cm) with a gravel substrate, and eggs were kept in 2 L plastic containers beneath a layer of soil. All were kept at temperatures of 20° to 22° C in the same controlled temperature room. Snails were fed fresh organically grown lettuce and supplied with calcium carbonate powder each week. The aquaria of adult and juvenile snails and the soil covering the eggs were kept damp with distilled water using an atomiser.

Snail mortality was monitored weekly for 149 days after spraying. Empty shells or shells containing dead snails were removed and their sizes recorded. Overall mortality figures for adult and juvenile snails were taken after omitting any accidental deaths or snails that were lost.

Egg containers were checked weekly for 4 weeks for any snails that hatched. At four weeks, the soil was then removed and a search was made for any remaining eggs, snails that had hatched or empty shells. Eggs that had failed to hatch by this time were considered to be infertile or dead.

Powelliphanta traversi traversi

Ten *P. t. traversi* fa. tararuaensis/traversi hybrids were collected from Shannon Forest (Ernslaw One Ltd.) (NZMS 260 S25 147 639). Snails were housed individually in 44 L plastic tanks and kept at between 14 - 17°C, with a light cycle of 11:13 hrs (light:dark) in a controlled temperature room. These tanks were provided with a layer of gravel covered with a layer of soil and native leaf litter, and were kept damp with distilled water from an atomiser. Snails were acclimatised to the conditions for 43 days before the experiment commenced.

Five of the tanks with *P. t. traversi* were sprayed with a 1.4% Grazon solution using a knapsack sprayer. One snail died before the experiment began so only 4 tanks with these snails were sprayed with tap water as a control group. In both cases the snails were not

sprayed directly but were present under the leaf litter during the spraying. Spraying was done until the leaf litter was thoroughly wet.

Once a week, each snail was placed in a 2 L plastic ice-cream container with some leaf litter for two consecutive nights and presented with at least 2 garden earthworms. The snails readily ate the worms after one or two nights. The snails were also weighed before and after being fed and the maximum shell length, maximum shell width, and maximum aperture width were measured each week.

Results were analysed using SAS (SAS Inc. 1996). A general linear model with repeated measures was used to compare mean mortality between the different treatments. Multiple comparisons between treatment and life history stage combinations were made using Tukey's, Duncan's, and least squares means procedures.

5.3 **RESULTS**

Cantareus aspersus

Juvenile snails suffered some mortality after 6 days in all three treatments, but the highest mortality occurred amongst those in the environmental spray treatment. By comparison, the first adult deaths occurred after 11 days in the environmental spray treatment, and not until after 52 days in the direct spray treatment, or 59 days in the control. The initial average mortality rate was also higher for juveniles than for adults in both the direct and environmental treatments. There was a steady increase in the cumulative mean mortality of adults and juveniles in the direct and environmental spray treatments (Figure 5.3.1 and 5.3.2).

The mean cumulative mortality of adult snails in the environmental spray treatment was consistently higher than any other treatment throughout the 149 days of the experiment (Figure 5.3.1). In contrast, the mean cumulative mortality of juvenile snails remained highest in the direct spray treatment (Figure 5.3.2).

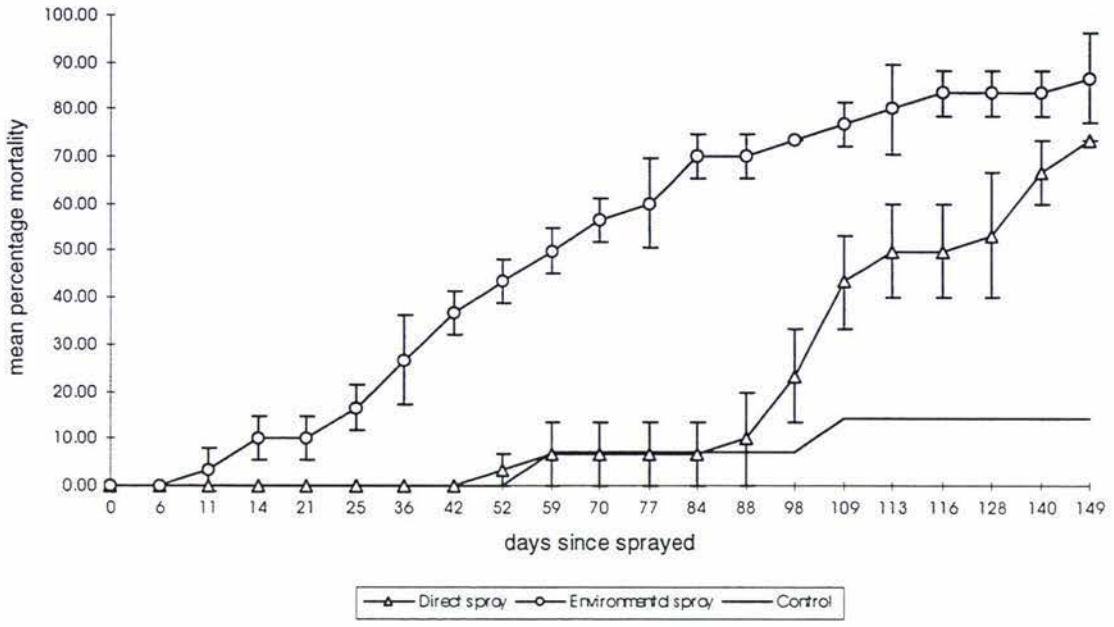

Figure 5.3.1 Cumulative mean (± S.E.) percentage mortality of adult *C. aspersus* in each treatment over 149 days.

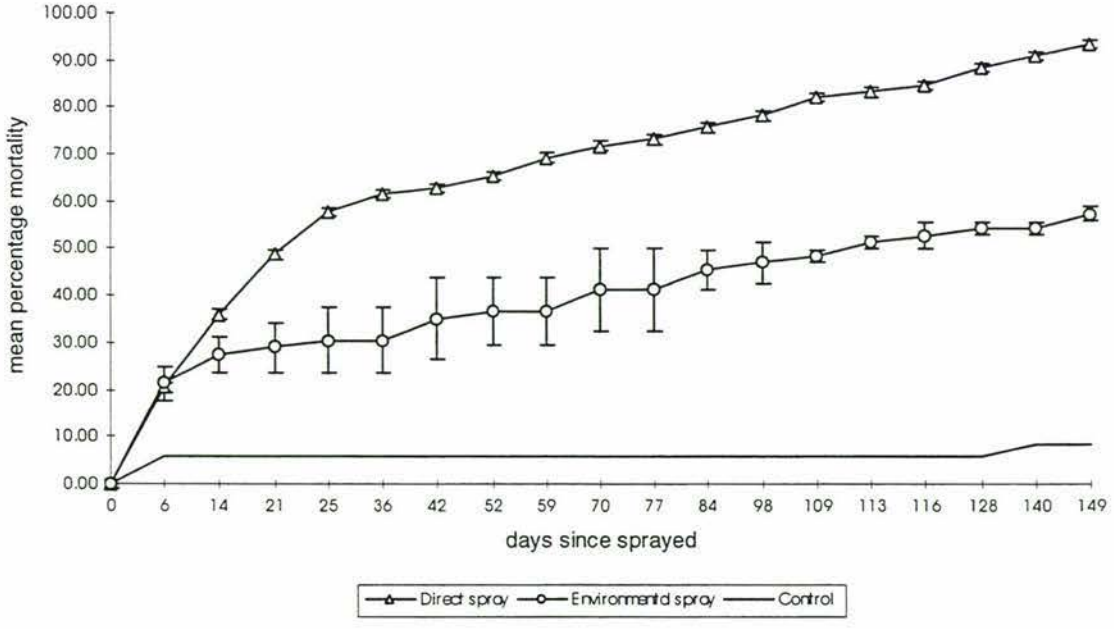

Figure 5.3.2 Cumulative mean (± S.E.) percentage mortality of juvenile *C. aspersus* in each treatment over 149 days.

After 149 days, mean percentage mortality was highest for adult snails and eggs in the environmental spray treatment (86.67% and 91.00% respectively) (Figure 5.3.3), whereas mean percentage mortality of juvenile snails was greatest in the direct spray treatment (93.69%). The lowest mortality of adult and juvenile snails was in the control (14.29% and 8.57% respectively). Egg mortality was high across all three treatments (76.66, 91.00, and 88.00% respectively) (Figure 5.3.3).

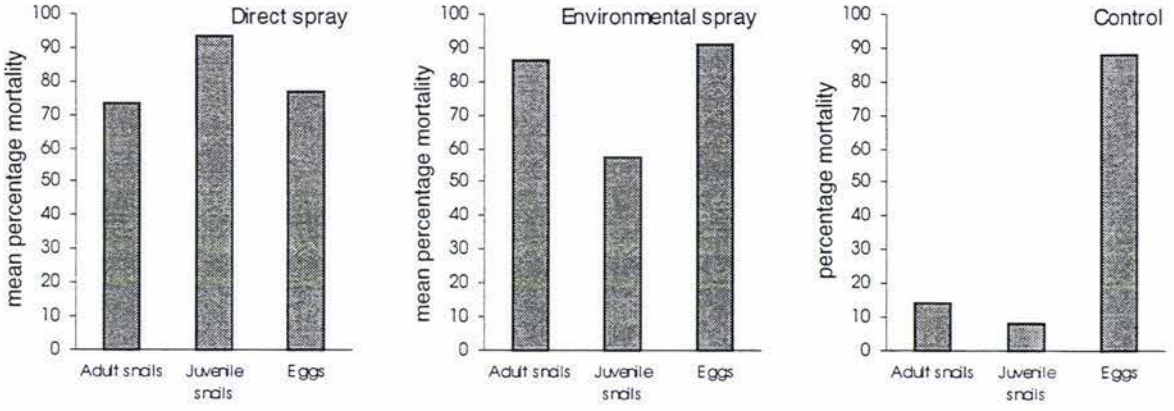

Figure 5.3.3 Percentage mortality of *C. aspersus* adult snails, juvenile snails, and eggs after 149 days. Mean percentage mortality is shown for the direct and environmental spray treatments (averaged over replicates).

Of the three treatments, overall mean percentage mortality was greatest for the direct spray treatment (82.22%), followed by the environmental spray (79.39%) and the control (32.96%) (Figure 5.3.4). Of the three life history stages, overall mean percentage mortality was greatest for eggs (84.55%), followed by adult snails (57.42%) and juveniles snails (52.59%).

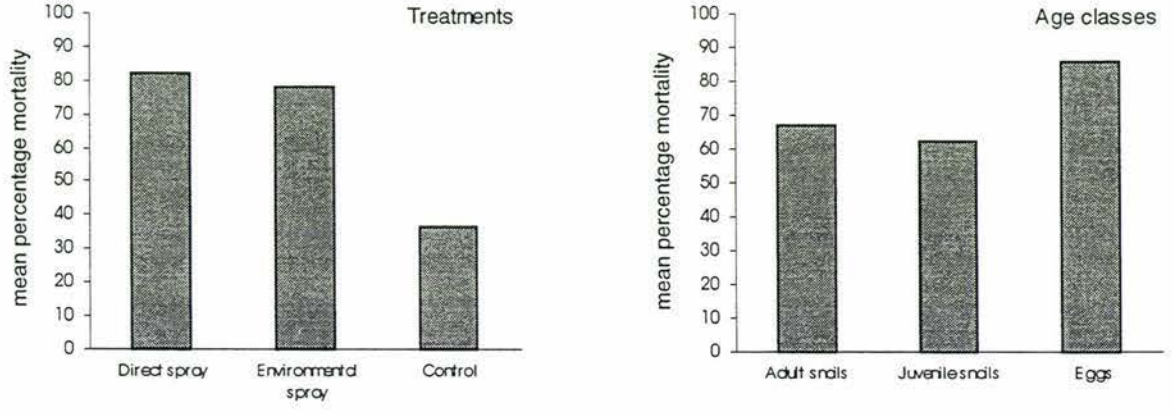

Figure 5.3.4 Overall mean percentage mortality for each treatment and *C. aspersus* life history stage.

A general linear model was fitted to the mortality data (Table 5.3.1). This explained most of the variation (r^2 = 93.7%) and showed a slight significant age class effect (P=0.04) and a highly significant treatment effect (P<0.01). In addition, there was a significant interaction effect between treatment and age class (P<0.05), confirming that the three age classes reacted differently to the three treatments. There was no significant replicate effect.

Table 5.3.1 Results of general liner model (repeated measures ANOVA) fitted to *C. aspersus* mortality data.

Factor	F-value	Pr > F
Treatment	37.96	P = 0.0025
Age class	9.39	P = 0.0375
Treatment-age class interaction	28.11	P = 0.0044

The significant interaction effect was confirmed by comparison of the three treatments for each life history stage (Tukey's test). The high mortality of adult snails in the direct and environmental spray treatments (73.33% and 86.66% respectively) were not significantly different (P<0.05) but both were significantly greater than mortality in the control treatment (14.20%) (P<0.05). In contrast, juvenile mortality was only significantly different from the control treatment (8.57%) in the direct spray treatment (93.68%). Egg mortality showed no significant difference between the three treatments suggesting some other factor was present in all three egg treatments that was not present in juvenile and adult treatments. As overall mortality was very high, the eggs may have been inviable or kept under unsuitable conditions.

Comparison of *C. aspersus* mortality within each treatment (least squares means procedure) showed that there was no significant difference (P>0.05) between the three life stages in the direct and environmental spray treatments. However, egg mortality in the control treatment (88.00%) was significantly greater than that observed for adult and juvenile snails (14.29 and 8.57% respectively) (P<0.05).

Comparison of the three treatments (Tukey's test) showed that the mean mortality in the direct and environmental spray treatments (82.22 and 79.39% respectively) was significantly greater than in the control (32.96%) (P<0.05) and that there was no significant difference between the direct and environmental spray treatments (P>0.05). Of the life history stages,

eggs suffered significantly greater mean mortality (84.55%) than adult (57.42%) and juvenile snails (52.59%) ($P<0.05$). Mean mortality of adult and juvenile snails did not differ significantly ($P<0.05$).

Powelliphanta traversi traversi

No deaths occurred in either treatment group after the 9 snails were sprayed. After being sprayed, all snails gained weight but not all grew in maximum shell length (ML) or width (MW) (Table 5.3.2). Most notably, three snails in the control group and one snail in the Grazon spray trial showed a decrease in their maximum aperture size (MA) after being sprayed. This is apparently not uncommon for *P. t. traversi* (see Chapter 2, Figure 2.2.2(a)) and appears to be a natural phenomenon in some populations, so it is not suspected to be due to any adverse effects of exposure to Grazon. The other decreases in maximum shell length and width (Table 5.3.2) are relatively small and may have been in part due to measurement error.

Table 5.3.2 Summary of overall changes in measures of *P. t. traversi* snails (% change rounded to nearest whole number, negl. = <1% change).

Snail	Treatment	Initial weight (g)	% change in weight	initial ML (mm)	% change in ML	initial MW (mm)	% change in MW	Initial MA (mm)	% change in MA
1	Control	4.34	54	26.59	17	21.74	15	15.04	15
2	Control	18.45	4	48.49	negl.	39.60	negl.	23.06	-3
3	Control	10.71	3	41.33	negl.	33.72	negl.	20.43	-3
4	Control	17.07	17	44.82	5	36.77	3	22.25	-3
5	Grazon	7.66	21	36.80	4	30.92	3	17.14	2
6	Grazon	16.02	3	45.71	negl.	37.45	negl.	21.05	-2
7	Grazon	15.30	20	43.68	4	35.07	2	21.46	2
8	Grazon	16.83	20	45.01	9	36.78	negl.	21.25	negl.
9	Grazon	19.50	10	49.35	1	40.00	negl.	23.00	2

Snails did not feed every night after worms were presented to them. There were always several snails that did not eat all the worms supplied and frequently several did not eat any. Despite this, a gain in weight was nearly always observed for all the snails after they had been placed overnight in the smaller containers to be fed. This may have been due to water

uptake or the consumption of other invertebrates, such as amphipods that were also present in the leaf litter.

Two snails in the control group produced eggs while three snails produced eggs after being sprayed with Grazon. In total, 21 eggs were produced during the experiment. Two of these were produced by snails in the control group before being sprayed with tap water. All eggs were found just beneath the leaf litter, and on two occasions more than one egg was produced by a single individual in a single night.

All but one of these eggs were found when the snails were being fed in the smaller containers so the exact date of their production is known. The remaining egg may have been up to 5 days old when it was found. Eggs were initially pale and greasy to touch, but soon lost this texture and darkened to brown after 1-2 weeks. Snails typically lost weight when eggs were produced, but on one occasion a gain in weight was observed.

The size and weight of the eggs produced varied both in relation to the snail and between snails. The lengths of four eggs (8.46, 8.48, 8.80, and 7.60 mm (all from the control group)) were smaller than previously described for *P. t. traversi* fa. tararuaensis (9.25-11 mm, (O'Connor, 1945)), and the widths of seven eggs (7.38, 7.49, 7.49, 7.67, and 6.64 mm from the control group; 7.59 and 7.93 mm from the Grazon treatment) were smaller than previous measures for *P. t. traversi* fa. tararuaensis (8-9 mm, (O'Connor, 1945) (Table 5.3.3). However, the mean egg length (9.84 mm) and width (8.12 mm) were within the range of measures described by O'Connor (1945).

Table 5.3.3 Maximum diameter (MD) of shell, maximum width (MW) of shell, and weight of *P. t. traversi* eggs (*outside range of measures for *P. t. traversi* fa. tararuaensis (O'Connor, 1945)).

Treatment	Egg	Snail	MD (mm)	MW (mm)	Weight (g)
Control	1	3	9.30	*7.67	0.35
	2	3	*8.46	*7.38	0.30
	3	3	*8.48	*7.49	0.30
	4	2	10.16	8.25	0.44
	5	3	*8.80	*7.49	0.31
	6	2	10.38	8.42	0.49
	7	3	*7.60	*6.64	0.21
	8	2	9.84	8.15	0.41
	9	2	10.58	8.09	0.43
	10	2	10.38	8.64	0.48
		Mean	*9.40*	*7.82*	*0.37*
Grazon	1	6	9.39	*7.59	0.33
	2	6	10.53	8.76	0.48
	3	6	10.26	8.20	0.43
	4	6	10.48	8.48	0.46
	5	8	9.78	*7.93	0.39
	6	8	9.96	8.64	0.43
	7	6	10.86	8.81	0.52
	8	6	10.51	8.16	0.44
	9	6	10.71	8.96	0.52
	10	9	10.15	8.29	0.45
	11	9	10.02	8.52	0.44
		Mean	*10.24*	*8.40*	*0.45*

5.4 **DISCUSSION**

Under the experimental conditions used, a 1.4% Grazon solution kills 82.34% of juvenile and adult *C. aspersus* irrespective of the exposure method. There was no significant difference between the direct spray and the environmental spray treatments. The common factors in both treatments were direct tissue contact with Grazon on the substrate, and the possibility that snails ingested Grazon with the lettuce that was sprayed. On the other hand, the results for *C. aspersus* eggs are not conclusive and the eggs appear to have been more affected by other unknown factor(s). Nematodes were never seen during successful egg rearing done before the experiment, but they were found in the soil of the experimental egg treatments. They are known to be able to affect the survival rate of *C. aspersus* eggs (Madec *et al.*, 2000). Humidity was not controlled and may have been another influencing factor because it does affect egg survival (Guéméné & Daguzan, 1983, cited in Barker, 1999). Cannibalism of unhatched conspecific eggs by newly hatched *C. aspersus* appears to be

common (Elmslie, 1988, cited in Barker, 1999; Desbuquois, 1997) and may have been yet another cause for the high mortality in all three treatments.

There is a possibility that chemical residues (including chemicals other than Grazon) were present in the knapsack sprayer and that these affected all treatment groups and life history stages. This could account for the deaths observed in the control treatments for all three life history stages. *C. aspersus* eats significant amounts of soil (Barker, 1999) and the soil was sprayed in the egg treatments. Consequently, the presence of chemical residues may have contributed to the high mortality seen in the control egg treatment if this life history stage was more sensitive to any chemicals present and mortality was principally of newly hatched snails.

It seems that the results of my experiment with *C. aspersus* do not apply to *P. t. traversi* snails. None of the five *P. t. traversi* sprayed with Grazon died or showed any signs of toxicological effects during the same period that resulted in high *C. aspersus* mortality. However, differences in the results of the two experiments are not unexpected considering the range of biological and ecological differences between these two species (most notably their size, life span and feeding habit), as well as any unknown differences between these two species in the metabolism of triclopyr. This emphasizes the importance of testing *P. t. traversi*, despite their protected and endangered status.

My experimental protocol was designed specifically to assess the toxicity of a single concentration of Grazon that represents a worst case field scenario so it differs fundamentally from other studies that establish LC50 values over a range of concentrations or time periods. Therefore, any conclusions inferred from the results of other organism toxicity studies may be of little value in relation to *P. t. traversi*. However, it may be worthwhile to consider the results of studies that have assessed the fate of triclopyr in the environment.

My study differs from other field investigations in that Grazon was applied directly to the leaf litter in tanks containing *P. t. traversi*, rather than to vegetation above this substrate. With foliar application in the field, triclopyr residues are typically reduced by the time they reach the soil because of processes such as translocation by plants, photodegradation, volatisation and metabolism (Norris *et al.*, 1987). In my study, maximum triclopyr

concentrations in the leaf litter would have occurred soon after the application of Grazon. Lower Grazon concentrations have been used to effectively control *T. fluminensis* (Brown & Rees, 1995; McCluggage, 1998) so the maximum triclopyr residues experienced by snails in my study are likely to have exceeded those that can be reasonably expected by using lower effective Grazon concentrations for controlling *T. fluminensis*.

The duration of my experiment was very much longer than typical toxicological investigations, and it was closer to a likely field situation. This is important considering that both concentration and duration of exposure determine the magnitude of the effect of triclopyr (Holmes *et al.*, 1994). Triclopyr causes delayed lethal effects in fish and aquatic insects, and has greater toxicity with longer exposure durations (Holmes *et al.*, 1994). Similarly, long-term exposure to sublethal concentrations of glyphosate (Roundup) has a delayed effect on growth, development, and hatching of the aquatic snail *Pseudosuccinea columella* (Tate *et al.*, 1997) but this did not occur until the third generation of continuously exposed snails. Hence, it is possible that *P. t. traversi* snails had insufficient time to develop toxic effects in my experiment, and it is also possible that the effects of Grazon may occur in subsequent generations.

Pesticide breakdown is generally faster in the field than it is in laboratory trials (Sparks, 1989) due to the greater number of factors influencing pesticide disappearance in the field (Rao & Davidson, 1980). In addition, the rate of decline of triclopyr may be inversely related to dose if increased concentrations saturate the microbial system responsible for the primary mode of degradation in soil (Holmes *et al.*, 1994). In such a case, the *P. t. traversi* sprayed with Grazon may not only have survived higher residue levels than can be reasonably expected in the field, they may also have experienced exposure to these levels for longer than is likely to occur in a field situation.

On the basis of my experiment, Grazon does not appear to represent a direct toxic threat to *P. t. traversi*, but the influence of this herbicide on aspects of the snail's environment, such as prey, also need to be considered. The intake of significant amounts of herbicide by terrestrial animals is most likely to result from the consumption of recently treated vegetation (Norris *et al.*, 1977). Because of their carnivorous nature, *P. t. traversi* should not be at risk by ingesting treated plant material. This may explain the difference in mortality compared to *C.*

aspersus. None the less, Lee (1985) reported likely toxic effects to earthworms of several herbicides at high rates used for completely suppressing plant growth, although he did not consider triclopyr. The toxicity of Grazon to earthworms will therefore also need to be tested to ensure that there is no indirect effect on *P. t. traversi*.

The results of my experiment lend weight to using Grazon as a preferred herbicide to control *T. fluminensis*, and supports the observations by Rachel Standish (*unpubl. data*) that a 0.66% Grazon solution had little effect on the abundance or taxa richness of native ground-dwelling invertebrates at one week and at seven weeks when applied to areas affected by *T. fluminensis*. Despite this, it would be prudent if the amount of Grazon applied to *T. fluminensis* was reduced as much as practicable because native invertebrates may be present upon which the effects of triclopyr have not been investigated. One method shown to achieve this is preherbicide manipulation of the target plant. Less spray was required for effective control of *T. fluminensis* when the bulk of this weed was reduced before spraying in trials by Kelly & Skipworth (1984b). Similarly, the level of control of Tropical Soda Apple (*Solanum viarum* Dunal) using triclopyr increases with the number of preherbicide mowings (Mislevy *et al.*, 1999). Mechanical destruction of *T. fluminensis* before application of Grazon could lower any risk to *P. t. traversi* by reducing the level of herbicide applied, but there could be negative effects of mechanical damage to these snails.

Consideration needs to be given to the effect spray-timing may have on both *T. fluminensis* and invertebrates beneath sprayed areas. For best results, application of Grazon is recommended during periods of active plant growth (Manufacturers Instructions). Shoot extension is greatest in the summer months (Maule *et al.*, 1995), but spraying in summer could have both advantages and disadvantages for *P. t. traversi*. These snails are typically more active following wet periods, so if spraying was done during dry periods the chance of snails being active and moving over contaminated soils could be lower. There would also be less contamination of the soil by rain-transported residues, and degradation of triclopyr may be nearly twice as rapid when applied in summer compared to an autumn application (Moseman & Merkle, 1977, cited in Norris *et al.*, 1987). On the other hand, the removal of *T. fluminensis* during drier months could increase snail mortality through dessication or predation if there is no alternate suitable habitat for them. Nothing is known about how a

sprayed and decaying layer of *T. fluminensis* affects *P. t. traversi* beneath it. Restricted field trials would be required to resolve these issues.

The results of my investigation represent the only information concerning the likely effect of Grazon on *P. t. traversi* if it is used to control *T. fluminensis* in native bush containing these snails. I recommend that Grazon be used to control *T. fluminensis* ahead of other herbicides currently available and that a spray concentration of 1.4% should be considered as an absolute upper limit. Further experimentation is necessary to resolve other issues in field situations. Lower concentrations have proven effective in controlling *T. fluminensis* (e.g. Brown & Rees, 1995; McCluggage, 1998), so similar spray concentrations to these should only be used. In the meantime, other more cautious methods (native plantings into *T. fluminensis* for example) could be used to reduce the bulk of *T. fluminensis* needed to be sprayed and provide shelter for invertebrates when the weed is controlled.

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CHAPTER 6

Summary and Recommendations

The Department of Conservation has done much to determine the status of the *Powelliphanta traversi traversi* (Powell) populations in the Horowhenua District, including supporting my own work. However, any study like mine can only capture information during a limited time. What such studies do provide is a starting point for continued monitoring. Long term investigations of the dynamics of specific populations of *P. t. traversi* and how they are affected by introduced flora and fauna, together with their habitat, is lacking and needs to be addressed. In addition, because of their presence in easily accessible sites, *P. t. traversi* snails provide an ideal opportunity to investigate the effects of introduced pests on them and to formulate management options that may be applied to other snails in the genus *Powelliphanta*.

Rodent control and monitoring were two recommendations made by Devine (1997). I have shown that rodent poisoning may benefit *P. t. traversi* and that monitoring is needed in conjunction with rodent control to elucidate the true benefit of poisoning. The apparent benefits to *P. t. traversi* at Lake Papaitonga Scenic Reserve in one area that was poisoned should prompt consideration of continuing rodent control at this site for several more years to determine longer term effects. This could further determine our ability to maintain low rodent densities and increase *P. t. traversi* recruitment there. In addition, it would be worthwhile to eventually switch rodent control between my experimental areas to validate the observed effects of rodent control on *P. t. traversi*.

If rodent control is continued at Lake Papaitonga, it would also be interesting to compare the benefits to *P. t. traversi* with the results of rodent control at other areas containing snail populations. These have largely been restricted to smaller snail populations than is present at Lake Papaitonga, and often with a less regular poisoning regime. Continuing rodent control at Lake Papaitonga could also provide an opportunity for others to study the effects on other native invertebrates present in the reserve.

There were large natural variations in snail numbers in the areas searched at Lake Papaitonga and Prouse Bush. If permanent snail survey quadrats are to be established consideration needs to be given to the possibility that snail density can vary greatly within relatively small areas. It would be interesting to understand the causes of these differences but this may be very difficult. It may also be interesting to see if more frequent searches for snails can establish short-term changes in snail numbers that are not revealed by annual searches. However, care would be needed to minimise the risk to snails of desiccation or increased exposure to predators by the disruption that is caused to the leaf litter habitat.

Powelliphanta traversi traversi certainly uses areas affected by *Tradescantia fluminensis* (Vell.) and evidence was gathered that conclusively showed a direct positive effect on these snails. More live snails were found in *T. fluminensis* than in leaf litter at Prouse Bush but it is not clear whether this result shows that the snails are truly benefiting from the presence of *T. fluminensis*, or whether *T. fluminensis* alone can provide suitable snail habitat. What is clear is that *T. fluminensis* threatens native habitat at Prouse Bush and other natural areas where *P. t. traversi* exists, and that herbicides may be used in order to control *T. fluminensis* in these areas. It is therefore important that Grazon® is not toxic to *P. t. traversi*. However, further testing of the effects of Grazon on *P. t. traversi*, especially eggs and small juvenile snails, is warranted. Its effects on other soil invertebrates, particularly the prey of these snails, should also be investigated.

It is clear that predation of *P. t. traversi* by introduced vertebrates (particularly rodents, blackbirds, and song thrushes) will continue to threaten many of these snails, and that *T. fluminensis* will continue to spread and threaten native forest regeneration. Therefore, action must continue to be taken to mitigate their impacts. I agree that prioritisation of particular populations is needed because of the limited resources available, but advocate the maintenance of as many populations as possible because of the unique environmental and biological influences each currently experience.

Recommendations

⇒ Continue rodent control (using poison or a combination of methods) and monitoring at Lake Papaitonga, and consider the reversal of rodent control between areas to reinforce conclusions about the effects on *P. t. traversi*.

⇒ Investigate the extent of predation on *Powelliphanta* snails by introduced bird species.

⇒ Investigate the use of *T. fluminensis* as habitat by other invertebrates.

⇒ Conduct restricted field experiments in areas containing *P. traversi* snails using Grazon herbicide to control areas of *T. fluminensis*.

⇒ Investigate the effect of repeated applications of Grazon on *P. traversi*.

⇒ Support further investigations of *Powelliphanta* ecology and biology, especially those that can establish long-term projects.

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APPENDIX 1

Shell damage caused by predators of *Powelliphanta traversi traversi* (Powell) snails.

The predatory behaviour of several animal species which have access to *Powelliphanta* snails naturally was assessed using wild captured and already captive individuals. Each animal was presented with an empty whole shell filled with a suitable attractant, and allowed to 'attack' the shell. It was hoped that this would result in a collection of shells showing diagnostic shell damage.

Stoat (*Mustela erminea*)

Four stoats (two males and two females) held captive by Lloyd Robbins, were each presented with one *P. t. traversi*. The four shells (40-50 mm maximum length) and were baited with minced beef. All shells were ignored the first night but were cached by the stoats on the second night. After six nights the shells were recovered from each cage. Three shells were damaged in different ways which appear to show different stages. This began at the aperture lip, followed by removal of the outermost whorl, and then some damage to inner whorls (Figure A1.1).

Figure A1.1 Damage to *P. t. traversi* shells caused by Stoats.

Kaka (*Nestor meridionalis*)

Two trials were performed using one captive breeding pair of North Island Kaka held at Mount Bruce Wildlife Sanctuary. For each trial, two *P. t. traversi* shells (40-50 mm maximum length) were baited with crunchy peanut butter (Sanitarium Co.). One was placed with their normal food supply at a feeding station, the other was placed on the ground within their enclosure.

The birds were observed for the first hour of the first trial. On its first visit to the feeder, the female immediately picked up the shell and attempted to lick out some of the peanut butter before dropping the shell to the ground. No other interest was shown during the first hour, so both shells were left with the birds for four days. On the morning of the forth day, the male picked up the shell that was originally in the feeder and tore it apart, removing much of the outermost whorl and inner whorls. It then licked the peanut butter from the shell (Raelene Berry, 2000, *pers. comm.*) (Figure A1.2). Only smaller fragments of the second shell were found.

The second trial was performed to determine if shell damage caused by the Kaka would be consistent between attacks. The shells were of similar size to those used in the first trial and left for the same period of time. Both shells received some damage but this was not as sever as that done during the first trial, and it was restricted to the aperture lip and outermost whorl (Figure A1.3).

Figure A1.2 Damage to *P. t. traversi* shells caused by North Island Kaka during first trial. The shell on the left was attacked by the male Kaka.

Figure A1.3 Damage to *P. t. traversi* shells caused by North Island Kaka during second trial.

Mouse (*Mus musculus*)

One mouse (captured at 8 Mountain View Road, Palmerston North) was presented with two *P. t. traversi* shells baited with peanut butter. Each shell was left with the mouse for one day and night.

No damage was inflicted on the first shell (40.18 mm maximum length). The second shell (27.77 mm maximum length) was severely damaged. The mouse gnawed the shell back from the aperture lip (Figure A1.4). This damage was similar to that done by rats.

Figure A1.4 Damage to *P. t. traversi* shell caused by a mouse.

Hedgehog (*Erinaceus europaeus*)

Two hedgehogs (captured at 8 Mountain View Road, Palmerston North) were kept separately overnight and each was presented with two *P. t. traversi* shells. One shell was baited with crunchy peanut butter, the other with catfood. Neither animal showed any interest in the shells when observed, and no shell damage was done.

APPENDIX 2

Summary of relocations using harmonic radar

Table A2 Summary of data collected from snails with harmonic radar transponders attached.

Snail	No. of relocations	No. days followed	Mean displacement	Total distance travelled	Overall displacement	% total distance explained by overall displacement
*12	19	662	1.59	30.25	6.97	23.04
6	17	692	4.35	73.99	2.43	3.28
*13	17	532	7.53	128.03	46.22	36.10
*2	16	600	3.50	55.98	25.57	45.68
16	16	594	1.96	31.38	16.65	53.06
*11	16	503	2.92	46.70	13.39	28.67
*A4	15	627	14.29	214.42	87.59	40.85
30	14	538	4.42	61.87	3.07	4.96
*5	14	491	8.79	123.11	6.60	5.36
*17	14	483	1.80	25.23	12.24	48.51
*19	14	483	3.42	47.94	12.46	25.99
*B2	14	463	2.65	37.07	10.62	28.65
D3	13	402	6.88	89.48	11.36	12.70
32	12	434	3.87	46.50	3.93	8.45
D2	12	402	5.47	65.64	8.95	13.63
*D1	12	359	4.65	55.79	14.29	25.61
31	11	380	4.16	45.76	10.39	22.71
4	11	347	5.99	65.92	13.06	19.81
B1	10	313	2.33	23.34	9.48	40.62
61	9	397	1.89	17.05	7.30	42.82
*60	9	373	5.77	51.93	14.13	27.21
3	9	295	5.39	48.51	12.83	26.45
10	9	285	1.22	11.00	6.68	60.73
8	8	379	6.85	54.78	10.30	18.80
34	8	373	7.35	58.84	4.83	8.21
35	8	373	4.70	37.57	4.25	11.31
*50	8	286	1.33	10.65	2.10	19.72
*A1	6	198	0.62	3.75	2.06	54.93
D4	5	214	6.18	30.88	8.03	26.00
33	5	150	1.91	9.56	6.19	64.75
72	3	157	2.25	6.76	6.20	91.72
7	3	98	2.95	8.86	6.58	74.27
14	3	88	0.31	0.94	0.51	54.26
*71	2	157	0.19	0.37	0.20	54.05
Qb10	2	130	1.82	3.64	2.86	78.57
18	2	66	0.03	0.05	0.05	100.00
*70	2	68	1.09	2.18	2.16	99.08
D5	1	34	0.29	0.29	0.29	100.00
Qb5	1	30	82.91	82.91	82.91	100.00
15	1	22	0.49	0.49	0.49	100.00

*snails determined to have moved between habitats.