

SHORT COMMUNICATION

Response of seedling communities to mammalian pest eradication on Ulva Island, Rakiura National Park, New Zealand

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Abstract: Norway rats (*Rattus norvegicus*) were eradicated from Ulva Island, Rakiura National Park, in 1996. The aim of our work was to determine if seedlings and saplings increased in density and/or species richness following this eradication. In 2003, we took advantage of eight permanent plots (5 × 5 m) that had been established on Ulva Island in 1991, by counting seedlings and saplings of woody species, including tree ferns. Over this period, total numbers of woody seedlings (< 30 cm tall), and saplings (30 cm – 2 m tall) did not increase significantly ($P > 0.05$). However, seedlings of two species, *Dicksonia squarrosa* and *Pittosporum colensoi*, and saplings of one species, *Coprosma colensoi*, did increase in numbers significantly. Little change in the species richness of seedlings or saplings was recorded. The removal of Norway rats may have been partly responsible for the recorded increases in density, though the eradication of white-tailed deer (*Odocoileus virginianus borealis*) from Ulva Island approximately two decades prior to this work is an important confounding factor.

Keywords: forest regeneration; herbivory; *Odocoileus virginianus borealis*; *Rattus norvegicus*; seedling establishment; seedling recruitment; seed predation; Stewart Island

Introduction

Invasive species are an increasing problem worldwide, with oceanic islands suggested by some to be more vulnerable than continental regions (Veitch & Clout 2002). Establishing the effects of such invasions is often hindered by unavailability of pre-invasion data. Eradication methods used for invasive mammalian species on oceanic and offshore islands are increasing in sophistication as well as size of area being successfully tackled (Towns & Broome 2003). Following eradication the ecosystem will recover to some degree, but it is questionable as to how similar that state will be to the pre-invasion situation (Caughley 1989). Nevertheless, post-removal measurements of key elements of the ecosystem are crucial to understanding the recovery trajectory of a particular habitat especially when considering reintroductions of native species.

New Zealand is regarded by many as a world leader in eradication technologies and the New Zealand Department of Conservation (DOC) has completed a series of eradication programmes on islands around New Zealand over the last c. 40 years (Towns & Broome

2003). This work has been done to provide refugia for native birds, invertebrates, and herpetofauna, at the same time as allowing plant communities to recover from the influence of introduced browsing mammals (see Atkinson 1973). Although the conservation benefits from these programmes are often very clear for fauna (e.g. Veitch & Clout 2002), fewer studies have measured changes to plant communities on islands following eradication of introduced animals, particularly rats (*Rattus* spp.). As well as eating a variety of seeds and fruits, rats eat seedlings, fleshy roots, leaf laminae and petioles, flowers, apical buds and bark (Campbell 1978). Also, by consuming native herbivores such as invertebrates, reptiles and birds, rats may indirectly affect vegetation structure and composition by modifying plant–herbivore interactions (Clout & Hay 1989; King 2005).

Several studies do show changes in seedling communities after eradication of introduced mammals on islands. Five years after Norway rats (*Rattus norvegicus*) were removed from Breaksea Island in Fiordland, Ralph Allen and colleagues (1994) found an increase in numbers of seedlings (< 30 cm tall) of 10 out of 24 common woody

plant species. Following the eradication of brushtail possums (*Trichosurus vulpecula*), Norway rats (*Rattus norvegicus*) and Pacific rats (*R. exulans*) from Kapiti Island, seedlings of four out of the five plant species expected to increase in density (based on known palatability) did so over a 4-year period (Campbell 2002). However, Campbell (2002) concluded that pest eradication could not be separated from other factors contributing to the apparent recovery of seedlings of many species.

Ulva Island is the largest (226 ha) of the approximately 20 islands in Paterson Inlet, Stewart Island (46°55' S, 168°08' E). The island was declared a reserve for the preservation of nature, game and flora in 1899 and is now managed as an 'open sanctuary' and part of Rakiura National Park (DOC 2004). The two remaining species of exotic vertebrates (i.e. white-tailed deer, *Odocoileus virginianus borealis* and Norway rats) were eradicated from Ulva Island in 1976 and 1996 respectively. Rance (1991–DOC Invercargill unpubl. report) established eight permanent vegetation plots on Ulva Island in 1991 in order to study vegetation changes after rat eradication. Changes in seedling communities were minimal in the subsequent few years (Rance 1994–DOC Invercargill unpubl. report). We revisited these plots in 2003 to determine whether the density of seedlings on Ulva Island had changed since plots were established there 12 years previously. We expected that the density of seedlings of some plant species known to be palatable to rats would have increased.

Methods

Study site

Meurk and Wilson (1989) described the vegetation of Ulva Island and identified two major forest types (nomenclature follows Allan Herbarium (2000)): (1) rimu-rātā-kāmahi forest (*Dacrydium cupressinum*, *Metrosideros umbellata* and *Weinmannia racemosa*) covering the majority of the island (86%); and (2) coastal 'muttonbird scrub'–leatherwood-inaka low forest (*Brachyglottis rotundifolia*, *Olearia colensoi* and *Dracophyllum longifolium*) dominating most of the remaining area (12%).

Field data collection

Eight plots (5 × 5 m) were established at random points along a public track network at the western end of Ulva Island (Rance 1991 unpubl.). These plots were all placed in the rimu-rātā-kāmahi forest; the coastal vegetation type was excluded as it had little plant species diversity and the major plant species present there were not expected to be influenced by rats. Vegetation on the plots was measured in 1991 and the seedling tiers (< 30 cm tall) were remeasured in 1993 and 1994 (Rance 1991 & 1994 unpubl.). All data were collected in late spring or early summer. In December 2003, we remeasured the vegetation on the same eight plots in accordance with the original methods.

Woody plants that were rooted in the plots and had true leaves (i.e. non-cotyledonous seedlings) were counted in two height categories: < 30 cm tall (seedlings) and 30 cm – 2 m tall (saplings). The tree ferns *Dicksonia squarrosa* and *Cyathea smithii* were counted in the same fashion (as 'seedlings' and 'saplings'), with height recorded to the base of the crown.

Numbers of seedlings and saplings of all species (combined and separate) were compared before and after rat eradication (i.e. 1991 and 2003 data) with two-tailed, paired *t*-tests, with a significance level of 0.05. Species richness in the seedling and sapling height classes, defined as the number of species present per plot, was compared before and after rat eradication in the same way.

Results

Seedlings

When counts of seedlings (< 30 cm) of all woody and tree fern species were combined, there were on average 9.1 seedlings m⁻² in 1991 and 11.0 seedlings m⁻² in 2003 (Table 1). This increase was not statistically significant ($t_7 = 0.94$, $P = 0.38$). However, increases in numbers of seedlings were statistically significant in two species, *Dicksonia squarrosa* ($t_7 = 2.56$, $P = 0.038$) and *Pittosporum colensoi* ($t_7 = 4.58$, $P = 0.003$). The number of *Myrsine divaricata* seedlings decreased significantly ($t_7 = 2.7$, $P = 0.03$). Two species not present in 1991 appeared in 2003 (*Coprosma grandifolia* and *Metrosideros umbellata*). Neither was present in large numbers, although 18 small seedlings of *C. grandifolia* were found on one plot.

An average of 14.6 woody and tree fern species per plot was recorded in the seedling height class in 1991 and 15.5 species per plot in 2003. This difference was not statistically significant ($t_7 = 2.0$, $P = 0.087$).

Saplings

When all species of saplings (30 cm – 2 m tall) were combined, there were on average 1.29 saplings m⁻² present in 1991 and 1.96 saplings m⁻² in 2003 ($t_7 = 2.32$, $P = 0.058$) (Table 1). Of 18 species of saplings that were present in both 1991 and 2003, only *Coprosma colensoi* increased significantly in density during this period ($P < 0.01$). *Pseudopanax colensoi* was present in 2003 but not in 1991, while *Pittosporum colensoi* had disappeared between the years.

An average of 7.8 species per plot were present in the sapling height class in 1991 and this number did not change significantly in 2003 with 8.1 species per plot ($t_7 = 0.48$, $P = 0.32$).

Table 1. Mean densities of seedlings (< 30 cm tall) and saplings (30 cm – 2 m tall) of species recorded in eight plots on Ulva Island, with associated standard errors. Significant changes ($P \leq 0.05$) indicated in bold.

Species	Seedlings m ⁻²					Saplings m ⁻²				
	1991	SE	2003	SE	<i>P</i>	1991	SE	2003	SE	<i>P</i>
<i>Pseudopanax crassifolius</i>	1.51	0.45	1.96	0.52	0.44	0.25	0.11	0.36	0.09	0.39
<i>Raukawa simplex</i>	1.41	0.31	1.99	0.50	0.21	0.12	0.04	0.39	0.18	0.10
<i>Coprosma colensoi</i>	0.99	0.16	1.40	0.32	0.22	0.14	0.07	0.24	0.08	0.01
<i>Dacrydium cupressinum</i>	0.76	0.34	0.85	0.17	0.68	0.03	0.02	0.04	0.02	0.35
<i>Coprosma foetidissima</i>	0.76	0.26	1.07	0.28	0.30	0.06	0.01	0.10	0.04	0.32
<i>Neomyrtus pedunculatus</i>	0.60	0.23	0.86	0.31	0.34	0.15	0.05	0.13	0.05	0.68
<i>Griselinia littoralis</i>	0.57	0.32	0.42	0.17	0.42	0.09	0.05	0.16	0.04	0.13
<i>Myrsine australis</i>	0.56	0.28	0.33	0.10	0.37	0.01	0.01	0.06	0.03	0.09
<i>Coprosma rhamnoides</i>	0.42	0.07	0.56	0.14	0.40	0.17	0.14	0.18	0.17	0.79
<i>Weinmannia racemosa</i>	0.39	0.32	0.28	0.11	0.75	0.01	0.01	0.01	0.01	0.35
<i>Ripogonum scandens</i>	0.34	0.15	0.12	0.03	0.12	0.00	0.00	0.00	0.00	1.00
<i>Myrsine divaricata</i>	0.31	0.15	0.16	0.13	0.03	0.02	0.01	0.02	0.02	1.00
<i>Podocarpus hallii</i>	0.19	0.10	0.22	0.15	0.70	0.13	0.09	0.12	0.09	0.40
<i>Coprosma lucida</i>	0.12	0.04	0.12	0.05	0.94	0.02	0.01	0.05	0.04	0.35
<i>Prumnopitys ferruginea</i>	0.09	0.04	0.19	0.09	0.17	0.00	0.00	0.00	0.00	1.00
<i>Dicksonia squarrosa</i>	0.09	0.03	0.32	0.11	0.04	0.10	0.04	0.11	0.03	0.63
<i>Brachyglottis rotundifolia</i>	0.05	0.04	0.02	0.01	0.43	0.02	0.02	0.02	0.02	0.35
<i>Pittosporum colensoi</i>	0.02	0.02	0.11	0.03	0.00	0.01	0.01	0.00	0.00	0.35
<i>Coprosma areolata</i>	0.01	0.01	0.00	0.00	0.35	0.00	0.00	0.00	0.00	1.00
<i>Coprosma grandifolia</i>	0.00	0.00	0.10	0.09	0.32	0.01	0.01	0.01	0.01	1.00
<i>Metrosideros umbellata</i>	0.00	0.00	0.01	0.01	0.35	0.00	0.00	0.00	0.00	1.00
<i>Pseudopanax colensoi</i>	0.00	0.00	0.00	0.00	1.00	0.00	0.00	0.01	0.01	0.35
Total	9.14	1.97	11.03	2.11	0.38	1.29	0.40	1.96	0.41	0.06

Discussion

The three plant species that increased after rat eradication on Ulva Island may be palatable to both Norway rats and white-tailed deer. Leaf fragments of *Coprosma colensoi* have been noted in the stomach contents of Norway rats (Sturmer 1988), and the fleshy fruits of this genus are eaten by the three species of rats in New Zealand (Campbell 1978). Rats ate tissue of many fern species at Robertson River, Stewart Island, although *Dicksonia squarrosa* was not among them (Sturmer 1988). No record could be found of rats consuming *Pittosporum colensoi* but Pacific rats (*Rattus exulans*) depressed recruitment of *P. crassifolium* on New Zealand's northern offshore islands (Campbell & Atkinson 2002). Although small-leaved *Coprosma* shrubs such as *C. colensoi* were classed as 'less preferred' forage for white-tailed deer in New Zealand (Nugent & Challies 1988), substantial deer browse on this species has been noted in forest health surveys on Stewart Island (DOC Stewart Island Field Centre 2003, unpubl. report). White-tailed deer eat notable amounts of *Dicksonia squarrosa* (Challies & Burrows 1984) and can impede regeneration of this species on surrounding Stewart Island (Veblen & Stewart 1980). *Pittosporum* spp. were also 'preferred species' in the diet of white-tailed deer on Stewart Island, i.e. eaten more often than expected based on their availability in the forest (Nugent & Challies 1988).

Because of these overlaps in the diets of rats and deer, the observed increases in seedling or sapling density on Ulva Island may have been related both to release from browsing by white-tailed deer and to release from browsing and/or seed predation by Norway rats.

The timing of the two eradications is important. Deer were absent from Ulva Island for at least 20 years before the rat eradication, and further, their numbers had been kept low for some decades prior to their eventual eradication, mainly by Roy Traill who was the resident naturalist on Ulva (R. Tindal, formerly of DOC, pers. comm., 2004). Therefore it is likely that many of the changes to the sapling tier (including our result of increased sapling numbers of *Coprosma colensoi*) were due to older seedlings that began growing after deer were removed but before 1991 when permanent plot monitoring was set up. Increased numbers of woody saplings and seedlings within deer enclosures have been documented in many New Zealand studies (e.g. Stewart & Burrows 1989). Further, the significant decrease in *Myrsine divaricata* seedling numbers appears to continue an existing pattern not directly related to the removal of rats, as plot measurements made in 1993 and 1994 (Rance 1991 & 1994 unpubl.; Clayton 2005) showed that a decline in seedling numbers of this species was apparent prior to rat eradication (1991: 0.31; 1993: 0.23; 1994: 0.17; 2003: 0.16 seedlings m⁻²).

In summary, few species of seedlings or saplings

have increased significantly in density on eight plots established on Ulva Island prior to rat eradication. This result contrasts with studies by Allen et al. (1994), Wilson et al. (2003), and to a lesser extent Campbell (2002), who all measured a clearer increase in woody seedlings following the eradication or exclusion of rats. Release from the browsing pressure of white-tailed deer was a confounding factor in this study, and probably explains most of the observed increases in density of seedlings of *Pittosporum colensoi* and *Dicksonia squarrosa* and saplings of *Coprosma colensoi*. However, it is impossible to separate the effects of the two introduced animals and both may have been important in their influence on plant species composition, structure, and abundance. In future, we suggest that to reliably detect changes in seedling density due to rodent herbivory, a broader network of plots would be needed. Rat effects on forest regeneration may be more subtle compared with those of deer, but a more robust study design would be needed to detect such relationships, e.g. rodent exclosures nested within deer exclosures (Wilson et al. 2006), and plots established prior to removal of deer. This study was constrained by the original sampling design, and a larger sample size would have delivered greater statistical power. Most studies on forest regeneration in New Zealand use multiple 20 × 20-m plots containing 24 systematically located seedling subplots of 0.49-m radius (Rob Allen 1993), a design likely to better capture variability in light levels and other factors. However, we were fortunate that the initiative was taken to collect data prior to rat eradication, and our results make a useful contribution to regeneration ecology in a system previously occupied by more than one introduced vertebrate herbivore.

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