Predictors of relative abundance of tree weta (*Hemideina thoracica*) in an urban forest remnant

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Abstract: Restoration of urban forest remnants is an increasing activity worldwide, but the effects of restoration efforts on local wildlife in urban remnants remain poorly understood. Understanding the benefits of restoration can also be confounded because of difficulties in monitoring the abundance of representative species, or understanding their ecological requirements. We studied tree weta (*Hemideina thoracica*) in an urban kahikatea (*Dacrycarpus dacrydioides*) forest remnant in Hamilton City: we estimated relative abundance of tree weta, and examined the relationship of weta occupancy of artificial refuges to tree size, bark depth, distance from a centrally lit path, and distance from the edge of the remnant. Estimates of tree weta abundance were comparable with those at other mainland sites. A range of tree species supported tree weta activity, but more weta were encountered on large trees, and fewer on kahikatea relative to other tree species. Occupancy records from 40 artificial refuges on kahikatea trees over 28 consecutive days revealed that tree stem diameter and proximity to the central path predicted occupancy on the dominant kahikatea trees. The data indicate constructed features of urban remnants, such as paths, can negatively affect habitat quality in urban forest remnants. Maintaining large trees in urban parks can provide critical habitat through the provision of natural cavities for weta. Erecting artificial cavities on these trees to gain estimates of tree weta density can also provide indications of ecosystem recovery, for example, after pest removal.

Keywords: forest fragment; indicator species; New Zealand; Orthoptera; restoration

Introduction

Restoration and conservation of small forest remnants is important for urban communities, despite debate about their biodiversity value (McDonnell 2007). Urban remnants can act as ecological reservoirs for species, provide habitat sites so that metacommunities are linked by dispersal (Leibold et al. 2004), and provide opportunities for people to interact with nature (Crane & Kinzig 2005; Miller 2005). However, many urban remnants are modified or degraded, for example by large-tree felling (e.g. Whaley et al. 1997), and have characteristics such as paths and artificial lighting that can have implications for species' reproductive biology, foraging and behavioural responses (e.g. in nocturnal species; Navara & Nelson 2007). There is now a large body of literature indicating roads influence dispersal and mortality patterns for a range of species (Trombulak & Frissell 2000; Laurance et al. 2004; Fahrig & Rytwinski 2009). However, the effect of paths on the distribution of invertebrates, in particular, within small remnants is largely unknown, despite the potential to mitigate effects with effective urban design during restoration.

Degraded urban remnants are often the focus of restoration by community groups, whose choice of plant species for revegetation will ultimately influence the composition of both flora and fauna, ecosystem functioning, and the future trajectory of the ecosystem (Hobbs & Norton 1996). However, restoration efforts can proceed with little monitoring of objectives, or knowledge of local species abundance and habitat (Aronson et al. 1995). Artificial nest boxes, or refuges, are increasingly attached to trees in managed ecosystems as a conservation tool (Jackson et al. 2013), to monitor species and provide increased habitat for cavity dwelling species. Despite this, it is often unclear which plant species provide important habitat for local fauna, particularly for cryptic invertebrates.

Tree wētā (*Hemideina* spp., Anostostomatidae) have been used as indicators to monitor the effects of habitat restoration in New Zealand (Spurr & Berben 2004; Fisher et al. 2007; Watts et al. 2011), and are part of an iconic group of endemic Orthoptera that are of high conservation value (Watts et al. 2008). Within New Zealand, the anostostomatids are represented by tree wētā *Hemideina* spp., giant wētā *Deinacrida* spp., ground wētā *Hemiandrus* spp., and two species of tusked wētā (*Anisoura nicobarica* and *Motuweta isolata*). The Auckland tree wētā (*Hemideina thoracica*) is abundant throughout most of the northern North Island. It is a flightless, large-bodied (adults 3–7 g) generalist herbivore (Wehi & Hicks 2010; Wehi et al. 2013b) that rests during the day in tree cavities before emerging to feed at night. However, despite the use of tree wētā when monitoring restoration efforts, population estimates are difficult to achieve as most tree wētā are cryptic, and it is extremely difficult to sight tree wētā in cavities during the day without destructive sampling of trees. This limitation has led to the frequent use of artificial refuges in ecological studies of these wētā, with tree wētā quickly colonising artificial refuges that are placed close to natural refuges (Kelly 2006a; Wehi et al. 2013a).

We investigated patterns of natural refuge occupancy in the Auckland tree wētā in an urban forest remnant dominated

by mature kahikatea trees (*Dacrycarpus dacrydioides*). Large trees generally have more natural cavities than small trees (Blakely & Didham 2008). Accordingly, we expected that large trees in this urban remnant would provide both a greater number of cavities and a greater range of cavity sizes than small trees. We therefore predicted that tree wētā numbers would increase with tree size. Because tree wētā may take refuge in cavities that closely match their size (e.g. Kelly 2006b), we also predicted that the range of tree wētā sizes would be greater on large trees, concomitant with the likely broader range of refuge sizes. In addition, we estimated relative abundance of wētā within the remnant, and analysed observations of tree wētā emergence from natural cavities in relation to tree species, to investigate whether tree species was an important factor in habitat provision. Finally, to examine how artificial refuges may be best utilised to supplement wētā habitat in forest remnants, we investigated the influence of four structural characteristics on tree wētā occupancy of the artificial refuges: tree size, bark depth, proximity to a central artificially lit path, and proximity to the unlit perimeter edge.

Materials and methods

Field site

The study was conducted at Hillcrest Park (37º47′46″ S; 175º19′18″ E), a 1.5-ha urban remnant in Hamilton, New Zealand. Hillcrest Park is a remnant of the kahikatea lowland forest that previously covered this district, and is dominated by mature (20–25 m tall) kahikatea trees >100 years old. Few epiphytes were observed in these trees. During the study period, a small number of broadleaved species were present (e.g. *Aristotelia serrata*, *Melicytus ramiflorus* and *Laurelia novae-zelandiae*), but few were mature, as previous understorey growth had been removed by the city council for many years. A hall located within the forest remnant was accessed via a path that is very brightly lit in two places with gooseneck lights at a height of 7.5 m and a pair of spotlights at 10.5 m (K. Pudney, Hamilton City Council, pers. comm.). The park is bordered by sports fields, residential and commercial buildings with no connectivity to other forest remnants. Arboreal pest mammals such as ship rats (*Rattus rattus*) and mice (*Mus musculus*) were rare (Morgan et al. 2009).

Availability of natural cavities and tree wētā activity

We established a 10×120 m (1200 m^2) transect in the study site from 5 to 10 January 2010; summer is when peak mating and oviposition occur, and tree wētā are thus very active (Rufaut & Gibbs 2003). Within this transect, we identified all individual trees to species, and then measured diameter at breast height (dbh) for stems with dbh > 3 cm. Trees were tagged with individual number codes. Tree wētā emerge backwards from their cavities shortly after dusk, with some variation in timing (Rufaut & Gibbs 2003). We recorded tree wētā seen on all stems up to a height of approximately 5 m within the transect for five consecutive nights, for a 2.5-h period from dusk, thus concentrating our effort during the period when almost all tree wētā emerge. Observations were made by two groups of trained observers using torches moving from one end of the transect to the other before returning to the start and sweeping through a second time so as to capture both early and late emerging tree wētā. We captured all accessible tree wētā and marked them uniquely on their back legs. We also

recorded sex and morphological measurements. The right tibia was measured to estimate body size, as these measurements are strongly correlated in this species (Wehi & Hicks 2010). At times we could not remove wētā from cavities without potentially damaging them. In these cases, wētā were sexed and marked with a small spot on the abdomen to identify them as encountered but 'unmarked' wētā (also see Rufaut & Gibbs 2003). For all wētā observed, we recorded distance from the ground to each wētā location, and tree identification number.

The effect of tree species (kahikatea or other) and tree size on the tree count of wētā was tested using a generalised linear model with Poisson distribution and a log-link function with respect to the *F* distribution, because of overdispersion in the Poisson model (d.f. = 36, residual deviance = 76.3). The predictors in the model were tree species and dbh. For each group of trees, we estimated intercept and a slope parameter, and we tested the equality of these parameters between the two groups, and whether the slope could be taken as zero. Using Cook's distance we identified two trees that differed substantially from others in their respective groups (tree 19, a kahikatea with 7 wētā, and tree 36, a tōtara (*Podocarpus totara*) with 12 wētā) that could be highly influential on the analyses reported (Fox & Weisberg 2011). We therefore repeated the analysis without these two trees to see whether the presence of these two influential trees affected our conclusions.

Relative abundance estimates at natural sites

An estimate of abundance for this population was calculated using data collected from a 5-night wētā survey within the transect. We assumed a closed wētā population because (1) wētā are relatively long-lived invertebrates (e.g. Leisnham et al. 2003), (2) the survey period was 5 nights only, and (3) predation pressure within the study site was likely to be low, owing to a limited predatory guild; a predator survey conducted in 2007 did not detect mustelids, rodents or brushtail possums (*Trichosurus vulpecula*) (Morgan et al. 2009). However, hedgehogs (*Erinaceus europaeus*) were detected in low numbers, and cats (*Felis catus*) were living in some of the houses adjacent to the park (Morgan et al. 2009), although these were not observed in the park during predator surveys or during site visits for the current study. Both species prey on wētā (Gillies & Clout 2003; Jones et al. 2005). Because all the uniquely marked wētā that were recaptured over the survey period were only detected on the same tree (see Results), we assumed that any tree wētā only detected on a single night would also most likely have been present on that tree over the entire survey period (PMW & M Morgan-Richards unpubl. data). Therefore, these wētā were probably not detected on other occasions because they did not emerge, emerged late, or were not detected by the observation teams.

We limited our analysis to estimating relative abundance, despite appropriate protocols being in place to do a mark–recapture analysis, as wētā recapture numbers were too low for a full mark–recapture analysis to be robust. In addition, we were unable to sample any wētā emerging from cavities or epiphytes that may have been present at distances > 5 m from the ground. It is clear from this limitation that our estimates are based on a subsample of all wētā in the remnant. However, similar distance limitations occur in all published estimates of wētā abundance (e.g. Moller 1985; Rufaut & Gibbs 2003), and we therefore present our data so they can be compared with published data in a consistent manner.

We calculated maximum and minimum wētā abundance estimates within the transect. Maximum abundance estimates

were calculated by combining the total number of marked wētā with the total number of unmarked wētā detected over the survey period, and then dividing by transect area (for wētā m^{-2}) and the number of trees within the transect (for wētā per tree). To determine the minimum wētā abundance, the total number of marked wētā was combined with the greatest number of unmarked wētā found on each tree on a given night over the survey period, and then divided by the area or number of trees within the transect. Accordingly, if a given tree had two unmarked wētā on one night and three the next, for the maximum abundance estimate the total number of wētā for that tree was five, while for the minimum estimate the total was three (as we assumed that we were recounting two of the wētā on the second night). Therefore, it is likely that the minimum estimate under-represented the actual number of wētā while the opposite was true for the maximum estimate.

Artificial cavities on kahikatea trees

We randomly selected 20 mature kahikatea trees within Hillcrest Park as host trees for artificial refuges. Few other large trees were available at the study site, so we decided to standardise to the one species. Forty artificial refuges constructed from untreated and aged tōtara timbers (described in Wehi et al. (2013a)) were attached to marked kahikatea trees at 2.5 m in height (i.e. each tree had two refuges). Each refuge had two unlinked but same-sized cavities, with one entrance top left and one bottom right, and could accommodate up to eight adult tree wētā per cavity. The 80 artificial cavities were monitored daily for tree wētā occupancy for 28 consecutive days from the day they were first attached, and occupancy numbers recorded. Tree dbh, bark thickness, distance to edge of remnant, and distance to the nearest path were recorded for each tree. To estimate artificial light intensity at night, we measured light intensity at half-metre distances along three transects perpendicular to the central path, and also at the perimeter edge, using an ILM-1335 lightmeter, 400000 lx set on 30-s-interval timing. Maximum and minimum light readings were averaged for each distance point. To determine if tree characteristics (dbh, mean bark thickness) or distance from the internal path or perimeter could explain variations in artificial refuge occupancy, we constructed a correlation matrix using our artificial cavity dataset from the 28 days of monitoring. We then conducted a manual stepwise regression analysis by progressively adding and removing factors to determine which might best predict occupancy. To confirm the results of this model, we used the R step() function for stepwise AIC variable selection, and tested whether any of the terms could be dropped, following the method of Venables & Ripley (2002, p. 175). All statistical analyses were carried out in R (version 2.12.2; R Development Core Team 2011).

Results

Natural cavities and tree wētā activity

Thirty-nine trees > 3 cm dbh occurred in the transect, and stem density was $0.033\,m^{-2}$. Twenty-five trees were kahikatea and the remaining 14 were broadleaved or podocarp species (*Alectron excelsus* (1), *Aristotelia serrata* (6), *Beilschmiedia tawa* (1), *Kunzea ericoides* (1), *Melicytus ramiflorus* (2), *Laurelia novae-zelandiae* (1), *Plagianthus regius* (1), and *Podocarpus totara* (1)). The tree with the highest number of tree wētā recorded (*n* = 12) was a mature tōtara. We individually marked and measured

50 tree wētā (35 females, 15 males), and made 44 observations of unmarked tree wētā, over the 5-night survey period. All 44 unmarked wētā were observed at heights less than 4.5 m, but were partially emerged in entrance holes and impossible to mark. We calculated a minimum abundance estimate of 1.92 wētā per stem or 0.063 m^{-2}, based on a sampling height of up to 5 m above the ground. The maximum abundance estimate was 2.46 per stem, or 0.08 m^{-2}, using this sampling distance.

Wētā were recorded on 56.4% (22/39) of the trees >3 cm dbh in the transect. We observed tree wētā on the trunks of more broadleaved trees (71%) than kahikatea trees (48%); no tree wētā were observed on the stems < 3 cm dbh that were present within the transect. There were no differences by sex in the mean (±SD) distance at which they were observed from the ground (males 2.09 ± 1.34 m; females 1.65 ± 0.84 m). When we grouped trees into 'kahikatea' and 'other' varieties it was evident from the model that wētā were observed more often on 'other' trees, but whatever the species type, larger trees had more wētā on them (Fig. 1). The best-fitting model had separate intercepts for the two tree categories ($P = 0.013$) but a common slope coefficient (r = 0.039, SE = 0.013, $P = 0.193$). When we repeated the analysis excluding the two trees identified by Cook's distance as influential (kahikatea tree-19 and tōtara tree-36), the fitted curves were similar and the significance of tree size and group remained (despite a lack of data in the large tree-group for 'other' species, which weakens the model predictions). Captured tree wētā were from a range of instars (tibia length 9.61–23.81 mm). Tibia length was not correlated with tree dbh ($r^2 = 0.003$, $P = 0.29$); instead, tree wētā size was highly variable on large trees (Fig. 2). Because all uniquely marked tree wētā that were recaptured were only detected on the same tree, it appears that wētā movement from one refuge cavity to another does not occur on a nightly basis.
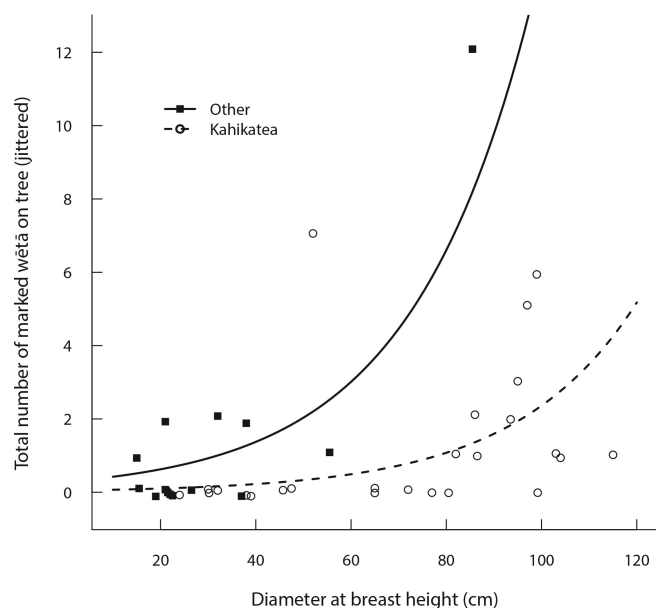

Figure 1. Effect of tree type (kahikatea (*Dacrycarpus dacrydioides*) or 'other') and tree size (dbh) on observations of wētā occurrence. As tree dbh increased, the number of wētā observed on trees also increased. 'Other' trees are primarily broadleaved species, with one *Podocarpus totara* in this category. A high number of tree wētā were observed on the individual tōtara. Outliers for both categories are shown here, but their inclusion or exclusion from the analysis did not change the overall result.
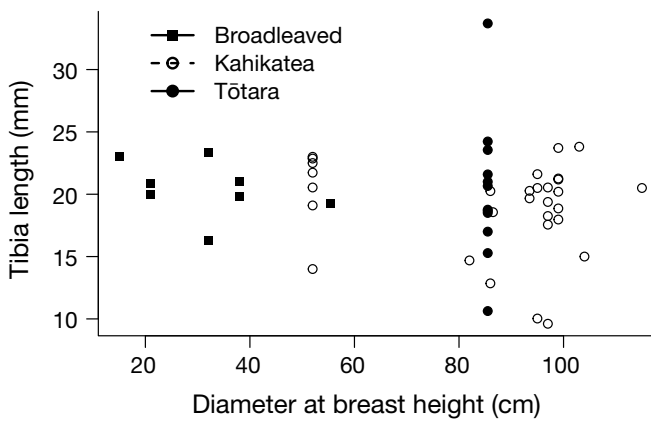
Figure 2. Tree wētā size, as estimated by tibia length, was not correlated with tree dbh, although the number of captured tree wētā increased with tree dbh. Each symbol represents a uniquely marked individual. Symbols differ for kahikatea (*Dacrycarpus dacrydioides*), tōtara (*Podocarpus totara*), and broadleaved trees.
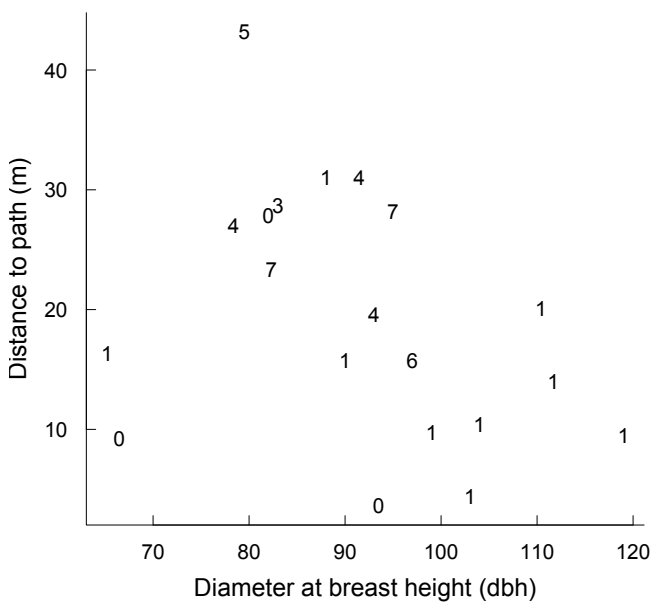
Figure 3. Tree wētā occupancy in relation to the two most important variables identified by stepwise regression analysis for the first 28 days after artificial refuges were installed. Plotting symbol is related to the mean number of wētā on a given tree (mean number of wētā on the tree over the 28 days, multiplied by 3 prior to rounding to make use of plotting symbols from 0 to 9). That is, the digits on the graph are a whole-number representation proportional to the number of tree wētā in the artificial cavities.

Artificial cavities and structural characteristics

Regression modelling indicated that dbh (r = 0.052, SE = 0.014) and distance to the internal path within the forest patch (r = 0.070, SE = 0.020) were the best predictors of occupancy in the first 28 days after the refuges were installed (r² = 0.53, Fig. 3). Wētā were slow to occupy artificial refuges near paths, but colonised artificial refuges on large trees more quickly than those on trees with smaller dbh. Light intensity was not included in the model, as it varied predictably with distance

from the central path. Both light intensity and variability were high within the first 10 m from the path, before low light conditions predominated. Lux readings of 31.25 were recorded near the artificial lights in the middle of the park, 1.30 lux at the park entrance, and mostly 0 lux from 15 to 20 m from the central path, on the park perimeter. Bark thickness and distance from the forest perimeter did not significantly improve occupancy predictions when added to the model ($F_{2,15}$ = 0.47, r² = 0.55, *P* = 0.64).

Discussion

In this urban forest remnant, the marking trial showed that tree wētā use a range of tree species, and that restoration planting of a range of tree species has successfully provided habitat for wētā. However, the lack of tree wētā observations on small stems indicates time is required for these benefits to occur. Tree wētā size was unrelated to tree dbh in the trial, indicating that tree wētā from a range of instars use large trees as habitat. However, tree wētā were most often observed on medium- and large-sized trees, with a higher number of wētā observations on 'other' tree species compared with kahikatea. The model results suggest that both tree size and species are likely to be important in the provision of habitat for tree wētā, and that some species such as tōtara could provide particularly favourable habitat for wētā. Nonetheless, further data exploring these trends would be useful as there were few large broadleaved or 'other' trees in the transect at Hillcrest Park, and hence our conclusions can only be tentative. Tree wētā populations may be limited by a lack of suitable cavities at some sites (Moller 1985; Field 1993) suggesting that, where possible, retention of large trees with more natural cavities (Blakely & Didham 2008) will help maintain healthy tree wētā populations. Results from the artificial refuge study are consistent with the estimates of relative abundance, with tree dbh similarly predicting refuge occupancy. The study further indicates that taking account of habitat factors when placing artificial cavities can maximise their worth as potential habitat in restoration efforts. In addition, installation of artificial cavities could promote a dense wētā population if large trees need to be removed.

We estimated tree wētā abundance at Hillcrest Park as between 0.06 and 0.08 m⁻². This figure is based on wētā observations from 0 to 5 m height within the transect, as with most other tree wētā abundance estimates (see below). Although detectability is likely to be incomplete, and hence estimates potentially low, all tree wētā studies using nocturnal observations have similar methodologies, with detectability and height restriction issues, and are thus broadly comparable. Our estimate is lower than the approximately 5 m⁻² on a predator-free offshore island (Rufaut & Gibbs 2003) and 0.53 ± 0.22 m⁻² estimated on Stephens Island (also mammal-free) using nocturnal counts (Moller 1985). It is also lower than the estimated abundance of 1.2 m⁻² from nocturnal counts of a mainland population of *Hemideina crassidens* at Cable Bay, in a small mixed broadleaved coastal forest remnant (Rufaut 1995; Gibbs 1998). It is higher, however, than two other recorded abundance estimates, of 0.031 m⁻² and 0.018 m⁻² respectively, for *Hemideina ricta* in severely modified habitat on the mainland (Townsend et al. 1997). Rufaut (1995) reported that tree wētā density was depressed in small, recently modified habitats and in the presence of rodents. Estimated wētā abundance in the modified urban forest remnant at Hillcrest Park, with one dominant podocarp species and a depauperate

understorey but relatively few rodents, is consistent with this hypothesis. However, further exploration of the effects of forest composition on tree wētā abundance could be enlightening.

Monitoring of cavity-dwelling invertebrates such as tree wētā can provide one indicator of restoration success in urban forest remnants. However, the regression model of tree wētā occupancy, using data from the artificial refuges, showed that tree wētā are not uniformly distributed through the remnant. Both tree size and distance from the internal path predicted occupancy of artificial refuges. Although ecological boundaries are a dominant feature of human-modified ecosystems, and can be an important predictor of species abundance (Ewers & Didham 2006; Harris & Burns 2000), the insignificant relationship between artificial cavity occupancy and perimeter distance suggests that more important modifiers of wētā behaviour are occurring within the remnant itself. Distance from the internal path was a strong predictor of artificial cavity occupancy. It is unclear exactly what characteristic associated with the internal path is responsible for this result. Human activity in and around the forest remnant is generally low at night when tree wētā are active. However, lux readings showed a consistent overall pattern in relation to distance from the central path, with very little light present around the perimeter of the forest fragment. It is thus possible that artificial lighting could be an important factor in wētā cavity occupancy. Artificial night lighting affects a wide suite of behaviours across many taxa (e.g. Bird et al. 2004; Baker & Richardson 2006; Kempenaers et al. 2010) and wētā are certainly less active on moonlit nights (Rufaut & Gibbs 2003). Although the abundance of nocturnal predators such as ship rats and morepork (*Ninox novaeseelandiae*) appeared to be relatively low at this site, this is unlikely to have always been the case. We suggest further research on the potential effects of artificial lighting on wildlife would be useful, especially in urban remnants.

Finally, we urge an increased emphasis on ecological studies of urban remnants. Tree felling, lighting, and path placement are essential elements of urban forest maintenance and planning. The predictors in this study imply that path proximity to high quality habitat (e.g. large trees) in urban remnants can affect the distribution of animals within the remnant. Yet although fauna in urban remnants can be source populations for locally important species and dispersal nodes in a network of connected wildlife sites (Hanski 1999), these remnants are also subject to pressures from local residents, housing, and council policy. In 2009, local neighbourhood groups petitioned the Hamilton City Council to extend both paving and lighting around the perimeter of the urban remnant studied here, and within the forest remnant itself. We suspect these requests are representative of city residents worldwide. In addition, large trees, particularly exotics, are sometimes removed from remnants without consideration of their wildlife value to native invertebrates. We conclude that the placement of paths, lighting and other structures and the removal of trees need to be carefully considered in relation to quality habitat for local fauna, if species and ecosystem conservation is a priority in urban remnants.

Acknowledgements

References

Aronson J, Dhillion S, Le Floc'h E 1995. On the need to select an ecosystem of reference, however imperfect: a reply to Pickett and Parker. Restoration Ecology 3: 1–3.

Baker BJ, Richardson JML 2006. The effect of artificial light on male breeding-season behaviour in green frogs, *Rana clamitans melanota*. Canadian Journal of Zoology 84: 1528–1532.

Bird BL, Branch LC, Miller DL 2004. Effects of coastal lighting on foraging behavior of beach mice. Conservation Biology 18: 1435–1439.

Blakely TJ, Didham RK 2008. Tree holes in a mixed broad-leaf–podocarp rain forest, New Zealand. New Zealand Journal of Ecology 32: 197–208.

Crane P, Kinzig A 2005. Nature in the metropolis. Science 308: 1225.

Ewers RM, Didham RK 2006. Confounding factors in the detection of species responses to habitat fragmentation. Biological Reviews 81: 117–142.

Fahrig L, Rytwinski T 2009. Effects of roads on animal abundance: an empirical review and synthesis. Ecology and Society 14 (1): 21. http://www.ecologyandsociety. org/vol14/iss1/art21/

Field LH 1993. Observations on stridulatory, agonistic, and mating behaviour of *Hemideina ricta* (Stenopelmatidae: Orthoptera), the rare Banks Peninsula weta. New Zealand Entomologist 16: 68–74.

Fisher P, Spurr EB, Ogilvie SC, Eason CT 2007. Bait consumption and residual concentrations of diphacinone in the Wellington tree weta (*Hemideina crassidens*) (Orthoptera: Anostostomatidae). New Zealand Journal of Ecology 31: 104–110.

Fox J, Weisberg S 2011. An R companion to applied regression. 2nd edn. Thousand Oaks, CA, Sage. 449 p.

Gibbs GW 1998. Why are some weta (Orthoptera: Stenopelmatidae) vulnerable yet others are common? Journal of Insect Conservation 2: 161–166.

Gillies C, Clout M 2003. The prey of domestic cats (*Felis catus*) in two suburbs of Auckland City, New Zealand. Journal of Zoology 259: 309–315.

Hanski I 1999. Metapopulation ecology. Oxford, Oxford University Press. 313 p.

Harris RJ, Burns BR 2000. Beetle assemblages of kahikatea forest fragments in a pasture-dominated landscape. New Zealand Journal of Ecology 24: 57–67.

Hobbs RJ, Norton DA 1996. Towards a conceptual framework for restoration ecology. Restoration Ecology 4: 93–110.

Jackson AK, Froneberger JP, Cristol DA 2013. Habitat near nest

boxes correlated with fate of eastern bluebird fledglings in an urban landscape. Urban Ecosystems 16: 367–376.

Jones C, Moss K, Sanders M 2005. Diet of hedgehogs (*Erinaceus europaeus*) in the upper Waitaki Basin, New Zealand: Implications for conservation. New Zealand Journal of Ecology 29: 29–35.

Kelly CD 2006a. Resource quality or harem size: what influences male tenure at refuge sites in tree weta (Orthoptera: Anostostomatidae)? Behavioral Ecology and Sociobiology 60: 175–183.

Kelly CD 2006b. The relationship between resource control, association with females and male weapon size in a male dominance insect. Ethology 112: 362–369.

Kempenaers B, Borgström P, Loës P, Schlicht E, Valcu M 2010. Artificial night lighting affects dawn song, extra-pair siring success, and lay date in songbirds. Current Biology 20: 1735–1739.

Laurance SGW, Stouffer PC, Laurance WF 2004. Effects of road clearings on movement patterns of understory rainforest birds in Central Amazonia. Conservation Biology 18: 1099–1109.

Leibold, MA, Holyoak M, Mouquet N, Amarasekare P, Chase JM, Hoopes MF, Holt RD, Shurin JB, Law R, Tilman D, Loreau M, Gonzalez A 2004. The metacommunity concept: a framework for multi-scale community ecology. Ecology Letters 7: 601–613.

Leisnham PT, Cameron C, Jamieson IG 2003. Life cycle, survival rates and longevity of an alpine weta *Hemideina maori* (Orthoptera: Anostostomatidae) determined using mark-recapture analysis. New Zealand Journal of Ecology 27: 191–200.

McDonnell MJ 2007. Restoring and managing biodiversity in an urbanizing world filled with tensions. Ecological Restoration 8: 83–84.

Miller JR 2005. Biodiversity conservation and the extinction of experience. Trends in Ecology and Evolution 20: 430–434.

Moller H 1985. Tree wetas (*Hemideina crassicruris*) (Orthoptera: Stenopelmatidae) of Stephens Island, Cook Strait. New Zealand Journal of Zoology 12: 55–69.

Morgan DKJ, Waas JR, Innes J 2009. An inventory of mammalian pests in a New Zealand city. New Zealand Journal of Zoology 36: 23–33.

Navara KJ, Nelson RJ 2007. The dark side of light at night: physiological, epidemiological, and ecological consequences. Journal of Pineal Research 43: 215–224.

R Development Core Team 2011. R: A language and environment for statistical computing. Vienna, Austria, R Foundation for Statistical Computing. www.r-project.org/

Rufaut CG 1995. A comparative study of the Wellington tree weta, *Hemideina crassidens* (Blanchard 1951) in the presence and absence of rodents. MSc thesis, Victoria University of Wellington, New Zealand.

Rufaut CG, Gibbs GW 2003. Response of a tree weta population (*Hemideina crassidens*) after eradication of the Polynesian rat from a New Zealand island. Restoration Ecology 11: 13–19.

Spurr EB, Berben PH 2004. Assessment of non-target impact of 1080-poisoning for vertebrate pest control on weta (Orthoptera: Anostostomatidae and Rhaphidophoridae) and other invertebrates in artificial refuges. New Zealand Journal of Ecology 28: 63–72.

Townsend JA, Brown B, Stringer IAN, Potter MA 1997. Distribution, habitat and conservation status of *Hemideina ricta* and *H. femorata* on Banks Peninsula, New Zealand. New Zealand Journal of Ecology 21: 43–49.

Trombulak SC, Frissell CA 2000. Review of ecological effects of roads on terrestrial and aquatic communities. Conservation Biology 14: 18–30.

Venables WN, Ripley BD 2002. Modern applied statistics with S. 4th edn. New York, Springer. 495 p.

Watts CH, Thornburrow D, Green CJ, Agnew WR 2008. Tracking tunnels: a novel method for detecting a threatened New Zealand giant weta (Orthoptera: Anostostomatidae). New Zealand Journal of Ecology 32: 92–07.

Watts CH, Armstrong DP, Innes J, Thornburrow D 2011. Dramatic increases in weta (Orthoptera) following mammal eradication on Maungatautari – evidence from pitfalls and tracking tunnels. New Zealand Journal of Ecology 35: 261–272.

Wehi PM, Hicks BJ 2010. Isotopic fractionation in a large herbivorous insect, the Auckland tree weta. Journal of Insect Physiology 56: 1877–1882.

Wehi PM, Jorgensen M, Morgan-Richards M 2013a. Sex- and season-dependent behaviour in a flightless insect, the Auckland tree weta (*Hemideina thoracica*). New Zealand Journal of Ecology 37: 75–83.

Wehi PM, Raubenheimer D, Morgan-Richards M 2013b. Tolerance for nutrient imbalance in an intermittently feeding herbivorous cricket, the Wellington tree weta. PLoS ONE 8(12): e84641. doi:10.1371/journal.pone.0084641.

Whaley PT, Clarkson BD, Smale MC 1997. Claudelands Bush: ecology of an urban kahikatea (*Dacrycarpus dacrydioides*) forest remnant in Hamilton, New Zealand. Tane 36: 131–155.