Milestone 3.3 Linking predator camera trap monitoring to biodiversity and economic benefits: density-impact functions in principle

Scoping report to Hawke's Bay Regional Council

Grant Norbury, Al Glen and Roger Pech
Landcare Research

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Introduction

One of the fundamental tenets of cost-effective pest management is knowing the control effort required to achieve the desired benefits. This knowledge is lacking for many pest systems (Caughley and Gunn, 1996; Grice, 2009), and consequently managers risk not applying enough effort, or conversely, overcommitting scarce resources. The critical knowledge gap is the form of the relationship between pest density and pest impact, which we call a 'density-impact function' or DIF. Impacts are measured as changes in a 'response variable' that can be either economic or ecological, e.g. increase in pasture biomass (when rabbits are controlled) or better survival of native birds (when rats and possums are controlled) (Hone, 2007; Yokomizo et al., 2009). Depending on the shape of this function, a DIF can identify a pest density threshold that managers can use to set a benchmark for expenditure on pest control (see Edge et al., 2011). An example for Cape-to-City is the reduction in cat density required to reduce toxoplasmosis in sheep to an acceptable level.

Theoretical DIFs and empirical evidence

The shape of a DIF has important implications for management. Six types of theoretical DIFs are illustrated in Fig. 1.
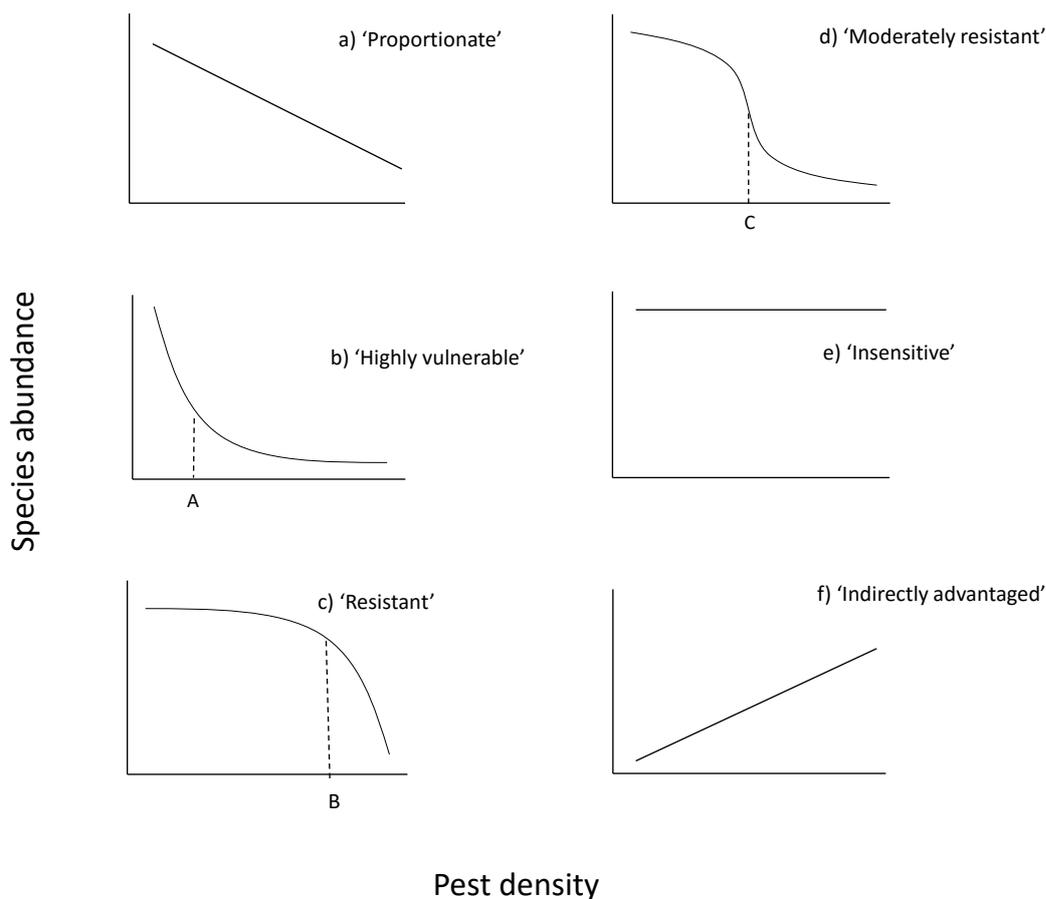
Fig. 1. Six theoretical relationships between pest abundance and a conservation 'response variable', such as the abundance of a native species. Thresholds (A, B and C) indicate where a small change in pest density results in a large change in the response variable. Modified from Norbury et al. (2015).

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'Proportionate' DIFs represent the default condition where there are consistent, incremental changes in the response of impacted biota to incremental changes in pest density (Fig. 1a). In this case, any level of pest control will achieve some benefit. Norbury et al. (2015) reviewed empirical evidence for DIFs in New Zealand and overseas and found that, for New Zealand, proportionate functions comprised 19% of 21 DIFs published. Proportionate DIFs applied to ship rats and robins in a podocarp-broadleaved forest in the lower central North Island (Armstrong et al., 2006) (Fig. 2), and to possums and tree fuchsia in native beech forest in South Westland, South Island (Pekelharing et al., 1998).
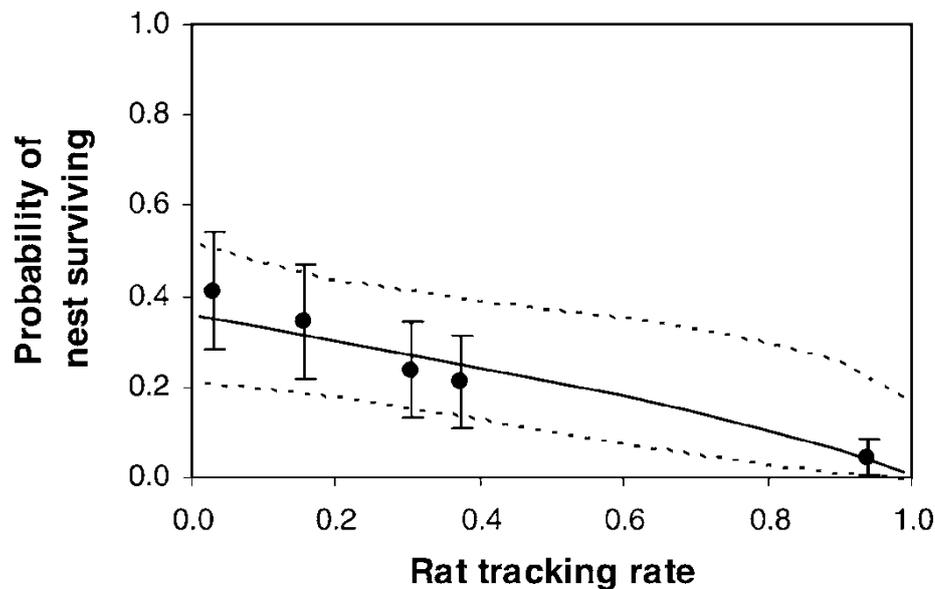
Fig. 2. Relationship between the probability of North Island robin nests surviving until one or more young fledge, and rat tracking rate (index of rat abundance based on data from tracking tunnels). Points show yearly estimates of nest success compared with rat tracking rates from August–February of the same years. Vertical bars are standard errors. The solid line shows the fitted function, and broken lines show 95% confidence intervals. From Armstrong et al. (2006).

Non-linear DIFs (Figs. 1b-d) indicate thresholds where the response variable changes rapidly with small changes in pest density (Suding and Hobbs, 2009). Three basic types of non-linear relationships are shown. 'Highly vulnerable' biota show positive responses only when pests are suppressed to very low densities or, in extreme cases, when all pests are removed (Fig. 1b). This was the most common DIF (43%) published for New Zealand systems. Pest density thresholds were found for stoats (<1-2 per km^2 to maintain northern brown kiwi populations in mixed beech-podocarp forest) (Basse et al., 1999), ship rats (<3 per ha to maintain high numbers of Auckland tree weta in mixed beech-podocarp forest) (Ruscoe et al., 2013) and hedgehogs (<1 per ha to increase numbers of McCann's skinks and ground weta in native tussock grassland) (Jones et al., 2013) (Fig. 3). These DIFs imply that pest control will generate little conservation benefit unless it is applied at an intensive level.
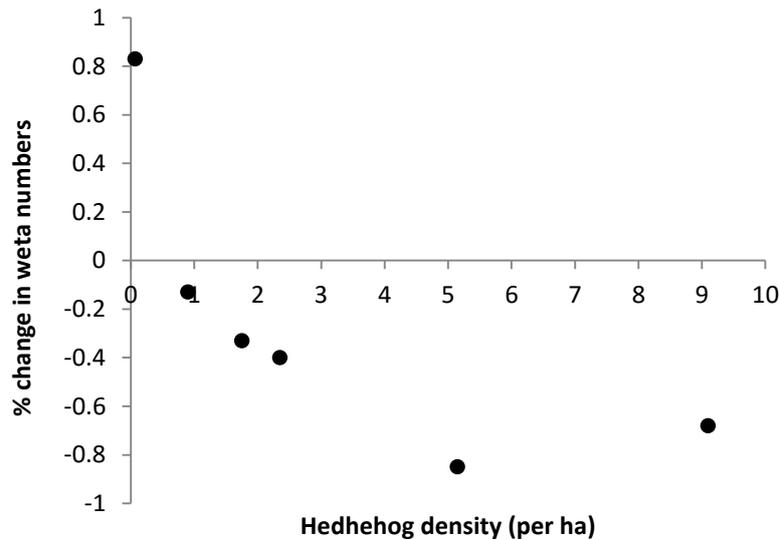
Fig. 3. Relationship between hedgehog density and proportional changes in numbers of ground weta before and after 3 months of exposure to hedgehog predation. Redrawn from Jones et al. (2013).

'Resistant' biota suffer major negative impacts only at high pest densities (Fig. 1c). This DIF (14% of New Zealand examples) applied to changes in the survival of kamahi and toro seedlings in response to possum control (Nugent et al., 2001), implying that only modest levels of pest control are required to promote regeneration of these tree species, and that further control generates little extra benefit. 'Moderately resistant' biota (10% of New Zealand examples) are intermediate between the highly vulnerable and resistant conditions – in this scenario, major negative impacts occur at intermediate-to-high pest densities (Fig. 1d). These applied to browsing by possums on tree fuchsia and five finger in podocarp-broadleaved forest (Nugent et al., 2001), and to predation by stoats and possums on bellbirds in beech forest (Masuda et al., 2014).

'Insensitive' biota reflect the null hypothesis of no detectable impact across all pest densities (Fig. 1e). This applied to ship rats and adult male robins in podocarp-broadleaved forest (Armstrong et al., 2006), and to a range of predator species and fantails in beech forest (Elliott et al., 2010). It also applied to possums and canopies of rewarewa and silver beech (Gormley et al., 2012). Whilst only 5% of published DIFs in New Zealand were categorised as 'insensitive', their scarcity may be due to a research emphasis towards high-profile, vulnerable species in need of intensive restoration. We know, for example, that New Zealand contains an abundance of indigenous biota that persists in the presence of pests. This is especially true for many plant species whose structural and/or chemical defence mechanisms are effective against invasive herbivores (McGlone and Clarkson, 1993).

A sixth DIF (10% of New Zealand examples) represents biota that respond *favourably* with increasing pest density ('Indirectly advantaged' species; Fig. 1f). For example, some indigenous species are unaffected directly, but benefit from the removal of indigenous competitors by pests. Such species may increase with increasing pest abundance, and decline with pest control. This appears to be the case for a range of predator species and the grey warbler. Grey warbler, a recent endemic to New

Zealand, while sometimes initially increasing in abundance after pest control, in the longer-term can decline if outcompeted by older endemic bird species that also respond favourably to pest control (Innes et al., 2010; Masuda et al., 2014).

How reliable are DIFs for identifying pest abundance targets?

One of the problems with DIFs is that they tend to be place- and time-specific, and do not include the effects of factors other than pests (see Ricciardi et al., 2013). Cruz et al. (2016) reduced the confounding influence of these other factors by measuring some of them in the field (e.g. site aspect and elevation) and accounting for them with statistical modelling. In the case of Cape-to-City and Poutiri Ao ō Tāne, such factors might include weather at the time of sampling, and habitat type. An alternative and more robust approach than DIFs is to use detailed numerical models of the interactions between consumers (pests) and resources (native biota) to predict pest impacts (Choquenot, 2006; Choquenot and Parkes, 2001). However, these models can be complex and difficult to parameterise because they require additional, often long-term, data that are logistically difficult or expensive to obtain, and we know of no conservation managers who use them. Managers are more likely to make use of pest abundance targets in a DIF as a 'rule of thumb' rather than fully-fledged consumer-resource models. We recommend that managers derive DIFs as robustly as possible for their particular system (see below), and interpret them in light of the potentially distorting factors discussed later. While some of the published New Zealand DIFs identify pest density targets that may apply to habitat in Cape-to-City and Poutiri Ao ō Tāne, we recommend deriving region- or site-specific DIFs using the pest abundance and biodiversity data collected during these projects.

A second potential problem is that the simple DIFs in Fig. 1 can be obscured by lags in species' responses. This is especially important for highly dynamic systems where populations of pests and indigenous species change at very different rates (e.g. rats and skinks). Data should be gathered at time scales long enough for impacted species to adjust to the new level of pest abundance. Density-impact functions may therefore take several years to derive in some cases.

A third factor affecting the reliability of DIFs is the complex issue of trophic interactions among pest species. For example, reducing numbers of top predators (e.g. ferrets or feral cats) can benefit sub-ordinate, usually smaller predators (e.g. rodents). This 'mesopredator release' can nullify any benefits for indigenous biota (Courchamp et al., 1999; Norbury et al., 2013). Not understanding these interactions poses a risk that management based on a DIF for a single pest species can be ineffective due to impacts of other pest species. The recommended management solution in these cases is usually simultaneous control of multiple pest species (Burns et al., 2011; Saunders and Norton, 2001), but this requires an understanding of which species cause the greatest damage.

Deriving DIFs for Cape-to-City and Poutiri Ao ō Tāne

The data collected in the Cape-to-City and Poutiri Ao ō Tāne projects have the potential to generate DIFs for invertebrates, lizards, birds and toxoplasmosis, provided they are interpreted in light of the other potential influences described above. The following issues also need to be addressed specifically for these projects.

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Which pest species to use?
The pest species to use as the predictor variable on the x-axis of a DIF depends on the outcome of interest. For benefits such as low toxoplasmosis levels in sheep, the pest species is clear (cats). Similarly for the survival of translocated robins and tomtits, the relevant pests are rats (and perhaps stoats). For other desired benefits, such as increased abundance of ground-dwelling invertebrates, lizards and some bird species, the pest species is less clear and more likely to be several species, e.g. cats, ferrets, stoats, hedgehogs, and rats, which are all known to depredate ground-dwelling fauna. In this case, a composite measure of multiple pest species is required and the data may need analysis by a specialist biometrician.

Which measure of pest abundance?
Estimates of true pest abundance or density are often expensive and difficult. Indices of relative abundance are easier and cheaper to obtain. Indices can be derived from data recorded with cameras and tracking tunnels, or from trap-catch data. Trap-catch data are collected as part of the trapping programme, so they do not incur extra cost, but the number of pests removed with kill-traps does not provide any information about the untrappable part of the pest population. It is much better to have an independent measure of pest abundance, e.g. using cameras that can record residual pest numbers after control operations. While tracking tunnels are a standard monitoring tool for many pests, the tunnels used at Cape-to-City and Poutiri Ao ō Tāne are not particularly good at detecting some species, such as stoats and cats. Cameras are much more effective for detecting the larger pest species (cats, ferrets and stoats) and provide the best measure of pest abundance for DIFs, particularly as farmers and agencies are most likely to use them for monitoring in the long-term. While cameras also detect rodents reasonably well, we recommend tracking tunnels for monitoring rodents, which are cheap and usually record clearly-identifiable rodent tracks. An additional advantage of both cameras and tracking tunnels is that they also detect native fauna, depending on body size.

Generating a pest abundance spectrum
A DIF is most useful if it represents the full spectrum of pest densities and benefits likely to be encountered. This spectrum can be generated from both temporal data (i.e. pre and post pest control) and spatial data (i.e. spatial variation in pest abundance). However, most pest control programmes initiate pest control without monitoring pests or expected response variables (e.g. native biota) beforehand, and so a full spectrum can be difficult to achieve, especially for high pest densities. The Cape-to-City and Poutiri Ao ō Tāne projects are in a reasonably strong position in this regard as monitoring preceded the start of pest control, and the project design includes treatment and non-treatment areas.

Scaling benefits to pest abundance
Response variables need to be recorded at the same scale as indices of pest abundance. For example, toxoplasmosis levels in sheep are recorded on only six properties. Therefore, indices of cat abundance should be derived from a subset of cameras that are local to each toxoplasmosis site. Similarly, for birds translocated to forest patches, site-based measures of birds should be coupled with site-based indices of rat abundance. Invertebrates, lizards and some bird species are monitored

across a much wider area, allowing more options for pest and response data to be aggregated at various levels across a large area.

Scaling and aggregating data in a biologically meaningful way requires careful planning and consideration. For example, how should the pest index data from the widely-spaced camera grid be coupled with biodiversity measures that are recorded at different locations? This will depend to some extent on which biodiversity measures are aggregated or pooled across monitoring transects. If the level of aggregation is low (i.e. coupling pest and biodiversity data at single sites), only a few cameras will provide pest information that pertains to those sites, thereby providing potentially imprecise estimates. More robust estimates will ensue as data are aggregated using clusters of sites, but if aggregation is too high it can potentially obscure valuable site-based variation that can be used to generate a DIF. Statistical modelling may be required to optimise the level of data aggregation.

Another way of using cameras to generate pest indices for DIFs is to deploy them strategically in clusters when and where more robust indices are required. This might be a better approach for generating a toxoplasmosis DIF, for example.

A more simplistic approach than DIFs

Rather than deriving a complete density-impact relationship across a full range of pest abundances, a simpler but less useful approach is to measure the expected benefit at some appropriate point in time after pest control has been maintained to see whether it is at a satisfactory level. If it is, the concurrent pest index could be an indicative management target. The risk with this approach is that, depending on the shape of the DIF, the same benefit might be achieved with less control effort and a higher pest abundance – in which case there would be over commitment of pest control resources. A worst case scenario is that what is thought to be a successful level of management could correspond to a pest density close to a threshold, as illustrated in Figures 1(b), (c) and (d). In this case there would be major adverse impacts from a small increase in pest abundance. If the expected benefit is not at a satisfactory level, further reductions in pest abundance will be required as well as further monitoring, and so on, until the desired result is achieved. This iterative process is likely to be less efficient and more protracted and expensive than generating a DIF.

Conclusion

Density-impact functions provide a framework for assessing the control effort required to achieve the required level of environmental and economic benefits. The data collected in the Cape-to-City and Poutiri Ao ō Tāne projects have the potential to generate DIFs for invertebrates, lizards, birds and toxoplasmosis, provided they are interpreted in light of other potential influences. Cameras are likely to generate the most robust indices of pest abundance, especially for large species, although tracking tunnels are often more appropriate for rodents. Both techniques can also detect native fauna, depending on body size. Ideally, indices of pest abundance and response variables should be measured across the full range of values likely to be encountered. This range can be generated by differences between sites, differences over time, and by comparisons before and after pest control is initiated. Pest and impact data need to be scaled correctly so they represent biologically meaningful comparisons. Data

should be gathered at time scales long enough for the impacted resources to adjust to the new level of pest abundance. Density-impact functions may therefore take several years to derive in some cases.

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