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Potential for invasive mammalian herbivore control to result in measurable carbon gains

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Abstract: Invasive mammalian herbivores (e.g. deer, feral goats and brushtail possums; hereafter ‘herbivores’) are widespread throughout New Zealand and their control is important for conservation. In addition to known biodiversity benefits, it has recently been suggested that herbivore control could lead to measurable carbon gains when aggregated across a large area of conservation land. However, a significant amount of uncertainty exists regarding the potential effects of herbivore control on carbon, and the practicalities of successfully implementing such projects. This paper provides a general basis for managers and ecologists to design scientifically robust herbivore control projects for carbon gain in New Zealand. Although there are few direct data on changes in carbon sequestration rates following herbivore control, the data that are available suggest that effect sizes are likely to be small in magnitude, variable in direction, and to occur primarily through complex indirect mechanisms. The largest positive effects of herbivore control (carbon sequestration rate of 1–2 Mg C ha⁻¹ year⁻¹) are likely to occur in localised areas of highly palatable early-successional vegetation and high herbivore densities where control initiates rapid development of woody vegetation. Project location is therefore critical in determining the potential for carbon gain in herbivore control projects. A power analysis reveals that the ability to monitor changes in carbon stock using plot-based methods is limited to effect sizes of > 0.5 Mg C ha⁻¹ year⁻¹, as smaller effect sizes would require an impractically large number of plots (i.e. >100), and the financial and carbon costs of implementing the control and quantifying the effects are likely to outweigh any potential carbon gains. Although more research is urgently required to quantify potential gains, and the mechanisms that underlie them, our findings suggest that with careful site selection, implementation, and monitoring, control of invasive mammalian herbivores could sometimes provide carbon gains in certain areas of New Zealand’s indigenous vegetation.

Keywords: additionality; biodiversity; carbon sequestration; conservation; invasive species; mammalian herbivory; power analysis; project design; succession

Introduction

Invasive mammalian herbivores (hereafter ‘herbivores’), such as deer (7 species), feral goats (*Capra hircus*), and brushtail possums (*Trichosurus vulpecula*), are widespread throughout New Zealand (Fraser et al. 2000; Nugent et al. 2001; Allen & Lee 2006), and can have significant direct and indirect effects on forest structure and composition, successional trajectories, nutrient cycling, and below-ground processes (e.g. Smale et al. 1995; Nugent et al. 2001; Wardle et al. 2001; Forsyth et al. 2010a). For example, red deer (*Cervus elaphus scoticus*) have been shown to inhibit tree seedling regeneration (e.g. Wilson et al. 2006) and dramatically reduce the abundance of palatable species in the forest understorey (e.g. Allen et al. 1984; Mark et al. 1991; Forsyth et al. 2003). Preferential browsing by both red deer and fallow deer (*Dama dama*) can lead to altered successional trajectories due to the failure of palatable broadleaved species such as *Melicactus ramiflorus* to establish within early-successional kānuka (*Kunzea ericoides*) forests (Payton et al. 1984, Smale et al. 1995). The observed impacts of invasive herbivores on the structure and function of New Zealand’s natural ecosystems justify the importance of controlling their population for conservation management (Allen & Lee 2006).

Recent research has focused on whether herbivore control could potentially benefit carbon sequestration (Allen & Carswell 2008; Peltzer et al. 2010; Tanentzap & Coomes 2012); an environmentally and politically desirable outcome for New Zealand that would contribute to international climate change responsibilities to enhance and preserve natural carbon sinks (Article 4.1(d) of the United Nations Framework Convention on Climate Change), while simultaneously supporting national biodiversity conservation objectives (Allen et al. 2003; Diaz et al. 2009; Carswell et al. 2012). Given the large area of land currently under conservation management in New Zealand (approximately 8 Mha, 30% of New Zealand’s total land area), even a small per-hectare increase in carbon stock across all public conservation land would result in significant net gains at national scale (Carswell et al. 2008), potentially providing a source of additional revenue for conservation management through the generation of tradable carbon credits. These factors have recently led to considerable interest and investment occurring in herbivore control projects for carbon gain (e.g. Solid Energy New Zealand 2009). However, a large amount of uncertainty exists regarding the magnitude of the potential effects and the practicalities involved in successfully implementing such projects. This uncertainty is driven by a lack of primary data on the effects of invasive herbivores

on forest carbon stocks (Tanentzap & Coomes 2012), the complexity of the possible mechanisms (Wardle et al. 2001; Coomes et al. 2003; Peltzer et al. 2010), the costs involved in reducing animal numbers to sufficiently low densities to elicit a positive ecosystem carbon response, and the methodological challenges involved in quantifying the additional carbon gains (or losses) attributable to the management activity (Allen & Carswell 2008), especially within the existing carbon accounting frameworks (e.g. Fahey et al. 2010). Here, we do not account for the emissions associated with the achievement of wild animal control (such as helicopter fuel) as these have been deliberately separated from forest carbon gains within the New Zealand policy context and are implicitly accounted for within other sections of the New Zealand Emissions Trading Scheme.

This paper provides a general basis from which land managers and ecologists can design scientifically robust invasive herbivore control projects for carbon gain (hereafter 'herbivore control projects') in natural forests, shrubland and tussock grasslands. First, we briefly outline the potential ecological mechanisms that could result in carbon change following herbivore control, and identify the most likely situations where significant gains may be achieved. We then examine the design of herbivore control projects, focusing on the importance of selecting an appropriate project location, the monitoring regime that would be required to robustly quantify changes in carbon within the project area, and the factors to consider when implementing herbivore control. Finally, we discuss the use of appropriate baseline sequestration rates to separate additional carbon gains (i.e. those that are directly attributable to the management intervention) from gains that would have occurred without intervention. Many of the design concepts mentioned here apply to other forms of carbon-related management intervention, not just herbivore control, and as such are applicable to most terrestrial carbon sequestration projects. Although more research is urgently required to quantify potential gains and the mechanisms that underlie them, we suggest that with careful site selection, effective implementation, and appropriate monitoring, control of invasive herbivores does sometimes have the potential to provide both carbon gains and conservation benefits in New Zealand's indigenous vegetation.

Effects of invasive herbivores on carbon

There are currently few data quantifying the effects of invasive herbivores, or herbivore control, on carbon stocks (Wardle et al. 2007; Forsyth et al. 2010a; Peltzer et al. 2010; Tanentzap & Coomes 2012). There are, however, considerable data documenting the effects of invasive herbivores on other community and ecosystem properties (e.g. Wardle et al. 2001; Allen & Lee 2006), and such data can be used to make predictions regarding the effects of wild animals on community-level carbon stocks (Peltzer et al. 2010). In this section we outline the potential ecological mechanisms that could result in carbon change following herbivore control, identifying the most likely situations where carbon gains may occur. We focus our attention on the effects of the most abundant large to medium-sized invasive herbivores (deer, feral goats, brushtail possums, rabbits (*Oryctolagus cuniculus*), and hares (*Lepus europaeus occidentalis*)), but include the effects of omnivorous/carnivorous invasive species such as feral pigs (*Sus scrofa*), mustelids, and rodents where they are likely to be significant.

Direct consumption of biomass

Herbivores have a direct effect on carbon stocks through the consumption of vegetation (primarily leaves) and its metabolic conversion to CO₂ and methane (Hollinger & Hunt 1990; Swainson et al. 2008). Typical annual dry matter consumption per individual is approximately 240–490 kg C for red deer, 90–180 kg C for goats and 3–40 kg C for possums (Nugent et al. 1997; Jorritsma et al. 1999; King 2005; Cowan 2007). Not all plant material eaten is lost to the atmosphere – a significant proportion of this ingested carbon passes intact through the animal to become part of the soil carbon pool (King 2005). Plants may offset a portion of the direct consumption of biomass through compensatory growth (McNaughton 1983), further reducing the direct effects of browsing on carbon stocks. Moreover, in forested situations only a small fraction of the total biomass is typically contained in palatable vegetation, and an even smaller fraction of this is located within the 'browse' tier (within 2 m of the ground for terrestrial herbivores). The direct effects of biomass consumption (by both terrestrial and arboreal herbivores) on carbon stocks in forested systems are therefore likely to be low (Peltzer et al. 2010; Tanentzap & Coomes 2012). The direct effects of biomass consumption are likely to be greater in shorter, primarily herbaceous, vegetation (e.g. tussock grassland or grasslands transitioning to forests) where a much larger fraction of the total biomass pool is palatable (Rose & Platt 1992; Forsyth et al. 2010a).

Ruminant animals such as deer, goats and pigs also convert some of the ingested carbon into methane. Marsupial mammals such as possums emit little or no methane due to their different intestinal physiology compared with placental mammals (Kempton et al. 1976). Although methane emissions from wild deer are estimated at only 6 kg CH₄ per individual per year (King 2005; Swainson et al. 2008), this is equivalent to annual emissions of 138 kg CO₂ per individual due to the high global warming potential of methane (Lassey 2007). Assuming an average age of 4 years for controlled animals, an average life expectancy without control of 10 years (King 2005), and no compensatory population growth, this translates into reduced emissions of approximately 0.8 Mg CO₂e equivalents (CO₂e) per deer. A similar calculation performed for goats gives methane-related savings of approximately 0.12 Mg CO₂e per goat (Hollinger & Hunt 1990; King 2005). Due to the existence of compensatory population growth, actual reductions in methane emissions are more appropriately calculated at the population level, with emissions savings that are proportional to the difference between peak and controlled densities. For example, intensive ongoing control can reduce red deer numbers from peak densities of 30–50 deer km⁻² to 3–4 deer km⁻² (King 2005; Forsyth et al. 2010a). If these controlled densities are maintained, then the annual methane emissions savings would be 26–47 deer km⁻² × 0.8 Mg CO₂e per deer = 20.8–37.6 Mg CO₂e km⁻² year⁻¹, regardless of the actual number of animals shot each year. For goats, peak densities of approximately 70–100 animals km⁻² and controlled densities of 5–13 animals km⁻² have been observed on mainland New Zealand (King 2005); equating to emissions savings of 57–95 goats km⁻² × 0.12 Mg CO₂e per goat = 6.84–11.4 Mg CO₂e km⁻² year⁻¹, provided that controlled densities are maintained. Avoidance of methane emissions is therefore a potentially important direct effect of wild animal control, especially for large-scale control programmes that successfully result in large reductions in ruminant animal densities.

Increased seedling mortality and successional change

Palatable seedlings and saplings within the browse tier are thought to suffer reduced growth and survival rates in the presence of wild animal herbivores (Husheer et al. 2006; Forsyth et al. 2010a). Over time, these demographic changes can lead to changes to species composition, with increased abundance of unpalatable species and decreased abundance of palatable species commonly observed (Allen et al. 1984; Mark et al. 1991; Wardle et al. 2001; Forsyth et al. 2005; Mason et al. 2010). For forest carbon, the biggest effects are likely to be seen in broadleaved forests where a large portion of the total basal area is comprised of palatable species (Coomes et al. 2003); however, to date no study has shown a herbivore-induced lack of tree canopy replacement in old-growth forest (Forsyth et al. 2010a). In most other forest types, palatable species (e.g. *Coprosma grandifolia*) typically make up a low portion of the total biomass (Beets et al. 2009), and are often replaced following heavy browsing by equivalently sized unpalatable species (e.g. *Pseudowintera colorata*; Coomes et al. 2003; Forsyth et al. 2005). Stand-level carbon is likely to be unaffected by replacement of palatable species with non-palatable species. However, in some regenerating forests, seedling herbivory has been shown to prevent the establishment of high-biomass late-successional species, thus altering the course of forest development and potentially reducing the long-term carbon sequestration potential. An example of this is the prevention of broadleaved tree establishment in regenerating kānuka caused by fallow deer on south Kaipara spit (Smale et al. 1995). Browsing of low-biomass understorey species may also have disproportionately large effects on soil properties (Peltzer et al. 2009), and could, through competitive release, lead to increases in canopy-tree growth in the presence of herbivores. The effects of browsing-induced seedling mortality on carbon are therefore potentially important but highly variable depending on the age of the forest, local successional processes, and the traits of the species present at any particular site.

Increased canopy mortality

Herbivores can increase the mortality rates of canopy trees through direct browsing (possums; Bellingham et al. 1999) or through ring-barking (deer, goats; Akashi & Nakashizuka 1999). Extensive canopy dieback has been attributed primarily to possum browsing in *Weinmannia racemosa* (kāmahī) dominated forests on the Urewera Ranges (Payton et al. 1984), rātā-kāmahī forest in South Westland (Payton 1988) and mixed broadleaved forests in the southern Ruahine Range (Rogers & Leathwick 1997). However, in some situations perceived dieback caused by possums does not equate to increased mortality beyond background levels (Bellingham et al. 1999). The effects of canopy dieback on ecosystem carbon are uncertain. As a tree dies, its carbon becomes part of the coarse woody debris pool and is not immediately lost to the atmosphere (Richardson et al. 2009). Dead wood of some high-biomass tree species with high wood-density (e.g. southern rātā *Metrosideros umbellata*) may take many decades to break down due to slow decay rates (Beets et al. 2008, D. Peltzer unpubl. data). In the meantime new growth may have occurred in the space previously occupied by the dead tree, potentially increasing ecosystem carbon stock. The effect of herbivore-induced canopy mortality on carbon is therefore highly variable, with the largest declines in carbon due to herbivore activity likely to occur in forests with high possum

numbers dominated by canopy trees that decay quickly (e.g. kāmahī; Rose et al. 1992) and are not rapidly replaced by other high-biomass species.

Soil feedbacks

The indirect effects of herbivores on soil carbon are numerous and complex (Wardle et al. 2001; Bardgett & Wardle 2003). Soil disturbance from pig rooting is likely to result in increased respiration and loss of soil carbon. Browsing-induced replacement of high-nutrient species with low-nutrient species can lead to reduced litter quality, slowing the carbon cycle and resulting in increased carbon storage in the soil and litter layers (Wardle et al. 2002; Bardgett & Wardle 2003). Trampling by large herbivores can result in soil compaction and reduced soil biota and litter layers, potentially slowing accumulation of carbon in the mineral soil layers (Basher & Lynn 1996; Wardle et al. 2001; Drewry 2006), while nutrient deposition from animal faeces is likely to increase nutrient cycling and plant growth rates (Bardgett & Wardle 2003; Fukami et al. 2006). Wardle et al. (2007) showed that islands with abundant burrowing seabird populations had increased soil carbon but decreased understorey plant growth, and lower ecosystem carbon stocks, when compared with rat-invaded islands with low seabird populations (Fukami et al. 2006; Wardle et al. 2007). In a study of 30 long-term fenced enclosure plots, Wardle et al. (2001) found that terrestrial herbivores did affect soil carbon in some situations, but that the direction of the effects was idiosyncratic. These results reflect the complex and variable mechanisms by which herbivores affect soil processes, and how these in turn influence stand-level carbon storage.

Seed predation and dispersal

Seed predation has the potential to limit establishment of seed-limited species and therefore reduce the rate of forest development (and thus carbon sequestration) during secondary succession in situations where seed sources for high-biomass species are scarce (Nepstad et al. 1996; Wijdeven & Kuzeev 2000; Wilson et al. 2003). Invasive mammals such as pigs, rats and mice are known to be important seed predators of canopy trees such as *Prumnopitys* spp. and *Nothofagus* spp.; however, the extent to which seed predation by these animals limits seedling establishment is uncertain (Moles & Drake 1999; Wilson et al. 2007). Conversely, invasive animals such as pigs, deer and possums may also be acting as important seed dispersers for some fleshy-fruited species (Williams et al. 2000; Dungan et al. 2002; L.M. Young unpubl. data). Invasive-animal-assisted seed dispersal could therefore be increasing the rate of forest regeneration in certain situations, especially in highly fragmented habitats with low bird abundance and a predominance of fleshy-fruited species. Although the magnitude of these effects in terms of carbon is likely to be small (i.e. $<0.1 \text{ Mg C ha}^{-1} \text{ year}^{-1}$), these studies highlight the complexity of between-animal interactions and their potential importance for understanding the net changes in ecosystem carbon stocks following herbivore control.

Synthesis of likely effects

A number of underlying patterns emerge from the hypothesised effects of invasive herbivores on terrestrial carbon stocks described above.

- Effects are likely to be small in magnitude (relative to carbon stocks) and occur primarily through complex indirect mechanisms.

- Multiple above- and below-ground mechanisms are likely to be operating at any one time, often increasing carbon stocks in one pool and decreasing them in another.
- There are few data relating the individual or population-level effects to community-level carbon stocks.
- What data there are suggest that the direction of the effect of herbivores on carbon could be negative, positive, or neutral.
- Effects are highly context-dependent, with the largest positive effects of herbivore control likely in areas of highly palatable early-successional vegetation and high herbivore densities.

Design of herbivore control projects for carbon gain

There are a number of key design criteria that should be taken into account when setting up a carbon-oriented herbivore control project, to ensure the best possible chance of a successful outcome. In this section, we discuss criteria relating to selection of an appropriate project location, the extent of the monitoring regime required to quantify the expected changes in carbon, and factors to consider when implementing the management activity (including the associated carbon emissions). We use power analyses to determine the number of plots required to (a) estimate carbon stock to a certain level of precision and (b) detect a range of expected changes in carbon stock over time. We also use a power analysis based on simulated plot data to explore the effects of measurement interval and measurement error on the ability to detect changes in carbon. Many of the design criteria discussed here are general and can be applied to any type of carbon sequestration project in forests.

Selection of project location

The ecological context provided by the location of a project determines the baseline rate of carbon gain within the project area, the ecological mechanisms that are likely to operate following herbivore control, and the magnitude of the potential effect sizes (i.e. carbon gains) that are likely following the management intervention (Coomes et al. 2003; Mason et al. 2010; Tanentzap & Coomes 2012). Potential effect sizes are influenced by a number of factors including the successional stage of the community (e.g. grassland, shrubland, regenerating forest, or mature forest), the natural and anthropogenic disturbance history of the site, current and historical animal densities (Forsyth et al. 2011), the presence of other factors limiting plant growth, and the ability to cost-effectively reduce animal densities to low levels. The selection of an appropriate project location is therefore perhaps the most critical step in designing successful herbivore control projects for carbon gain.

The biggest effect sizes following herbivore control are likely in areas of young (regenerating) forest or shrubland with good seed sources and high animal numbers – especially those where succession is currently halted due to the presence of the target herbivore(s). In best-case situations where animal control initiates rapid forest development, effect sizes could be in the range of 1–2 Mg C ha⁻¹ year⁻¹ (Carswell et al. 2009). In contrast, despite sequestering significant amounts of carbon, many tall regenerating forests, or those comprised primarily of unpalatable species (e.g. kānuka forest or young beech–podocarp forest), are unlikely to show a large herbivore effect. This is because any carbon sequestration that occurs in these forests will be primarily due to growth of existing canopy trees or unpalatable tree species, and therefore not likely to be strongly influenced by the presence or absence of herbivores.

An exception would be when browsing of seedlings prevented the establishment of high-biomass tree species that would alter the trajectory of a succession, for example the establishment of broadleaved–podocarp tree species under senescing kānuka canopies (Payton et al. 1984; Smale et al. 1995).

Although mature forests occupy a large portion of public conservation land, the potential for carbon gains following herbivore control in these areas is limited. This is because mature forests have relatively low net gains in carbon over time (Luyssaert et al. 2008), are often at or near carbon exchange steady-state (Jarvis et al. 1989), and only a small portion of the carbon stock is affected by animal activity. Furthermore, most canopy-tree biomass in New Zealand consists of species that are generally unpalatable to invasive herbivores (Forsyth et al. 2002; Coomes et al. 2003). Any change within these areas due to animal control is therefore likely to be small (e.g. < 0.1 Mg C ha⁻¹ year⁻¹), to occur over long timescales through complex mechanisms, and require large investment in plots to quantify such small changes. Mature forests are therefore not ideal locations for establishment of herbivore control projects.

Other factors, such as temperature, rainfall, and seed availability, are also important to consider when selecting a project location. In general, the productivity of the ecosystem (and therefore the potential rate of ecosystem recovery following herbivore control) is lower at cold temperatures (i.e. high elevations), low rainfall, and in the absence of seed sources of high-biomass species. To avoid these limitations it has been recommended that project areas should be at <1000 m altitude and have >1000 mm of rainfall and adequate existing seed sources (Carswell et al. 2003). Other abiotic factors, including poor drainage, ultramafic geology and areas of cold-air drainage in valley floors, constrain carbon sequestration by affecting establishment and growth rate of trees. The practical and economic limitations of small effect sizes are also likely to play an important part in selection of project location, as is the current policy context (see ‘Demonstrating additional carbon gains’).

Design of monitoring network

A monitoring network should be established within the project area to measure in a verifiable and defensible way any carbon changes that occur. Plot-based methods of measuring actual carbon stocks are well developed (Coomes et al. 2002; Davis et al. 2004; Payton et al. 2004) and, ideally, the same monitoring framework could also be used for biodiversity monitoring (Allen et al. 2003). Representative sampling strategies, such as grids or randomly located transects, should be employed to ensure unbiased sampling of the entire project area (Clark & Clark 2000). Randomly located plots are preferred from a statistical perspective (Greig-Smith 1983); however, grid-based methods (with the grid size dependent on the area of the project and the total number of plots required) are commonly employed to monitor carbon stocks as they provide reliable geographic coverage and are easily scalable from plot to whole-project scales (Coomes et al. 2002; Kurz & Apps 2006). Stratification of the project area into vegetation types can be employed to help reduce sample variability, although this is likely to increase monitoring costs.

The number of plots required depends upon the variability in carbon stocks within the study area (or more importantly variability in their rates of change) and the likely effect size. We used natural forest data from 1239 plots sampled by New Zealand’s Land Use and Carbon Analysis System

(LUCAS) to illustrate the sample size required to estimate carbon stocks to within a certain level of accuracy. The LUCAS plots are 0.04-ha square plots located on an 8-km grid overlaid across mainland New Zealand (Coomes et al. 2002; Wisser et al. 2011). Plots were established between 2002 and 2007 on all grid intersections mapped as forest or shrubland according to New Zealand's Land Cover Database (LCDB1), using the methods described in Payton et al. (2004). For our analysis, plots were split into forest types, using the classification system developed by Wisser et al. (2011). Carbon stocks for each plot were estimated using the methods outlined in Coomes et al. (2002) for live-stem and coarse-woody-debris (CWD) pools only. We then calculated the sample sizes required to have 95% confidence that the mean stock estimate is within 5%, 10% and 20% of the true mean (Gelman & Hill 2007). The results (Table 1) show that for most community types, 20% accuracy is likely to be the best that could be achieved with a realistic number of plots (i.e. <100). These results do not incorporate potentially important sources of uncertainty associated with the allometric equations used to calculate tree biomass, and the landscape-scale representation of the sampling network (Chave et al. 2004; Hall & Case 2008). Including these sources of uncertainty would increase the number of plots required to estimate carbon stocks to a certain level of precision.

In terms of carbon sequestration, however, it is the rate of change in carbon stocks over time that is most important. This can be quantified with greater accuracy through repeat measurements on permanent plots. We conducted a separate power analysis to assess the number of plots required to detect a range of changes in sequestration rates (Table 2). Since many of the parameters required for the power analysis are currently unknown, we ran simulations using a range of potential parameter values (see Appendix 1 for detailed methodology). The absolute effect sizes used (0.05–2.0 Mg C ha⁻¹year⁻¹) reflect the most likely range of potential values (Burrows et al. 2008; Beets et al. 2009; Carswell et al. 2009), and translate into 0.18–7.33 Mg CO₂e ha⁻¹ year⁻¹. Standard deviations of 0.5–4.0 Mg C ha⁻¹year⁻¹ were used for each potential effect size, encompassing the full range of published data (Coomes et al. 2002; Beets et al. 2009; Carswell et al. 2009). Our results show that to have a reasonable (80%) chance of detecting a < 0.5 Mg C ha⁻¹ year⁻¹ change in carbon sequestration rate, impractically large numbers of plots are required (Table 2). Even for larger effect sizes, up to 100 plots are required, depending on the level of variability in the data.

At small project scales the monitoring costs associated with demonstrating a change in carbon are likely to be considerably larger than the potential revenue from carbon

Table 1. Estimated number of monitoring plots (*N*) needed to obtain a given level of precision (percentage of the mean) in carbon stock estimates for different forest types. Forest types are based on Wisser et al. (2011). Indicative mean carbon stocks and standard deviations are calculated from the LUCAS natural forest dataset.

Vegetation type	Carbon stock (Mg C ha ⁻¹)			Sample size (<i>N</i>) required for varying levels of accuracy			
	<i>N</i> Plots	Mean	SD	<i>N</i> (5%)	<i>N</i> (10%)	<i>N</i> (20%)	<i>N</i> (30%)
Shrubland							
Mānuka shrubland	31	42	32	923	231	58	26
<i>Schoenus-Dracophyllum</i> subalpine shrubland	19	40	62	3666	917	229	102
<i>Ozothamnus-Dracophyllum</i> montane shrubland	31	20	19	1330	333	83	37
Matagouri shrubland	28	12	18	3493	873	218	97
Sweet vernal – Yorkshire fog successional shrubland	61	33	35	1699	425	106	47
Whēki-mānuka shrubland/low forest	26	49	38	914	228	57	25
Broadleaved-podocarp forest							
Kāmahi-podocarp forest	105	236	138	525	131	33	15
Māhoe forest	52	182	121	674	168	42	19
Silver fern – māhoe forest	79	149	94	612	153	38	17
Tawa forest	69	235	139	540	135	34	15
Broadleaved forest							
Kānuka forest and tall shrubland	51	66	49	857	214	54	24
Beech-broadleaved-podocarp forest							
Kāmahi – Southern rātā forest and tall shrubland	49	166	90	451	113	28	13
Kāmahi forest	48	287	149	417	104	26	12
Pepperwood-hardwood forest and successional shrubland	58	242	124	402	100	25	11
Kāmahi – silver fern forest	46	199	97	371	93	23	10
Beech-broadleaved forest							
Silver beech – broadleaved forest	64	237	119	391	98	24	11
Silver beech – red beech – kāmahi forest	94	281	93	168	42	10	5
Marbleleaf-pepperwood-wineberry forest and successional shrubland	22	184	125	709	177	44	20
Kāmahi-hardwood forest	80	270	157	521	130	33	14
Beech forest							
Black/mountain beech forest	21	150	60	245	61	15	7
Silver beech – red beech – black/mountain beech forest	19	275	106	229	57	14	6
Black/mountain beech forest (subalpine)	20	166	40	87	22	5	2
<i>Hoheria glabrata</i> – <i>Olearia ilicifolia</i> – hard fern low forest and successional subalpine shrubland	25	89	76	1110	277	69	31
Black/mountain beech – silver beech forest/subalpine shrubland	63	186	85	322	80	20	9
All forest and shrubland (including unclassified plots)	1239	172	137	981	245	61	27

Table 2. Number of monitoring plots required to have a 80% chance of detecting a significant effect (at 95% confidence) for a range of potential effect sizes and levels of data variability (SD = standard deviation of the observed rates of change), assuming variability in rates of change is entirely due to process error (i.e. measurement error assumed to be zero), and is independent of measurement interval, initial carbon stock, and expected effect size.

SD (Mg C ha ⁻¹ year ⁻¹)	Effect size (Mg C ha ⁻¹ year ⁻¹)					
	0.05	0.1	0.2	0.5	1	2
0.5	787	199	52	10	5	3
1	3142	787	199	34	10	5
1.5	7066	1768	444	73	20	7
2	12561	3142	787	128	34	10
2.5	19625	4908	1229	199	52	15
3	28258	7066	1768	285	73	20
3.5	38462	9617	2406	387	99	27
4	50235	12561	3142	505	128	34

gains. For example, establishing and remeasuring 100 plots in a 250-ha study area is likely to cost at least \$4,000 per plot, or \$1,600 per hectare, compared with the potential carbon revenue ranging from \$4 to \$916 per hectare (based on a 5-year measurement interval and the range of effect sizes given in Table 2, assuming \$25 per tonne CO₂e).

Our power analysis in Table 2 assumes that variability in rates of change is entirely due to process error (Bolker 2008), and is independent of measurement interval, initial carbon stock, and expected effect size. These assumptions are not necessarily true, as younger, actively regenerating stands are more likely to consistently increase in carbon (and thus have a lower variability) compared with larger-biomass stands, which may have greater potential for carbon loss due to major disturbance events. Also the chance of major disturbance increases as measurement interval increases, and this is likely to increase variability in rates of change at longer measurement intervals. Further data on rates of change and their variability across a range of environments are required in order to estimate more precisely the number of monitoring plots required.

Measurement interval

The appropriate measurement interval for carbon monitoring plots is dependent on the magnitude of the likely effect size and the sources of uncertainty present in the data. The larger the hypothesised effect, the more cost effective it will be to employ short measurement intervals. The ability to detect a true effect (i.e. power), however, will be more affected by measurement error at shorter measurement intervals. We used a simulated power analysis to explore the effect of (unbiased) measurement error, variability in actual sequestration rates across the landscape (process variability), and uncertainty associated with the estimation of the business as usual (BAU) rate of change (described in ‘Demonstrating additional carbon gains’ section), on the probability of detecting a true effect at 1-, 2-, 5-, or 10-year measurement intervals (Fig. 1). Simulations were based on an effect size 0.5 Mg C ha⁻¹ year⁻¹, an effect standard deviation of 2 Mg C ha⁻¹ year⁻¹, a normally distributed measurement error with a mean of zero and a standard deviation of 2% of the actual carbon stock (estimated from Dickie et al. (2009)), and a baseline rate with a mean of zero and standard deviation of 2 Mg C ha⁻¹ year⁻¹ (for more details see Appendix 1). Measurement error occurs

at time of measurement and at the time of remeasurement, and is independent of the measurement interval (Chave et al. 2004; Hall & Case 2008). In contrast, the effect size and associated process variability both increase with time since last measurement. This means that the reduction in power caused by measurement error decreases with increasing measurement interval, as shown in Fig. 1. For our simulations, at time intervals shorter than 5 years the increased variability due to measurement error results in large reductions in the statistical power, significantly reducing the ability to detect changes in carbon sequestration rates (Fig. 1). Experience has shown that a measurement interval of at least 5 years is appropriate, and this is supported by our results. Further data on measurement errors, effect sizes, and other sources of uncertainty are required to assess the generality of the recommended minimum 5-year measurement interval.

Implementation of the management activity

There are a number of excellent sources of information regarding the practical implementation of effective herbivore control strategies (e.g. Parkes & Murphy 2003; Clayton & Cowan 2010 and references therein), so we will not go into detail regarding these here. It is important to note, however, that financial cost of undertaking the control activities (including the associated carbon emissions) should be balanced against the hypothesised carbon gains. The control activity must also be cost-effective in reducing herbivore numbers below a certain threshold such that their effects are reduced sufficiently to allow a positive response in the community of interest (Nugent et al. 1997; Coomes et al. 2003). It will also be necessary to maintain herbivores at low densities for a number of years to allow the system time to recover (Coomes et al. 2003). Exactly how low herbivore numbers need to be to result in carbon gains is unknown, but the threshold density is likely to be context-specific, depending on the local species composition, current animal densities relative to the carrying capacity of the site, and history of control in the area (Forsyth et al. 2011). For example, areas of historically high deer densities could have undergone significant changes in vegetation composition, leading to altered successional pathways that may now limit the capacity of that area to recover following animal control (Coomes et al. 2003; Forsyth et al. 2011). The scale of the herbivore control also needs to be sufficient to prevent reinvasion from outside the control area (Forsyth et al. 2010b),

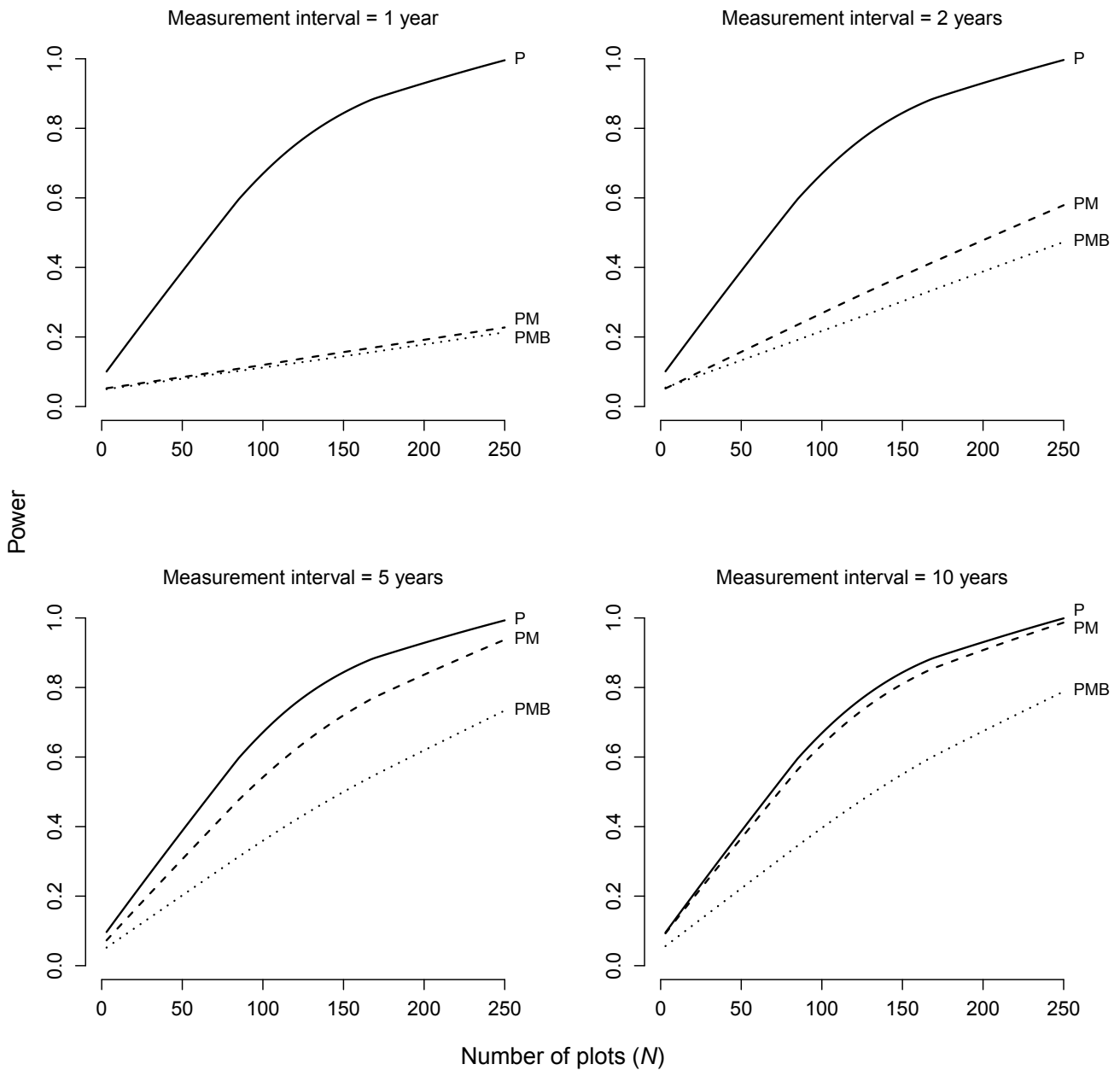


Figure 1. Relationship between sample size (N) and the probability of detecting a true effect (power) for three different error scenarios, based on measurement intervals of 1, 2, 5, or 10 years. Error scenarios are within-site process error only (P, solid line), within-site process error plus measurement error (PM, dashed line), and within-site process error, plus measurement error, plus process error in the baseline rate estimates (PMB, dotted line). The reduction in power due to measurement error decreases with increasing measurement interval. Power estimates are based on an effect size of $0.5 \text{ Mg C ha}^{-1} \text{ year}^{-1}$, an effect standard deviation of $2 \text{ Mg C ha}^{-1} \text{ year}^{-1}$, a normally distributed measurement error with a mean of zero and a standard deviation of 2% of the actual carbon stock, and a baseline rate with a mean of zero and standard deviation of $2 \text{ Mg C ha}^{-1} \text{ year}^{-1}$.

and should encompass all the major herbivores present within the study area (Forsyth et al. 2000).

In many situations, an integrated approach to managing for carbon gain is likely to lead to greater carbon gains than herbivore control alone. This is especially the case when multiple barriers to forest establishment exist. For example, in highly modified seed-limited areas it may be important to supplement the herbivore control with seed addition, hotspot

planting of key tree species, and targeted weed control in order to achieve sufficient establishment of high-biomass tree species (Manning et al. 2006; Ledgard et al. 2008). In these situations, the effect of integrated management is likely to be non-additive in that gains due to implementing multiple management activities at one location are likely to be much greater than the gains achieved by implementing each management activity independently.

Summary of key design criteria

- Projects should be located in areas that have the potential to show a significant carbon response to wild animal control. Best locations are likely to be areas of young regenerating forest/shrubland with good seed sources and high animal numbers – preferably to the point where succession is currently inhibited due to the presence of wild animal herbivores.
- Monitoring should be established to measure in a verifiable and defensible way any carbon changes that occur within the project area.
- The number of plots required to detect an effect is strongly dependent on the likely effect size, and the estimated variability among plots. In practice, effect sizes of $>0.5 \text{ Mg C ha}^{-1} \text{ year}^{-1}$ can be detected with a reasonable number of plots (i.e. <100). Smaller effect sizes are likely to be undetectable.
- Measurement error can severely reduce the ability to detect changes over time, especially for annual measurement intervals. A minimum 5-year measurement interval is recommended to avoid these effects.
- The financial costs of implementing wild animal control (including the associated carbon emissions) should be considered in relation to the potential carbon gains.
- Integrated carbon management techniques may be required to remove multiple barriers to regeneration where these exist.

Demonstrating additional carbon gains

In order to fulfil the terms of the Kyoto Protocol, carbon offset projects have to demonstrate additionality, i.e. that the carbon gains observed were due to the management intervention and would not have occurred without it. The practicalities involved in demonstrating additional carbon gains based on data from within-site monitoring are heavily dependent upon

the current policy context. Within both the Kyoto Protocol and the New Zealand Emissions Trading Scheme (ETS) there is a clear distinction made between forests that already existed at 1990 ('pre-1990 forest'), forests that have established after that date (post-1989 forest), and non-forest land. This split involves differential treatment of baseline or business as usual (BAU) rates of change (Fig. 2). In post-1989 forest, BAU rates of change are assumed to be zero (or near zero), and thus all carbon gain that occurs in such forests can be attributable to the primary management intervention – land-use change. For pre-1990 forest, however, a non-zero BAU rate of change must be used. This is because it is possible that a significant fraction of the observed carbon accumulated since 1990 would have occurred anyway without the need for management intervention (Fig. 2b). Currently, carbon accumulated through management of pre-1990 forests is not recognised within the New Zealand ETS, primarily because New Zealand elected not to sign up to Article 3.4 of the Kyoto Protocol. However, these carbon gains, as well as those within non-forest vegetation (e.g. tussock grassland), are tradable on the voluntary market and could become part of an ETS in the future depending on the nature of any post-Kyoto agreement.

Post-1989 forest

The calculation of net carbon removals/emissions from post-1989 forest requires estimating carbon stocks for each plot for each time period, using standard allometric functions based on tree height, stem diameter, and wood density (e.g. Coomes et al. 2002) and then subtracting the initial stocks from the current stocks for each plot. This provides an estimate of the amount of carbon gained/lost at each plot. Using data from every plot surveyed, a simple paired *t*-test can be done to see if the carbon gained/lost is significantly different from zero (at $\alpha = 0.05$), and if so the mean difference, multiplied by the area of the pre-1989 forest, equals the amount of carbon that has accumulated (or been lost) since the change in management.

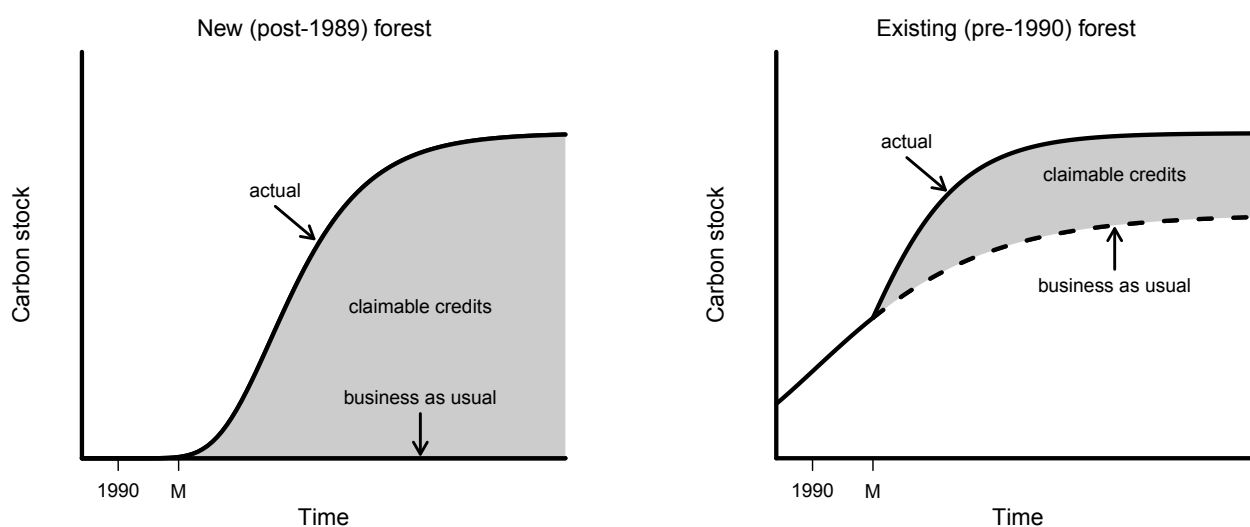


Figure 2. Business as usual (BAU) concept illustrated using hypothetical carbon stock trajectories following management intervention for new (post-1989) Kyoto-compliant forest and existing (pre-1990) forest. The BAU trajectory is zero for new (post-1989) forest, but non-zero for existing (pre-1990) forest. The point of management intervention (M) occurs sometime after 1990, and in both cases the claimable credits are equivalent to the difference between actual and BAU trajectories following this point in time.

Pre-1990 forest

Calculating gross carbon gain for pre-1990 forest follows the same steps as for post-1989 forest; however, the BAU rates of change are likely to be non-zero and therefore must also be estimated in order to fulfil the terms of the Kyoto Protocol. This can be done in a number of ways, all of which are subject to limitations and are relatively difficult to achieve in practice. The first and most cost effective technique is to use other data sources to generate an approximate BAU rate of change for your study area. This could be achieved using data stored in the National Vegetation Survey Databank (NVS, curated by Landcare Research), or using data from other carbon monitoring plots (e.g. LUCAS data held by the Ministry for the Environment). The drawbacks of these are that the existing data are likely to be a poor match to the study site, thus prone to over- or under-estimating the actual BAU carbon sequestration rates. These inaccuracies could have a very significant impact on the ability to detect changes in carbon within the study area, especially when effect sizes are likely to be small (e.g. $<0.5 \text{ Mg C ha}^{-1} \text{ year}^{-1}$).

A more accurate representation of BAU rates of change could be achieved through establishment of an experimental control site in an area of similar vegetation, climate, topography and animal densities to the study area. Monitoring plots would need to be established in this area, at equivalent densities to the main treatment area, which would effectively double the monitoring costs. In addition to the extra cost, experience has shown that it is difficult to get a perfect large-scale control site due to the many uncontrollable variables that can influence stand dynamics (Underwood 1992; Murtaugh 2000). The use of only a single pair of experimental control and treatment sites also results in pseudoreplication, which should be avoided (Hurlbert 1984); multiple sites are therefore recommended to ensure a more powerful design (e.g. Jacobson et al. 2009). BAU rates of change could also be estimated using plots located within the project area providing these were established and remeasured prior to the management intervention occurring. However, the accuracy of these BAU rate estimates would be strongly influenced by uncontrollable temporal dynamics such as disturbance events (Murtaugh 2000; Mason et al. 2010).

The third possible way to estimate BAU rates of change is to use a model of forest development to predict carbon changes with and without browsing animals (Allen & Carswell 2008). Such a model could be developed nationally, and parameterised for specific projects using a combination of national-level data (e.g. from NVS) and easily collectable data specific to the project area (e.g. species composition, seed availability, climate). Existing models of forest dynamics such as SORTIE/NZ (Kunstler et al. 2009) and PPA (Purves et al. 2008) have the ability to predict changes in biomass (and thus carbon) over time but require further modification and parameterisation to predict carbon sequestration accurately under different management regimes in early-successional environments. Despite the considerable investment involved in its development, a national-level early-successional carbon change model would be a powerful future management tool as it could potentially be used to estimate both BAU and actual rates of change following management intervention.

Summary of additional requirements

- All carbon offset projects have to demonstrate additionality, i.e. that the carbon gains observed were due to the management intervention and would not have occurred without it.

- For post-1989 forest, a zero BAU rate is assumed. All gains in carbon are therefore directly attributable to the management intervention.
- For pre-1990 forest, non-zero BAU rates of change need to be estimated. Various methods are capable of doing this, all of which require further trialling and research.

Concluding remarks

The potential for invasive herbivore control to result in measureable carbon gains depends on the ecological context of the project area. In general, carbon gains are likely to be small, occur through complex indirect mechanisms, and difficult to quantify in practice. However, significant carbon gains may occur in certain situations such as when herbivore control promotes the establishment of high-biomass woody species. Primary data on carbon changes following wild animal control are scarce, and further research is required to quantify potential gains (and their variability across the landscape), and to properly understand the underlying ecological mechanisms. Projects involving herbivore control for carbon gain therefore contain a significant element of risk. To ensure the best chance of demonstrating significant carbon gains, projects should be carefully designed to ensure that appropriate site-selection and monitoring takes place, and that the potential carbon gains outweigh the costs involved in implementing the project. While carbon gains from herbivore control are unlikely to provide a silver bullet for conservation funding at a national scale, there is potential in certain areas of New Zealand's indigenous vegetation for both conservation benefits and carbon gains to go hand in hand.

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Appendix 1. Description of methods used for power analyses

We conducted a power analysis to assess the number of plots required to detect a range of changes in sequestration rates (Table 2) using the *power.t.test* function in R (version 2.11.1; R Foundation for Statistical Computing: www.R-project.org). Mean effect sizes (representative of differences between observed and expected sequestration rates) of 0.05, 0.1, 0.2, 0.5, 1 and 2 Mg C ha⁻¹ year⁻¹ and effect standard deviations of 0.5, 1, 1.5, 2, 2.5, 3, 3.5, and 4 Mg C ha⁻¹ year⁻¹ were used to reflect the full range of published values (Coomes et al. 2002; Burrows et al. 2008; Beets et al. 2009; Carswell et al. 2009). For every combination of mean effect size and associated variability (standard deviation), a one-sided two-sample *t*-test was used to determine the number of plots required to have a good chance of detecting the true effect (power = 0.8, α = 0.05). For simplicity this analysis does not explicitly separate measurement error from process error, and assumes that the mean sequestration rate and its variance are independent of current carbon stock and are constant over time.

To explore the effects of measurement error (Fig. 1), we simulated a sample of N plots, with initial observed carbon stock (C_{obs, t_1}) generated from a normal distribution with a mean of 217.5 Mg C ha⁻¹ and a standard deviation of 21.5 Mg C ha⁻¹ (Marburg et al. 2010). Plot-level measurement error was simulated with a mean of zero and a standard deviation of $\sigma_{\text{measurement}}$, and this was added to the observed carbon stock (C_{obs, t_1}) to give an approximation of the true initial carbon stock at each plot (C_{true, t_1}). An annual treatment effect was simulated (mean = μ_{effect} , standard deviation = σ_{effect}), multiplied by the yearly measurement interval (y), and added to the true initial carbon stock estimate to give the true carbon stock at time 2 (C_{true, t_2}). A second measurement error was simulated (mean = 0, SD = $\sigma_{\text{measurement}}$) and added to C_{true, t_2} to give observed carbon stock at time 2 (C_{obs, t_2}). The ability to detect a significant increase in carbon stock (effect size = μ_{effect}) was tested using a paired *t*-test to see if the difference between C_{obs, t_2} and C_{obs, t_1} was different from the BAU estimate (mean = 0, SD = σ_{BAUerror}). We ran the simulation 10 000 times and calculated the power of the design as the proportion of simulations where $P < 0.05$. Three scenarios were simulated, all based on an effect size (μ_{effect}) of 0.5 Mg C ha⁻¹ year⁻¹, effect standard deviation (process error) of 2 Mg C ha⁻¹ year⁻¹, and measurement intervals of 1, 2, 5, and 10 years. For the first simulation (P) we set $\sigma_{\text{measurement}}$ and σ_{BAUerror} to zero to represent only process error. The second simulation (PM) included an unbiased measurement error, with $\sigma_{\text{measurement}}$ equivalent to 2% of the actual carbon stocks (Dickie et al. 2009). The third scenario (PMB) included error in the BAU estimate, with σ_{BAUerror} set to 2 Mg C ha⁻¹ year⁻¹.