Report Cover Page

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Evaluating vegetation condition measures for cost effective biodiversity investment planning
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Summary

This report summarises the three papers that are the main outputs of this project.

These papers examine the uncertainty associated with measures of vegetation condition on patches of remnant vegetation and on sites managed as offsets for clearing or other vegetation modification. In particular the papers focus on the BioMetric and Habitat Hectares measures of site quality, developed for use in New South Wales and Victoria respectively, and on the Cumberland Plain Woodland vegetation type. The papers find that uncertainty regarding current and expected future vegetation quality is a significant issue for investment planning. Inaccurate measures can lead to declines in biodiversity value and reduce the cost effectiveness of investments in vegetation restoration. Improving the accuracy of the measures and making planning processes more robust to failures of restoration actions to enhance biodiversity outcomes are important implications. There appears to be less scope for improving the precision of the measures and this is likely to be costly. A useful extension of the work would be to undertake a spatially explicit landscape scale analysis although this would require landscape scale measures of vegetation quality. The software developed for specifying the uncertainty and calculating robust offset ratios are additional project outputs.

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Evaluating vegetation condition measures for cost effective biodiversity investment planning; ACERA Project No. 0706

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Final project report

Summarising research by Emma Gorrod, David Keith, Atte Moilanen, Andrew Robinson, Mark Burgman, Astrid van Teeffelen, Yakov Ben-haim, Simon Ferrier and Kenton Lawson.

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As this report summarises three project reports the reader is directed to those papers (attached) for full references and acknowledgements.

Disclaimer

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1. Executive Summary

Quantitative measures of biodiversity value are a critical input to biodiversity investment planning. Such measures allow one to compare different investments and find which offer the best value for money and to determine which offsetting actions are required to compensate for development actions that reduce biodiversity value.

Vegetation condition measures are increasingly used as a surrogate measure of biodiversity value. Any such measures will be limited in how well they reflect biodiversity values and this project has examined some of the issues associated with uncertainty in these measures.

Errors in the measurement of vegetation condition may lead to losses associated with particular trades, and any biases may lead to landscape-wide declines or shifts in biodiversity values. The first project report summarised here explores this issue of observer error in the measurement of vegetation condition. It finds there is significant and probably largely unavoidable measurement error and thus planning processes need to be robust to this uncertainty.

The second report discussed is a theoretical exploration of the implications of uncertainty in vegetation condition measures for vegetation offsets. The paper develops a robust offset ratio and examines how this is affected by the risk of restoration failure and correlation of this failure across restoration sites. The paper introduces the concept of discounting across time to accommodate the fact that restoration actions take many decades to generate biodiversity values whilst actions such as clearing cause an immediate loss. In the examples given these factors can lead to very large offset ratios. This work highlights the possibility that restoration actions may fail to produce improved biodiversity outcomes because plantings may fail to grow or because they do not complement other vegetation in the landscape. Vegetation planning processes need to reduce or account for these risks.

The third project report undertakes a simulation analysis of uncertainty in vegetation condition measures across a range of representative sites from a particular woodland ecosystem and over a 150-year time period. The most cost-effective actions are likely to involve restoration and development of the more degraded sites. Losses are greatest and hardest to offset on high quality sites. Fencing and excluding stock from one type of site, the *high herbaceous* site, appears to be a particularly attractive restoration action when these sites are available. For the other types of sites more active restoration actions are required. The study also finds that it takes considerable time to recover biodiversity values and that offset ratios calculated in the early decades tend to be very large.

This project also supports the finding that relying on a single index for vegetation quality in restoration and offsetting activities may lead to a shift towards those vegetation attributes that are more readily restored. This may indicate that the vegetation condition measure no longer accurately represents the biodiversity value of restoration actions. Ongoing monitoring of restoration sites could provide the stimulus to revise the measure in such cases. Alternatively the measure could be modified to account for the distribution of the different vegetation attributes across a landscape.

This study has focused on measures of vegetation condition at the site level and has largely ignored the landscape context. A useful extension of this work would be to undertake an explicitly spatial analysis that also adds the costs of the different actions in order to better identify cost-effective biodiversity investments at a landscape scale.

2. Introduction

In order to determine whether biodiversity investments are cost-effective it is necessary to be able to quantify the change in biodiversity value (BV) under changed vegetation management. Cost-effective investments must be effective in improving BV and must be more effective per unit cost than alternate investments. So it is critical that the measures of BV used are reliable and robust to the uncertainties of ecosystem and management processes and responses.

This paper presents a summary of the three main papers produced under ACERA project number 0706: *Evaluating vegetation condition measures for cost effective biodiversity investment planning*. These papers are attached at the end of this document. The reader is directed to these papers for the full analyses.

This project has focussed on the uncertainty involved with two vegetation condition measures currently in use. These are the BioMetric (Gibbons et al. 2005; Gibbons et al. 2009) site index that is used in New South Wales' property vegetation planning process and the Habitat Hectares (Parkes et al. 2003; DSE 2004) site index that is in use in Victoria.

The project has also focussed on a particular vegetation type that occurs near Sydney in Australia: the Cumberland Plain Woodland (CPW). Woodlands are one of the more modified vegetation types in Australia and especially near Sydney. These BV indexes require field surveys that estimate a range of vegetation attributes at the survey site. The attributes compare the current state of the vegetation with benchmark data for the same vegetation type. The attribute measures are then added up using pre-specified weights to give the BioMetric or Habitat Hectares site index.

Similar indexes are in use or development in other states and are documented in ESCAVI (2007). The need to undertake field surveys makes these indexes, in their current form, impractical for regional planning processes. However current efforts to create regional vegetation quality data sets will necessarily extrapolate from these survey based measures.

The question of how well these indexes relate to BV was not examined in the project. BV is often represented as an aggregation of the viability of all the species that constitute an ecosystem. Vegetation indexes are a surrogate measure that may work well for some species groups and less well for others. Factors such as the presence of feral predators, water availability, rocky habitats or fire patterns are often important in determining a population's viability. This is an important research question as identifying cost-effective biodiversity investments depends critically on the particular metric used – but it is not analysed further here. In practice some accounting is also carried out relating to the spatial configuration of the vegetation in the landscape such as with the BioMetric landscape index which complements the site index that is examined here. Also biodiversity investments may involve a number of different vegetation types and these are usually weighted according to the current conservation status of each vegetation type. The BioMetric regional index performs this function. However, in the case of offsets for clearing, the offset areas are generally required to be of the same vegetation type.

The first paper discussed below examines the uncertainty inherent in the field assessment and calculation of the indexes. The second paper is a more theoretical study of the robustness of BV measures for a range of sources of uncertainty. The third paper simulates the uncertainty in the metrics for a range of CPW sites under a range of management scenarios.

After the initial project planning meetings the scope of the original project objectives was reduced from examining the robustness of the vegetation metrics for vegetation planning at a landscape scale to examining the robustness of the metrics for vegetation management decisions at representative sites. This also limited our ability to explicitly examine tradeoffs between biodiversity and commercial objectives in vegetation planning for the CPW.

3. Methodology

3.1 Measurement error

The first paper, Gorrod and Keith (2008), examines observer variation in estimating the BioMetric and Habitat Hectares site indexes. Table 1 (all figures and tables are reproduced from the relevant report) lists the vegetation attributes that must be estimated in order to calculate the indexes.

Table 1. BioMetric and Habitat Hectares component attributes, weightings and Cumberland Plain Woodland benchmarks.

BioMetric				Habitat Hectares		
Attribute		**Benchmark**	**Weight**	**Pairs of attributes**	**Benchmark**	**Weight**
H	Number of hollow bearing trees	≥1 tree	30	Number of large trees per hectare	>15 trees with dbh ≥50cm	10
				Large tree canopy health	>70%	
B	Native over-storey cover	19-24%	5	Tree canopy cover	7-22%	5
				Tree canopy health	>70%	
A	Native plant species richness	>29	20	Diversity & cover of understorey lifeforms	≥9 lifeforms present	25
C	Native mid-storey cover	20-30%	10			
D	Native ground cover (grasses)	23-31%	5	Proportion of understorey lifeforms that are substantially modified	0 lifeforms modified	
E	Native ground cover (shrubs)	0-5%	5			
F	Native ground cover (other)	12-20%	5			
g	Cover of weeds	0-5%	5	Cover of weeds	0-5%	15
				Proportion of weed species that are considered high threat	0	
i	Proportion of overstorey species regenerating	100%	10	Total number of woody species	≥5	10
				Proportion of woody species recruiting	>70%	
j	Total length of logs	≥5 m of logs ≥10cm diameter	5	Total length of logs	≥7.5 m of logs of ≥10cm diameter	5
				Proportion of logs that are large	≥2.5 m of logs of ≥25cm diameter	
				Litter cover	5-15%	5
				Dominance of native/exotic litter	Native litter	
	Total		**100**	**Total**		**75**

Ten observers each carried out these assessments at the same twenty sites that contain either remnant, restored or cleared examples of CPW. The distribution of the vegetation attributes and indexes across observers and sites was then compared.

3.2 Robust offset ratios

The second paper, Moilanen et al. (2007), develops a framework for accounting for a range of sources of uncertainty in the calculation of offset ratios. Offset ratios are usually specified as the BV per hectare that would be lost by a development action such as vegetation clearing divided by the predicted future BV per hectare at a proposed restoration site. This approach is referred to as the matching mean expected utility (MMU) approach.

The paper discusses the effect of a range of sources of uncertainty on the offset ratios. These include

- uncertainty in the level of BV achieved at restoration and development sites,
- an additional chance that restoration actions may fail and
- correlation between the success of restoration actions at different sites.

The first source of uncertainty means that in particular cases the net effect of the offset may be greater or less than the BV lost at the development site. This may not be problematic at a broader regional level if the measures of BV are not systematically biased. It may be a problem if there are particular components of biodiversity where losses may be irreversible. In this case one may adopt a more risk-averse approach. The paper develops a robust offset ratio to deal with this case. This is defined as the offset ratio required to compensate a BV loss with a given level of certainty. In this project a 95% probability was used.

The formulae used for calculating the MMU and robust offset ratios, R_{MMU} and R_{ROB} respectively, at time t_1 relative to a start time of t_o are

$$R_{MMU} = \frac{BV_l(t_0) - BV_l(t_1)}{BV_o(t_1) - BV_o(t_0)} \quad (1)$$

$$R_{ROB} : \Pr(R_{ROB} * (BV_o(t_1) - BV_o(t_0)) > (BV_l(t_0) - BV_l(t_1))) = 0.95 \quad (2)$$

where BV_o is the biodiversity value of the restoration or offset site and BV_l is the biodiversity value of the development or loss site. As the uncertainty is reduced R_{ROB} approaches R_{MMU} in value.

The paper also includes an approach to time discounting. This is important because of the long periods required for restoration sites to provide particular site attributes such as tree hollows. Such attributes are generally lost immediately with certainty following the development action. Biodiversity banking schemes may address this issue by requiring developers to purchase already restored areas (Berkessy and Wintle 2008). However, appropriate discounting of expected BV gains can in principle reduce this difference between the vegetation offsets and biobanking approaches.

The paper examines the effect of these factors on the robust offset ratio using the info-gap decision theory (Ben-Haim 2006) in a series of stylised examples. Here the time discounting approach discounts the time series of offset ratios. In practice a more realistic approach would first calculate the discounted sum of the time series of BV indexes on both sites with and without the development and restoration actions and then use these sums in the offset ratio calculation.

3.3 CPW simulations

The third paper, Gorrod et al (in prep), uses a simulation approach to predict the time path of the BV indexes on five hypothetical CPW sites under six hypothetical management scenarios. Table 2 lists the vegetation attributes at each hypothetical site at the start of the simulation period. The sites represent a range of ecological states ranging quite degraded, the *all low* site, to reasonably intact, the *all high* site. Intermediate sites vary in their woody and herbaceous attributes.

Table 2. Vegetation attributes of hypothetical sites at t_0

Vegetation attribute	Hypothetical site				
	All low	All moderate	High woody	High herbaceous	All high
Native plant species richness [1]	15	20	20	45	40
Number of lifeforms present (out of ten) [2]	4	6	4	9	10
Proportion of lifeforms modified [2]	1	0.33	0.75	0.11	0
Native overstorey cover (%) [1,2]	5	10	20	2	15

Native midstorey cover (%) [1]	5	10	25	1	15
Native ground cover (%) [1]	15	25	15	80	50
Exotic plant cover (%) [1,2]	55	15	35	2	2
Proportion of woody species regenerating[1,2]	0	0.5	0.2	0.4	1
Number of large trees per hectare[2]	2	8	18	0	14
Number of hollow bearing trees per 0.1 hectare[1]	0	0	1	0	1
Length of logs per 0.1 hectare[1,2]	0	10	25	1	25
Litter cover (%) [2]	1	4	12	5	10
BioMetric score (%)	**20**	**48**	**65**	**39**	**90**
Habitat Hectares score (scaled to %)	**24**	**68**	**53**	**65**	**97**

1: Component of the BioMetric protocol; 2: Component of the Habitat Hectares protocol

Table 3 lists the management scenarios. The three development or biodiversity loss scenarios range from total clearing to introducing grazing. The three restoration scenarios range stock exclusion to the mixed management scenario.

Table 3. Details of actions undertaken in different management scenarios

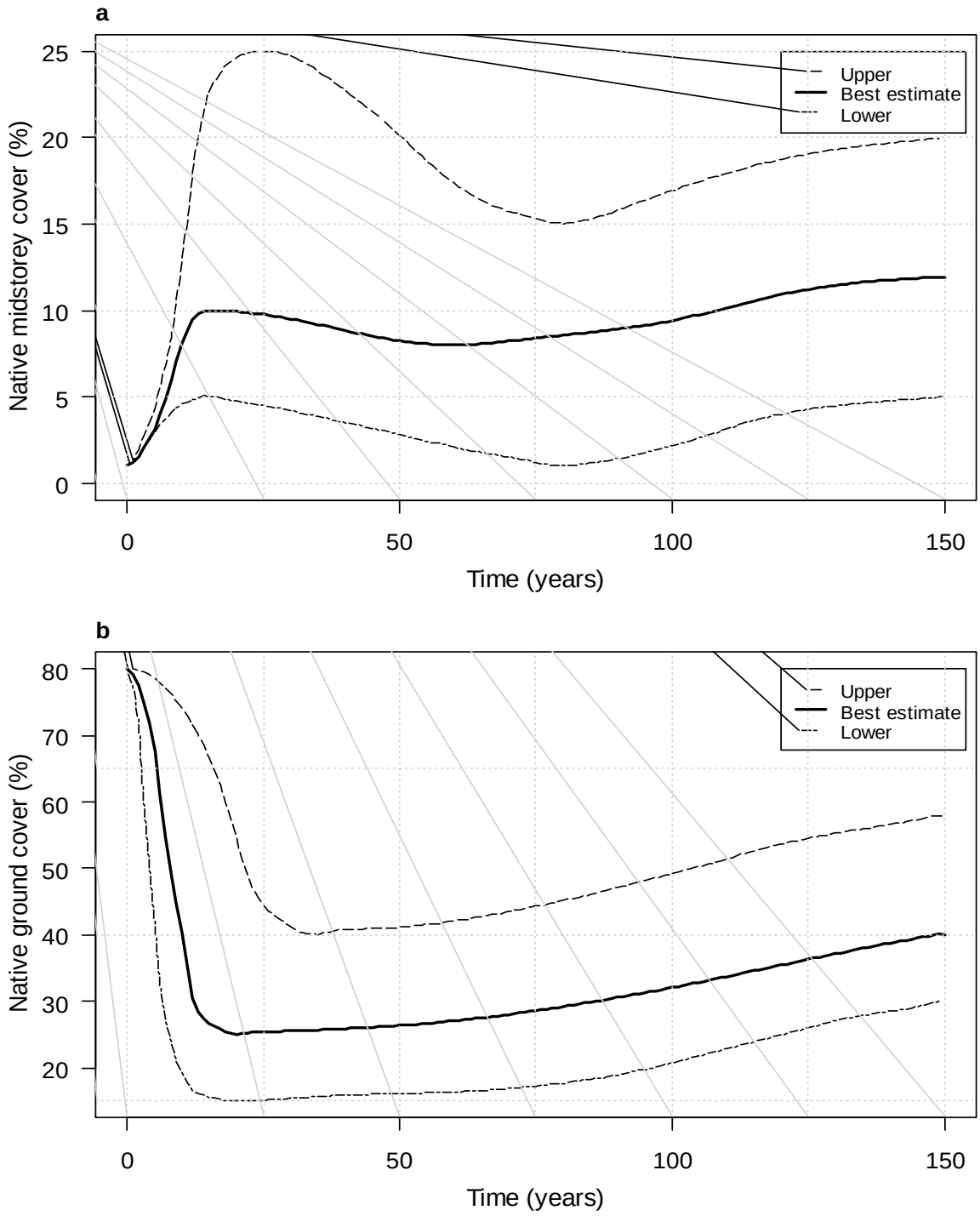
Management scenario		**Actions at t_1**	**Actions t_2-t_{150}**
L1	**Total clearing**	All vegetation permanently removed.	
L2	**Partial clearing**	The shrub layer and all logs are initially removed from the site, and the canopy trees are thinned by half.	Over time, the site is mown or slashed regularly (preventing recruitment) and any new logs are removed.
L3	**Grazing**	Moderate levels of grazing introduced (or continued if grazed prior to t_1) and maintained at levels that remove 50% of potential groundcover biomass and virtually all seedlings of palatable woody species.	
0	**No action**	No management actions undertaken (or grazers excluded if present prior to t_1).	
R 1	**Planting**	Four woody species are planted at moderate to high densities. Only conducted in sites with low canopy cover at t_1. Grazing excluded.	No further management actions undertaken.
R 2	**Mixed management**	Restoration actions including planting, weed control, grazer exclusion and fire management are undertaken as specified by best management advice.	

A best estimate and upper and lower bounds for the trajectory of each of the vegetation attributes on each site under each management scenario over the 150 year time frame was specified using expert judgement and informed by empirical data where available. Figure 1 illustrates the trajectories for midstorey and ground cover on the *high herbaceous* site under the planting restoration scenario, R1. The planting action increases the midstorey cover and reduces the ground cover as the woody plantings are established. These trajectories were constructed using tools developed in the open-source statistical environment R (R Development Core Team 2007) that allowed the user to plot unique trajectories by specifying at least five values in a graphic interface.

These trajectories were then used to construct 200 stochastic simulations of the change in the vegetation attributes and hence the BV indexes for each site and management scenario. These values were then used to calculate the offset ratios using both the MMU and robust offset ratio approaches at selected time periods.

The starting conditions of the sites are generally assumed to be known with certainty. However due to measurement error the actual values may differ. Robust offset ratios are also calculated allowing an observer error coefficient of variation of 25% based on Gorrod and Keith (2008).

Figure 1. Examples of models for change in vegetation attributes: a) native midstorey cover and b) native ground cover on the *high herbaceous* site under the planting management scenario.



4. Results

4.1 Measurement error

Gorrod and Keith (2008) reports the variation in measurement of the BioMetric and Habitat Hectares BV site indexes at a range of CPW sites across a range of observers. If the estimated BV at the development and restoration areas are measured with error then BV may decline. This will be important if such declines compromise the viability of vulnerable species. For other species it is likely that underestimates in some areas will be compensated for by overestimates in other areas. However, if the measurement is systematically biased then the cost of biodiversity offsets for development actions will be biased and so there will either be too much or too little of such actions with the risk of a decline in BV in the landscape.

Figure 2 shows the average BV scores for the two indexes. The two indexes produced similar rankings of sites. The different weightings used in the two indexes do lead to some differences that may be significant for some vegetation types. These differences are also apparent in Figure 3 below.

Figure 2. Average BioMetric and scaled Habitat Hectares scores across observers for all sites.

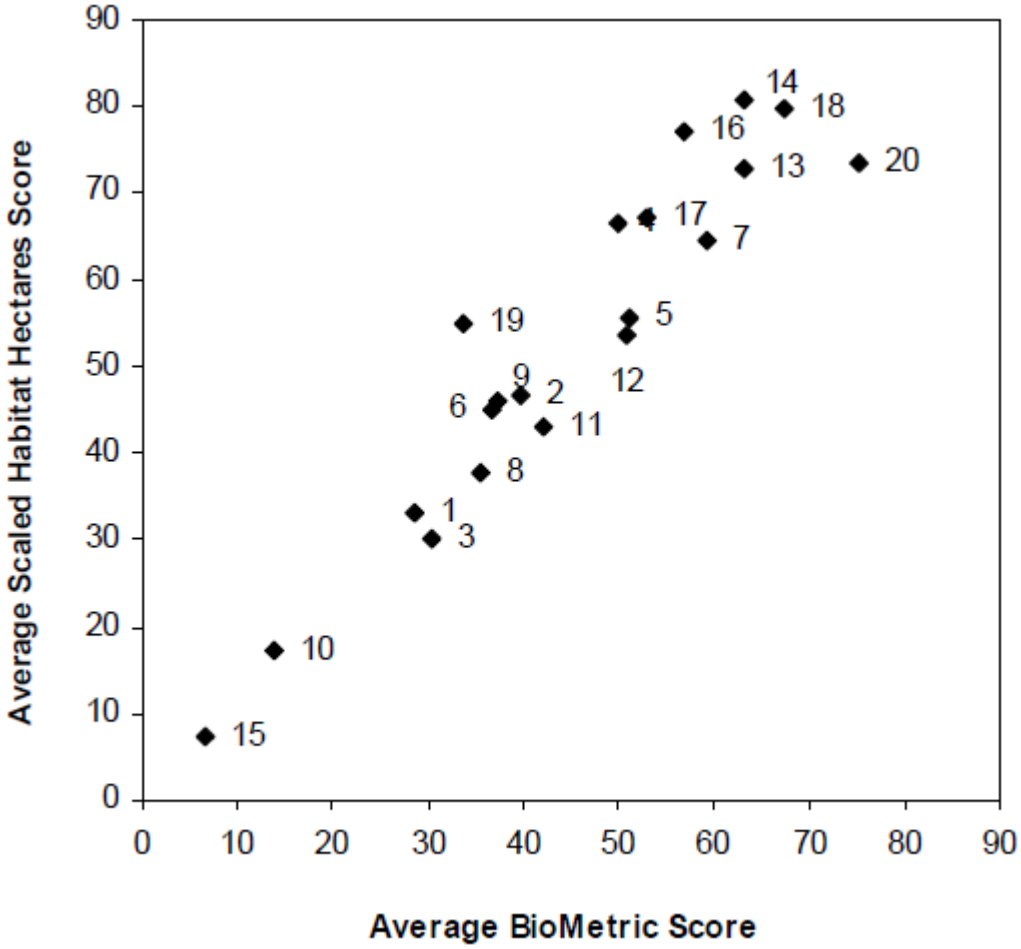
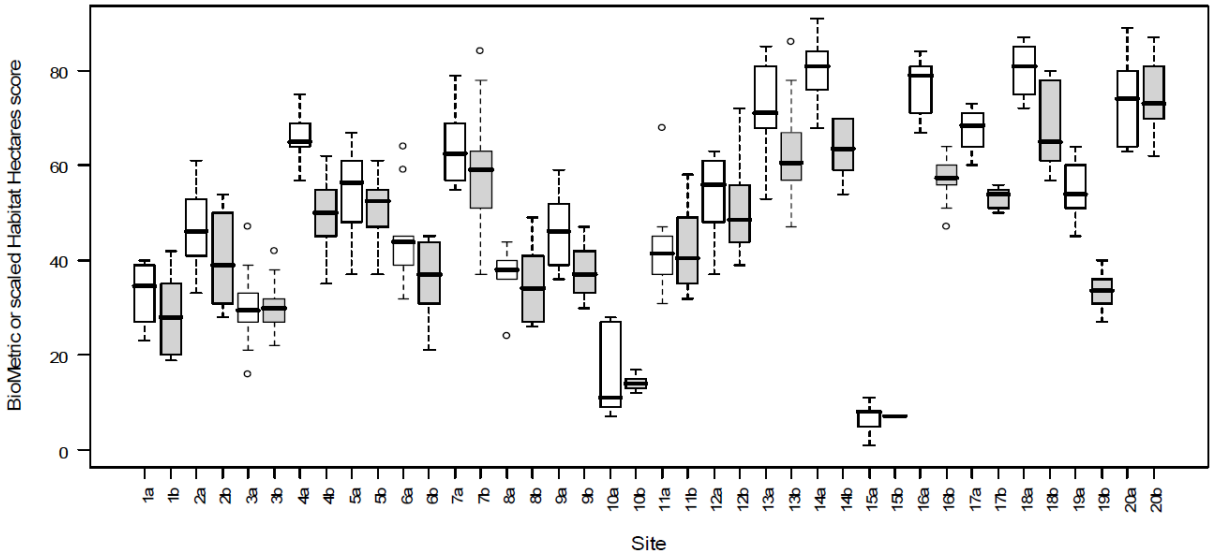


Figure 3 shows the distribution of the BV indexes across the 10 observers for each site.

Figure 3. Boxplots of total vegetation condition scores for each site, as measured by ten observers using the Habitat Hectares (open boxplots – subscript a) and BioMetric (grey boxplots – subscript b) methods. Boxplots show median, quartiles and outliers.



Observers' estimates varied substantially for all vegetation attributes on almost all sites. Observers generally agreed on the total scores and ranks of highly degraded (pasture) sites, but were less consistent on other sites. Across all sites, the average coefficient of variation (the ratio of the standard deviation to the mean) was 18% for BioMetric and 15% for Habitat Hectares, and the maximum was 60%. All observers estimated vegetation condition scores that were substantially different from the group mean on at least some sites. For example, each observer made an estimate of a vegetation index that was the furthest from the group mean on at least one site.

The results indicate that uncertainty in field estimates of site attributes may cause vegetation condition to be under- or over-estimated on all but highly degraded sites. The primary cause of observer variation in total vegetation condition scores was random error in raw estimates of vegetation attributes, rather than differences in the index structures or sampling methods. However, in cases where observer attributes are close to category boundaries the use of categorised scores can increase the uncertainty of the BV index – especially if such attributes have a large weight in the index.

Note that this study may have underestimated the variation in real-world assessments. This is because

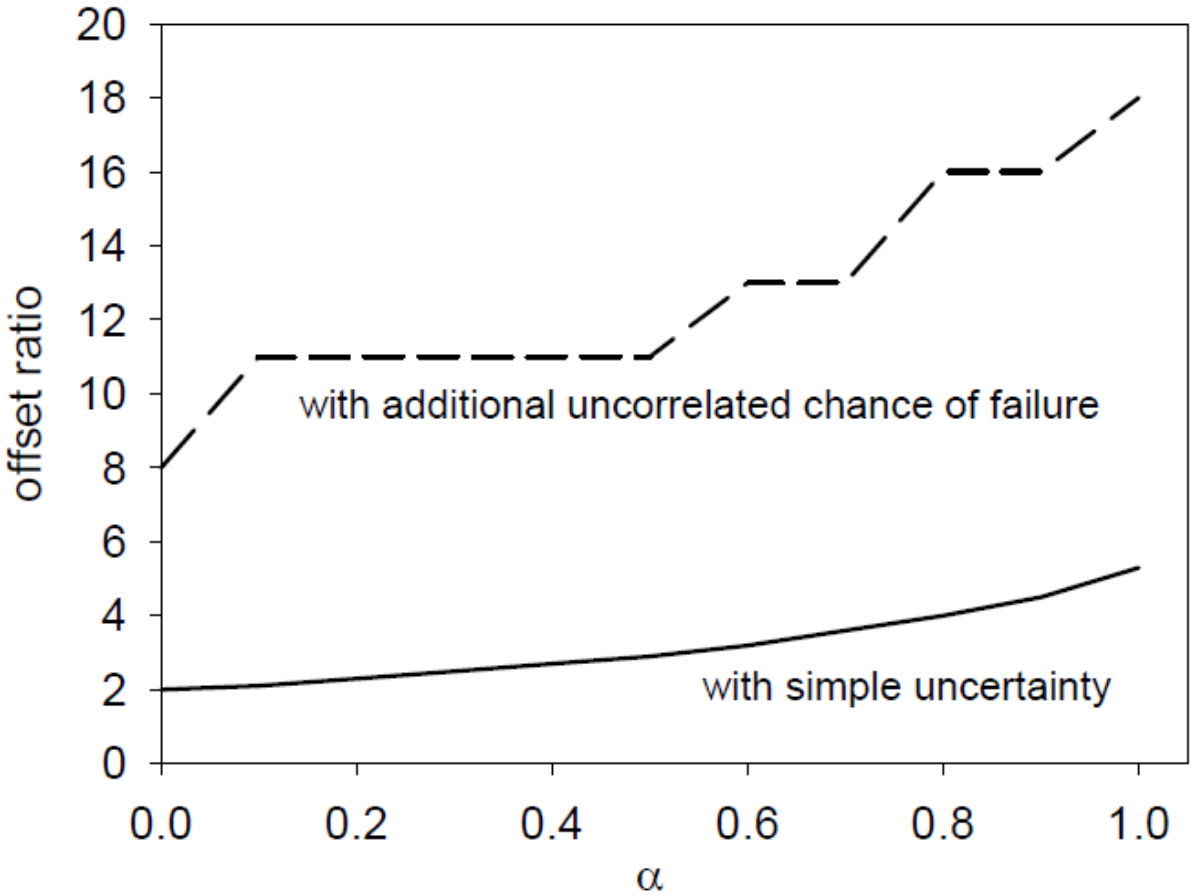
- observer variation in one of the attributes (species richness) was excluded;
- most observers were trained by the same person;
- all observers surveyed exactly the same area of each site;
- the seasonal conditions and time of year of the survey varied little; and
- the order in which observers conducted assessments was not random.

While there may be some scope for reducing observer error by enhancing management protocols an appreciable level of error appears unavoidable without significantly increasing the cost. Replicating the field surveys with different observers would reduce the error. However the results of this study indicate that in order to estimate the BV index within 10% of the true value around 30 observers would be required for most sites. Thus the processes associated with the use of such indexes need to be robust to this uncertainty. This may be facilitated by having field observers estimate uncertainty around point estimates of vegetation condition and using this information in the index calculation.

4.2 Robust offset ratios

Moilanen et al. (2007) use hypothetical calculations to explore the effect of various sources of uncertainty on the robust offset ratio $R_{ROB.}$ In these calculations an area of high conservation is being developed while the restoration area initially has poor quality. The MMU offset ratio, R_{MMU}, is assumed to be two. The solid line in figure 4 shows how R_{ROB} increases with the level of uncertainty, represented by α. When α is zero there is no uncertainty and the robust offset ratio equals the MMU ratio value of 2. As α increases to one the ratio increase to 5.25 in order to offset the loss with 95% certainty.

Figure 4. Robust offset ratios with simple uncertainty and with an additional chance of offset failure.

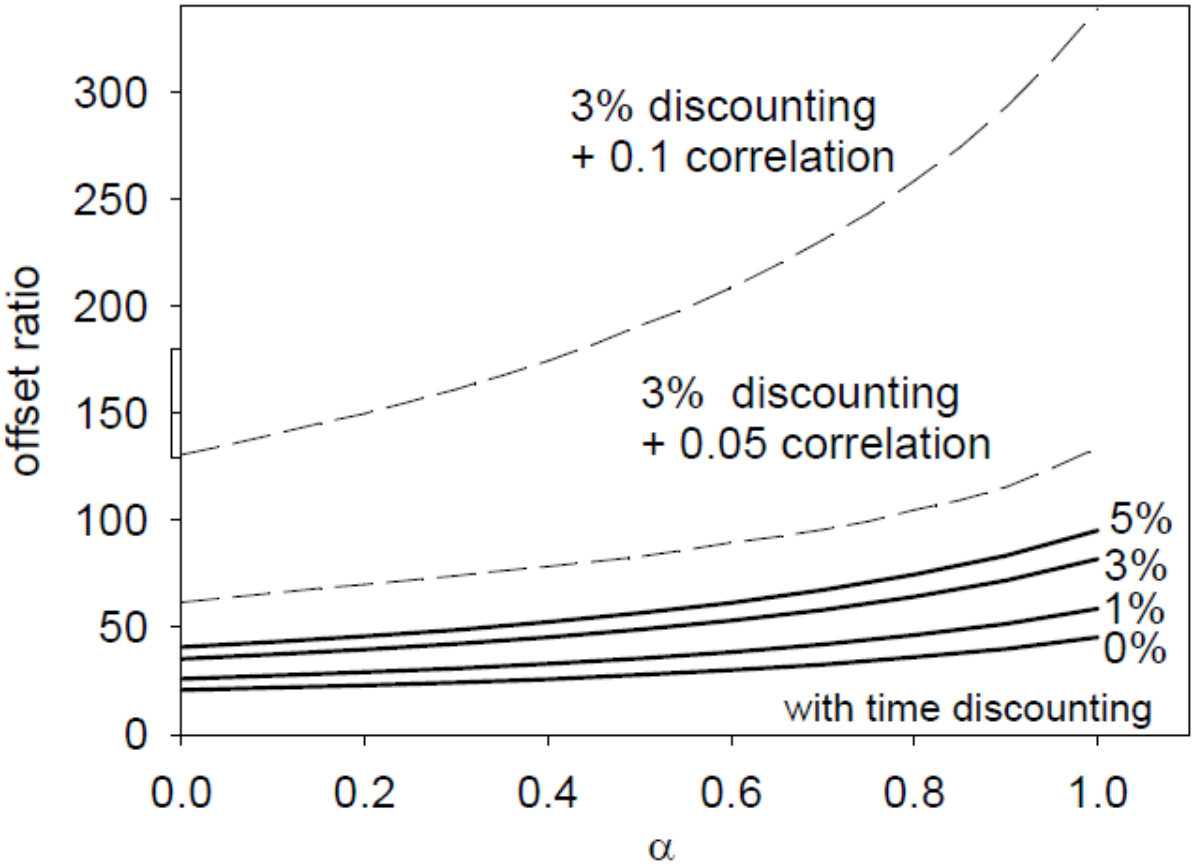


The dashed line in figure 4 shows the robust offset ratios when there is an additional (uncorrelated) chance that the restoration activity will fail with a probability of 0.5. This increases the MMU fourfold and increases the robust ratios by a similar amount as the level of uncertainty increases.

In practice such a quadrupling in the cost of offsetting a development action is likely to be sufficient to justify additional restoration actions in the case of restoration failure in order to reduce this cost.

The above offset ratios compare the gains on the offset site at some point in the future with the immediate losses at the development site. Figure 5 illustrates the effects of discounting over a 150 year time period – in addition to the types of uncertainty included above. This reduces the measured value of the restoration actions as the increase in BV occurs in the future. The dashed lines also assume that the failure probability is correlated across sites. In the extreme case that correlation is perfect then adding additional sites does not increase the chance of having a successful restoration site. With the assumptions used these dramatically increase the offset ratios.

Figure 5. Robust offset ratios with discounting and correlation in addition to the uncorrelated chance of failure.



If the MMU offset ratios are also calculated using the full information available then they also increase significantly as these additional factors are included. However, if substantial sources of uncertainty are ignored in the estimation of BV then these estimates are likely to be biased. In the above examples this would be the case if offset ratios are calculated without taking into account the chance that the restoration sites may entirely fail to provide any increase in BV. If these sites are not subject to ongoing management to ensure that they don't fail then such offset failures would lead to a net loss in BV. One simple alternative would be to delay full accreditation of offsets until they have been successfully established.

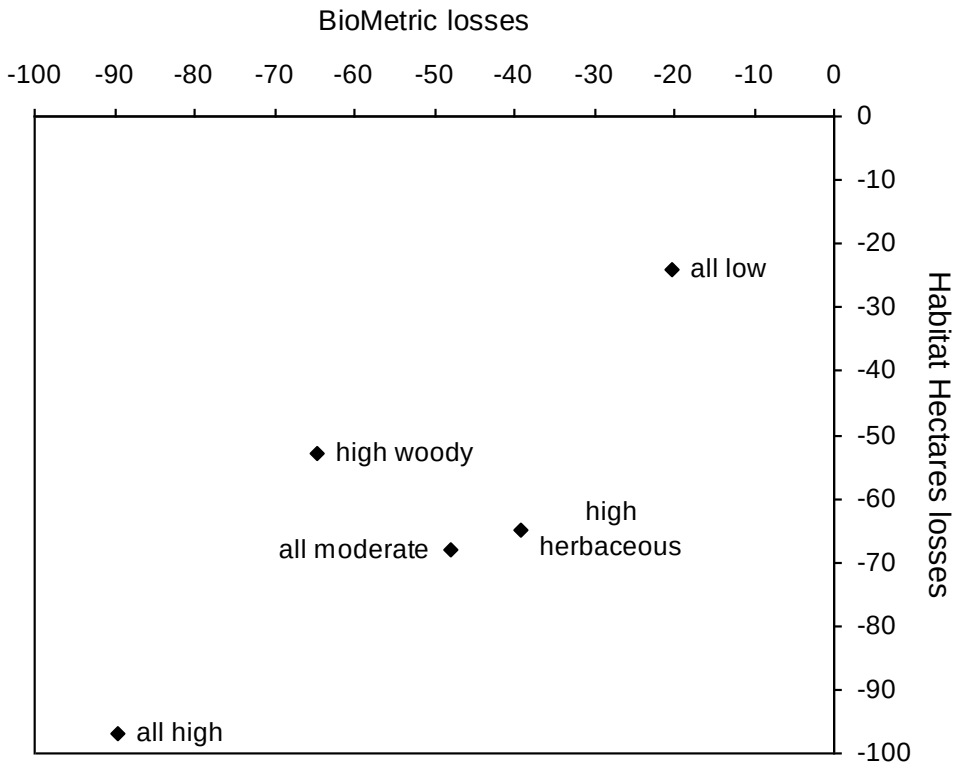
Some correlation of restoration failure across sites is likely although it need not result in total failure of the activity as assumed here. Exploring this issue fully would require an explicitly spatial analysis. Different configurations of restoration areas may increase their BV while correlated risks of offset failure may decrease the benefits of additional sites. Using a spatially explicit model of the BV of offset areas would be a valuable extension of this research. The final paper examines MMU and robust offset ratios in the context of representative sites of the CPW.

4.3 CPW simulations

Given the large number of cases examined by Gorrod et al. (in rep), only a subset and the broad conclusions are described here. Figure 6 presents the BV loss estimates under total clearing at each of the sites. These losses are assumed to occur in the year of clearing. The Habitat Hectares and BioMetric site indexes give similar loss measures although there are

some differences corresponding to the different weightings used. As was noted in the measurement error report Biometric weights the woody attributes more highly and Habitat Hectares weights the herbaceous attributes more highly. Under the other development scenarios the predicted losses were less and smallest for the grazing option. Under partial clearing most of the losses occurred immediately. The losses in BV due to grazing were more gradual with most of the losses accruing by year 50.

Figure 6. Predicted losses due to total clearing on five hypothetical site types, according to the BioMetric and Habitat Hectares protocols.



Turning to the restoration actions, the grazing exclusion/no action scenario produced little response or small declines on most type of sites. These responses are shown in figure 7. The exception is the *high herbaceous* site where grazing exclusion is likely to result in near benchmark states after 150 years (panels (d) and (i) in figure 7). Such sites are thus likely to be very attractive for cost effective biodiversity investments. However these sites may be fairly rare and the more costly restoration actions are likely to be required.

Figure 7. Predicted changes over time for BioMetric (a-e) and Habitat Hectares (f-j) scores under the No action scenario for all site types: all low (1st row), all moderate (2nd row), high woody (3rd row), high herbaceous (4th row), all high (5th row). Unbroken line is best estimate, dashed line is median and dotted lines are 95% confidence intervals.

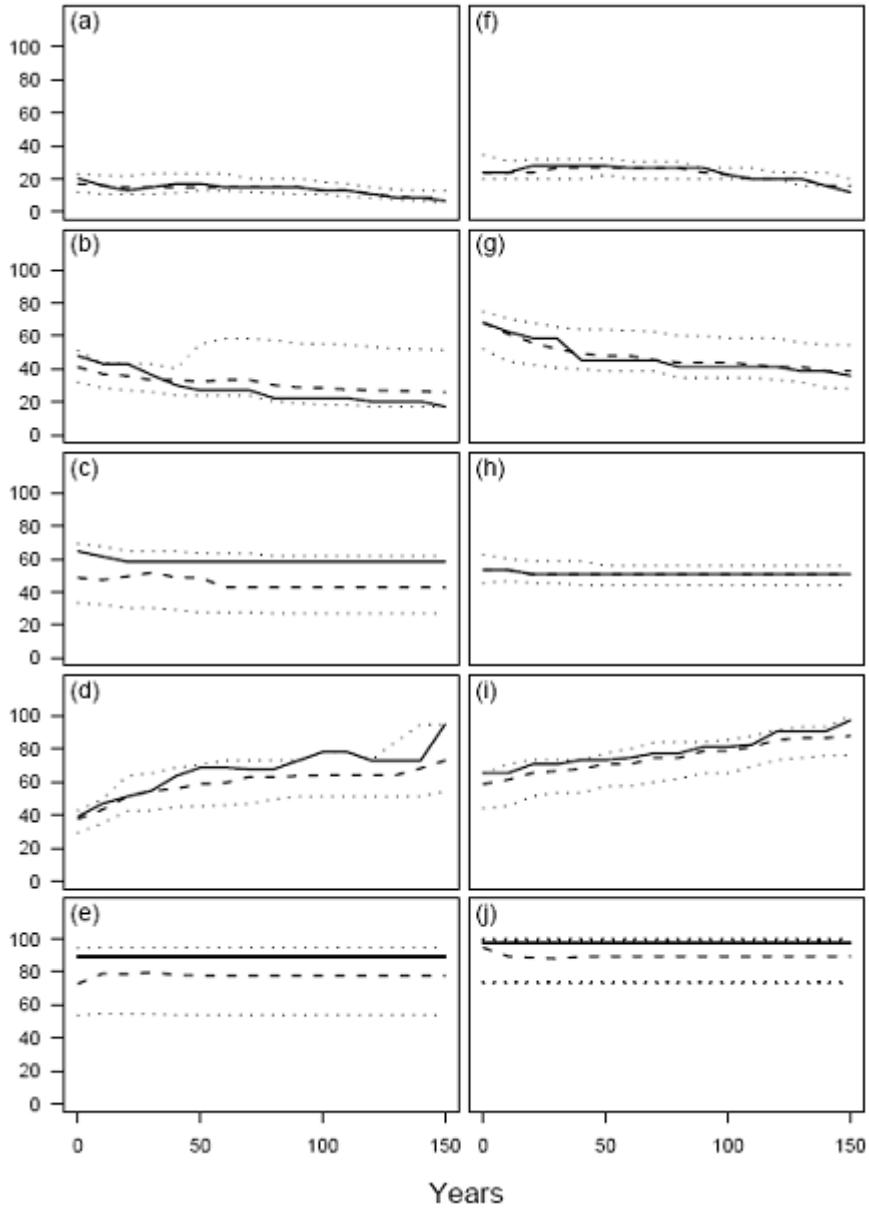
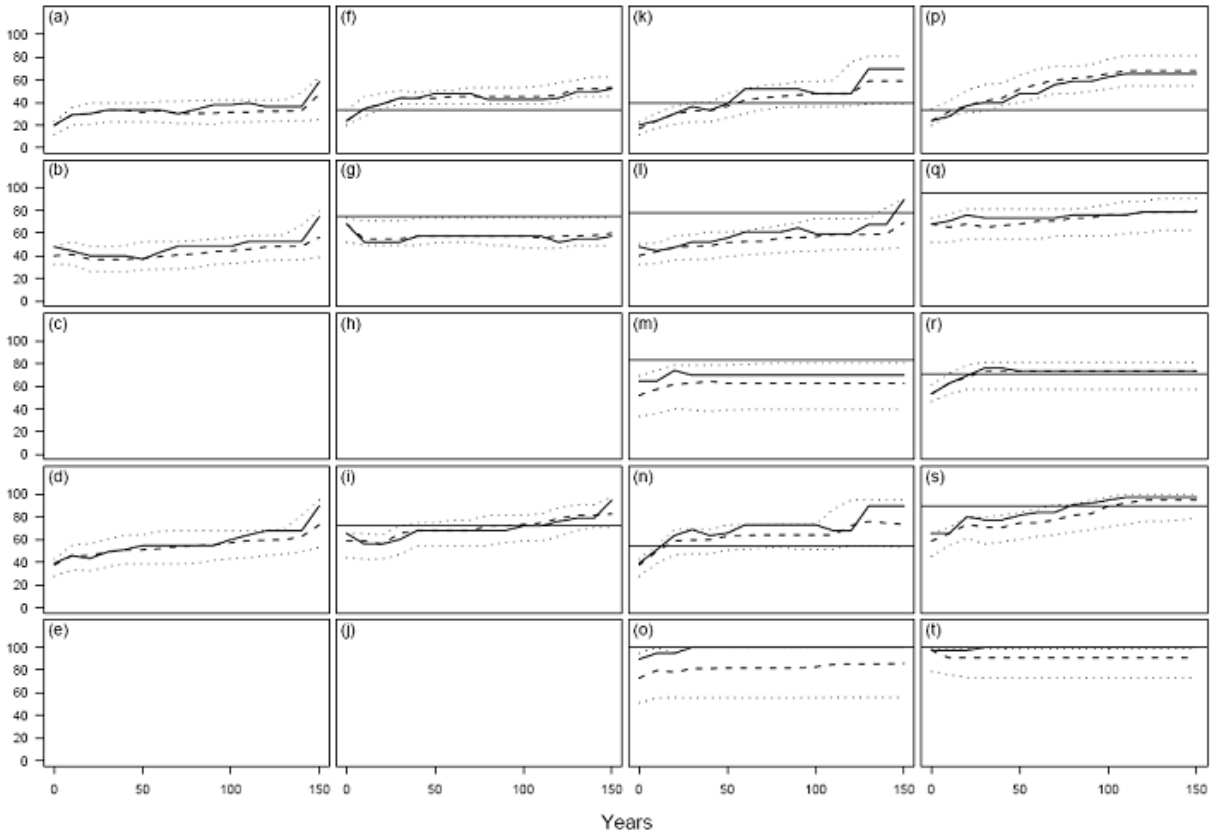


Figure 8 shows the responses to planting (the two leftmost columns) and mixed management (the two columns on the right) for the five sites. This figure also shows the improvements predicted using the established gain scoring protocols for these actions (DECC 2007 and DSE 2006) – the solid horizontal lines. Here the greatest simulated gains were on those sites with a degraded woody component with these restoration actions improving canopy cover, large trees and logs.

The herbaceous attribute gains were smaller and found to be achieved more slowly. Thus in the case of significant offsetting activity, losses of herbaceous attributes at development sites are likely to be compensated by increases in woody attributes on the offset sites. This could lead to a landscape-wide decline in the herbaceous components of the vegetation and of species particularly dependent on these attributes. There is scope for the development of restoration actions that specifically target the herbaceous attributes. This also reflects a limitation with these univariate indexes because attributes making up the index are fully substitutable. Declines in one attribute can be compensated by increases in another (McCarthy et al. 2004).

This could be remedied by also reporting an herbaceous attributes index in addition to the overall index. For particular vegetation types it may be worthwhile increasing the dependence of the BV index on these attributes.

Figure 8. Predicted changes over time for BioMetric (a-e; k-o) and Habitat Hectares (f-j; p-t) scores under the Planting (a-j) and Mixed management (k-t) scenarios for all site types: all low (1st row), all moderate (2nd row), high woody (3rd row), high herbaceous (4th row), all high (5th row). Gains predicted by DECC (2007) and DSE (2006) are shown as horizontal lines on BioMetric and Habitat Hectares graphs respectively. Graphs c, e, h and j are blank as planting was not simulated for the high woody or all high sites. Unbroken line is best estimate, dashed line is median and dotted lines are 95% confidence intervals.



The distributions of simulated gains, shown by the various non-straight lines in figure 8, are in many cases below the established prediction protocols (the straight horizontal lines) – such as with the treatment of the *all moderate* site (row two). However, the established protocols are too pessimistic in some cases such as mixed management on the most degraded site (row one). These differences are significant as they will affect the allocation of investments across the different restoration actions and types of restoration sites. If we take the simulations as being more accurate, then using the established prediction protocols overestimates the BV gain per dollar of restoration investment on the *all moderate* site and underestimates the BV gain per dollar spent on restoring the *all low* site. This would lead to too many *all moderate* sites being treated compared with the number of *all low* sites treated with the overall increase in BV across restoration sites being lower than would be possible for the same overall level of investment. As a result such biases would mean that restoration investments are not cost-effective.

The paper reports MMU and robust offset ratios for a number of combinations of development and restoration actions. The offset ratios varied from below one, where both actions occurred on the more degraded sites, to over one hundred. The highest offset ratios correspond to the cases where the development occurred on the *all high* site. The MMU offset ratios are shown in table 4.

Table 4. Offset ratios calculated by Matching Mean Utilities to offset 1 ha of Total clearing with No action, Planting or Mixed Management according to BioMetric and Habitat Hectares at years 10, 50, 100 and 150. (-) indicates that it was not possible to calculate a ratio because gains were insufficient on the offset site.

		Offset Action: No Action								Offset Action: Planting only								Offset Action: Mixed management							
Offset site	Total clearing site	BioMetric				Habitat Hectares				BioMetric				Habitat Hectares				BioMetric				Habitat Hectares			
		10	50	100	150	10	50	100	150	10	50	100	150	10	50	100	150	10	50	100	150	10	50	100	150
All low	All low	-	-	-	-	-	4.5	-	-	2.3	1.6	1.1	0.5	1.7	0.8	1.0	0.6	5.6	1.1	0.7	0.4	4.5	0.8	0.5	0.4
	All mod	-	-	-	-	-	12.8	-	-	5.4	3.7	2.7	1.3	4.8	2.1	2.7	1.8	13.1	2.6	1.7	1.0	12.8	2.1	1.3	1.2
	High wood	-	-	-	-	-	10.0	-	-	7.3	5.0	3.6	1.7	3.8	1.7	2.1	1.4	17.7	3.4	2.3	1.3	10.0	1.7	1.0	1.0
	High herb	-	-	-	-	-	12.3	-	-	4.4	3.0	2.2	1.0	4.6	2.0	2.6	1.8	10.7	2.1	1.4	0.8	12.3	2.0	1.3	1.2
	All high	-	-	-	-	-	18.3	-	-	10.1	6.9	5.1	2.4	6.8	3.0	3.9	2.6	24.6	4.8	3.2	1.8	18.3	3.0	1.9	1.8
All mod	All low	-	-	-	-	-	-	-	-	-	-	39.0	0.8	-	-	-	-	-	2.6	1.8	0.5	6.8	3.4	2.3	1.7
	All mod	-	-	-	-	-	-	-	-	-	-	92.0	1.8	-	-	-	-	-	6.1	4.2	1.2	19.1	9.6	6.4	4.8
	High wood	-	-	-	-	-	-	-	-	-	-	124.0	2.4	-	-	-	-	-	8.3	5.6	1.6	15.0	7.5	5.0	3.8
	High herb	-	-	-	-	-	-	-	-	-	-	75.0	1.5	-	-	-	-	-	5.0	3.4	0.9	18.4	9.2	6.1	4.6
	All high	-	-	-	-	-	-	-	-	-	-	172.0	3.4	-	-	-	-	-	11.5	7.8	2.2	27.4	13.7	9.1	6.8
High woody	All low	-	-	-	-	-	-	-	-	N/A								-	3.9	3.9	3.9	1.9	0.9	0.9	0.9
	All mod	-	-	-	-	-	-	-	-									-	9.2	9.2	9.2	5.5	2.6	2.6	2.6
	High wood	-	-	-	-	-	-	-	-									-	12.4	12.4	12.4	4.3	2.0	2.0	2.0
	High herb	-	-	-	-	-	-	-	-									-	7.5	7.5	7.5	5.3	2.5	2.5	2.5
	All high	-	-	-	-	-	-	-	-									-	17.2	17.2	17.2	7.8	3.7	3.7	3.7
High herb	All low	2.6	0.7	0.5	0.4	-	2.3	1.1	0.6	3.0	1.3	1.0	0.4	-	6.8	2.7	0.6	1.7	0.8	0.6	0.4	-	1.1	0.6	0.6
	All mod	6.1	1.6	1.2	0.9	-	6.4	3.2	1.6	7.1	3.1	2.3	0.9	-	19.1	7.7	1.7	4.0	1.8	1.4	0.9	-	3.2	1.7	1.6
	High wood	8.3	2.2	1.7	1.2	-	5.0	2.5	1.3	9.5	4.1	3.1	1.3	-	15.0	6.0	1.4	5.4	2.4	1.9	1.3	-	2.5	1.4	1.3
	High herb	5.0	1.3	1.0	0.7	-	6.1	3.1	1.5	5.8	2.5	1.9	0.8	-	18.4	7.4	1.7	3.3	1.5	1.2	0.8	-	3.1	1.7	1.5
	All high	11.5	3.0	2.3	1.6	-	9.1	4.6	2.3	13.2	5.7	4.3	1.8	-	27.4	11.0	2.5	7.5	3.4	2.6	1.8	-	4.6	2.5	2.3
All high	All low	-	-	-	-	-	-	-	-	N/A								3.9	1.9	1.9	1.9	-	6.8	6.8	6.8
	All mod	-	-	-	-	-	-	-	-									9.2	4.6	4.6	4.6	-	19.1	19.1	19.1
	High wood	-	-	-	-	-	-	-	-									12.4	6.2	6.2	6.2	-	15.0	15.0	15.0
	High herb	-	-	-	-	-	-	-	-									7.5	3.7	3.7	3.7	-	18.4	18.4	18.4
	All high	-	-	-	-	-	-	-	-									17.2	8.6	8.6	8.6	-	27.4	27.4	27.4

The robust offset ratios are generally higher but in many cases not dramatically so. The more extreme values occur in the earlier years or where it was not possible to calculate a ratio that compensated for a loss with 95% confidence. Interestingly the cases where this occurred differed between the two indexes. The robust offset ratios for the planting and mixed management (referred to as optimal management in the table) scenarios are shown in table 5.

Table 5. Robust offset ratios (95% probability of No Net Loss) to offset 1 ha of Total clearing, Partial clearing or Grazing with Planting or Mixed management, according to

BioMetric and Habitat Hectares at years 10, 50, 100 and 150. (-) indicates it was not possible to calculate a robust offset ratio due to insufficient gains on the offset site.

			Offset action: Planting only								Offset action: Optimal management							
			BioMetric				Habitat Hectares				BioMetric				Habitat Hectares			
Offset Site	Loss Action	Loss Site	10	50	100	150	10	50	100	150	10	50	100	150	10	50	100	150
All low	Grazing	All low	0.8	0.7	1.9	2.9	0.6	0.2	0.5	0.5	2.0	0.7	0.5	0.6	1.0	0.2	0.2	0.3
		All mod	1.0	2.6	5.4	5.9	2.0	1.8	2.3	1.7	2.4	2.1	1.3	1.4	4.0	1.5	1.0	1.1
		High wood	3.2	3.6	6.4	6.1	0.0	0.5	0.7	0.5	7.7	3.1	1.8	1.8	0.0	0.4	0.3	0.3
		High herb	0.8	1.3	2.5	2.7	2.8	1.8	2.4	1.7	2.0	1.1	0.7	0.7	5.7	1.5	1.0	1.1
		All high	4.6	4.2	7.9	6.9	3.6	3.0	2.8	1.9	11.1	3.8	2.4	2.2	7.0	2.3	1.3	1.2
	Partial clearing	All low	1.7	1.5	2.6	2.9	1.2	0.5	0.8	0.5	4.1	1.3	0.6	0.6	2.0	0.5	0.4	0.3
		All mod	4.6	4.1	7.5	8.2	7.0	3.3	3.5	2.2	11.1	3.4	1.8	1.8	11.7	2.7	1.6	1.4
		High wood	6.5	4.6	10.4	11.4	4.2	2.3	2.5	1.6	15.7	4.2	2.5	2.5	7.3	1.9	1.1	1.0
		High herb	3.6	3.1	5.5	6.1	8.0	3.4	2.8	1.9	8.9	2.7	1.4	1.4	13.3	2.6	1.3	1.3
		All high	9.1	7.3	14.7	16.2	10.8	5.5	5.5	3.4	22.1	5.9	3.6	3.6	19.0	4.5	2.5	2.3
	Total clearing	All low	2.3	2.1	3.5	3.9	3.6	1.6	1.6	1.0	5.6	1.7	0.9	0.9	6.0	1.4	0.8	0.7
		All mod	5.4	4.8	8.4	9.2	10.2	4.6	4.6	2.8	13.1	4.0	2.0	2.0	17.0	3.9	2.2	1.9
		High wood	7.3	6.5	11.3	12.4	8.0	3.6	3.6	2.2	17.7	5.4	2.8	2.8	13.3	3.1	1.7	1.5
		High herb	4.4	3.9	6.8	7.5	9.8	4.5	4.5	2.7	10.7	3.3	1.7	1.7	16.3	3.8	2.1	1.8
		All high	10.1	9.1	15.6	17.2	14.6	6.6	6.6	4.1	24.6	7.5	3.8	3.8	24.3	5.6	3.2	2.7
All mod	Grazing	All low	-	-	-	-	-	-	-	-	-	17.0	2.3	1.8	-	-	1.0	1.1
		All mod	-	-	-	-	-	-	-	-	-	49.0	5.9	3.9	-	-	6.3	4.1
		High wood	-	-	-	-	-	-	-	-	-	54.0	5.4	4.3	-	-	1.7	1.2
		High herb	-	-	-	-	-	-	-	-	-	15.0	2.8	1.9	-	-	5.5	4.1
		All high	-	-	-	-	-	-	-	-	-	106.0	7.9	6.1	-	-	7.8	4.3
	Partial clearing	All low	-	-	-	-	-	-	-	-	-	29.0	3.2	1.8	-	-	1.5	1.1
		All mod	-	-	-	-	-	-	-	-	-	75.0	9.1	5.1	-	-	9.0	4.9
		High wood	-	-	-	-	-	-	-	-	-	86.0	12.7	7.1	-	-	6.3	3.5
		High herb	-	-	-	-	-	-	-	-	-	61.0	6.8	3.8	-	-	7.8	4.3
		All high	-	-	-	-	-	-	-	-	-	155.0	17.7	10.1	-	-	14.5	7.6
	Total clearing	All low	-	-	-	-	-	-	-	-	-	39.0	4.3	2.4	-	-	4.5	2.3
		All mod	-	-	-	-	-	-	-	-	-	92.0	10.2	5.8	-	-	12.7	6.4
		High wood	-	-	-	-	-	-	-	-	-	124.0	13.8	7.8	-	-	10.0	5.0
		High herb	-	-	-	-	-	-	-	-	-	75.0	8.3	4.7	-	-	12.2	6.1
		All high	-	-	-	-	-	-	-	-	-	172.0	19.1	10.8	-	-	18.2	9.1
High woody	Grazing	All low	N/A								-	-	-	-	1.0	0.2	0.9	1.3
		All mod									-	-	-	-	2.3	2.9	3.9	4.3
		High wood									-	-	-	-	0.0	0.7	1.0	1.0
		High herb									-	-	-	-	3.4	2.9	3.4	4.0
		All high									-	-	-	-	4.7	4.2	4.4	4.4
	Partial clearing	All low									-	-	-	-	2.0	0.9	1.2	1.3
		All mod									-	-	-	-	11.7	5.1	5.6	5.6
		High wood									-	-	-	-	6.3	3.6	4.0	4.0
		High herb									-	-	-	-	12.3	4.9	4.4	5.3
		All high									-	-	-	-	18.0	8.7	8.7	8.7
	Total clearing	All low									-	-	-	-	6.0	2.6	2.6	2.6
		All mod									-	-	-	-	17.0	7.3	7.3	7.3
		High wood									-	-	-	-	13.3	5.7	5.7	5.7
		High herb									-	-	-	-	16.3	7.0	7.0	7.0
		All high									-	-	-	-	24.3	10.4	10.4	10.4
High herb	Grazing	All low	4.0	1.2	0.7	0.6	-	1.5	1.2	1.5	0.6	0.4	0.4	0.6	-	0.4	0.3	0.4
		All mod	8.5	3.5	2.0	1.3	-	11.5	5.6	4.5	1.1	1.2	1.2	1.3	-	4.0	1.3	1.4
		High wood	24.0	4.7	2.7	1.7	-	4.0	2.0	1.0	3.2	1.8	1.6	1.7	-	1.0	0.5	0.4
		High herb	4.0	1.4	1.0	0.7	-	11.5	6.6	3.4	0.9	0.6	0.6	0.7	-	4.0	1.5	1.4
		All high	13.5	6.6	3.4	2.3	-	18.0	7.2	4.3	4.3	2.2	2.0	2.0	-	6.2	1.7	1.5
	Partial clearing	All low	14.5	2.1	1.0	0.6	-	3.0	1.8	1.5	1.9	0.7	0.6	0.6	-	1.2	0.5	0.4
		All mod	37.5	5.4	2.7	1.8	-	19.5	7.8	6.0	5.0	1.9	1.7	1.8	-	7.2	2.0	1.6
		High wood	53.5	7.9	3.8	2.5	-	14.0	5.6	4.7	7.1	2.4	2.4	2.5	-	5.0	1.4	1.2
		High herb	31.0	4.1	2.0	1.3	-	18.5	6.8	5.2	4.1	1.4	1.3	1.4	-	6.8	1.7	1.4
		All high	65.5	10.6	5.4	3.5	-	30.5	12.2	9.7	9.7	3.6	3.4	3.5	-	12.2	3.1	2.5
	Total clearing	All low	19.5	2.8	1.3	0.9	-	9.0	3.6	3.0	2.6	0.9	0.8	0.9	-	3.6	0.9	0.8
		All mod	46.0	6.6	3.1	2.0	-	25.5	10.2	8.5	6.1	2.2	1.9	2.0	-	10.2	2.6	2.1
		High wood	62.0	8.9	4.1	2.8	-	20.0	8.0	6.7	8.3	3.0	2.6	2.8	-	8.0	2.0	1.7
		High herb	37.5	5.4	2.5	1.7	-	24.5	9.8	8.2	5.0	1.8	1.6	1.7	-	9.8	2.5	2.0
		All high	86.0	12.3	5.7	3.8	-	36.5	14.6	12.2	11.5	4.1	3.6	3.8	-	14.6	3.7	3.0
All high	Grazing	All low	N/A								-	-	-	9.7	-	-	-	-
		All mod									-	-	-	22.3	-	-	-	-
		High wood									-	-	-	13.0	-	-	-	-
		High herb									-	-	-	8.3	-	-	-	-
		All high									-	-	-	35.3	-	-	-	-
	Partial clearing	All low									-	-	-	9.7	-	-	-	-
		All mod									-	-	-	27.3	-	-	-	-
		High wood									-	-	-	38.0	-	-	-	-
		High herb									-	-	-	20.3	-	-	-	-
		All high									-	-	-	54.0	-	-	-	-
	Total clearing	All low									-	-	-	13.0	-	-	-	-
		All mod									-	-	-	30.7	-	-	-	-
		High wood									-	-	-	41.3	-	-	-	-
		High herb									-	-	-	25.0	-	-	-	-
		All high									-	-	-	57.3	-	-	-	-

5. Conclusion

Achieving and demonstrating efficiency in biodiversity investments is a difficult task. The research summarised here explores some of the issues associated with estimating and using site level vegetation condition measures for biodiversity investment planning. This research highlights the preliminary nature of current procedures for biodiversity investment planning. The use of vegetation condition metrics as a surrogate for BV is a practical approach that allows the biodiversity costs and benefits of alternate management actions to be compared. This will help planners and developers to identify cost-effective investments. However these metrics are unlikely to be equally informative for all vulnerable species. This is reflected in the differences between the two indexes examined in these studies.

The current studies focus on the site level measurement of vegetation condition. In practice both metrics are complemented by measures relating to the spatial context of the vegetation and the conservation status of the vegetation type. Undertaking a spatially explicit analysis of biodiversity investments at a landscape scale would be a useful extension to this work. This would also allow the heterogeneous costs of these investments to be incorporated and the correlation of restoration failure issue identified in the theoretical study to be explored. A practical issue here is the lack of landscape scale information on vegetation quality. There is a need for reliable estimates of vegetation quality in order to better target restoration actions.

Both the theoretical analysis and the simulations illustrate the importance of the time taken to achieve biodiversity gains. Investments need to be long term as little benefit is achieved in the first decade. To maintain or increase the viability of species it is necessary that vegetation of sufficient quality is maintained in landscapes. Additionally it is important to reduce the possibility that individual restoration actions fail. This would require some ongoing management and monitoring such as by delaying full accreditation or payments.

The simulations indicate that the MMU offset ratios generally underestimate the amount of restoration required to achieve no net loss for any particular development. At a broader scale, greater than expected benefits associated with other offsets may compensate for these losses. However, there may be a shift in the distribution of vegetation attributes towards those that are more readily restored such as the woody attributes and this may have detrimental effects for some components of biodiversity. Similarly, if the standard protocols for predicting biodiversity gains are systematically biased – as suggested particularly for the *all low* site by the simulation analysis – then lesser gains in biodiversity are likely to be achieved and investments may be directed towards less cost-effective actions. Some ongoing monitoring could be targeted to detecting these outcomes and trigger changes to the measurement or planning processes.

The studies indicate that that the most cost-effective actions are likely to involve restoration and development of the more degraded sites. Losses are greatest and hardest to offset on high quality sites. Fencing and excluding stock from the *high herbaceous* site appears to be a particularly attractive restoration action when these sites are available. Identifying such sites could be a useful planning exercise. For the other types of sites more active restoration actions are required.

All of the papers summarised here indicate that uncertainty in vegetation condition measures is significant and needs to be accounted for in biodiversity investment planning.

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7. Attachments

Paper 1

Report Cover Page

ACERA Project
0706
Title
Evaluating vegetation condition measures for cost effective biodiversity investment planning
Author(s) / Address (es)
Emma Gorrod[1,2] and David A. Keith[2] [1] School of Botany, University of Melbourne, Parkville, Victoria 3010, Australia. [2] NSW Department of Environment and Climate Change, PO Box 1967, Hurstville, NSW 2220, Australia.
Material Type and Status (Internal draft, Final Technical or Project report, Manuscript, Manual, Software)
Manuscript (final report)
Summary
This study was motivated by the call from Auditor General to provide reliable feedback on NHT investments. This study summarises the magnitude of observer error in measures of vegetation condition developed by the states. Errors are random and appreciable, and the study indicates the most important areas where improved field protocols will substantially improve the quality of monitoring and reporting.

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	DAFF Endorsement: ( ) Yes ( ) No	Date:

Evaluating vegetation condition measures for cost-effective biodiversity investment planning; ACERA Project No.0706

Kenton Lawson
ABARE

Observer variation in field assessments of vegetation condition: Implications for biodiversity conservation

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Manuscript (final report)

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Disclaimer

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Executive Summary

Assessments of vegetation condition are often used to inform land management and planning decisions for biodiversity conservation, such as allocation of incentive funding and determination of offset actions to compensate for biodiversity losses. Uncertainty in assessments of vegetation condition may lead to poor decisions or unexpected outcomes, including the loss of biodiversity. This study investigates uncertainty in assessments of vegetation condition due to observer error in field estimates of vegetation attributes. Ten observers conducted vegetation condition assessments using two assessment protocols (BioMetric and Habitat Hectares) on 20 sites in a grassy woodland community. Observers' estimates varied substantially across multiple scoring categories for all vegetation attributes on almost all sites. Observers generally agreed on the total scores and ranks of highly degraded (pasture) sites, but were less consistent on other sites. Across all sites, the average coefficient of variation was 18% for BioMetric and 15% for Habitat Hectares, and the maximum was 60%. All observers estimated vegetation condition scores that were substantially different from the group mean on at least some sites. The results indicate that uncertainty in field estimates of site attributes may cause vegetation condition to be under- or over-estimated on all but highly degraded sites. The primary cause of observer variation in total vegetation condition scores was random error in raw estimates of vegetation attributes, rather than differences in the index structures or sampling methods. It is recommended that: research is undertaken into methods for reducing observer error; field observers estimate uncertainty around point estimates of vegetation condition; the sensitivity of index scoring structures to observer error is reviewed; and that decision makers explicitly incorporate uncertainty into the decision making processes and aim for outcomes that are robust to this uncertainty.

Introduction

Quantitative estimates of vegetation attributes are frequently required to inform land management decisions for biodiversity conservation. Relevant decisions include the allocation of incentive funding to manage private land for biodiversity conservation (USDA 2003; Oliver et al. 2005) and determination of offset actions to compensate for unavoidable biodiversity losses (ten Kate et al. 2004; DEC 2005a; DSE 2006). Decisions that fail to consider uncertainty in field assessments of vegetation attributes may lead to unexpected outcomes, including loss of biodiversity. Observer error in field estimates of vegetation attributes may be an important cause of uncertainty in land management decisions for biodiversity conservation (for others see Gorrod et al. *in review*). Yet decisions in the context of biodiversity conservation are usually made as though the values of vegetation attributes are known without error.

Observer error may occur due to inaccurate estimation of a quantity (measurement error) or failure to correctly identify or interpret the feature to be estimated (identification error). Measurement error arises from observational techniques or instrument error, and varies randomly about the true value (Regan et al. 2002). Identification errors may occur as a result of linguistic uncertainty (imprecise language) in the protocol's survey methodology. Identification error may occur, for example, if an observer estimates the length of all logs rather than only those logs that exceed a specified diameter, or

assesses projective foliage cover rather than canopy cover (e.g. Gray and Azuma 2005). Some observers may systematically over- or under-estimate attribute values. Systematic bias and identification error may confound measurement error such that variation across estimates of multiple observers does not centre on the true value of the parameter.

Observer error is reflected in imprecision amongst the estimates of multiple observers. Research into precision of multiple observers estimating vegetation cover has reported coefficients of variation between 10 and 200% (Sykes et al. 1983; van Hees and Mead 2000; Klimes 2003). As true values of vegetation attributes are seldom known, it is difficult to estimate accuracy of observers' estimates, particularly for cover estimates. Underestimation of counts (detectability) has been documented for attributes such as hollow bearing trees (Harper et al. 2004) and plant species richness (e.g. Hellmann & Fowler 1990; Ringvall et al. 2005).

Observer error in field estimates of vegetation attributes may be exacerbated by the mathematical structure of multivariate indices used to quantify biodiversity value at the site scale. Conversion of raw attribute estimates into categorical scores may exacerbate error, for instance, if the placement of scoring thresholds is such that small errors in raw estimates cause changes in class membership. Scoring errors may be compounded if combined multiplicatively.

Different protocols for quantifying biodiversity value at the site scale, including field survey techniques and multivariate index structures, are developed by different jurisdictions to meet their policy requirements. The Environmental Benefits Index, for example, is used to rank sites for incentive funding in the Conservation Reserve Program in the United States (USDA 2003). In Australia, vegetation condition indices are used to allocate incentive funding and determine offset requirements. The BioMetric (Gibbons et al. 2005) and Habitat Hectares (Parkes et al. 2003; DSE 2004) protocols are used in New South Wales and Victoria, respectively. Both protocols require field assessments of vegetation structural and composition attributes (listed in Table 1), which are selected on the basis that they are surrogates for habitat features required by indigenous species or indicators of ecological processes. Both protocols allocate scores for each site attribute relative to reference (or benchmark) values. Scores for individual attributes are combined to yield a total vegetation condition score that represents the similarity of the site to a benchmark stand of the same vegetation community.

We aimed to address the following questions:

(1) What is the magnitude of variation among assessors in their assessment of attributes contained within contemporary vegetation condition assessment tools?
(2) What is the impact of this variation on vegetation condition metrics generated by two contemporary vegetation condition assessment tools?
(3) What are the implications for biodiversity conservation of the results for (1) and (2)?

Methodology

Protocols for assessing vegetation condition

The BioMetric and Habitat Hectares indices contain similar sets of vegetation attributes, which are estimated, weighted and combined using different methods (Oliver et al. 2007) (Table 1). Though both use relatively broad scoring intervals for each attribute, in part to accommodate a margin of observer error, the size and number of scoring intervals differs (see Table 3 in Gibbons et al. 2005 and Appendix 8 in DSE 2004). The total BioMetric score is calculated by combining the individual attribute scores using Equation 1 to yield a score out of 100, whereas Habitat Hectares total score is the sum of attribute scores with a maximum score of 75. In this study, Habitat Hectares scores have been standardised to a maximum value of 100.

$$\text{BioMetric score} = \frac{\left(\sum_{v=a}^{j} s_v \cdot w_v\right) + 5\left(s_a \cdot s_g + s_b \cdot s_i + s_h \cdot s_j + s_c \cdot s_k\right) \times 100}{480} \qquad (1)$$

S_v is the score for attribute a-j (see Table 1), w_v is the weight of the attribute and S_k is the average of scores for attributes d, e and f. Selected pairs of attributes are combined as products to reflect ecological relationships (S. Briggs, pers. comm., 2007). The multiplication of scores for *number of native plant species* and *lack of exotic cover*, for example, implies that they are not directly substitutable and co-occurrence of both attributes would substantially improve the value of a site (Gibbons and Freudenberger 2006). The denominator, 480, is the maximum possible score for a community in which values for all ten attributes fall within the benchmark.

Table 1. BioMetric and Habitat Hectares component attributes, weightings and Cumberland Plain Woodland benchmarks.

BioMetric				Habitat Hectares		
Attribute		**Benchmark**	**Weight**	**Pairs of attributes**	**Benchmark**	**Weight**
H	Number of hollow bearing trees	≥1 tree	30	Number of large trees per hectare	>15 trees with dbh ≥50cm	10
				Large tree canopy health	>70%	
B	Native over-storey cover	19-24%	5	Tree canopy cover	7-22%	5
				Tree canopy health	>70%	
A	Native plant species richness	>29	20	Diversity & cover of understorey lifeforms	≥9 lifeforms present	25
C	Native mid-storey cover	20-30%	10			
D	Native ground cover (grasses)	23-31%	5	Proportion of understorey lifeforms that are substantially modified	0 lifeforms modified	
E	Native ground cover (shrubs)	0-5%	5			
F	Native ground cover (other)	12-20%	5			
g	Cover of weeds	0-5%	5	Cover of weeds	0-5%	15
				Proportion of weed species that are considered high threat	0	
i	Proportion of overstorey species regenerating	100%	10	Total number of woody species	≥5	10
				Proportion of woody species recruiting	>70%	
j	Total length of logs	≥5 m of logs ≥10cm diameter	5	Total length of logs	≥7.5 m of logs of ≥10cm diameter	5
				Proportion of logs that are large	≥2.5 m of logs of ≥25cm diameter	
				Litter cover	5-15%	5
				Dominance of native/exotic litter	Native litter	
Total			**100**	**Total**		**75**

BioMetric assessments are conducted in 20 x 50 m plots whereas Habitat Hectares assessments are conducted within an area of unlimited size containing relatively homogeneous vegetation. Cover attributes are visually estimated for Habitat Hectares, whereas observers may choose to either visually estimate or use point count techniques for estimating native shrub cover, native ground cover and total exotic cover for BioMetric. Both BioMetric and Habitat Hectares protocols supply operational manuals (Gibbons et al. 2005 and DSE 2004 respectively), which contain different schematics for assisting observers to make cover estimates. The BioMetric protocol requires the observer to specify a point estimate which is then converted into a score, whereas the Habitat Hectares protocol requires the observer only to select a scoring category. The landscape context components of the indices were not addressed in this study.

Observers

Ten observers were selected to represent a sample of all possible observers that may carry out vegetation condition assessments. The observers all had relevant tertiary qualifications and previous experience conducting vegetation surveys, and included the

authors (Observers A and B) (Table 2). General experience ranged from approximately two years experience as an environmental consultant (Observer J) through to 26 years experience as a plant ecologist (Observer B). Observers E and F were relatively more experienced in the use of the Habitat Hectares and BioMetric methodologies respectively.

Table 2. Characteristics of observers in the field trial.

CCode	Current occupation	Years of general vegetation survey experience	Familiarity with Cumberland Plain Woodland[1]	Familiarity with survey methods (BioMetric/Habitat Hectares) [2]
A	PhD candidate	5	Mod	low/mod
B	Principle Research Scientist, DECC[3]	26	High	low/low
C	Vegetation dynamics Research Officer, DECC	4	Low	low/low
D	Vegetation dynamics Project Officer, DECC	7	Low	low/low
E	Native Vegetation Project Officer, DSE[4]	4.5	Nil	low/high
F	Information and assessment Officer, DECC	10	Mod	mod/low
G	Vegetation dynamics Research Officer, DECC	8	Mod	low/low
H	Environmental consultant (flora)	5	High	low/low
I	Environmental consultant (flora)	8	Mod	low/low
J	Environmental consultant (flora, fauna & hydrology)	2	Mod	low/low

1: Self rated familiarity with Cumberland Plain Woodland. 2: Relative familiarity with (i.e. field experience using) BioMetric and Habitat Hectares. 3: New South Wales Department of Environment and Climate Change. 4: Victorian Department of Sustainability and Environment.

Field trial

All observers independently conducted vegetation condition assessments using BioMetric and Habitat Hectares protocols on 20 sites in Cumberland Plain Woodland (CPW), west of Sydney metropolitan area, Australia. CPW is a grassy woodland community that occurs on shale derived soils and is listed as an Endangered Ecological Community on state and commonwealth legislation (ESSS 2000; NPWS 2004). Sites were selected to represent a spectrum of structural and compositional variants of CPW, in which each of the canopy, shrub and ground strata were a) either structurally intact or modified and b) dominated by either native or exotic plant species (Table 3). All sites were located in reserves at the time of survey, but had been subject to a range of management histories including grazing, clearing, logging, fertiliser application, planting, weed control and conservation. Each site consisted of an area of relatively homogeneous vegetation and ranged in size from approximately 0.5 to 3 hectares. The available BioMetric benchmark for CPW was unmodified from the published version (DEC 2006). The benchmark for Habitat Hectares was composed on the basis of expert opinion (DK), and available floristic and structural survey data (Tozer 2003) (Table 1).

Table 3. Structural and compositional characteristics of sites 1 to 20, and their locations (Western Sydney Regional Parklands (WSRP), Planning NSW, Prospect Reservoir and Scheyville National Park).

Tree canopy	**Absent**		**Present (Planted)**		**Present (Non planted)**			
Shrub and ground layer	Absent	Dense native	Absent or sparse	Dense native	Absent or sparse	Dense exotic	Dense native	Open native
WSRP	10, 15		1, 3, 8, 11	2		5, 12		7, 20
PlanningNSW					4			
Prospect		19			16, 17		18	
Scheyville		9				6	13	14
Total	**2**	**2**	**4**	**1**	**3**	**3**	**2**	**3**

At each site, a floristic survey in which the occurrence of all native and exotic plant species were recorded within a randomly located 20 x 20 m quadrat was conducted by Observer A and at least one other observer. Due to observers' time constraints, the floristic data were then available to all observers conducting vegetation condition assessments. This reduced variation between observers for estimates of lifeform richness that may have been attributable to differences in plant identification skills. The native plant species richness attribute of the BioMetric method was estimated for all observers from the common floristic data set.

Assessments of vegetation condition were conducted by up to five independent observers on the same day between January and November 2006. The 20m x 50m quadrat used in the BioMetric method, and the boundary of the 'zone of relatively homogeneous vegetation' used in the Habitat Hectares method, was positioned in the same location for all observers, and encompassed the 20m x 20m floristic plot. Observers B, C and D conducted most assessments concurrently between January and October 2006; Observers E, F and G conducted all assessments concurrently over two weeks in May 2006; and Observers H, I and J conducted most assessments concurrently between May and October 2006. Weather conditions were reasonably stable throughout the duration of the surveys, and the vegetation attributes of CPW are not prone to significant seasonal fluctuations. The order in which the sites were visited was randomised amongst groups and the order in which each protocol was used was randomized amongst observers and sites. In accordance with the field assessment protocols, observers recorded point estimates for BioMetric attributes and selected scoring categories for Habitat Hectares attributes.

Observer A was trained by an officer from the Victorian Department of Sustainability and Environment in conducting field assessments using the Habitat Hectares method. The first time that other observers conducted assessments, Observer A trained them in the use of each assessment protocol, using the supporting documentation and guidelines provided (DSE 2004 and Gibbons et al. 2005). The documentation was available at all times for observers to consult for the remainder of the assessments. No attempt was made to calibrate estimates of percent cover amongst observers. Observers were not permitted to discuss their estimates with others, including the interpretation of terms beyond definitions provided in the protocols.

Data analysis

We analysed the magnitude of variation amongst observers' estimates of the attributes contained within BioMetric by plotting the range of observer estimates, and calculating Coefficients of Variation (CV), on each site. CV is calculated by dividing the standard deviation by the mean (of ten observers' estimates) and provides a unitless measure of the variation that is comparable across different types of vegetation attributes. CV is sensitive to small changes in the mean when the mean value is near zero. Variation was not calculated in this way for Habitat Hectares attributes as point estimates were not specified in field assessments.

The spread of observer estimates of vegetation attributes across BioMetric scoring categories was plotted, as was the spread of scores for Habitat Hectares attributes. CV of total scores was calculated to determine whether the magnitude of variation amongst observers was related to site types. The total score of each site averaged across observers was plotted for BioMetric and Habitat Hectares, and spearman rank correlation calculated, to determine how well the two metrics correlated. Spearman rank correlations were calculated for total score estimates between all pairs of observers, to test the direction and strength of relationships amongst observers.

Results

Variation in attribute estimates amongst observers

For all attributes recorded in the BioMetric assessment on each site, we calculated the mean and CV of the ten observers' estimates. For all attributes, CV declined with increasing mean, and for any given mean the magnitude of CV tended to vary more across sites than attributes. However, CV of *native ground cover (shrubs)* tended to be comparatively higher, and *native overstorey cover* comparatively lower, than other attributes (Figure 1). For means of 5 and under CVs ranged from 40% to 300%, and CV was not less than 20% for means up to 75 (Figure 1).

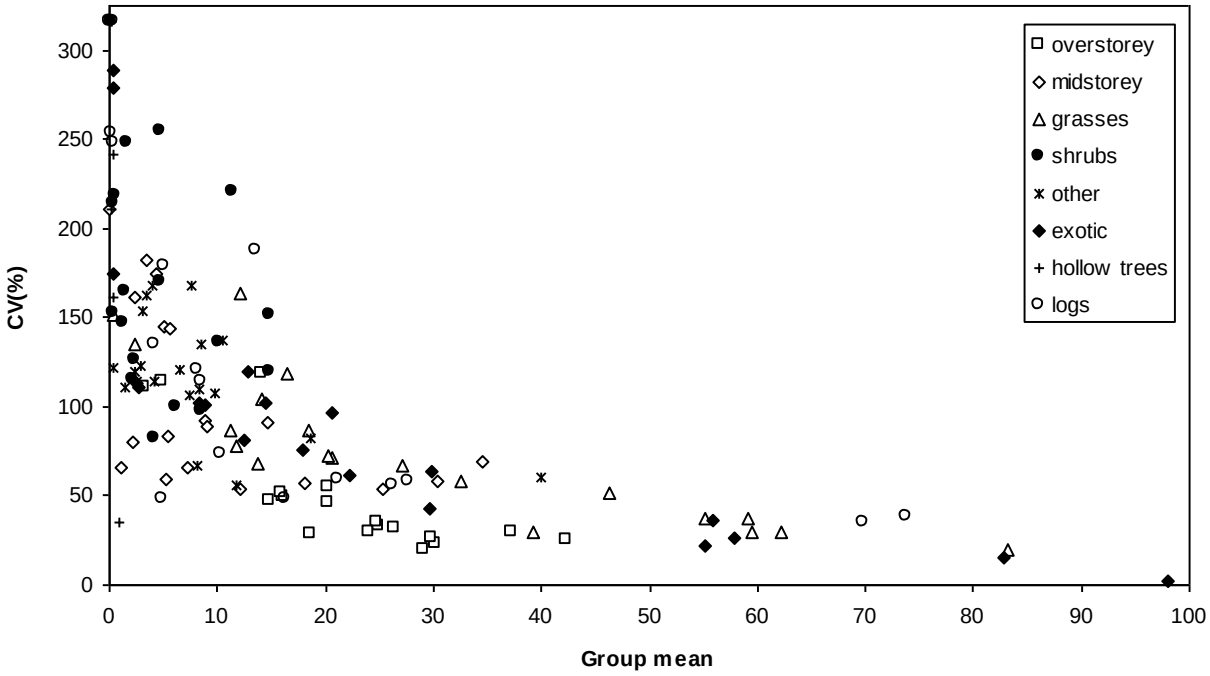
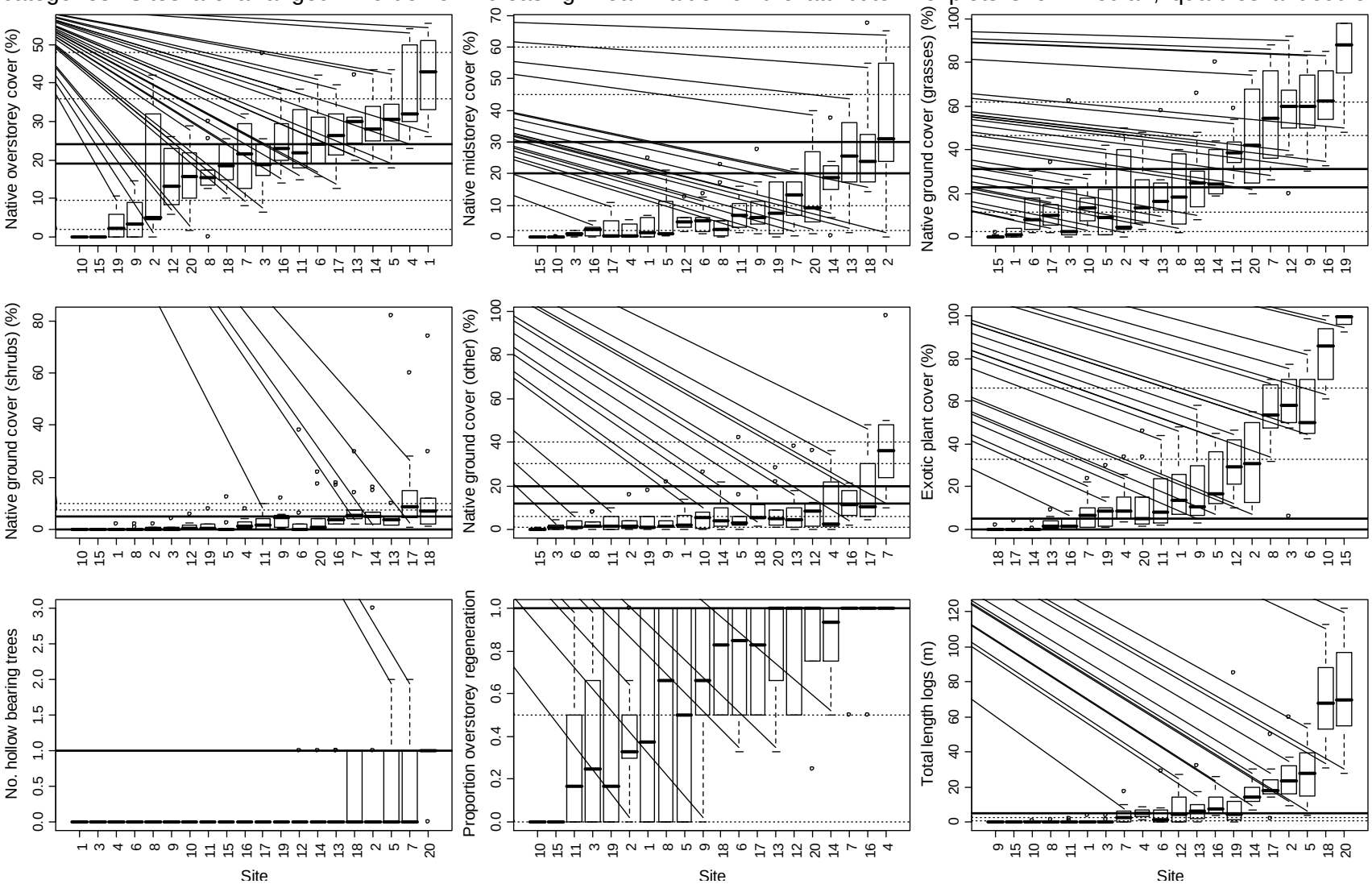


Figure 1. Coefficients of variation for vegetation attributes estimated by ten observers on twenty sites for BioMetric assessments.

The range of raw estimates for each attribute on each site almost always spread across multiple BioMetric scoring categories (Figure 2). Exceptions (in which all observations were within the same scoring category) were for most BioMetric attributes on the pasture sites (Sites 10 and 15); sites on which no observers recorded a *hollow bearing tree*; and sites for which mean values of *exotic plant cover*, *proportion of overstorey species regenerating* and *length of logs* were high (Figure 2). Outliers in estimates of *native ground cover (shrubs)* were primarily from one observer (Observer J). The BioMetric attributes for which raw estimates spread across the greatest number of scoring categories on average were: proportion of *overstorey species regenerating*, and the lowest was for *number of hollow bearing trees* (because there are only two categories).

Figure 2. Boxplots of observer's estimates of vegetation attributes included in the BioMetric vegetation condition metrics. Thick horizontal lines indicate the benchmark scoring category for the attribute, and thin horizontal lines indicate the other scoring categories. Sites are arranged in order of increasing mean value for the attribute. Boxplots show median, quartiles andoutliers.

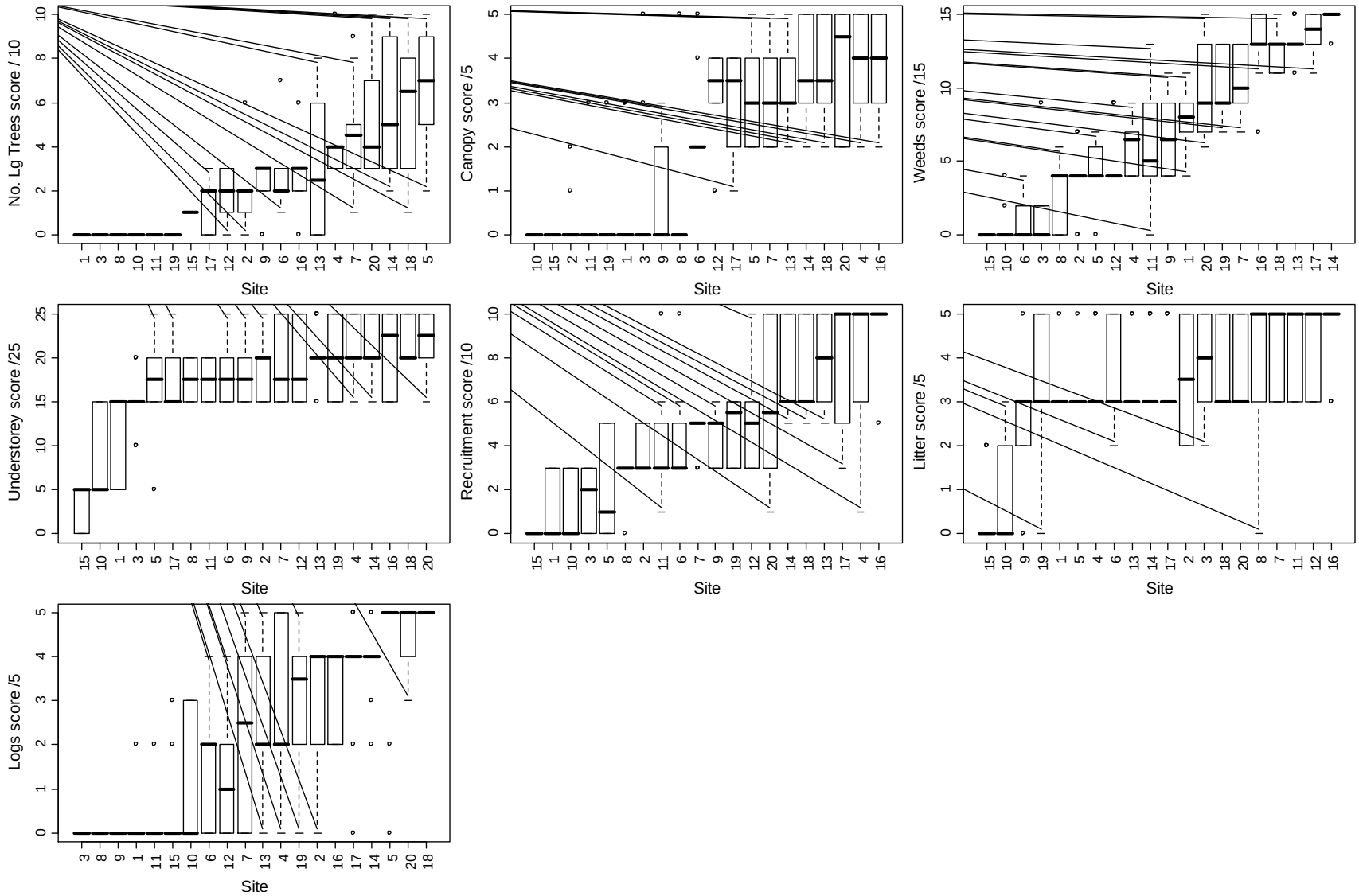


Observers' estimates also generally spanned multiple scoring categories for Habitat Hectares attributes, except for those pasture sites (10 and 15) and when there were no large trees, canopy (though not always), or logs (Figure 3). Observer estimates of Habitat Hectares' *understorey* score span ten points (two scoring categories) for all sites, except site 15. The other highly weighted attributes varied substantially too (but less than understorey), the range of scores for *number of large trees* increased as mean score increased (with range of 7-9 out of 10 points for the five sites with highest mean scores), *weeds* had greatest spread of scores for intermediate mean scores, and recruitment.

Though our study was not designed to distinguish between measurement and identification errors, the latter were apparent where some observers recorded *native overstorey cover* in the BioMetric assessment as zero (absent) when most observers recorded it as present (Figure 2).

BioMetric scoring structure may have exacerbated the expression of observer variation for some attributes. This was most apparent for *native ground cover (other)*, in which raw estimates below 10% cover that differed by only a few percentage points frequently spanned three scoring thresholds (Figure 2). For most attributes, including *native ground cover (grasses)* and *proportion of overstorey species regenerating* (Figure 2), raw estimates varied substantially and were scattered across multiple scoring thresholds. Point estimates were not recorded for Habitat Hectares.

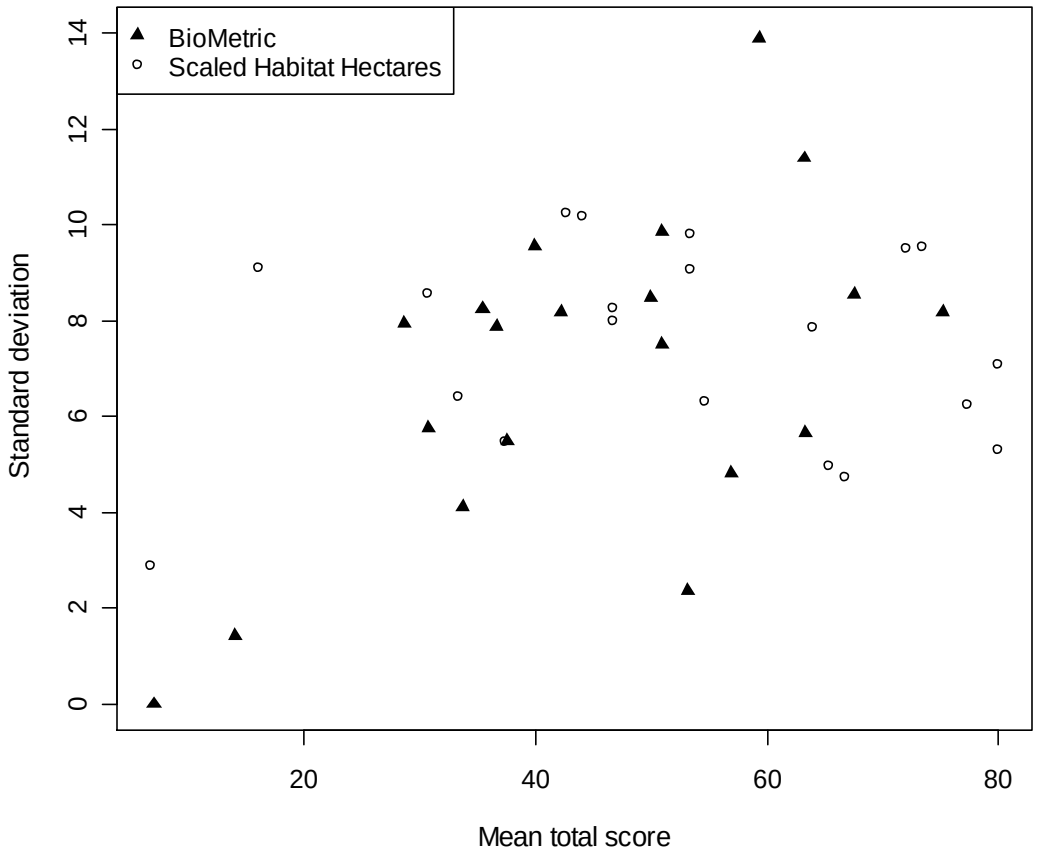
Figure 3. Boxplots of scores recorded by ten observers for Habitat Hectares assessments on 20 sites. Sites are arranged in order of increasing mean score for the attribute. Boxplots show median, quartiles and outliers.



Variation in total BioMetric and Habitat Hectares scores

The average standard deviation around the mean across all sites was 7 for BioMetric and 5.5 for Habitat Hectares. For almost all sites with mean total BioMetric or Habitat Hectares scores greater than 30, the standard deviation was between approximately 4 and 10 (Figure 4). Two remnant sites had slightly higher standard deviations for BioMetric: Sites 7 and 13 had mean scores of about 60 and standard deviations of 14 and 11 respectively. The two pasture sites (Sites 10 and 15) had low mean scores and low standard deviations, with the exception of Site 10 for Habitat Hectares which had a higher standard deviation due largely to differences in *understorey* scores. The CV of total scores declined from approximately 30% to approximately 10% as mean scores increased from 30 to 80 for both metrics (data not shown). The average CV of total scores across observers was 15% for BioMetric and 18% for Habitat Hectares, and the maximum was 60% (Site 10 according to Habitat Hectares).

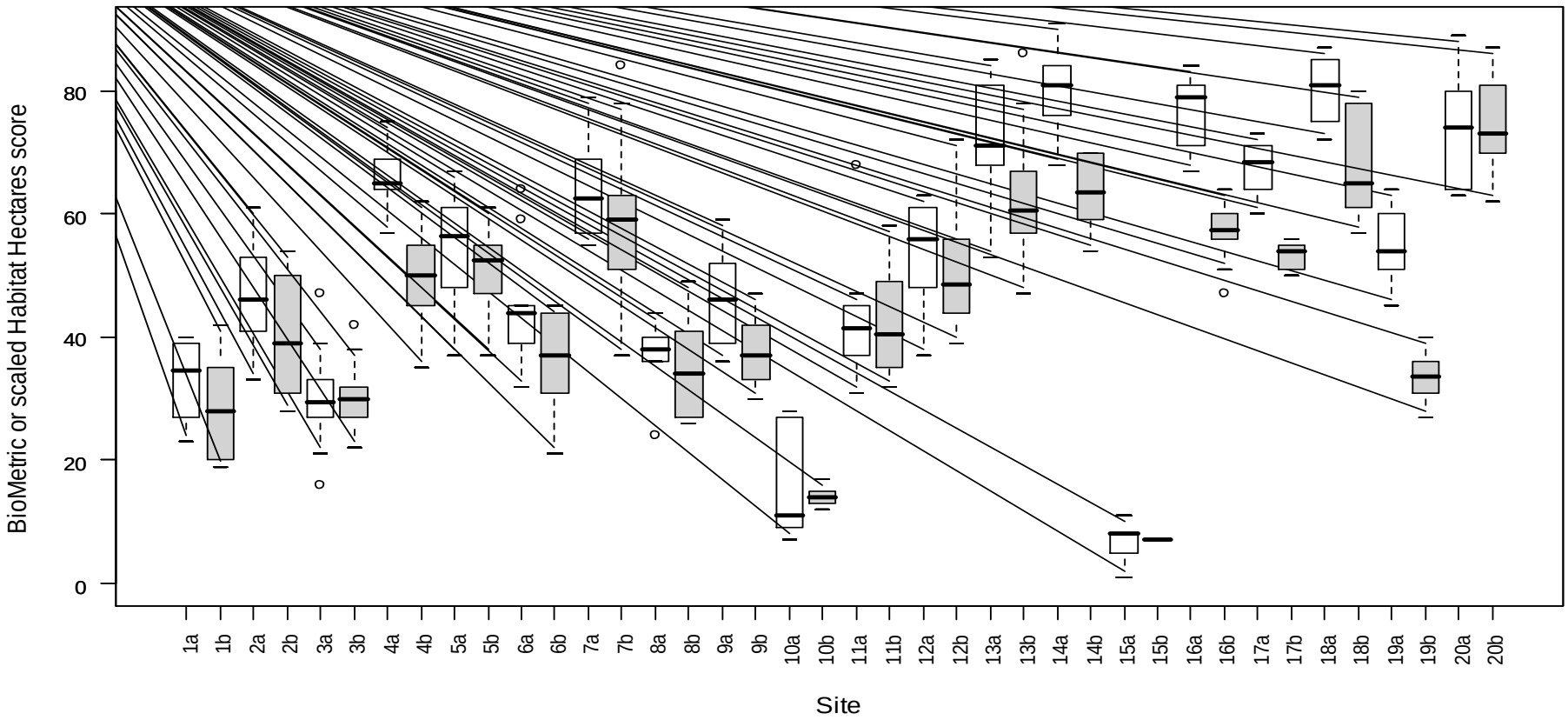
Figure 4. Standard deviations of total vegetation condition scores, as measured by ten observers using the BioMetric and Habitat Hectares methods.



Variation in total scores for both BioMetric and Habitat Hectares were not explained by single factors. In some instances, variation arose from small errors between observers in heavily weighted attributes (e.g. *hollow bearing trees* in Sites 2, 7, 12 and 13 for BioMetric and *understorey* in Sites 5, 7, 10 and 20 for Habitat Hectares). In other cases, one or two observers estimated attributes very differently for multiple moderately

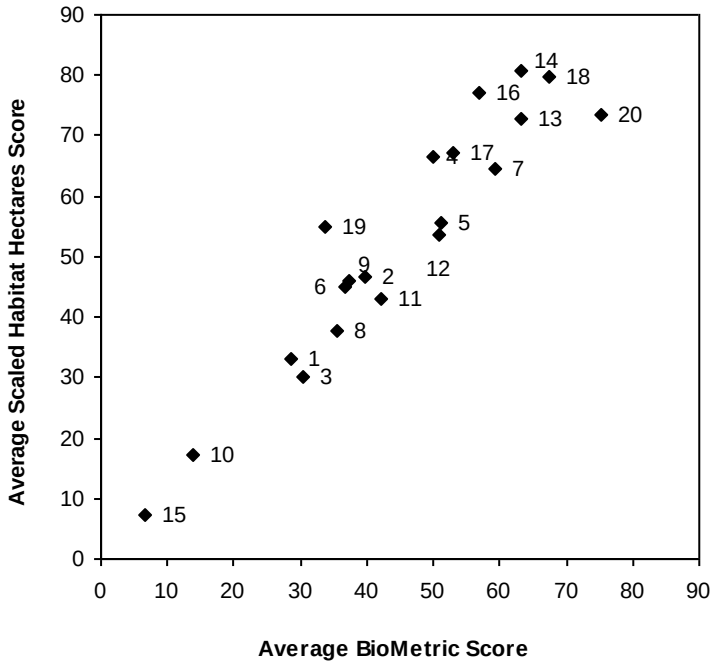
weighted attributes (e.g. Site 11 for Habitat Hectares). In most cases, observer estimates varied above and below the group mean over all attributes consistent with random independent errors unexplained by observer experience, attribute or the details of field conditions.

Figure 5. Boxplots of total vegetation condition scores for each site, as measured by ten observers using the Habitat Hectares (open boxplots) and BioMetric (grey boxplots) methods. Boxplots show median, quartiles and outliers.



Based on the average total score across observers for each site, the BioMetric and Habitat Hectares protocols ranked sites similarly, with a Spearman correlation coefficient of 0.91. The lowest ranking sites in both methods lacked both a tree canopy and a shrub stratum, and had a ground stratum dominated by weeds (Table 3, Figure 6). The sites with low to moderate scores were either planted or lacked a tree canopy or native shrub stratum. The sites with higher scores in both methods were all remnant vegetation, and the highest ranking sites were structurally complex remnant vegetation with all three major strata and relatively few weeds (Table 3, Figure 6). No sites scored above an average of 80 points using either method.

Figure 6. Average BioMetric and scaled Habitat Hectares scores across observers for all sites.



Patterns amongst observers

The average rank correlation of all pairs of observers was calculated for each metric; as was the average correlation of each individual observer with all other observers. The average rank correlation across all pairs of observers was 0.82 (range 0.57 - 0.94) for BioMetric and 0.91 (range 0.82 - 0.97) for Habitat Hectares (Table 4). There was no apparent cause for the particularly low correlation between Observers G and H for BioMetric (0.57), and their Habitat Hectares correlation was high (0.92). The lowest rank correlation of an individual observer with other observers was 0.74 for Observer G, and this was only for BioMetric. All observers agreed on the lowest ranking site using BioMetric (Site 15), while two different sites were ranked lowest by observers using Habitat Hectares (Sites 10 and 15). There was less agreement amongst observers as to the ranks of moderate to high ranking sites using both protocols, with between 4 and 9 sites at each rank (data not shown).

Table 4. Spearman rank correlation coefficients of pairs of observers for total scores of BioMetric and *Habitat Hectares*.

	A	B	C	D	E	F	G	H	I	J
B	0.93 *0.94*									
C	0.8 *0.9*	0.8 *0.9*								

	0	*2*	8	*1*															
D	0.92	*0.89*	0.91	*0.93*	0.88	*0.84*													
E	0.86	*0.95*	0.87	*0.94*	0.74	*0.88*	0.85	*0.91*											
F	0.94	*0.88*	0.94	*0.96*	0.79	*0.83*	0.89	*0.93*	0.86	*0.89*									
G	0.73	*0.96*	0.78	*0.92*	0.82	*0.82*	0.75	*0.90*	0.69	*0.97*	0.74	*0.90*							
H	0.89	*0.93*	0.80	*0.92*	0.74	*0.90*	0.87	*0.91*	0.88	*0.94*	0.81	*0.88*	0.57 #	*0.92*					
I	0.84	*0.93*	0.78	*0.90*	0.66	*0.92*	0.83	*0.85*	0.85	*0.93*	0.85	*0.84*	0.71	*0.89*	0.79	*0.93*			
J	0.85	*0.94*	0.81	*0.92*	0.82	*0.87*	0.86	*0.87*	0.86	*0.92*	0.80	*0.87*	0.84	*0.92*	0.86	*0.91*	0.81	*0.88*	
Observer Mean	**0.86**	***0.93***	**0.86**	***0.93***	**0.79**	***0.88***	**0.86**	***0.89***	**0.83**	***0.92***	**0.85**	***0.89***	**0.74**	***0.91***	**0.8**	***0.92***	**0.79**	***0.9***	**0.83**	***0.9***
Observer mean – Total mean	+0.04	*+0.02*	+0.04	*+0.02*	-0.03	*-0.03*	+0.04	*-0.02*	+0.01	*+0.01*	+0.03	*-0.02*	-0.08	*0.00*	-0.02	*+0.01*	-0.03	*-0.01*	+0.01	*-0.01*

#Not significantly different from zero, for two tailed probabilities at □= 0.01 (Zar 1972).

Only one observer's total site scores were, on average, consistently different from the group mean using both protocols: Observer B had slightly higher estimates (Figure 7). Each observer made the most extreme estimate (i.e. the absolute difference from the group mean, regardless of direction) on at least one site using BioMetric, except Observer A. Similarly, all observers made the most extreme estimate on at least one site using Habitat Hectares, except Observer E. The average absolute difference from the group mean was higher in Habitat Hectares than BioMetric.

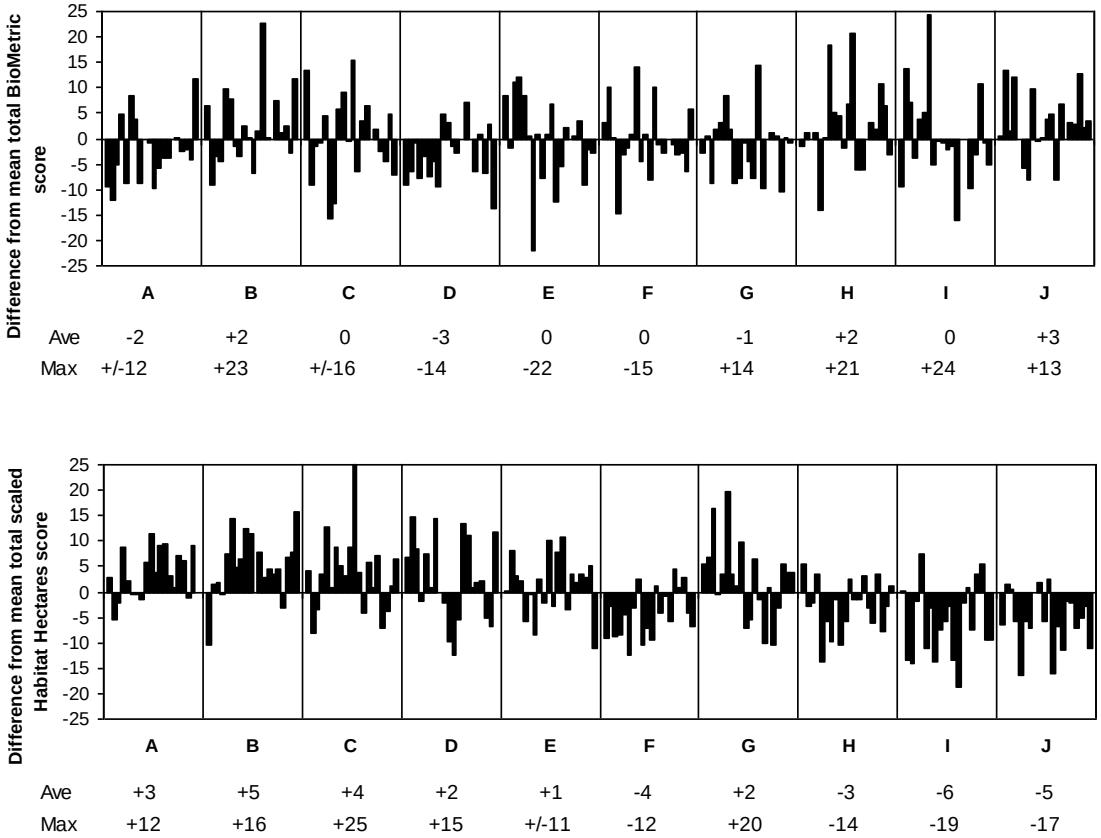


Figure 7. Difference of total a) BioMetric scores and b) scaled Habitat Hectares scores for Observers A-J from the group mean.

BioMetric and Habitat Hectares had quite different patterns of variation in estimates (Figure 7). Habitat Hectares had a larger proportion of variation between observers, with Observers A, B and C recording higher scores on average than the group mean and Observers H, I and J recording lower scores on average than the group mean (Figure 7). Using the BioMetric protocol, a greater proportion of variation occurred between sites within observers.

Discussion

Observer variation in vegetation condition scores

Observer variation in total scores was similar for BioMetric and Habitat Hectares, and does not appear to have been greatly affected by differences in the index structures or sampling methods. Variation in total scores was primarily caused by random observer variation in all vegetation attributes. The magnitude of variation detected (average CV of 15-18%) may have underestimated variation in real-world assessments as: observer variation in species richness was excluded; almost all observers were trained by the same person; all observers surveyed the exact same sites; and the order in which observers conducted assessments was non random.

There was general, but imperfect, agreement amongst observers about the rank order of sites. Most observers agreed as to the scores and ranks of very highly degraded (or poor condition) sites in which many attributes had zero values. At less degraded (moderate and high condition) sites, there was substantially more disagreement among observers as to site scores and ranks. Protocols for assessing vegetation condition therefore may not reliably distinguish between sites of moderate to high condition, though they should distinguish low condition sites with certainty.

Potential implications for decision making and biodiversity conservation

The results indicate that there may be considerable uncertainty as to which site is in best condition or whether any given site exceeds some threshold condition value for a management decision. In the context of offsetting decisions, underestimation of a development site and/or overestimation of an offset site may cause greater than expected loss of biodiversity. Suppose it is proposed to remove all vegetation from Site 14, and offset it with management actions on Site 4. Based on the average BioMetric scores of the two sites (50 and 63 respectively) and ignoring uncertainty in these estimates, management actions would be required to increase the value of Site 4 by 13 BioMetric points. However, the spread of observer scores about the mean values suggest that this decision could still lead to a loss of between 6 and 22 points (or alternatively a windfall of between 5 and 19 points) based on the highest and lowest estimated scores for these sites.

Dealing with observer variation: Increasing sample size

One possible method for dealing with observer variation in vegetation condition assessments for decision making would be to increase the sample size. Given observer errors, Block et al. (1987) calculated that sample sizes of greater than 75 may be required for precise point estimates for vegetation attributes. The results of this study indicate that ten observers may be insufficient to reliably estimate the biodiversity value of a site using two vegetation condition assessment protocols. Suppose decisions for offset assessment required vegetation condition to be estimated within 10% of the true value. Using Equation 2, the number of independent observers required would be up to 30 for both BioMetric and Habitat Hectares for most sites. Many hundreds of observers would be required for vegetation condition to be estimated within 1% of the true mean. Time and monetary constraints will therefore prevent the use of sample sizes sufficient to deal with observer variation.

$$\text{Number of observers required} = \frac{CV^2 \cdot t^2}{E^2} \qquad (2)$$

Where CV is the Coefficient of Variation, *t* is the standard value from the students t distribution and *E* is the standard error of the mean.

Dealing with observer variation: Improving precision

The scatter of extreme estimates across all observers suggested a tendency for any observer to make inaccurate estimates on occasion, and this was true regardless of the kind of attribute

measured or the condition of the vegetation. These results emphasise the opportunity to improve operator performance by enhancing measurement protocols.

While empirical evidence of observer error has long been reported in relevant literature, conclusive evidence of causative factors and potential remedial actions is surprisingly scarce. Potential causes of variability within individual observers may include: complexity of the vegetation (an observer may be more or less accurate in particular vegetation structures or the vegetation may be heterogeneous); survey conditions such as weather and time of day (which may cause fatigue); and survey techniques (which may be subjective). Variability amongst different observers may be caused by different levels of training in the use of a particular survey technique or familiarity with a given vegetation structure.

There is some evidence that precision of observer's estimates may be improved, but not eradicated, through use of small sampling units (Leps and Handicova 1992; McCune and Lesica 1992; Klimes 2003; Archaux et al. 2007), or more objective survey techniques (Brakenhielm and Qinghong 1995; Zhou et al. 1998; Ringvall et al. 2005). Training may also improve precision, although evidence is limited and equivocal (Smith 1944; Sykes et al. 1983). The extent to which precision of observers can be improved through smaller sampling units, improved survey techniques (including minimising fatigue and reducing any linguistic uncertainty in operating manuals) and regular training and calibration against standards needs to be further researched.

Dealing with observer variation: Modifying scoring structures

Categorical scoring structures may have exacerbated the expression of observer error for some attributes. Moderate levels of observer variation across scoring categories in heavily weighted attributes can have very large impacts on total score. An example is the sensitivity of BioMetric to tree hollows. The variation in number of hollow bearing trees detected on any given site was low (maximum range of 0 to 3), yet total BioMetric score can be altered up to 22 percentage points for this attribute. These sensitivities to weighting suggest that, while observer error appears to be the main driver of uncertainty in vegetation condition scores, the heavy weighting of some attributes can exacerbate its effects.

Incorporating uncertainty into decision making

Assuming observer error is unlikely to be eradicated, and few independent observers are likely to be able to conduct assessments, estimates of vegetation condition may always be uncertain. Use of available insights on the magnitude and direction of uncertainty in field assessments can be explicitly considered in the decision making process, and will usually reduce the risk of decisions that lead to poor conservation outcomes. New methods for quantifying uncertainty and incorporating it into decision making processes are emerging (e.g. Ben-Haim 2001; Burgman et al. 2001). Most involve specifying a best estimate and describing the uncertainty around that estimate in the form of an interval, a fuzzy number or a probability distribution. Statistical probabilities should be used with caution in the absence of adequate knowledge of the nature of uncertainty. Valuable information about uncertainty in field assessments of vegetation condition may be obtained simply by requiring individual observers to provide bounded estimates for all parameters. Additional information may be obtained by requiring multiple observers to independently conduct assessments if there are Occupational Health and Safety requirements to work in teams.

Regardless of the method used to describe uncertainty around best estimates, uncertainty should be propagated through the calculation of vegetation condition scores. Taking the example of offsetting clearing of Site 14 with gains on Site 14, a more robust offset that minimised the risk of loss of biodiversity may require the biodiversity value of Site 4 to be increased by 35 points based on the highest estimated score for Site 14 and the lowest estimate for Site 4. More sophisticated methods could potentially be incorporated into software tools used to assist decision making, such as the BioMetric Decision Support Tool (DEC 2005b). Decisions may then be made that are more robust to uncertainty (unlikely to deliver unexpected bad outcomes), and less likely to cause loss of biodiversity than if the decisions were made on best estimates alone.

Conclusion

This study has shown appreciable levels of uncertainty in field assessments of vegetation condition, which may cause difficulty distinguishing between moderate and high value sites. Broadly similar levels of uncertainty were recorded for two protocols for assessing vegetation condition, due to imprecision in estimates of all vegetation attributes and potentially exacerbated by the protocols' scoring structures. It is recommended that further research is conducted into methods for improving precision of observers' estimates through regular training and calibration and reducing fatigue effects. It is also recommended that observers provide bounded estimates for all parameters, and uncertainty is formally incorporated into management decisions for biodiversity conservation.

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Paper 2

Report Cover Page

ACERA Project
Round 2, Project 6 (Project Number 0706)
Title
Evaluating vegetation condition measures for cost-effective biodiversity investment planning
Author(s) / Address (es)
Atte Moilanen, Astrid van Teeffelen, Yakov Ben-Haim *and* Simon Ferrier
Material Type and Status (Internal draft, Final Technical or Project report, Manuscript, Manual, Software)
Project Report 1
Summary

The aim of this project is to explore the usefulness of vegetation condition indices to measure environmental and economic outcomes from biodiversity investments. This report documents stage 1 of the project, development of the theoretical framework for the analysis of tradeoffs. The framework outlined here uses decision theory to calculate a 'robustly fair' offset ratio, which guarantees a high probability of an exchange (an area offset or a biodiversity trade) producing at least as much conservation value in the offset areas than is lost from the target (development) site. The report analyzes how a fair offset is influenced by uncertainty in the effectiveness of restoration action, correlation between success of different compensation areas and time discounting.

This framework will form the basis of an analysis of a case study from the Cumberland Plain in Sydney. The case study will explore tradeoffs between biodiversity investment objectives and conventional economic objectives, and will evaluate tradeoffs for their robustness to uncertainty in the measures and the predicted response of the environment and economic returns. These analyses will be based on field data that quantifies uncertainties in the condition indices currently being implemented by state regulatory agencies, resulting in recommendations for methods that will improve the reliability of biodiversity and economic investments.

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How much compensation is enough? A framework for incorporating uncertainty and time discounting when calculating offset ratios for impacted habitat.

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Disclaimer

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1. Executive Summary

Biodiversity offset areas may compensate for ecological damage caused by human activity elsewhere. One way of determining the offset ratio, or the compensation area needed, is to divide the present per area unit conservation value of the development site by the predicted future conservation value of the compensation areas. Matching mean expected utility in this way is deficient because it ignores uncertainty and time lags in the growth of conservation value in compensation areas. Instead, we propose an uncertainty analytic framework for calculating what we call robustly fair offset ratios, which guarantee a high enough probability of the exchange producing at least as much conservation value in the offset areas than is lost from the development site. In particular, we analyze how the fair offset ratio is influenced by uncertainty in the effectiveness of restoration action, correlation between success of different compensation areas and time discounting. We find that very high offset ratios may be needed to guarantee a robustly fair exchange, compared to matching mean expected utilities. These results demonstrate that considerations of uncertainty, correlated success/failure, and time discounting should be included in the determination of the offset ratio, to avoid a significant risk that the exchange is unfavorable for conservation in the long run. This is essentially because the immediate loss is certain whereas future gain is uncertain. The proposed framework is also applicable to the case when offset areas already hold conservation value and do not require restoration action, in which case uncertainty about the conservation outcome will be lower.

2. Introduction

Several countries have adopted policy to regulate the impact of economic development on natural habitats. After estimating the expected damage that a particular development project will do to existing habitat and associated species, a hierarchy of measures can be employed to alleviate the impact (Cuperus et al. 2001; ten Kate et al. 2004). The first step in this hierarchy aims at avoidance of the impact, for example by looking for alternative locations for development, where impact will be less severe. Once the development location is decided, the second step concerns minimizing the impact. In the European context this step is often referred to as mitigation, whereas in North America the term mitigation often refers to the third step, the use of compensation measures for unavoidable damage to natural areas (Race & Fonseca 1996; ten Kate et al. 2004). Here, we use the term biodiversity offsets to indicate ecological compensation for unavoidable damage.

Biodiversity offsets involve the designation of compensation areas, which either hold significant conservation value already, or where habitat creation, re-creation or restoration practices are carried out in order to balance for biodiversity loss elsewhere. Typically, loss would be caused by direct anthropogenic action (urban expansion, forestry etc.), but offsets could also be used to compensate for the slow degradation of biodiversity from present reserve areas (Sinclair et al. 1995). As ten Kate et al. (2004) emphasize in their review, quantitative guidelines for determining offset ratios and types are generally lacking. Typically, rules of thumb are used to describe offset requirements, in terms of the location and the habitat type; compensation areas near the development site are preferred over sites further away, and compensation in habitat similar to the habitat at the development site is preferred over other types of habitat. Although the size of the affected areas is a quantitative measure, determining the conservation value of habitat remains difficult (ten Kate et al. 2004).

A similar concept, No Net Loss (NNL), has been developed for wetlands under the Fisheries Act in Canada, and the Clean Water Act in the United States. Under these regulations permits for development often require offsets to compensate for damaged wetlands. Harper and Quigley evaluate this approach for Canada (Harper & Quigley 2005; Quigley & Harper 2006a, b). Quigley & Harper (2006b) report that although compensation requirements did determine required offset ratios to be on average 6.8:1 (area gained : area lost), the mean offset ratio that was actually implemented was only 1.5:1, resulting in 10 out of 16 cases not reaching NNL in terms of habitat productivity. Poor compliance to offset agreements was also found a problem in Australia by Gibbons & Lindenmayer (2007). The principle of no net loss is similar to the concept of strong sustainability in capital theory, which requires that each form of capital, such as conservation value, is kept constant (Cowdy & Carbonell, 1999; Figge & Hahn, 2004). A related concept, weak sustainability, allows that different forms of capital can be substituted for each other (Figge & Hahn, 2004).

Habitat banking and Habitat Equivalency Analysis (HEA) are yet another two concepts used in the context of habitat compensation measures. Habitat banking, also referred to as ‘mitigation banking’ or ‘conservation banking’, aims at conservation practices which generate ‘biodiversity credits’ that can be traded for later habitat destruction elsewhere by development practices (Bruggeman et al. 2005; Morris et al. 2006). An explicit feature of banking is that credits are generated before damage is undertaken. In contrast, with offsets damage and credits are generated at best simultaneously. Due to inevitable

delays in the growth of conservation value at restoration areas, credits can be realized after a substantial time-delay (Morris et al. 2006).

Habitat Equivalency Analysis aims to compensate injured natural resources, and has in particular been applied to coastal and marine habitats (National Oceanic and Atmospheric Administration 2000). Although HEA is widely applied in practice (particularly in USA), very little has been published in peer-reviewed literature (Race & Fonseca 1996; Dunford et al. 2004). HEA involves quantitative measures to determine the amount of compensation required, potentially accounting for time delays in the process. Dunford et al. (2004) provide a thorough demonstration of the use of HEA in the context of oil spills. Framed in the context of conservation banking, Bruggeman et al. (2005) extended the concept of HEA to terrestrial habitats, and coined the term Landscape Equivalency Analysis (LEA). They incorporate spatial and population genetic aspects quantitatively into the valuation of habitats and species.

In this study we are interested in determining the offset ratio needed to achieve a fair exchange of areas. Fair could be defined in many ways. It could, at simplest, be defined as an exchange where the mean expected long-term value of the compensation areas equals the estimated value lost from the development site. We refer to this criterion as ‘matching mean expected utilities’; utility that is gained (eventually) from the compensation areas is estimated to exactly compensate for the immediate loss of utility from the development site. This criterion is deficient in that it ignores, for example, the time lag before the full value of compensation areas is realized, as well as uncertainty in the extent to which the expected conservation value at the compensation areas will be realized (Hilderbrand et al. 2005). Our criterion for a fair exchange accounts for uncertainty. We seek that the compensation areas together should have a less than *x*% (e.g., 5%) chance of producing less conservation value than is lost from the development site(s). As will be seen, this criterion is very different from the matching of expected utilities. The difference becomes apparent when (restoration) success of compensation areas cannot be taken for granted.

In this study, we investigate at a theoretical level what influence the following components have on the estimate of a fair offset ratio: 1) uncertainty in the effectiveness of restoration action and the growth of conservation value at compensation areas; 2) correlation between success of different compensation areas, and 3) time discounting. We develop a framework for the calculation of robustly fair offsets. Using a mathematically simple example, we demonstrate that our assumptions make a huge difference for the amount of compensation that should be perceived as adequate.

3. Methodology

3.1 The conceptual framework of robustly fair offsets

Our framework assumes that the goal of offsetting is no net loss, but not in the sense that present loss is exactly compensated by mean expected future gain. Rather, we intend that present loss is compensated by future gains, accounting for uncertainty and time lags in the development of these gains. We specify that the probability of incurring net loss must be small, thereby ensuring what we call ‘robustly fair offsets’. The framework we propose accounts for uncertainty in the development of conservation value, and time discounting. The uncertainty is a critical component when the aim is to avoid net loss due to unfavorable growth of conservation value at restoration areas.

We assume three components of uncertainty. (i) Future value could be less than estimated, which could, for example, represent the case that an area of forest develops fewer nesting holes than expected, or that forest understory develops a community which is less species-rich than expected. (ii) Some feature of conservation value might completely fail to be established, e.g. a focal species fail may fail to colonize the area. (iii) We also allow for the possibility that success and failure could be correlated between different restoration areas. The uncertainties in our analysis are most relevant where restoration action is applied at compensation areas. However, the proposed framework is equally applicable when compensation areas are such that they already hold substantial conservation value and some form of protection is applied rather than restoration action. In this case uncertainties are smaller (or even zero), but the structure of the proposed calculations need not be changed.

We account for uncertainty by adopting a decision theoretic approach to the calculation of offsets. If statistical models are available for the components above, one could use a statistical approach for identifying an offset ratio, which has, for example, a less than 5% chance of resulting in net loss. However, our formulation includes parameters, such as long-term success of restoration effort, for which it may be difficult to obtain reliable distributional information. In such a case, information-gap decision theory (Ben-Haim 2006), which we employ here, provides a straightforward way of analyzing the influence of uncertainty on the offset ratio.

Time discounting (Carpenter et al. 2007) of the offset ratio is included because it is not fair to compensate immediate loss by hypothetical distant-future gain, and thus omitting time discounting could place overall nature conservation efforts at a disadvantage. Presumably, the conversion of the development site would produce a relatively immediate economic return in the order of some percents per year. This revenue could plausibly be used for further environmentally harmful activity either directly or indirectly. On the other hand, conservation benefits arising from restoration effort may take a very long time to materialize fully, for example, if one needs to wait for forest to grow. Consequently, we find it reasonable that the offset ratio should be calculated as a time discounted weighted average across the planning frame – loss of conservation value from the development site is immediate but habitat restoration may produce expected conservation gains with a time delay of decades.

These components have been noted in prior work: The outcome of restoration is often different from expected, due to e.g. existence of alternative equilibria and differences in ecological dynamics between degraded and less-impacted systems (Suding et al. 2004; Hilderbrand et al. 2005). HEA explicitly includes time-discounting as an option (Dunford

et al. 2004; Bruggeman et al. 2005). Morris et al. (2006) and Roach and Wade (2006) both mention that there is a time lag between impact and compensation, although they do not present methods that explicitly take that into account in analysis. Several authors also note that there is uncertainty associated with the expected outcome of restoration (Cuperus et al. 2001; Bruggeman et al. 2005; Morris et al. 2006; Gibbons and Lindenmayer 2007), but did not explicitly account for it in analyses. Keagy et al. (2005) investigate the feasibility of compensation for maintaining overall population abundance in the study area, when the compensation areas are of inferior quality compared to the lost habitat. Gibbons and Lindenmayer (2007) summarize that offsets will only contribute to no net loss if clearing is restricted to vegetation that is simplified enough so that its functions can be restored elsewhere, if any temporary loss in habitat between clearing and the maturation of an offset does not represent significant risk to a species, population or ecosystem process, and if offsets are substantive enough and they are complied to. Here we combine all these factors together into the same quantitative theoretical analysis.

3.2 Evaluating offset solutions using an uncertainty-analytic approach

We use info-gap decision theory (Ben-Haim 2006) to analyze the consequences of uncertainty to the fair offset ratio. The main components of the info-gap theory are the goal (performance aspiration), the performance function, the nominal model, the uncertainty model, and the robustness function.

Our goal is to robustly achieve no net loss. The nominal model is our best estimate for the expected conservation value in the development area and compensation areas (thick lines in Fig. 1). We indicate nominal models by $\tilde{V}_0(t)$ and $\tilde{V}_i(t)$, for conservation value at time *t* at the development area and compensation area *i*, respectively. The nominal model represents our best understanding of how conservation value will change in these areas over time. However, this information may be quite uncertain, especially if the expectation of conservation value is based on expert opinion about likely future success of restoration action. Here, the info-gap theory utilizes another central component, the uncertainty model (thin lines in Fig. 1).

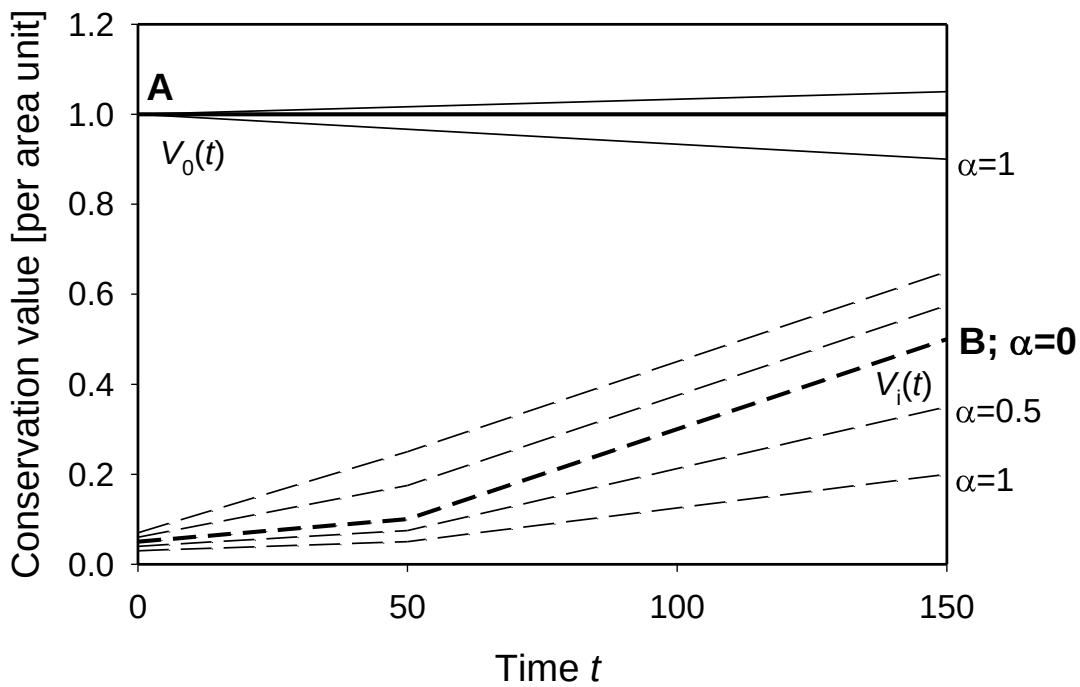

Figure 1. The assumed per area-unit change in conservation value at the development area (thick solid line) and at the restoration areas (thick dashed line). Thin lines represent uncertainty bounds around these estimates; the relative uncertainty about the growth of conservation value at the restoration area is in our example higher compared to uncertainty about maintenance of value at the development site. The width of the uncertainty bounds would depend on the info-gap horizon of uncertainty parameter, α. When α is zero, the estimate (thick line) is taken as certain. With increasing α, the range of values possible for conservation value at any point in time widens. Points A and B are used when calculating a naïve offset ratio based on mean expected value. Note that the conservation value of the development site is our estimate of what it would have if it was not developed. We assume that as a consequence of development, all conservation value is lost.

The info-gap uncertainty model does not simply place bounds around the nominal estimate, as it might appear from Fig. (1). Rather, the robustness of solution candidates are analyzed in terms of an uncertainty parameter, the horizon of uncertainty α. When this parameter is zero, it indicates full confidence in our nominal model and the nominal model is accepted as the true model. Higher values for α indicate less confidence in the nominal model: the true model could be somewhere within an expanding bound around the nominal model. In our example of Fig. (1), the uncertainty model is represented by the thin lines around the nominal model. When α=0, the thick line is taken as the truth, and increasing α implies expanding bounds of possible outcome. Importantly, different areas and restoration actions could have different nominal estimates as well as different levels of uncertainty (error weights). For example, smallest error weights could be associated with a presently high-quality area that has been well surveyed. A relatively higher error would go for an area that is apparently valuable but is poorly surveyed. Highest error weights would be associated with areas where there is substantial lack of knowledge concerning the growth of conservation value there, for example as a consequence of trying out a completely new restoration technique. Technically, when evaluating a solution at any given level of α, the solution is evaluated according to the most adverse choice of the model inside the uncertainty bounds. However, since the horizon of uncertainty, α, is unknown, a solution is evaluated according to the greatest horizon of uncertainty up to which that solution yields adequate outcomes.

The aim of our uncertainty analytic approach is to identify solutions that are robust in the sense that they achieve our performance aspiration even when allowing for high uncertainty. In the typical info-gap formulation, the robustness of a solution, α^*, is the highest α at which it is guaranteed to meet the performance target (Fig. 2a). A solution is not robust if it may fail to achieve the goal even at low α, indicating that a small deviation from expected restoration outcome might cause the target of no net loss to be missed.

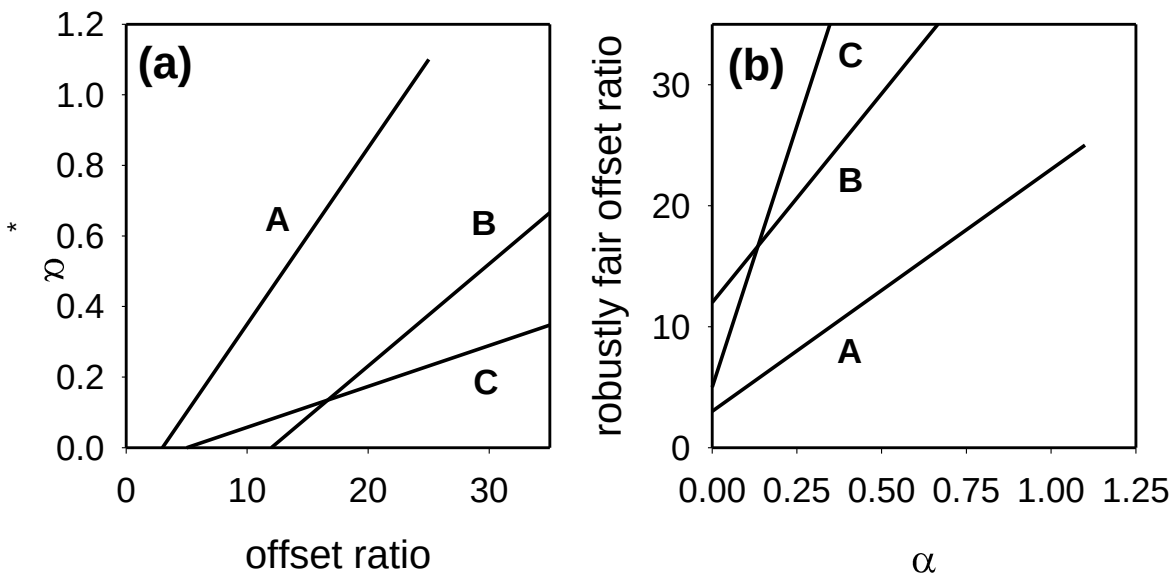

Figure 2. A schematic of how offset solutions would be compared in the info-gap approach. Each line is for one candidate solution, when uncertainty, α, increases. Of the three candidates, solution **A** is always best as it produces highest conservation value. Candidate **C** is better than **B** with low uncertainty, but with high uncertainty **B** is better. Preference between **B** and **C** would depend of the level of confidence required for the solution. These curves can be graphed in two alternative ways. Panel (a) is the typical info-gap representation, in which solutions are graphed in terms of the level of uncertainty they can allow while still guaranteeing the performance goal (no net loss with reliability). Panel (b) is the representation we use here, the offset ratio needed to guarantee no net loss at given level of uncertainty.

Each offset candidate solution would be examined in terms of its performance under increasing uncertainty. This is illustrated in figure 2. Assuming that offset candidates **A**, **B** and **C** have equal cost, then **A** is best option because it achieves goals while allowing for highest uncertainty (Fig. 2a). Candidate **C** is second best option assuming nominal models are correct. However, candidate **B** is more robust to increasing uncertainty than **C**.

The robust-optimal solution is the one solution that achieves goals while allowing for highest possible errors in nominal models. If only a few scenarios need to be compared, then solution performance and robustness can be evaluated for all candidates. If however the robust optimal solution needs to be identified from a large set of options (such as select 100 sites out of 1000 options), then some optimization method is needed. Below, we do not graph solutions in the traditional info-gap way. Rather, we turn the axes around and calculate the offset ratio that is sufficient for guaranteeing no net loss while accounting for the modeled uncertainties (Fig. 2b).

3.3 A simple example of the framework

We illustrate the proposed framework for the simple case where one area unit of land with relatively high conservation value is offset by a number of units of less valuable land that is restored. In this example, conservation value is treated as a one-dimensional construct. Table 1 gives a summary of symbols used in equations.

Table 1. Explanation of symbols used.

t_p	length of planning period
β	reliability requirement, the probability of net loss should be < (1-β)
p	failure probability of restoration action at an area
ρ	correlation coefficient for failure of restoration action between areas
d	time discounting rate
α	info-gap robustness parameter, horizon of uncertainty
$\tilde{v}_0(t)$	best estimate for per area unit conservation value of the development site at time t [per area unit]
$\tilde{v}_i(t)$	best estimate for per area unit value of compensation area option i at time t
$w_0(t)$	size of error envelope (weight) of $\tilde{v}_0(t)$
$w_i(t)$	error weight of $\tilde{v}_i(t)$; with restoration $w_i(t) >> w_0(t)$
$N_{method}(\alpha,t)$	number of equal-sized offset areas needed according to an offset calculation using the method indicated by subscript, N_{simple}, N_{IG} ,N_{prob}, N_{corr} and $N_{discounted}$, for Eqns. (1), (2), (3), (5) and (7), respectively. This quantity depends on both α and t via Eq. 2.

Assuming that all conservation value of the high-quality development area will be lost following the land exchange, a naive solution using matching of mean expected utility for the offset ratio is

$$N_{simple} = \frac{\tilde{V}_0(0)}{\tilde{V}_i(t_p)}, \qquad (1)$$

where $\tilde{V}_0(0)$ is the best estimate for the conservation value of the development area presently (at time 0), and $\tilde{V}_i(t_p)$ is the best estimate for the final conservation value of the restoration area at the end of the planning period at time t_p. This is the ratio A/B in Fig. (1). N_{simple} units of restoration land are eventually predicted to hold the same conservation value as the development area.

We extend this solution to consider two sources of uncertainty: 1) that the conservation value achieved at restoration areas could be less than expected; and 2) that the

conservation value of the development area could be even better than is thought. At simplest, to calculate the robustly fair offset ratio, $N_{IG}(\alpha, t)$, using the info-gap formulation only requires that $\tilde{V}_0(t)$ is replaced by $\tilde{V}_0(t)+\alpha w_0(t)$, and $\tilde{V}_i(t)$ by $\tilde{V}_i(t)-\alpha w_i(t)$ in Eq. (1):

$$N_{IG}(\alpha,t)=\frac{\tilde{V}_0(t)+\alpha w_0(t)}{\tilde{V}_i(t)-\alpha w_i(t)} \qquad (2)$$

Here, $w_0(t)$ and $w_i(t)$ are relative error weights for conservation value at the development area and compensation areas at time t in the future. For instance, these envelope functions may derive from the spread in expert opinion. Since other experts may have yet other opinions, or differently framed questions may elicit different expert responses, the uncertainty envelopes are multiplied by the unknown horizon of uncertainty, α. In our example $w_0(t)$ and $w_i(t)$ were calculated as the difference between the nominal estimate and the hypothetical error bounds of Fig. (1), indicating that at α=1 the uncertainty envelope has expanded to the outer thin lines.

We next allow for the possibility that conservation action in any one land unit could also fail altogether with a probability p. It is then logical to require that the even exchange would be achieved with a given reliability level β, say β=0.95. The number of area units where conservation action would succeed, N_S, is now distributed binomially as N_S~Bin(N, p). To satisfy the reliability requirement, we need Prob[N_S< $N_{IG}(\alpha, t)$] <(1-β). Denoting by $N_{prob}(\alpha,t)$ the minimum number of area units needed, this number can be determined by finding smallest $N_{prob}(\alpha, t) > N_{IG}(\alpha, t)$ for which

$$\sum_{k=0}^{N_{IG}(\alpha,t)-1}\binom{N_{prob}(\alpha,t)}{k}(1-p)^{N_{prob}(\alpha,t)-k}p^k<(1-\beta). \qquad (3)$$

Eq. (3) assumes statistical independence in success of restoration effort between different sites when calculating $N_{prob}(\alpha, t)$. The assumption of independence is a strong one, and in general restoration success between distinct restoration sites would be correlated to some degree (Fig. 3 illustrates effects of correlation). Ovaskainen and Hanski (2003) give a formula for the effective number of independent units, N_{eff}, when there is an uniform level of pairwise correlation, ρ, between N_{corr} sites,

$$N_{eff}=\frac{N_{corr}}{1+\rho(N_{corr}-1)}. \qquad (4)$$

This equation essentially states, that if the correlation is ρ, then there can at most be 1/ρ independent units irrespective of how many sites there are. Note that Eq. (4) ignores higher-order correlations, but even so, it provides useful insight into the influence of correlation on the fair offset ratio.

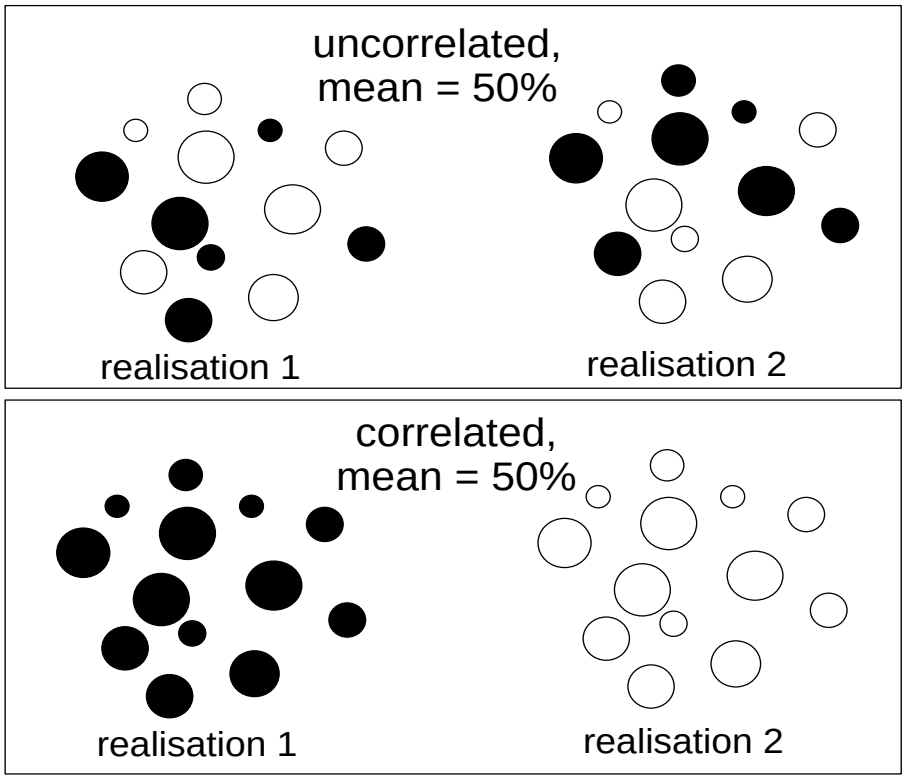

Figure 3. Illustrating effects of correlation. In both the uncorrelated and correlated cases the a-priori chance of restoration success is 50% per site, but the realized patterns are very different. Black and empty circles indicate sites with restoration success and failure, respectively.

Assuming N_{corr} correlated sites, we have only N_{eff} effective independent units, each of average size $S=N_{corr}/N_{eff}$. We then require that unit-size times the minimum number of units that succeed with reliability greater than β must be greater than $N_{IG}(\alpha, t)$. The number of effective units where conservation action would succeed, N_S, is now distributed N_S~Bin(N_{eff}, p). To satisfy the reliability requirement, we need Prob[$SN_S < N_{IG}(\alpha, t)$] <(1-β). The minimum number of real units needed for this relation to be true can be determined numerically by finding smallest $N_{corr}(\alpha, t)$, for which

$$\frac{N_{corr}(\alpha,t)}{N_{eff}} N_{\min} > N_{IG}(\alpha,t), \qquad (5)$$

where N_{eff} comes from Eq. (4) and N_{min} is the smallest number of units (out of N_{eff}) that succeed with a probability of at least β. N_{min} can be determined by inspecting the tail of the binomial distribution for the effective number of successful independent units. It is the largest number satisfying that out of N_{eff} units at maximum N_{min}-1 can fail with probability (1- β) or less, which implies that N_{min} or more units will succeed with probability greater than β:

$$\sum_{k=0}^{N_{\min}-1} \binom{N_{eff}}{k} (1-p)^{Neff-k} p^k < (1-\beta). \qquad (6)$$

Note that Eq. (6) will not be satisfied under all conditions. For example, with ρ=0.25 there can be at most four effective independent units. Then, if the failure probability of a unit is

0.5, a 95% reliability can never be achieved as $0.5^4 = 0.0625 > (1-0.95)$ meaning that the chance of all units failing is greater than the 5% allowed.

We add one final component, time discounting, to our analysis. A time discounted offset ratio can be obtained simply as

$$N_{discounted}(\alpha,t) = \frac{\sum_{t=0}^{t_p}(1-d)^t N_{method}(\alpha,t)}{\sum_{t=0}^{t_p}(1-d)^t}, \qquad (7)$$

in which d is the time discounting coefficient and $N_{method}(\alpha,t)$ represents any of the offset ratios from equations (1), (2), (3), or (5), where the offset calculations have been done at time t using given horizon of uncertainty α. For practical purposes this means that the offset ratio is weighted most heavily by the early years where the quality of the restoration areas is worst.

4. Results

We use our simple model to analyze the effects of uncertainty, correlation and time discounting on the offset ratio. In our example matching of mean expected utilities gives N_{simple}=2, implying that an exchange could indeed be feasible – i.e. by restoring an area twice the size of that lost to development. Figure (4) shows the effects of info-gap uncertainty analysis on the offset ratio (solid line). With α=0, the ratio $N_{IG}(\alpha, t_p)=N_{simple}$, but when α increases, the ratio increases substantially. In the present case, $N_{IG}(1, t_p)$=1.05/0.2=5.25. Hence, accounting for uncertainty in the growth of conservation value makes a large difference to the offset ratio.

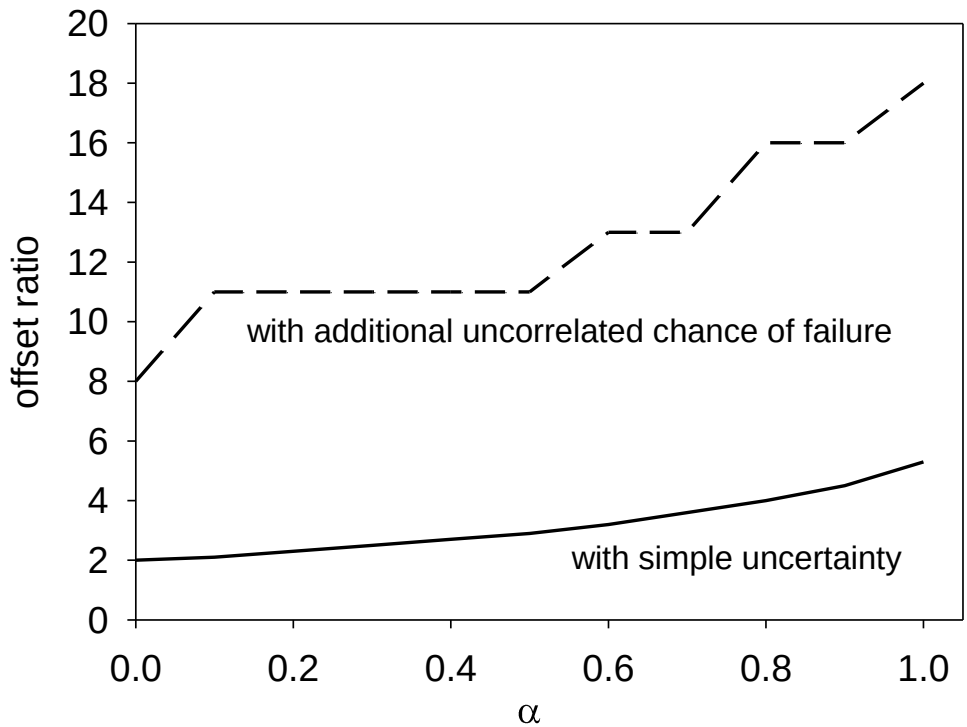

Figure 4. Offset ratio required to get "an even exchange" when exchanging one area unit of high conservation value development area to initially poor-quality restoration compensation areas. The solid line shows the ratio with simple effects of uncertainty ($N_{IG}(\alpha,t)$, with $t=t_p$; Eq. 2) and the dashed line the respective result assuming there is an additional uncorrelated per area-unit chance of complete failure of restoration activity (N_{prob} assuming p=0.5; Eq. 3). (Steps in the dashed line are due to rounding down to integer values when calculating the number of areas needed.)

So far we have assumed that each restoration area will produce (with certainty) at least some compensation, but we have allowed for the possibility that this compensation may be less than expected. Next we allow for the possibility that restoration fails completely in some of the restoration areas, for example, because the most important focal species fail to migrate/establish there (Suding et al. 2004). To explore the effect of this additional factor, we now assume that each area has a 0.5 possibility of complete failure. Thus p=0.5 in eqs.(3) and (6). The number of restoration area units needed for getting the conservation value of the development site with 95% reliability is given by the dashed line in Fig. (4). At α=0, the ratio becomes 1:8 and at α=1 the ratio becomes 1:18. Assuming uncertainty in the establishment of conservation value at restoration areas has

thus changed our perception of the number of area units needed from 2 to 18. Note that with 18 units the expected utility is 18*0.5*0.5=4.5, where the halves account for predicted restoration value and the chance of failure. In fact, the expected utility is one quarter of the number of restoration area units in all of our subsequent analyses.

With time discounting, the offset ratio is calculated as a time-discounted weighted average for the period t=1,2, ..., 150 (Eq. 7). The solid lines in Figure (5) show the offset ratios we arrive at now (assuming 50% chance of failure per area unit and a 95% reliability requirement). With 1%, 3% and 5% time discounting coefficients the α=1 offset ratios are now 1:59, 1:82 and 1:95, respectively. Even using no time discounting (0%) but calculating the ratio as an average over the 150 year planning horizon gives a ratio of 1:45 for α=1.

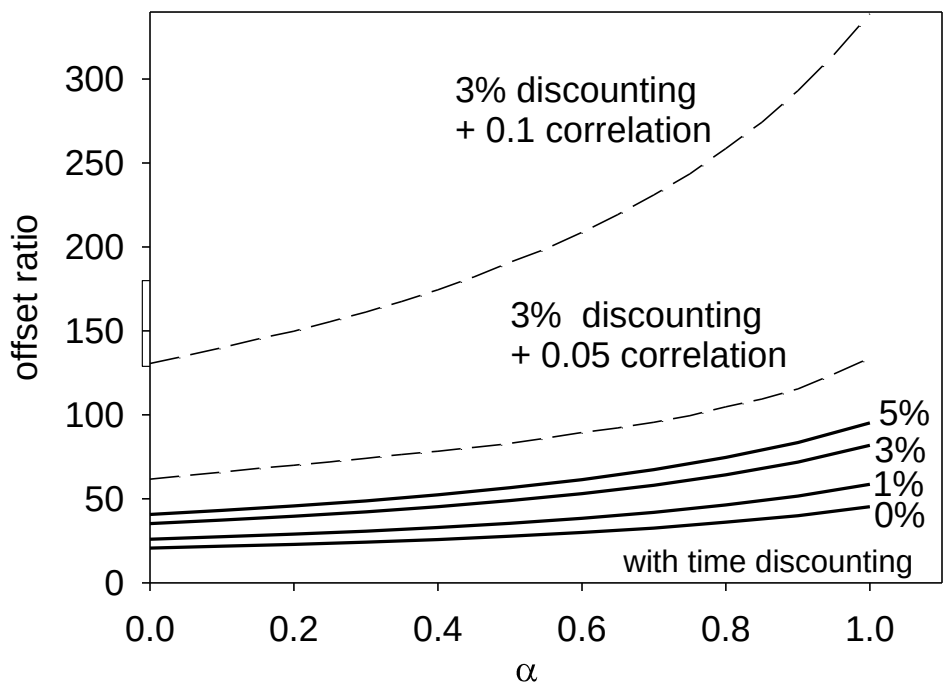

Figure 5. The robustly fair offset ratio when assuming time discounting on top of the uncorrelated chance of failure (solid lines; Eq. 7 applied on N_{prob}; cf. dashed line in Fig. 2). Offset ratio when adding a further 5% or 10% correlation on top of 3% time discounting (dashed lines; Eq. 7 applied on N_{corr}).

We have left for last the hardest factor in our analysis, i.e. correlation (dashed lines in Fig. 5). If the restoration success of individual sites is strongly correlated with the restoration success at other sites, then either action succeeds in (almost) all sites or fails simultaneously in all sites. Notably, with strong correlation, increasing the number of restoration sites does not notably decrease the probability of complete failure. Figure (5) shows the major influence of correlation on the offset ratio. For example, adding a minor 10% correlation increases the fair offset ratio from ~80 to ~340 when assuming 3% yearly time discounting.

5. Discussion

Using different assumptions, our estimate of the fair offset ratio increases quickly from two to hundreds in our simple example. This potentially surprising result is due to the criterion on which we have based our analyses. Instead of using the mean expected value of the restoration areas to determine the offset ratio, we look at the possibility of the proposed exchange turning out to be less than even. These criteria are completely different. The mean-expected-value criterion is based on the assumption that conservation value of restoration sites grows as expected. The criterion studied here takes an uncertainty analytic approach and looks at the possibility of restoration sites failing to deliver at least the same conservation value that is lost from the development site after the exchange. It is quite possible that, while a proposed exchange promises high expected conservation value, it at the same time has a high likelihood of (almost) complete failure. This would be the case, for example, when a large area of similar habitat is restored using a single method, which is not guaranteed to work. In this case the mean expectation for the conservation value of the restoration areas is high (because the area is large), but the probability of correlated failure across the entire region is large as well (because the effectiveness of the restoration action is not guaranteed).

The influence of time discounting is large as well. This reflects likely delays in the realization of conservation value due, for example, to the slow growth of trees. In fact, if the improvement of conservation value is slow enough, it is questionable whether the habitat should be considered restorable at all (Morris et al. 2006). Still, correlation in restoration success between different areas is the factor that has by far the greatest influence on the offset ratio in our analysis.

Is correlation, of the type we have simulated here, likely to be found in real-world planning situations? We believe so. Correlation in restoration success will be increased by (i) uniform habitat quality and environmental conditions across the restoration sites, (ii) the same restoration action being applied across all areas and (iii) physical proximity of restoration sites. All these conditions apply commonly in the real world. We would expect an effective absence of correlation only if different restoration actions are applied in different habitat types occurring in different regions. Substantial heterogeneity of habitat quality across the restoration sites would probably also decrease correlations substantially. However, if restoration areas are close to each other, some level of correlation is likely to be present. This is because according to the basic principles of spatial population ecology (see e.g. Hanski 1998), dispersal and establishment of species into the area will depend on the distance to nearby source areas and on the quality and species composition of these source areas (Donald & Evans 2006). If the restoration sites effectively share the same source areas, then it can be expected that a similar set of species will eventually colonize the restoration areas. Or, if sources are far away, some species of conservation value might fail to reach any of the restoration sites (Bakker et al. 2000). If the quality of the restoration areas becomes suitable for the species only after a lengthy maturation of vegetation, then it is possible that nearby population sources will disappear before the restoration areas become sufficiently suitable to allow colonization. This could be the case, for example, for species that require trees to mature enough to form nesting holes. Correlated failure can of course be avoided by selecting offset areas that already hold reasonable conservation value, and therefore require protection rather than restoration.

In summary, when calculating offsets one should recognize that loss is immediate but gain is uncertain and may not be achieved for a long time into the future. Accounting for uncertainty in offset calculations, and aiming at offsets that robustly avoid net loss, may suggest much higher offset ratios than recommended by matching of mean expected utilities. To obtain a reliably good offset solution, one should employ a bet-hedging strategy, where presently valuable offset areas are preferred, and restoration effort is split among an anti-correlated, or at least uncorrelated, set of sites – i.e. where different restoration actions are applied across environmentally different, and spatially dispersed, sites.

The present analysis is only a first theoretical step and it provides a starting point for further methodological development concerning the calculation of robustly fair offset ratios. For example, we have treated conservation value as an aggregate property of a site, whereas in reality one might wish to have separate estimates for a set of different species or biodiversity features. Then the objective would be to obtain a satisfactory outcome for a broad range of biodiversity features simultaneously, accounting for complementarity, retention of the features in the landscape, and certainty of species' occurrences in sites. There are alternatives for how offsetting would be done across many features. One safe alternative is to require that the offset is robustly fair for all features simultaneously, which may result in very large offset ratios. Search for an optimal offsetting solution under this assumption implies a strategy, which is analogous to target-based reserve selection (Margules & Pressey 2000) accounting for retention (Pressey et al. 2004; Moilanen & Cabeza 2007) in the landscape. An alternative is to require that summed conservation value across species does not decline, implying that a reduction for one feature may be compensated via increased representation for other features. This approach would allow much flexibility for offsetting, which has potential for both success and misuse. Search for an optimal offsetting solution under this assumption implies a strategy that is analogous to an additive benefit function approach (Arponen et al., 2005; Moilanen 2007).

Also, our analysis does not cover the involved mathematical details of how to handle partial correlation in restoration success between restoration options. We have assumed area units of equal size and cost. Uncertainty could be relevant for many other components of our model, such as the failure probability or correlation, instead of just the development of conservation value at compensation areas. We have also ignored questions of connectivity, spatial population dynamics and questions of persistence. Solving offset calculations involving such complications will allow for increasingly robust and realistic allocation of habitat restoration effort. Despite these complications, the basic message of the present work stands: guaranteeing no net loss assuming uncertainty and time delays suggests offset ratios that are far greater than what is needed if calculations are based on a mean expectation of restoration success.

5.1 Implications for practice

- Uncertainty in effectiveness of restoration action should be accounted for when calculating offsets, otherwise a long-term net loss for conservation is likely.
- Time discounting of conservation value, with a rate comparable to the economic return expected from the development site, should be used in offset calculation when conservation value grows slowly in the compensation areas.

- If the same restoration action is applied to a set of environmentally similar sites that are either close to each other, or combine to effectively form one or more larger compensation areas, then success of restoration action is likely to be highly correlated between sites. Correlation may lead to all-around unsatisfactory outcome of restoration effort and consequent net loss of conservation value. In this light, having a large area of compensation sites does not alone guarantee that a net loss could not occur, which could be accounted for when negotiating offsets.
- From an uncertainty-analytic view, the safest offset solution consists of a set of different areas that are treated in variable ways, catering for the needs of partially different groups of species. An informed bet-hedging strategy is more likely to satisfy a minimal performance requirement than a strategy that relies on the success of one particular action at one large compensation area.

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Paper 3 – draft manuscript

Predicting uncertain gains and losses in biodiversity value for conservation offsets and investments.

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Predicting uncertain gains and losses in biodiversity value for conservation offsets and investments

Abstract

Contemporary policies for biodiversity conservation increasingly rely on quantitative estimates of change in biodiversity value (BV) brought about by particular management actions. Biodiversity offsetting schemes, for example, aim to achieve 'no net loss' of biodiversity by requiring unavoidable biodiversity losses to be compensated for by gains elsewhere. The effect of uncertainty on the risk of failing to achieve no net loss is rarely considered. We investigated implications of uncertainty in predictions of BV using expert models of change in twelve vegetation attributes for five states of a grassy woodland ecosystem under six (gain and loss) management scenarios. BV was calculated using two metrics commonly employed in Australia. The quantity of gains that would be required to compensate for each loss scenario (i.e. the offset ratio) was calculated using one method that ignored uncertainty and another that was robust to it. Uncertainty increased the offset ratio up to 1400% where the magnitude of both gains and losses were uncertain, but increased the ratio only marginally where minimal losses were incurred on highly degraded site types and offset with maximal gains. Some results were highly dependent on the metric employed, primarily because the metrics weighted woody vegetation attributes differently. As results differed across site types, by metric, and across time, it is recommended that uncertainty analyses are incorporated into decision making processes. The study also revealed that both metrics of biodiversity value predicted greatest gains in woody vegetation attributes such as canopy and large trees. Consequently, policies that aim to maximise gains in BV may result in landscape wide decline in herbaceous components of vegetation and their associated biodiversity.

Introduction

A range of contemporary government policies for biodiversity conservation require quantitative estimates of current and future biodiversity value (BV) at the site scale (Regan et al. 2007; Hui et al. 2008). Policies that allocate incentive funding via auctions for conservation contracts, for example, award contracts to the landholders that predict greatest gains in BV for a given budget (Stoneham et al. 2003; Oliver et al. 2005; FSA 2006). Quantitative estimates of BV are also required for biodiversity offset and biobanking schemes that aim to compensate unavoidable biodiversity loss resulting from land development with gains elsewhere (e.g. ten Kate 2004; DSE 2006; DEA&DP 2007; DECC 2007a). A specific aim of biodiversity offset policies is to achieve 'no net loss' of biodiversity (a.k.a 'maintain or improve' outcomes), such that gains on the offset (gain) site must be at least equal to losses on the development (loss) site.

Biodiversity value may be defined in many ways, but in the context of biodiversity conservation policy it usually represents the extent to which a site can sustain viable populations of indigenous plant and animal species (Keith & Gorrod 2006). Biodiversity value is quantified on a univariate scale using metrics (quantitative valuation models) that represent causal relationships between site attributes and BV. To our knowledge, the metrics developed and implemented in Australia for the purpose of investment planning and biodiversity offsetting are the first of their kind, and are likely to be adopted internationally (BBOP 2008). BioMetric (Gibbons et al. 2005; Gibbons et al. 2009) and Habitat Hectares (Parkes et al. 2003; DSE 2004) are metrics of biodiversity value employed in Victoria and New South Wales, respectively. These metrics score several ecological attributes relative to a benchmark that represents a long-undisturbed patch of the same vegetation type, and aggregate the attribute scores to produce an estimate of overall BV. Predictions of future BV are based on assumptions about changes in each site attribute brought about by particular suites of management actions (DSE 2006 for Habitat Hectares and DECC 2007b for BioMetric).

Estimates of current and future BV may be uncertain for many reasons, which are reviewed in Gorrod et al. (in review) and briefly summarised here. The current BV of a given development or offset site may be uncertain due to: linguistic uncertainty in the definition of BV (Keith & Gorrod 2006); epistemic uncertainty as to relationships between

site characteristics and BV; model uncertainty in the quantitative representations of said relationships; or measurement error in field assessments (Gorrod & Keith in press). Estimates of the predicted future BV of a given site under a particular management scenario may be inaccurate due to uncertainty about the rate and direction of change in each attribute over time. Contributing factors to this uncertainty include: (i) the effectiveness of management actions (e.g. uncertainty as to the precise reduction in weed cover that will result from weed removal); (ii) natural variation depending on site conditions; (iii) influence of exogenous factors (including both natural variation such as weather, and anthropogenic activity on neighbouring land); (iv) the length of time required to bring about change in vegetation attributes (e.g. uncertainty as to when hollow bearing trees will be developed if trees are planted now); (v) the vegetation dynamics or underlying successional model of the ecosystem (e.g. uncertainty as to whether reducing weed cover will cause a transition to a different state, (vi) or whether there is an additional factor (such as moisture or seed availability) preventing transition); (vii) measurement error; and (viii) model uncertainty. Uncertainty about predicted gains is borne out in empirical evidence demonstrating that restoration projects carry considerable risks of failure in a range of ecosystems, even where projects are relatively well resourced (e.g. Zedler & Callaway 1999; Wilkins et al. 2003; Hilderbrand et al. 2005; Vesk & Dorrough 2006; Cunningham et al. 2007; Standish et al. 2007).

In practical terms, these uncertainties mean that there is considerable uncertainty as to which combination of actions will achieve the greatest gains, and the extent to which the magnitude of gains depends upon the initial conditions of the site. In the case of offsetting, there is also uncertainty as to the relative magnitudes of loss incurred by land use intensification. While biodiversity offsetting currently applies predominantly to total clearing (removal of vegetation), there is potential for offsetting losses due to grazing and partial clearing (e.g. removal of understorey).

To achieve no net loss in biodiversity offsetting, loss at the development site is compensated for by gains elsewhere. The ratio of loss in BV on the development to gain in BV on the offset site is the ‘offset ratio’, and it provides the area of gain required to offset loss on one unit of area (e.g. hectare) to achieve no net loss. Normally, offset ratios are calculated by ‘matching mean expected utilities’ (MMU) of the development and offset sites (e.g. US FWS 2003; DSE 2006; DECC 2007a). That is, the best

estimate of loss of utility from the development site is compensated by the estimated utility gained in offset sites. For example,
if it is possible to gain 20 points per hectare through management, then to offset a loss of 100 points, an offset ratio of 1:5 would be required. However, the uncertainties inherent in expected losses and gains result in risks that realised gains may not balance realised losses. To accommodate these risks in decision-making, Moilanen et al. (2008) suggested 'robustly fair' offsets as those in which offset sites must have a user-specified (e.g. 95%) chance of gaining equal to or greater biodiversity value than is lost from the development site. Robustly fair offsets therefore address uncertainty in predictions of gains and losses of BV due to epistemic uncertainty about the values that vegetation attributes will take, which MMU ignore.

Neither BioMetric nor Habitat Hectares explicitly takes uncertainty into consideration in estimating current BV or calculating future gains that result from management, although the use of broad scoring categories aims to reduce the sensitivity of scores to observer error and natural variation. The impacts of uncertainty on offset outcomes may vary, depending on initial site conditions and type of management actions undertaken and the ecological changes that they are expected to generate.

In this study we simulated the dynamics of a relatively well-understood grassy woodland ecosystem, in order to evaluate:

1. Which management strategies resulted in the greatest gains and losses in biodiversity value, and whether this varied with the initial state of the system or with the metric of biodiversity value employed.
2. Whether simulated predictions were consistent with expected gains calculated from the standard gain scoring protocols (DSE 2006 and DECC 2007b).
3. Which combinations of development and offset actions were capable of producing 'no net loss' outcomes for biodiversity value by matching mean utilities?
4. How much higher were offset ratios that achieved robustly fair 'no net loss' under predictive uncertainty, and how did this vary with the initial state of the system?
5. Which causes of predictive uncertainty appear to have the greatest influence on robustly fair 'no net loss' outcomes (model uncertainty for biodiversity value, natural variation and epistemic uncertainty about vegetation dynamics, or measurement error)?

We conclude by examining the landscape-scale implications of our results and identify some major constraints that offset decisions must address if they are to achieve no net loss outcomes for biodiversity conservation.

Methods

Simulations

Five hypothetical site types were defined to represent a range of ecological states in a grassy woodland ecosystem: *all low* (all attributes had low values relative to the benchmarks), *all moderate*, *high woody* (high values for woody attributes, such as large trees, canopy cover and log abundance), *high herbaceous* (high values for non-woody attributes, such as native plant species richness and understorey lifeform richness) and *all high*. Values of twelve vegetation attributes were assigned to each site type based on published descriptions (Benson 1992; NSW Scientific Committee 1997; Tozer 2003; Wilkins et al. 2003) and extensive field experience for Cumberland Plain Woodland, a grassy woodland community found in western Sydney, Australia (Table). Six hypothetical management scenarios were defined including three land-use intensification (or 'loss') scenarios (total clearing, partial clearing and grazing), one 'no action' scenario and two restoration (or 'gain') scenarios (planting only and mixed management) (Table). All six management scenarios assume native and feral grazers are excluded from the site by fencing.

For each of the vegetation attributes listed in Table , a best estimate of the trajectory of change over a 150 year time frame was made for each hypothetical site type under each management scenario. Upper and lower bounds (95% subjective probabilities, *sensu* Kyberg & Smokler 1964; Regan et al. 2002) were also specified to represent plausible values the attribute may take due to natural variation and lack of knowledge about alternative successional models for Cumberland Plain Woodland. The estimated trajectories were based on our expert judgement, and informed by empirical data. Data for cleared, planted and remnant Cumberland Plain Woodland sites were available for some attributes: native plant species richness, and native cover of overstorey, midstorey and ground layer (Wilkins et al. 2003; Nichols 2005; Gorrod & Keith in press) (Figure 1).

Information relevant to the development of hollow bearing trees (Gibbons & Lindenmayer 2002; NSW Scientific Committee 2007) and a range of vegetation attributes in revegetated sites in Victoria, Australia (Vesk et al. 2008) was also used. The best estimate, upper and lower bounds for each attribute were constructed using tools developed in the open-source statistical environment R (R Development Core Team 2007) that allowed the user to plot unique trajectories by specifying at least five values in a graphic interface (examples shown in Figure).

Best estimates and bounds for each attribute were used as an input to stochastic simulations of change over time. The difference between the upper bound and the best estimate, and the difference between the lower bound and the best estimate were each set to be equal to three standard deviations, so that the distance between the upper and lower bounds was interpreted as a 99% confidence interval. Two hundred uniformly distributed (0,1) pseudo-random numbers, u_i, were generated to represent possible trajectories the site might take. A number less than 0.5 was assigned a trajectory between the best estimate and the lower bound, and a number greater than 0.5 was assigned a trajectory between the best estimate and the upper bound. The specific trajectory was then selected to correspond to the quantile corresponding to u_i of the normal density assuming that the relevant bound was three standard deviations away from the best guess. Thus the distribution of possible trajectories consisted of two halves of possibly unalike normal distributions, joined such that the median was equal to the best estimate.

To account for our inability to measure ecological responses with perfect precision, observer error was simulated using the same method and added to the natural variation in trajectories. Levels of simulated observer error were set at 25% based on an empirical study of 10 observers applying the metrics in Cumberland Plain Woodland (Gorrod & Keith in press).

In each of the 200 simulations, raw attribute values were converted into BioMetric scores (Table 3, Gibbons et al. 2005) and Habitat Hectares scores (Table 4, DSE 2004) for each decade over a 150 year timeframe. The site condition component of Habitat Hectares is normally scored out of 75, but was scaled here to a maximum of 100 to enable direct comparison with BioMetric. Some attributes in the Habitat Hectares

protocol were not simulated but set as constant for all sites over time, based on empirical evidence (Gorrod & Keith in press): large tree health (>75%), canopy health (>75%) and large log length (<2.5 m). The regional and landscape context attributes of both indices, such as distance to nearest core area, were not included in these calculations. The net result for all site by action combinations on each pair of sites over time for both protocols were calculated and graphed. These analyses were conducted in the R statistical package (R Core Development Team 2007).

Comparison with standard gain scoring protocols

The standard protocols for calculating expected gains in BioMetric and Habitat Hectares scores were employed (DECC 2007b and DSE 2006, respectively). The DECC (2007b) gain scoring protocol specifies a suite of management actions equivalent to the mixed management simulations. Each BioMetric attribute increases by one scoring category (0 to 1, 1 to 2, 2 to 3, see Table 3) under mixed management, except that no gain for hollow bearing trees is scored if there were none present initially. DSE (2006) assumes active management for ten years only, with a commitment to maintain the gain in perpetuity if used as an offset. DSE (2006) has different gain scoring protocols depending on the management actions undertaken, existing entitlement rights, size of the offset site and security of the offset. Here, we used the gain predicted for offset sites smaller than five hectares, assuming no previous entitlements and without an 'on title agreement'. Management actions were equivalent either to our simulated planting or mixed management treatment. DSE (2006) also has different scoring protocols for regenerating an existing remnant patch (a minimum of three trees with at least 20% cover) or re-establishing vegetation on previously cleared land. The DSE (2006) revegetation gain scoring protocol was used for the *all low* site, and the regeneration scoring protocol for remaining sites. Gains in Habitat Hectares scores were not permitted to increase the site score to greater than 100%. The landscape context components of the DSE (2006) and DECC (2007b) scoring protocols (e.g. patch size) were not included in the predicted gain.

Data analyses

Offsets were calculated by matching mean utilities (MMU, Equation 1) and by calculating the ratio with a 95% probability of achieving no net loss (Equation 2). MMU offsets were calculated for losses due to total clearing (not partial clearing or grazing) at years 10, 50, 100 and 150. Robust offsets were calculated for all loss versus gain scenarios at the same time steps, and were also calculated under observer variation.

$$R_{MMU} = \frac{BV_l(t_0) - BV_l(t_x)}{BV_o(t_x) - BV_o(t_0)} \tag{1}$$

$$R_{ROB} = \Pr(R_{ROB} * (BV_o(t_x) - BV_o(t_0)) > (BV_l(t_0) - BV_l(t_x))) \geq 0.95 \tag{2}$$

R_{MMU} and R_{ROB} are Ratios calculated using Matching Mean Utilities and Robust methods, respectively. BV is the best estimate of biodiversity value calculated using either BioMetric (Gibbons et al. 2005) or Habitat Hectares (DSE 2004). Subscripts indicate BV at the loss site (l) and offset site (o) prior to development and offset management (t_0) at and after x years of management for gain (t_x).

Empirical cumulative density function (ecdf) worst-case curves were produced for each trade. These curves were constructed by: 1) determining the minimum across time for each of the 200 simulations, and 2) plotting the ecdf of those 200 minima. In this way, the curves showed the probability that the observed value declined relative to any given value (either of one site or the combined values of two sites) at any time during the simulation period. The probability of decline relative to the starting value was of particular interest.

Results

Losses

Losses due to total clearing were greatest on *all high* sites and least on the *all low* sites according to both metrics of BV (Figure 3). The relative magnitude of losses on *high woody* and *high herbaceous* sites differed between metrics: BioMetric predicted greater losses on *high woody* sites; whereas Habitat Hectares predicted greater losses on the

high herbaceous and *all moderate* sites (Figure 3). These differences reflected different attribute weightings in the two metrics.

Losses of BV due to partial clearing were estimated to be greater for Habitat Hectares than BioMetric on all sites except *high woody* sites (Figure 4), again reflecting differences in attribute weightings. In response to partial clearing BV initially declined rapidly, then either remained constant or continued to decline for up to 80 years and then became constant. The only exception was the *high herbaceous* site type according to Habitat Hectares, which increased by about 10 % between years 40 and 90 after the initial decline due to increases in canopy cover and large trees.

Losses of BV due to grazing were smaller than losses due to partial clearing on all sites for both metrics (Figure 4). Losses due to grazing were greater for Habitat Hectares than BioMetric on all sites except the *high woody* and *all low* sites. In response to grazing, all sites gradually declined in value, some reaching a minimum within 50 years (the *high woody* and *all high* sites), others taking longer or declining further after a period of being stable (Figure 4). The exception was the *all moderate* site according to BioMetric, which temporarily increased due to the development and subsequent loss of a hollow bearing tree (Figure 4). Values of both metrics were particularly uncertain for *all high* sites under the grazing treatment.

No Action

Best estimates of response to no action (or grazing exclusion) were similar for both metrics: the values of the *all low*, *all moderate* and *high woody* sites declined; *high herbaceous* sites gradually increased; and *all high* sites remained constant (Figure 5). Losses of BV under no action were small for the *all low* and *high woody* sites, and substantially higher (though uncertain) for *all moderate* sites. There were relatively large gains (40% by year 100) in BioMetric value for *high herbaceous* sites, but uncertainty increased over time. The lower bound for *all high* sites was approximately 40% lower than the best estimate for BioMetric, and 20% lower for Habitat Hectares.

Gains: Planting

The planting treatment caused substantial gains in BV on *all low* sites, with gains of 20% by year 20 for Habitat Hectares and year 90 for BioMetric (Figure 6). However, the lower bound was as much as 15% lower than the best estimate for this scenario. Gains due to planting on *all low* sites exceeded gains due to mixed management for the first 20 years according to BioMetric and first 50 years according to Habitat Hectares (Figure 6), due largely to more rapid increases in tree cover.

Planting resulted in the gradual increase of BioMetric value for *high herbaceous* sites, whereas the Habitat Hectares value declined for the first 40 years before increasing (Figure 6). Planting in *all moderate* sites produced a decline in BV for at least 50 years, with Habitat Hectares value continuing to decline and BioMetric value increasing after 50 years, reaching a net gain after 100 years. Planting was not simulated for the *high woody* or *all high* sites.

Gains: Mixed management

Greatest gains in BV were generated on the *all low* and *high herbaceous* sites under mixed management after year 100 (Figure 6). BioMetric predicted similar gains on both sites (about 50% at year 130) whereas Habitat Hectares predicted greater gains on *all low* sites (41% versus 32%) (Figure 6). Uncertainty in BioMetric values of *all low* and *high herbaceous* sites increased after about year 100 under the mixed management scenario, coincident with the development of tree hollows. Gains were primarily due to development of woody attributes: for example, the *all low* site's 'understorey' score was 5 out of a possible 25 (unscaled) under mixed management at all time steps.

Mixed management of the *all high* site type increased BV to 100% within 30 years, though uncertainty was high (Figure 6). Similar initial gains occurred on *high woody* sites, which subsequently declined within 50 years. BioMetric scores for *all moderate* sites declined initially then gradually increased to a maximum gain of 40%, whereas the Habitat Hectares score increased and plateaued at a gain of around 10% by year 30. Substantial gains in BioMetric scores on the *all low*, *all moderate* and *high herbaceous* sites between years 120 and 140 were due to hollow development.

Comparison of predictions with established gain scoring protocols

Relative to simulated gains in BV, DSE (2006) predictions of gains from planting were optimistic for all but the *all low* sites (Figure 6).

Simulated gains from mixed management were generally consistent with the predictions of established gain scoring protocols for *all low* and *high herbaceous* sites, with simulated best estimates exceeding predictions prior to year 80 (Figure 6). Simulated and predicted gains on *all high* sites were also generally consistent. Simulated gains in BVs from mixed management on *all moderate* sites, however, were below DSE (2006) and DECC (2007b) predictions (until year 140 for BioMetric). Results for the *high woody* sites were inconsistent across metrics: simulated Habitat Hectares gains approximately equalled the DSE (2006) prediction after year 30; whereas the BioMetric best estimate was 14% below DECC (2007b) prediction for the whole simulation period.

Risk of decline

Almost all trades were associated with some risk that gains in BV would not compensate losses. The only trades for which there was no risk of falling below the combined starting value were: offsetting grazing on *all low* site types with planting or mixed management on *all low* sites based on Habitat Hectares, or offsetting grazing on *all low* or *high herbaceous* sites with mixed management on the *high herbaceous* sites based on BioMetric. Risk (ecdf) curves for an example scenario are shown in Figure 7, in which accounting based on Habitat Hectares shows a greater risk of offset failure than that based on BioMetric.

Offset ratios: Matching Mean Utilities

MMU offset ratios were not calculable for scenarios where no gains in BV were generated by management actions (those with missing values in Table 5), including no action on *all moderate*, *high woody* and *high herbaceous* sites according to both metrics; and mixed management at year 10 on *high herbaceous* and *all high* sites according to Habitat Hectares (Table 5). In most scenarios, MMU offset ratios declined over time, though they were constant from year 50 on *high woody* and *all high* sites according to both metrics.

MMU offset ratios less than 1:1 occurred for trades involving loss and gain on *all low* sites, or at year 150 for some other scenarios, and never at year 10 (Table 5). MMU offset ratios were very high (up to 1:172) at year 100 for planting on *all moderate* sites according to BioMetric. Highest MMU offset ratios were required when losses were incurred on *all high* sites.

MMU ratios were generally lower for mixed management than no action or planting, but not always (Table 5). For example, offset ratios for mixed managing the *all low* site were higher than those for planting until year 50 according to both metrics. On *all low* sites, the same ratio was required for mixed management for 10 years or no action for 50 years according to Habitat Hectares.

Offset ratios: Robust

The only case where R_{ROB} and R_{MMU} were equal was at year 10 under planting or mixed management on the *all low* site type according to BioMetric (Tables 5 and 7). In some cases R_{ROB} was not substantially higher (17-50% higher) than R_{MMU}, particularly after at least 50 years of mixed management or planting on the *all low* and *high herbaceous* sites (Tables 5 and 7). Robust ratios were frequently substantially higher (200-600% higher) than MMU ratios for offsetting total clearing, including for gains on *all low* and *high herbaceous* sites due to no action, or planting before year 50 (Tables 5, 6 and 7). Robust offset ratios were 1400% greater than MMU offset ratios for mixed management on the *high woody* site at year 10 according to Habitat Hectares.

There were several cases for which it was possible to calculate R_{MMU} to offset total clearing, but it was not possible to calculate R_{ROB} (Figures 5, 6 and 7). These scenarios primarily involved gain actions on *all moderate*, *high woody* and *all high* sites, and often R_{MMU} was not calculable for earlier years or for the alternate metric.

In every case that it was or was not possible to robustly offset total clearing, it was also possible to offset partial clearing and grazing (Table 7). Robust ratios were always higher for offsetting total clearing than partial clearing, which were higher than those required to offset grazing. BioMetric ratios for total clearing were up to 25% larger than

those for partial clearing, and approximately 25-90% greater than grazing. Habitat Hectares ratios for total clearing were approximately 16-60% and 30-100% greater than those for partial clearing and grazing, respectively.

The most notable difference in robust offset ratios between the metrics was for mixed management on *high woody* sites (Table 7). According to BioMetric, it was untenable to robustly offset on *high woody* sites at any year, but according to Habitat Hectares, it was possible from year 10 with moderately high ratios which decreased over time.

Robust offsets: Observer error

R_{ROB} increased up to 25% due to observer variation for offsetting on *all low* sites according to both metrics; increased by 40-200% for offsetting on *high herbaceous* sites; and increased by 100-200% for offsetting on *high woody* sites using mixed management according to Habitat Hectares.

Observer variation increased R_{ROB} for mixed management on *all moderate* sites by 600-800% according to BioMetric, and made it untenable to use mixed management on *all moderate* sites according to Habitat Hectares. Observer variation also made it unfeasible to robustly offset using mixed management on *all high* sites according to BioMetric (note that it was not possible without observer variation according to Habitat Hectares).

Discussion

Landscape scale implications of maximising gains

Both of the metrics analysed predicted that the greatest gains in BV, and lowest offset ratios, could be achieved on sites that initially had a degraded woody component. That is, greatest gains were generated by development of attributes such as canopy cover, large trees and logs on the *all low* and *high herbaceous* site types. Gains in herbaceous attributes were generally smaller and achieved more slowly than gains in woody attributes, irrespective of site type, management strategy or metric. This result is consistent with Wilkins et al. (2003), who found that there was no significant difference between the plant species composition of untreated pasture and sites ten years after

tree planting when the planted trees themselves were excluded from the analysis. Conservation investment policies that aim to maximise gains in BV are likely to fund gains from establishment of woody attributes. Loss of herbaceous components from development sites may therefore be more likely to be offset by gains in woody attributes than gains in the same type of attributes that were lost. As a consequence of maximising gains in BV, then, there is a potential for landscape-wide decline in the herbaceous components of vegetation and their associated biodiversity. Neither of the metrics examined here would be able to detect this type of decline, which could only be addressed by separate accounting for different types of biodiversity attributes.

There may be other landscape scale implications of offsets that were not examined in this study, as we did not simulate data for the regional and landscape metrics that accompany BioMetric and Habitat Hectares site assessments. Existing offset schemes assume that BV scales with area, and that BV scores can be summed across different sites. Valuable future research would evaluate different spatially explicit models to estimate the BV of offset areas, which would have parallels with Moilanen et al.'s (2008) discussion of correlated risk of offset failure.

Metric matters

In many cases, the results depended on which metric was used to calculate BV, producing greater differences in results than differences attributable to natural variation and epistemic uncertainty or observer error. This difference between the metrics is indicative of uncertainty in the model of BV. In particular, the metrics differed in the relative weighting of woody and herbaceous attributes, which heavily influenced the predicted magnitude of gains and losses, and the calculation of offset ratios for different site types. The choice of metric will therefore affect decisions about where and how conservation resources are deployed, and hence the future biodiversity attributes of the landscape. It is recommended that model uncertainty is considered in addition to epistemic uncertainty about vegetation attribute values in predictions of BV for biodiversity conservation decisions. Preferably, the implicit assumptions and sensitivities of the metric would be evaluated and documented, using empirical data across a range of spatial and temporal scales where possible.

Fencing, Planting or Mixed management?

The planting management strategy was predicted to cause declines in BV of some sites, either temporarily or for the whole simulation period. This was due to the overabundance of canopy cover and its negative effects on understorey diversity (Specht & Specht 1993; Keith & Bradstock 1994). In particular, sites with moderate scores for all vegetation attributes were predicted to decline for at least 70 years according to one metric and at least 150 years according to the other. For this site type, mixed management, which involved tree-planting at lower densities, would generate greater gains than planting for any length of time. This is particularly important as many sites available for offsetting in grassy woodland ecosystems may be similar to the hypothetical *all moderate* sites. The planting strategy only resulted in greater gains and lower offset ratios than mixed management if implemented for 10 years or less on highly degraded sites. Over longer time frames, greater gains were always generated by mixed management.

Both metrics of BV predicted substantial gains from fencing sites with intact herbaceous attributes but degraded woody attributes. The gains from fencing such sites were almost as high as those for mixed management, though they were achieved over longer time frames. This is due to 'natural' recruitment of canopy trees in the absence of grazing pressure, and subsequent development of logs, large trees and hollows. Although the opportunity cost of stock exclusion may be considerable to the landholder, the most cost effective gains for investment planning are likely to be through fencing these sites with high herbaceous values. Such sites would approach benchmark states in time, with intact woody *and* herbaceous attributes. These sites may, however, be relatively rare in agricultural landscapes, particularly when pasture improvement has been employed (Prober & Thiele 1995; Dorrough et al. 2006).

Standard gain scoring protocols

Simulated predictions were not inconsistent with expected gains calculated from the standard gain scoring protocols. Standard protocols, however, tended to underestimate gains on *all low* sites, which may reflect a deliberate choice to down play the value of those gains. In contrast, standard protocols tended to overestimate gains on *all*

moderate sites, which may have significant landscape wide implications for achieving gains in BV where this is the most common site type in the region.

High quality sites should be maintained

The results of this study indicated that losses were greatest on high scoring grassy woodland sites (with intact woody and herbaceous components) compared with other sites, and also achieved minimal gains relative to other sites. Consequently, any offset scenarios involving high quality sites had very high offset ratios in comparison with other scenarios. Therefore it is likely to be more efficient to prevent damage to sites that currently possess high biodiversity values, rather than allow losses to be compensated by uncertain gains elsewhere. Also, high uncertainty under the 'no action' scenario for this site type indicates that it may be prudent to invest in maintaining the high values of these sites.

Time lags

The only case with no risk of decline in the combined biodiversity values of development and offset sites was when minimal losses resulting from grazing were offset with maximal gains through mixed management. It follows that offsetting losses from total or partial clearing will cause at least a temporary decline in combined biodiversity value (and combined abundance of site attributes), which may have negative implications for population viability and ecological processes in the landscape. Viable populations of indigenous species may be unable to persist if particular habitat features fall below some minimum density in the landscape, even if temporarily (Gibbons & Lindenmayer 2002; Vesk & MacNally 2006). Therefore it is important to not only consider the site-by-site tradeoffs, but also the contribution of the site to the abundance of features in the landscape over time.

Uncertainty and robust offsets

Given that current and future estimates of biodiversity value are uncertain, MMU offset ratios generally underestimated the magnitude of gains required to achieve no net loss. For some trades, incorporation of predictive uncertainty in the offset calculations made it

impossible to robustly achieve no net loss (for example, offsetting through planting on the *all moderate* site). But for other trades, robust offset ratios were of a similar magnitude to ratios that matched mean utilities. Observer error had minimal effects on the robust offset ratios required to offset using gains on the *all low* and *high herbaceous* sites, but it caused a substantial difference between the MMU and robust offset ratios for those scenarios that were already very uncertain, in particular those involving the *all moderate* or *all high* sites.

Given the differences in predictions for different site types by the two metrics, it is difficult to draw generalisations about which scenarios are most immune to uncertainty without conducting a thorough uncertainty analysis. An uncertainty analysis is additionally useful to weigh the benefits against the costs of alternative actions. The construction of risk curves may be a particularly useful for communicating the risk of net loss and informing decisions. The results indicate that it is possible, and relatively simple, to account for some aspects of predictive uncertainty with robust offset calculations, but that the implications of model uncertainty for biodiversity valuation may require further research and analysis.

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Table 5. Vegetation attributes of hypothetical sites at t_0

Vegetation attribute	Hypothetical site				
	All low	**All moderate**	**High woody**	**High herbaceous**	**All high**
Native plant species richness [1]	15	20	20	45	40
Number of lifeforms present (out of ten) [2]	4	6	4	9	10
Proportion of lifeforms modified [2]	1	0.33	0.75	0.11	0
Native overstorey cover (%) [1,2]	5	10	20	2	15
Native midstorey cover (%) [1]	5	10	25	1	15
Native ground cover (%) [1]	15	25	15	80	50
Exotic plant cover (%) [1,2]	55	15	35	2	2
Proportion of woody species regenerating[1,2]	0	0.5	0.2	0.4	1
Number of large trees per hectare[2]	2	8	18	0	14
Number of hollow bearing trees per 0.1 hectare[1]	0	0	1	0	1
Length of logs per 0.1 hectare[1,2]	0	10	25	1	25
Litter cover (%) [2]	1	4	12	5	10
BioMetric score (%)	**20**	**48**	**65**	**39**	**90**
Habitat Hectares score (scaled to %)	**24**	**68**	**53**	**65**	**97**

1: Component of the BioMetric protocol; 2: Component of the Habitat Hectares protocol

Table 6. Details of actions undertaken in different management scenarios

Management scenario		Actions at t_1	Actions t_2-t_{150}
L1	**Total clearing**	All vegetation permanently removed.	
L2	**Partial clearing**	The shrub layer and all logs are initially removed from the site, and the canopy trees are thinned by half.	Over time, the site is mown or slashed regularly (preventing recruitment) and any new logs are removed.
L3	**Grazing**	Moderate levels of grazing introduced (or continued if grazed prior to t_1) and maintained at levels that remove 50% of potential groundcover biomass and virtually all seedlings of palatable woody species.	
0	**No action**	No management actions undertaken (or grazers excluded if present prior to t_1).	
R 1	**Planting**	Four woody species are planted at moderate to high densities. Only conducted in sites with low canopy cover at t_1. Grazing excluded.	No further management actions undertaken.
R 2	**Mixed management**	Restoration actions including planting, weed control, grazer exclusion and fire management are undertaken as specified by best management advice.	

Table 7. BioMetric component attributes, weights and scores for Cumberland Plain Woodland. Benchmarks (which score 3) are from DEC (2006).

Vegetation attribute	Weight	Score			
		3	2	1	0
Native plant species richness	20	>29	15 – 28	1 – 14	0
Native over-storey cover	5	19-24%	9.5 – 18.9 24.1 – 36	2 – 9.4 36.1 – 48	0 – 1.9 >48
Native mid-storey cover	10	20-30%	10 – 19.9 30.1 – 45	2.1 – 9.9 45.1 – 60	0 – 2 >60
Native ground cover (grasses)	5	23-31%	11.5 – 22.9 31 – 46.5	2.4 – 11.4 46.4 – 62	0 – 2.3 >62
Native ground cover (shrubs)	5	0-5%	5 – 7.5	7.5 – 10	>10
Native ground cover (other)	5	12-20%	6 – 11.9 20.1 – 30	1.3 – 5.9 30.1 – 40	0 – 1.2 >40
Cover of weeds	5	0-5%	5 – 33	33 – 66	>66
Number of hollow bearing trees	30	≥1 tree	n/a	n/a	0 trees
Proportion of overstorey species regenerating	10	1	0.5 – 0.99	0 – 0.49	n/a
Total length of logs (m)	5	≥5 m of logs ≥10cm diameter	2.5 – 4.99	0.51 – 2.49	0 – 0.5

Table 8. Component attributes, weights and scores of Habitat Hectares for Cumberland Plain Woodland. Benchmarks were constructed from empirical data (NSW Scientific Committee 1997; Tozer 2003) and expert opinion.

Attribute		Benchmark	Weight
Large trees	Number of trees per hectare Large tree canopy health	15 trees with dbh[1] ≥ 50 cm >70%	10
Tree canopy cover	Tree canopy cover Canopy health	15% cover, trees >18 m tall >70%	5
Lack of weeds	Total cover of weeds Proportion that is high threat weed cover	<5% 0	15
Understorey life forms	Number of lifeforms present Proportion of present lifeforms that are substantially modified	10 0%	25
Recruitment	Proportion of native woody species present that have adequate recruitment Number of native woody species present	100% 5	10
Organic litter	Organic litter cover Proportion of litter that is native	10% >50%	5
Logs	Length of logs Length of large logs per hectare	150 m of logs ≥ 10 cm diameter 25 m of logs ≥25 cm diameter	5

[1] dbh = Diameter at breast height

Table 9. Offset ratios calculated by Matching Mean Utilities to offset 1 ha of Total clearing with No action, Planting or Mixed Management according to BioMetric and Habitat Hectares at years 10, 50, 100 and 150. (-) indicates that it was not possible to calculate a ratio because gains were insufficient on the offset site.

		Offset Action: No Action								Offset Action: Planting only								Offset Action: Mixed management							
		BioMetric				Habitat Hectares				BioMetric				Habitat Hectares				BioMetric				Habitat Hectares			
Offset site	**Total clearing site**	**10**	**50**	**100**	**150**	**10**	**50**	**100**	**150**	**10**	**50**	**100**	**150**	**10**	**50**	**100**	**150**	**10**	**50**	**100**	**150**	**10**	**50**	**100**	**150**
All low	All low	-	-	-	-	-	4.5	-	-	2.3	1.6	1.1	0.5	1.7	0.8	1.0	0.6	5.6	1.1	0.7	0.4	4.5	0.8	0.5	0.4
	All mod	-	-	-	-	-	12.8	-	-	5.4	3.7	2.7	1.3	4.8	2.1	2.7	1.8	13.1	2.6	1.7	1.0	12.8	2.1	1.3	1.2
	High wood	-	-	-	-	-	10.0	-	-	7.3	5.0	3.6	1.7	3.8	1.7	2.1	1.4	17.7	3.4	2.3	1.3	10.0	1.7	1.0	1.0
	High herb	-	-	-	-	-	12.3	-	-	4.4	3.0	2.2	1.0	4.6	2.0	2.6	1.8	10.7	2.1	1.4	0.8	12.3	2.0	1.3	1.2
	All high	-	-	-	-	-	18.3	-	-	10.1	6.9	5.1	2.4	6.8	3.0	3.9	2.6	24.6	4.8	3.2	1.8	18.3	3.0	1.9	1.8
All mod	All low	-	-	-	-	-	-	-	-	-	-	39.0	0.8	-	-	-	-	-	2.6	1.8	0.5	6.8	3.4	2.3	1.7
	All mod	-	-	-	-	-	-	-	-	-	-	92.0	1.8	-	-	-	-	-	6.1	4.2	1.2	19.1	9.6	6.4	4.8
	High wood	-	-	-	-	-	-	-	-	-	-	124.0	2.4	-	-	-	-	-	8.3	5.6	1.6	15.0	7.5	5.0	3.8
	High herb	-	-	-	-	-	-	-	-	-	-	75.0	1.5	-	-	-	-	-	5.0	3.4	0.9	18.4	9.2	6.1	4.6
	All high	-	-	-	-	-	-	-	-	-	-	172.0	3.4	-	-	-	-	-	11.5	7.8	2.2	27.4	13.7	9.1	6.8
High woody	All low	-	-	-	-	-	-	-	-	N/A								-	3.9	3.9	3.9	1.9	0.9	0.9	0.9
	All mod	-	-	-	-	-	-	-	-									-	9.2	9.2	9.2	5.5	2.6	2.6	2.6
	High wood	-	-	-	-	-	-	-	-									-	12.4	12.4	12.4	4.3	2.0	2.0	2.0
	High herb	-	-	-	-	-	-	-	-									-	7.5	7.5	7.5	5.3	2.5	2.5	2.5
	All high	-	-	-	-	-	-	-	-									-	17.2	17.2	17.2	7.8	3.7	3.7	3.7
High herb	All low	2.6	0.7	0.5	0.4	-	2.3	1.1	0.6	3.0	1.3	1.0	0.4	-	6.8	2.7	0.6	1.7	0.8	0.6	0.4	-	1.1	0.6	0.6
	All mod	6.1	1.6	1.2	0.9	-	6.4	3.2	1.6	7.1	3.1	2.3	0.9	-	19.1	7.7	1.7	4.0	1.8	1.4	0.9	-	3.2	1.7	1.6
	High wood	8.3	2.2	1.7	1.2	-	5.0	2.5	1.3	9.5	4.1	3.1	1.3	-	15.0	6.0	1.4	5.4	2.4	1.9	1.3	-	2.5	1.4	1.3
	High herb	5.0	1.3	1.0	0.7	-	6.1	3.1	1.5	5.8	2.5	1.9	0.8	-	18.4	7.4	1.7	3.3	1.5	1.2	0.8	-	3.1	1.7	1.5
	All high	11.5	3.0	2.3	1.6	-	9.1	4.6	2.3	13.2	5.7	4.3	1.8	-	27.4	11.0	2.5	7.5	3.4	2.6	1.8	-	4.6	2.5	2.3
All high	All low	-	-	-	-	-	-	-	-	N/A								3.9	1.9	1.9	1.9	-	6.8	6.8	6.8
	All mod	-	-	-	-	-	-	-	-									9.2	4.6	4.6	4.6	-	19.1	19.1	19.1
	High wood	-	-	-	-	-	-	-	-									12.4	6.2	6.2	6.2	-	15.0	15.0	15.0
	High herb	-	-	-	-	-	-	-	-									7.5	3.7	3.7	3.7	-	18.4	18.4	18.4
	All high	-	-	-	-	-	-	-	-									17.2	8.6	8.6	8.6	-	27.4	27.4	27.4

Table 10. Robust offset ratios (95% probability of No Net Loss) for offsetting loss of 1 ha due to Grazing, Partial or Total clearing with No action on *high herbaceous* sites, according to BioMetric and Habitat Hectares at years 10, 50, 100 and 150. (-) indicates it was not possible to calculate a robust offset ratio due to insufficient gains on the offset site.

Offset site	Loss action	Loss site	BioMetric				Habitat Hectares			
			10	50	100	150	10	50	100	150
High herb	Grazing	All low	1.8	0.4	0.4	0.5	-	-	0.6	0.5
		All mod	2.1	1.3	1.2	1.2	-	-	3.3	1.8
		High wood	6.8	1.8	1.6	1.5	-	-	1.0	0.5
		High herb	1.8	0.6	0.6	0.6	-	-	3.0	1.8
		All high	8.8	2.3	1.9	1.9	-	-	3.6	1.9
	Partial clearing	All low	3.6	0.8	0.6	0.5	-	-	0.9	0.5
		All mod	9.8	2.0	1.7	1.7	-	-	5.1	2.3
		High wood	13.4	2.7	2.4	2.4	-	-	3.5	1.6
		High herb	7.8	1.6	1.3	1.2	-	-	4.4	2.0
		All high	19.4	4.0	3.4	3.3	-	-	8.3	3.6
	Total clearing	All low	4.9	1.0	0.8	0.8	-	-	2.6	1.1
		All mod	11.5	2.4	1.9	1.9	-	-	7.3	3.0
		High wood	15.5	3.3	2.6	2.6	-	-	5.7	2.4
		High herb	9.4	2.0	1.6	1.6	-	-	7.0	2.9
		All high	21.5	4.5	3.6	3.6	-	-	10.4	4.3

Table 11. Robust offset ratios (95% probability of No Net Loss) to offset 1 ha of Total clearing, Partial clearing or Grazing with Planting or Mixed management, according to BioMetric and Habitat Hectares at years 10, 50, 100 and 150. (-) indicates it was not

			Offset action: Planting only								Offset action: Optimal management							
Offset Site	Loss Action	Loss Site	BioMetric				Habitat Hectares				BioMetric				Habitat Hectares			
			10	50	100	150	10	50	100	150	10	50	100	150	10	50	100	150
All low	Grazing	All low	0.8	0.7	1.9	2.9	0.6	0.2	0.5	0.5	2.0	0.7	0.5	0.6	1.0	0.2	0.2	0.3
		All mod	1.0	2.6	5.4	5.9	2.0	1.8	2.3	1.7	2.4	2.1	1.3	1.4	4.0	1.5	1.0	1.1
		High wood	3.2	3.6	6.4	6.1	0.0	0.5	0.7	0.5	7.7	3.1	1.8	1.8	0.0	0.4	0.3	0.3
		High herb	0.8	1.3	2.5	2.7	2.8	1.8	2.4	1.7	2.0	1.1	0.7	0.7	5.7	1.5	1.0	1.1
		All high	4.6	4.2	7.9	6.9	3.6	3.0	2.8	1.9	11.1	3.8	2.4	2.2	7.0	2.3	1.3	1.2
	Partial clearing	All low	1.7	1.5	2.6	2.9	1.2	0.5	0.8	0.5	4.1	1.3	0.6	0.6	2.0	0.5	0.4	0.3
		All mod	4.6	4.1	7.5	8.2	7.0	3.3	3.5	2.2	11.1	3.4	1.8	1.8	11.7	2.7	1.6	1.4
		High wood	6.5	4.6	10.4	11.4	4.2	2.3	2.5	1.6	15.7	4.2	2.5	2.5	7.3	1.9	1.1	1.0
		High herb	3.6	3.1	5.5	6.1	8.0	3.4	2.8	1.9	8.9	2.7	1.4	1.4	13.3	2.6	1.3	1.3
		All high	9.1	7.3	14.7	16.2	10.8	5.5	5.5	3.4	22.1	5.9	3.6	3.6	19.0	4.5	2.5	2.3
	Total clearing	All low	2.3	2.1	3.5	3.9	3.6	1.6	1.6	1.0	5.6	1.7	0.9	0.9	6.0	1.4	0.8	0.7
		All mod	5.4	4.8	8.4	9.2	10.2	4.6	4.6	2.8	13.1	4.0	2.0	2.0	17.0	3.9	2.2	1.9
		High wood	7.3	6.5	11.3	12.4	8.0	3.6	3.6	2.2	17.7	5.4	2.8	2.8	13.3	3.1	1.7	1.5
		High herb	4.4	3.9	6.8	7.5	9.8	4.5	4.5	2.7	10.7	3.3	1.7	1.7	16.3	3.8	2.1	1.8
		All high	10.1	9.1	15.6	17.2	14.6	6.6	6.6	4.1	24.6	7.5	3.8	3.8	24.3	5.6	3.2	2.7
All mod	Grazing	All low	-	-	-	-	-	-	-	-	-	17.0	2.3	1.8	-	-	1.0	1.1
		All mod	-	-	-	-	-	-	-	-	-	49.0	5.9	3.9	-	-	6.3	4.1
		High wood	-	-	-	-	-	-	-	-	-	54.0	5.4	4.3	-	-	1.7	1.2
		High herb	-	-	-	-	-	-	-	-	-	15.0	2.8	1.9	-	-	5.5	4.1
		All high	-	-	-	-	-	-	-	-	-	106.0	7.9	6.1	-	-	7.8	4.3
	Partial clearing	All low	-	-	-	-	-	-	-	-	-	29.0	3.2	1.8	-	-	1.5	1.1
		All mod	-	-	-	-	-	-	-	-	-	75.0	9.1	5.1	-	-	9.0	4.9
		High wood	-	-	-	-	-	-	-	-	-	86.0	12.7	7.1	-	-	6.3	3.5
		High herb	-	-	-	-	-	-	-	-	-	61.0	6.8	3.8	-	-	7.8	4.3
		All high	-	-	-	-	-	-	-	-	-	155.0	17.7	10.1	-	-	14.5	7.6
	Total clearing	All low	-	-	-	-	-	-	-	-	-	39.0	4.3	2.4	-	-	4.5	2.3
		All mod	-	-	-	-	-	-	-	-	-	92.0	10.2	5.8	-	-	12.7	6.4
		High wood	-	-	-	-	-	-	-	-	-	124.0	13.8	7.8	-	-	10.0	5.0
		High herb	-	-	-	-	-	-	-	-	-	75.0	8.3	4.7	-	-	12.2	6.1
		All high	-	-	-	-	-	-	-	-	-	172.0	19.1	10.8	-	-	18.2	9.1
High woody	Grazing	All low	N/A								-	-	-	-	1.0	0.2	0.9	1.3
		All mod									-	-	-	-	2.3	2.9	3.9	4.3
		High wood									-	-	-	-	0.0	0.7	1.0	1.0
		High herb									-	-	-	-	3.4	2.9	3.4	4.0
		All high									-	-	-	-	4.7	4.2	4.4	4.4
	Partial clearing	All low									-	-	-	-	2.0	0.9	1.2	1.3
		All mod									-	-	-	-	11.7	5.1	5.6	5.6
		High wood									-	-	-	-	6.3	3.6	4.0	4.0
		High herb									-	-	-	-	12.3	4.9	4.4	5.3
		All high									-	-	-	-	18.0	8.7	8.7	8.7
	Total clearing	All low									-	-	-	-	6.0	2.6	2.6	2.6
		All mod									-	-	-	-	17.0	7.3	7.3	7.3
		High wood									-	-	-	-	13.3	5.7	5.7	5.7
		High herb									-	-	-	-	16.3	7.0	7.0	7.0
		All high									-	-	-	-	24.3	10.4	10.4	10.4
High herb	Grazing	All low	4.0	1.2	0.7	0.6	-	1.5	1.2	1.5	0.6	0.4	0.4	0.6	-	0.4	0.3	0.4
		All mod	8.5	3.5	2.0	1.3	-	11.5	5.6	4.5	1.1	1.2	1.2	1.3	-	4.0	1.3	1.4
		High wood	24.0	4.7	2.7	1.7	-	4.0	2.0	1.0	3.2	1.8	1.6	1.7	-	1.0	0.5	0.4
		High herb	4.0	1.4	1.0	0.7	-	11.5	6.6	3.4	0.9	0.6	0.6	0.7	-	4.0	1.5	1.4
		All high	13.5	6.6	3.4	2.3	-	18.0	7.2	4.3	4.3	2.2	2.0	2.0	-	6.2	1.7	1.5
	Partial clearing	All low	14.5	2.1	1.0	0.6	-	3.0	1.8	1.5	1.9	0.7	0.6	0.6	-	1.2	0.5	0.4
		All mod	37.5	5.4	2.7	1.8	-	19.5	7.8	6.0	5.0	1.9	1.7	1.8	-	7.2	2.0	1.6
		High wood	53.5	7.9	3.8	2.5	-	14.0	5.6	4.7	7.1	2.4	2.4	2.5	-	5.0	1.4	1.2
		High herb	31.0	4.1	2.0	1.3	-	18.5	6.8	5.2	4.1	1.4	1.3	1.4	-	6.8	1.7	1.4
		All high	65.5	10.6	5.4	3.5	-	30.5	12.2	9.7	9.7	3.6	3.4	3.5	-	12.2	3.1	2.5
	Total clearing	All low	19.5	2.8	1.3	0.9	-	9.0	3.6	3.0	2.6	0.9	0.8	0.9	-	3.6	0.9	0.8
		All mod	46.0	6.6	3.1	2.0	-	25.5	10.2	8.5	6.1	2.2	1.9	2.0	-	10.2	2.6	2.1
		High wood	62.0	8.9	4.1	2.8	-	20.0	8.0	6.7	8.3	3.0	2.6	2.8	-	8.0	2.0	1.7
		High herb	37.5	5.4	2.5	1.7	-	24.5	9.8	8.2	5.0	1.8	1.6	1.7	-	9.8	2.5	2.0
		All high	86.0	12.3	5.7	3.8	-	36.5	14.6	12.2	11.5	4.1	3.6	3.8	-	14.6	3.7	3.0
All high	Grazing	All low	N/A								-	-	-	9.7	-	-	-	-
		All mod									-	-	-	22.3	-	-	-	-
		High wood									-	-	-	13.0	-	-	-	-
		High herb									-	-	-	8.3	-	-	-	-
		All high									-	-	-	35.3	-	-	-	-
	Partial clearing	All low									-	-	-	9.7	-	-	-	-
		All mod									-	-	-	27.3	-	-	-	-
		High wood									-	-	-	38.0	-	-	-	-
		High herb									-	-	-	20.3	-	-	-	-
		All high									-	-	-	54.0	-	-	-	-
	Total clearing	All low									-	-	-	13.0	-	-	-	-
		All mod									-	-	-	30.7	-	-	-	-
		High wood									-	-	-	41.3	-	-	-	-
		High herb									-	-	-	25.0	-	-	-	-
		All high									-	-	-	57.3	-	-	-	-

possible to calculate a robust offset ratio due to insufficient gains on the offset site.

Figure 8. Empirical data used to construct expert models for the dynamics of a) native canopy cover and b) species richness after restoration or disturbance. Fifty years is the nominal age for regrowth or previously disturbed remnant stands. A and C are unpublished data, B is data from Nichols (2005).

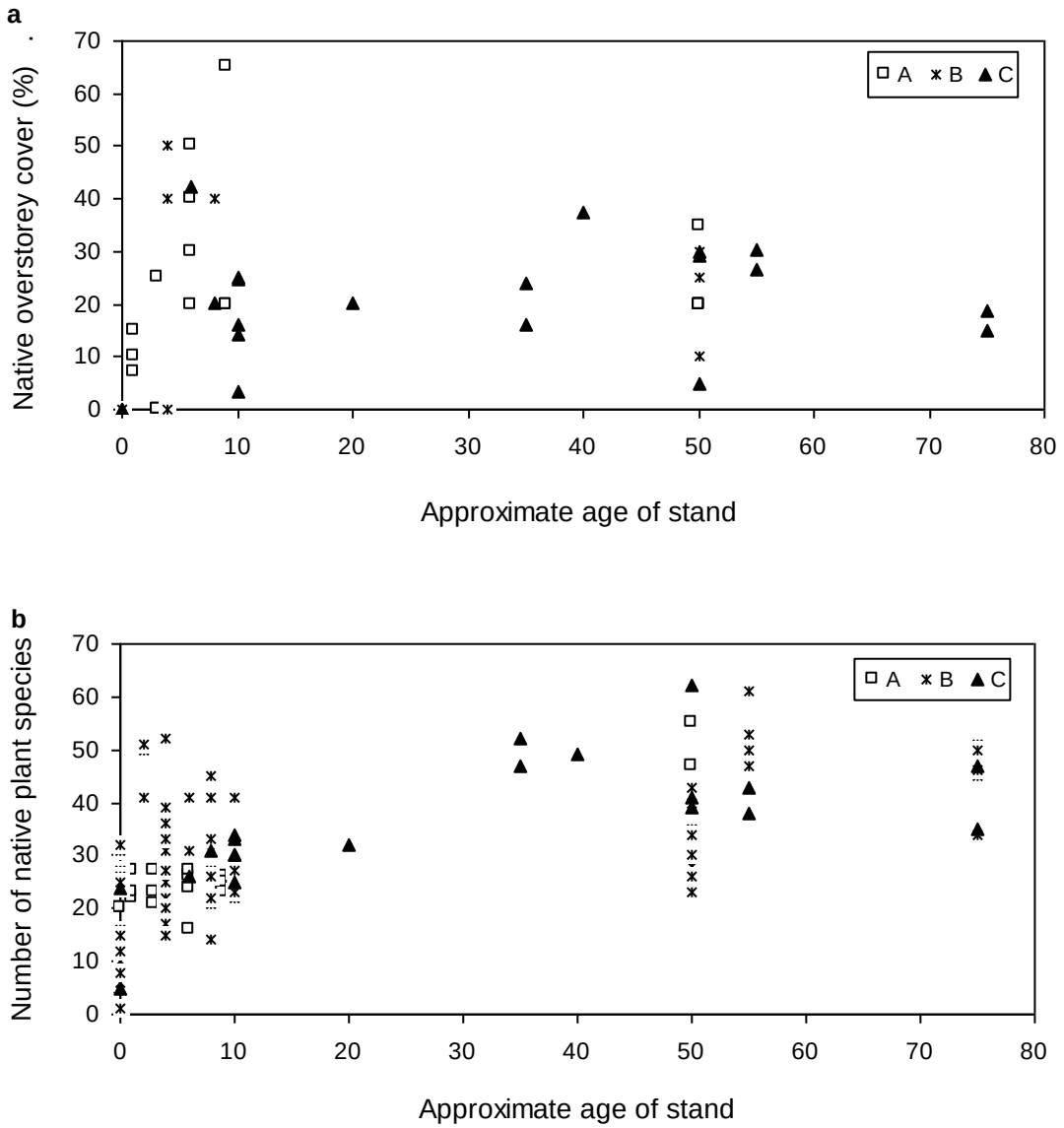


Figure 9. Examples of models for change in vegetation attributes: a) native midstorey cover and b) native ground cover on the *high herbaceous* site under the planting management scenario.

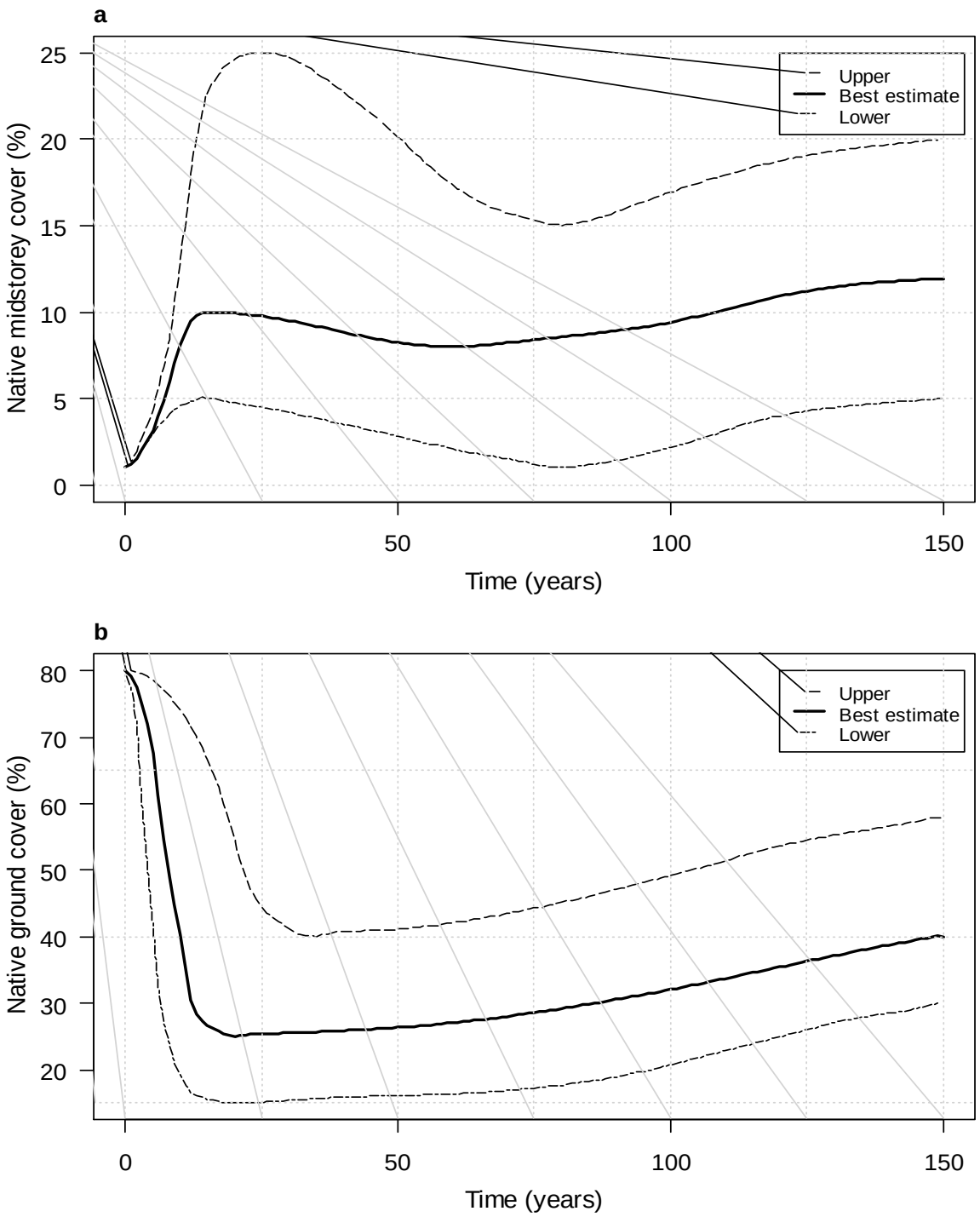


Figure 10. Predicted losses due to total clearing on five hypothetical site types, according to the BioMetric and Habitat Hectares protocols.

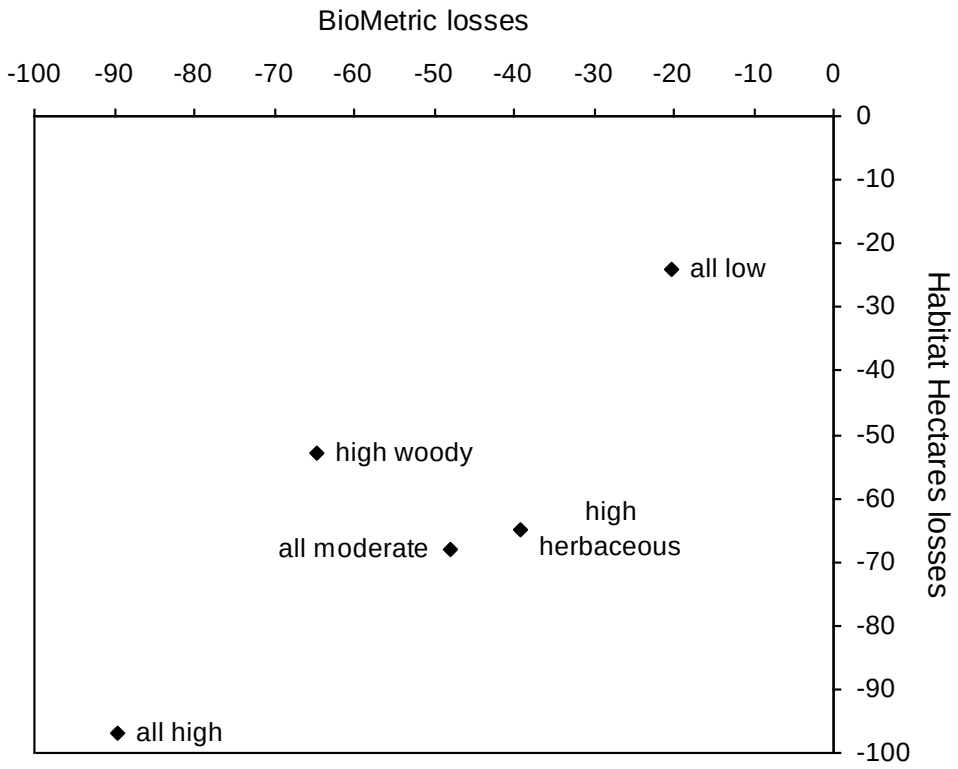
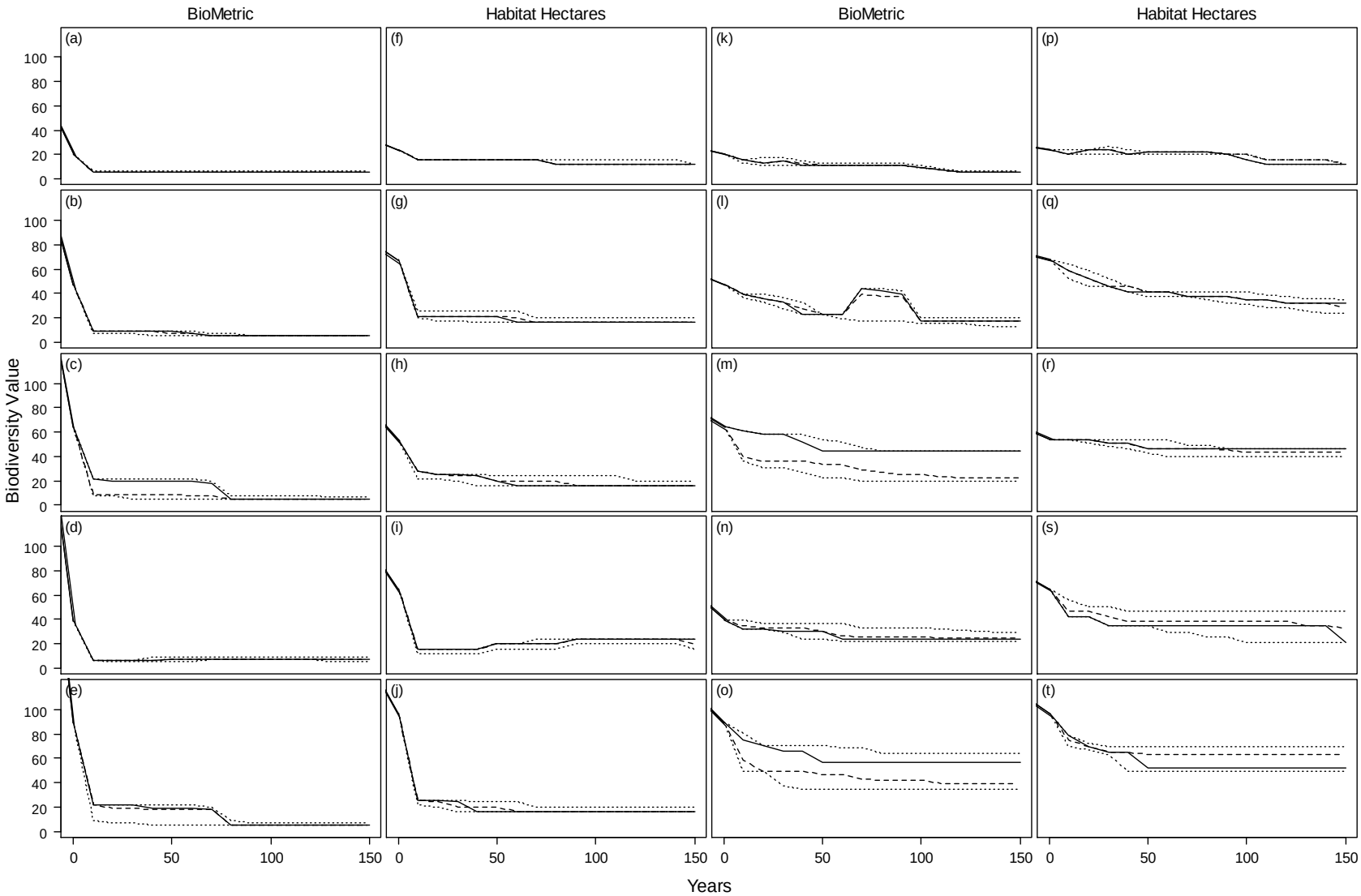


Figure 11. Predicted change over time in BioMetric (a-e; k-o) and Habitat Hectares (f-j; p-t) scores under Partial clearing (a-j) and Grazing (k-t) scenarios for all site types: *all low* (1st row) *all moderate* (2nd row), *high woody* (3rd row), *high herbaceous* (4th row), *all high* (5th row).



Unbroken line is best estimate, dashed line is median and dotted lines are 95% confidence intervals.

Figure 12. Predicted changes over time for BioMetric (a-e) and Habitat Hectares (f-j) scores under the No action scenario for all site types: *all low* (1st row), *all moderate* (2nd row), *high woody* (3rd row), *high herbaceous* (4th row), *all high* (5th row). Unbroken line is best estimate, dashed line is median and dotted lines are 95% confidence intervals.

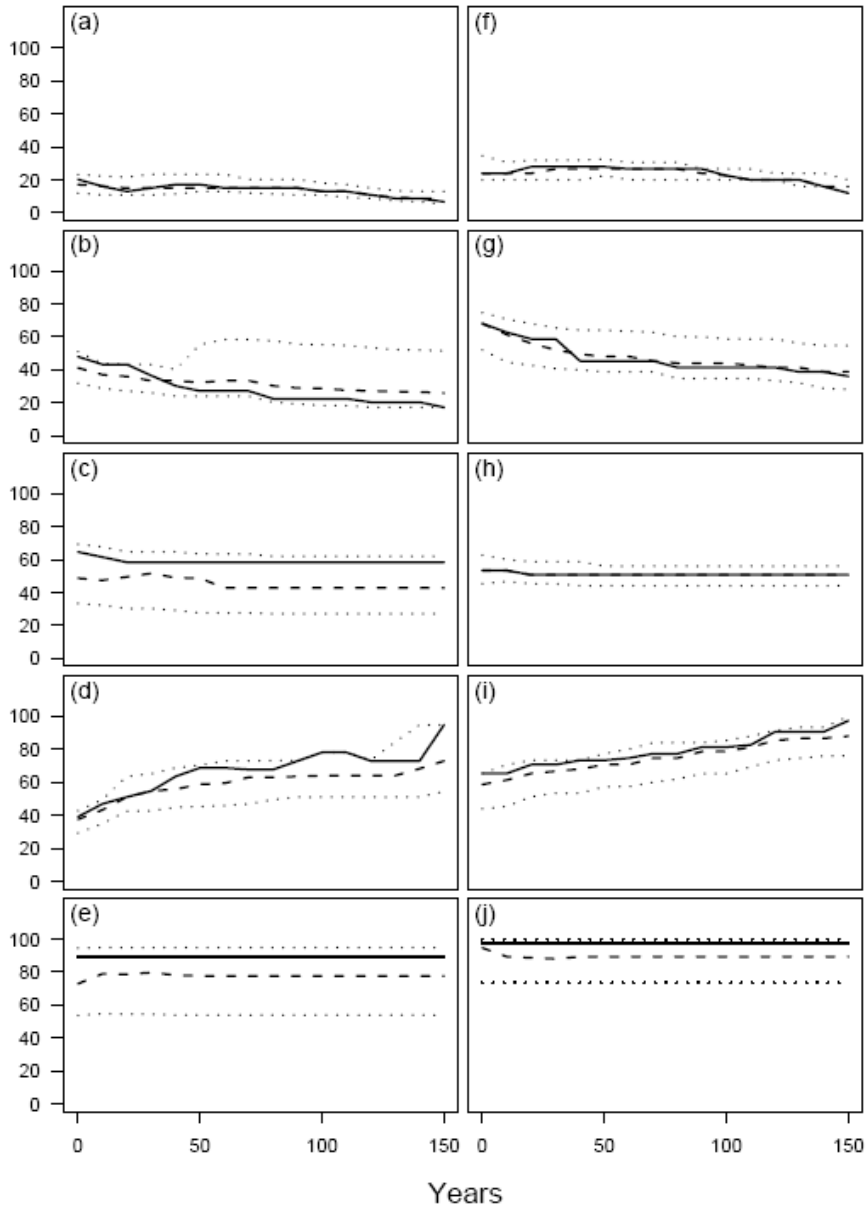


Figure 13. Predicted changes over time for BioMetric (a-e; k-o) and Habitat Hectares (f-j; p-t) scores under the Planting (a-j) and Mixed management (k-t) scenarios for all site types: *all low* (1st row), *all moderate* (2nd row), *high woody* (3rd row), *high herbaceous* (4th row), *all high* (5th row). Gains predicted by DECC (2007b) and DSE (2006) are shown as horizontal lines on BioMetric and Habitat Hectares graphs respectively. Graphs c, e, h and j are blank as planting was not simulated for the *high woody* or *all high* sites. Unbroken line is best estimate, dashed line is median and dotted lines are 95% confidence intervals.

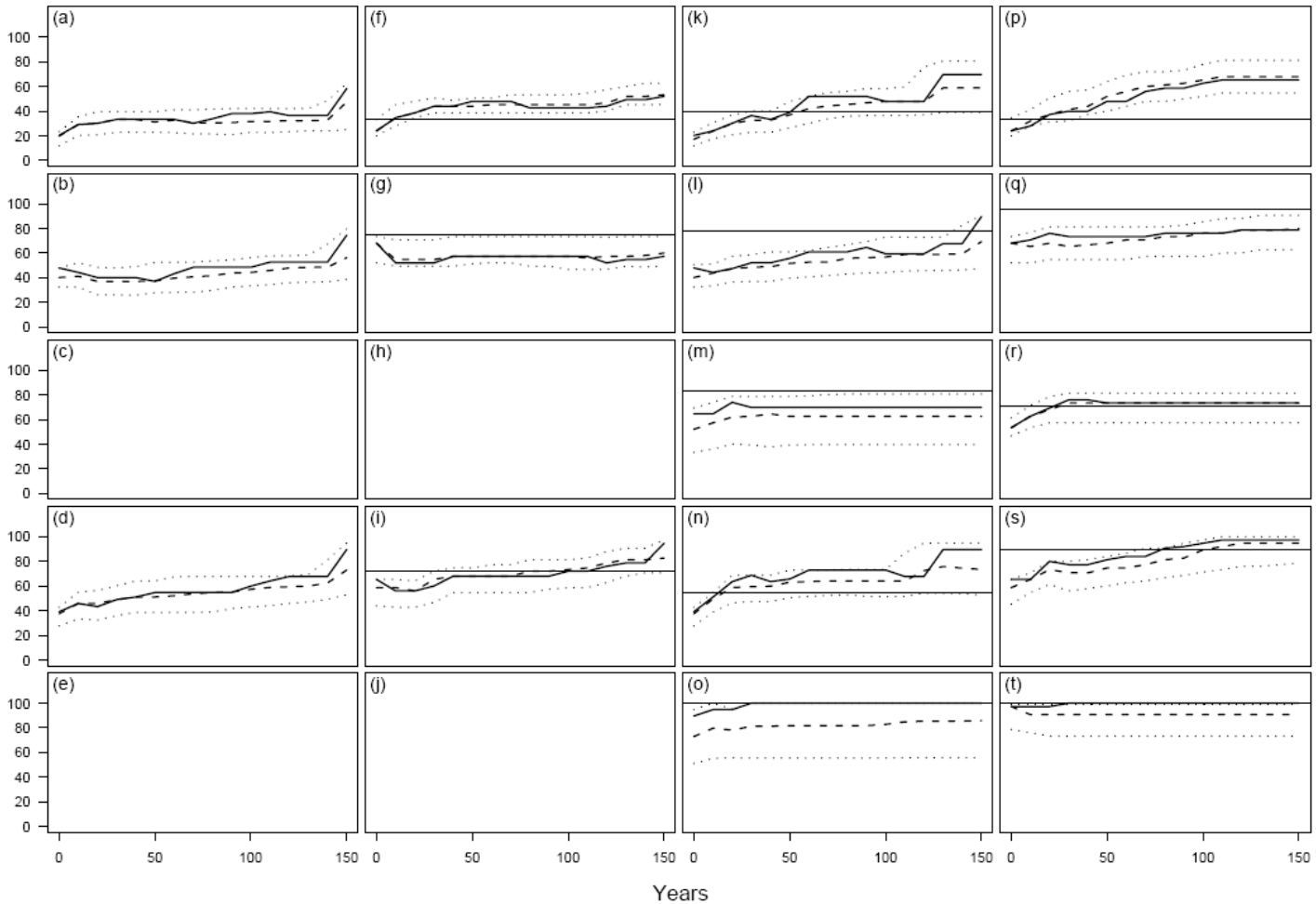


Figure 14. Examples of using empirical cumulative density function curves to examine the risk that the combined value of sites will fall below the initial score (vertical dotted line) at any time over 150 years. Here, the *all low* site is grazed and the *high herbaceous* site is managed using Planting, according to a) BioMetric and b) Habitat Hectares. For BioMetric, there is a 50% chance that the combined score will fall below the initial score of 59, and no possibility that it will fall below 45; whereas according to Habitat Hectares, there is a 100% chance that the combined value will be 6% less than the starting value of 89.