Evaluating off-site environmental mitigation using choice modelling*

Geoffrey N. Kerr and Basil M.H. Sharp[†]

Evaluation of off-site mitigation entails comparison of utility changes between two sites. Choice modelling has been used to identify community willingness to trade-off attributes for two different types of stream in New Zealand. Estimated utility functions are used to derive marginal rates of substitution and stream attribute part worths which can be used to design or evaluate both on-site and off-site mitigation policy. Latent class multinomial logit models identified classes of citizens who valued stream attributes quite differently. Significant differences in values for some attributes on different stream types imply heterogeneous mitigation ratios across environmental attributes.

Key words: choice modelling, mitigation, water management and policy.

1. Introduction

Environmental regulations frequently require offsetting mitigation for negative impacts of development activities. Mitigation can occur on-site or off-site. On-site mitigation entails enhancing environmental attributes at the development site to compensate for environmental harm caused at that site. In many situations on-site mitigation is not feasible and the focus transfers to off-site mitigation, in which environmental attributes are improved elsewhere to compensate for environmental harm at the development site. Choice modelling is a stated preference approach that can be used to assess community preferences and identify trade-offs to satisfy environmental managers' needs for information on community acceptance of proposed mitigation. Because choice modelling values the attributes associated with development and mitigation activities it offers distinct advantages over contingent valuation and is increasingly being applied in the appraisal and design of mitigation policy.

Wetland management can be used to illustrate the potential advantages of choice modelling for assessing mitigation options. It is recognised that wetlands provide significant economic benefits (Woodward and Wui 2001), and the economic dimension is important for wetland managers and for wetland management agencies (Stone 2002) even though economic factors are not

† Dr Geoffrey N. Kerr (email: kerr@lincoln.ac.nz), Christchurch. Lincoln University, Canterbury, New Zealand. Bail M.H. Sharp (email: b.sharp@auckland.ac.nz) Department of Economics, University of Auckland, Auckland, New Zealand.

always considered in making decisions about wetland management. Federal agencies in the USA require mitigation as a condition of issuing a permit to develop wetlands. Mitigation banking is used where on-site mitigation is not feasible or is not environmentally beneficial (Zinn 1997). One hectare of wetland at site A, deposited in a mitigation bank, may not be adequate compensation for the loss of one hectare at site B. Howorth (1991, p. 142) notes 'agencies argue that one-for-one is usually inadequate compensation, and that a method which evaluates functions of the area, such as [the US Fish and Wildlife Service habitat evaluation procedure] methodology, is far more appropriate'. Four arguments favour mitigation ratios greater than one. First, the probability of successful off-site restoration is less than unity, indicating that greater enhancement off-site might be required to compensate for a unit reduction on-site (Granger *et al*. 2005). Second, mitigation typically involves a lag before off-site benefits reach their design potential (Granger *et al*. 2005). Third, restoration of partly-degraded wetlands may not be sufficient to offset total loss of high quality wetlands because the degraded wetlands already provide some ecosystem services. In other words, restoration may be partial, but loss could be complete. Fourth, decreasing marginal benefits of environmental quality and/or loss aversion may mean that gains in environmental quality may be valued lower than losses in environmental quality of the same magnitude. Examples of agencies requiring wetland mitigation ratios in excess of one include Washington State (ratios between 1 and 24), US Environmental Protection Agency (ratios between 2 and 6), and the US Army Corps of Engineers (ratios between 1.5 and 4) (USACE 2003; USEPA 2001; Granger *et al*. 2005). Some agencies use lower ratios for mitigation that is proven to be successful (e.g. USACE 2003). However, even then, some ratios exceed unity in recognition of differences in service flows from enhanced and destroyed wetlands.

Economic studies have contributed to understanding of wetland mitigation. Bauer *et al*. (2004) and Johnston *et al*. (2002) used choice modelling to model benefits from, and community preferences for, mitigation of already damaged wetlands. Their results indicate that communities place high values on use of wetlands, an aspect typically not considered in determinations of ecological equivalence. For the community, provision of recreational access and associated facilities may be more important than size, type, or ecological benefits of mitigation proposals. Lupi *et al*. (2002), also using choice modelling, show communities may require wetland mitigation ratios greater than one to maintain economic benefits subsequent to wetland draining. Existing mitigation studies all presume complete loss of the developed wetland, but the policy context sometimes entails on-site or off-site mitigation of partial loss of environmental amenity subsequent to development actions. Stream management is an example of a situation where these types of changes occur.

Environmental agencies frequently make peri-urban development conditional upon mitigation, either on-site or off-site, to compensate for a reduction in ecosystem quality. When designing mitigation rules, planners must decide on the relevant bundle of attributes and acceptable mitigation ratios. For example, small

urban streams might be described according to a range of attributes; such as, vegetative cover, water clarity, fish habitat and so on. If urban development were expected to reduce the quality of fish habitat, what improvement in fish habitat off-site would be sufficient? Or, would requiring the developer to invest in off-site stream-bank planting adequately compensate for on-site diminution in fish habitat?

The research reported in this paper focuses on the application of choice modelling to a situation where mitigation is an instrument of environmental policy. The application goes beyond earlier studies because it measures willingness to trade site attributes both at and between damaged and mitigated sites. This advance allows mitigation to be tailored to the specific environmental changes proposed for the damaged site and does not assume complete site destruction. We assume that mitigation occurs instantaneously, with certainty, and illustrate how choice modelling can be used to quantify cross-site mitigation ratios in physical and/or monetary terms. The next section describes the case study. Section three describes the approach, choice model design, and data collection. Results are presented in Section four. Section five illustrates how the results can be used to measure the costs of environmental damage, the benefits of environmental enhancement and the effectiveness of on-site and off-site mitigation. Section six reports on mitigation ratios and conclusions are drawn in Section seven.

2. Case study

Every year hundreds of hectares of land in New Zealand's Auckland region are disturbed for transportation, housing, industrial, commercial and community amenity purposes. Major development activities are controlled by the Auckland Regional Council, which requires an earthworks consent for each development. Most applications for earthworks consents are associated with small first or second order soft-bottomed streams in retired pasture. These streams are usually ecologically degraded before any development occurs. Other developments involve activities in or near relatively pristine waterways, and disturbance or removal of native vegetation, which provides habitat and food sources for both terrestrial and aquatic species.

Sedimentation is a particular concern in the Auckland region because of the combination of soils, weather, topography and receiving environment attributes. There is up to 100 times the sediment yield per hectare from construction sites compared to pastoral land. Adverse ecological effects of sediment include: modified or destroyed instream values; smothering and abrading of fauna and flora; changes in food sources; and interruption of life cycles. In addition, the quality of water usually diminishes and there is loss of aesthetic appeal.

Projects in the Auckland Region involving land disturbance are required to incorporate best practice erosion and sediment controls. However best management practices are not 100 per cent effective and, even with appropriately designed and maintained systems in place, significant sediment discharges and other environmental impacts occur because of development. Residual sedimentation can lead to significant cumulative effects within catchments. In

addition to requiring best management practices, the Auckland Regional Council has the ability to place conditions on earthworks consents, including specific offsetting mitigation requirements. Offsetting mitigation may augment stream quality at one site to compensate for the adverse environmental effects associated with development at other sites. Enhancement could occur within the catchment undergoing development and/or in other catchments.

The method for establishing 'appropriate mitigation' in Auckland is far from clear and generally relies on a 'best professional judgement' approach based on ecological indicators such as species diversity, stream cover, flow rate, temperature, and so on. In order for offset mitigation to function effectively as required by New Zealand's Resource Management Act (1991) and Local Government Act (2002) the community needs to have confidence in the mitigation process. However, very little is known about community preferences regarding alternative states of Auckland streams. Without information on community preferences it is not possible for the Auckland Regional Council to identify mitigation that reflects the environmental outcomes the community desires. Choice modelling was employed to identify stream attributes that the community considers to be important and to quantify the rate at which the community is prepared to trade-off attributes across sites.

3. Approach

3.1 Choice model design

Choice models are based on random utility theory (Ben-Akiva and Lerman 1985). For some choice alternative (k) utility for any individual (i) consists of an observable component (V_{ik}) and an unobservable, or random, component (ε_{ik}).

$$U_{ik} = V_{ik} + \varepsilon_{ik}$$

Ben-Akiva and Lerman (1985) note that in real choices the unobservable component can arise for several reasons including unobserved attributes, unobserved taste variations, measurement errors and instrumental variables. In the choice modelling environment taste variations are probably most prominent because the analyst controls the attributes and there is no measurement error. The probability of individual i selecting alternative n is

$$P_i(n) = \text{Prob}[V_{in} + \varepsilon_{in} \geq V_{im} + \varepsilon_{im}] \forall m \neq n$$
$$= \text{Prob}[V_{in} - V_{im} \geq \varepsilon_{im} - \varepsilon_{in}] \forall m \neq n$$

The observable component of utility is specified as:

$$V_{ik} = V(Z_{ik}, Y_{ik}) = \beta_0 + \beta_1 Z_{1ik} + \beta_2 Z_{2ik} + \ldots + \beta_n Z_{nik} + \beta_Y Y_{ik}$$
$$= \beta Z' + \beta_Y Y_{ik} \tag{1}$$

Where Z_{jik} are environmental attributes (or transformations of attributes), and Y_{ik} is the cost to the individual. The attributes differ between alternatives,

but coefficients in the utility function do not. Assumptions about the nature of the error terms (ε) determine the form of the model used to estimate the indirect utility function, V.

Extending the indirect utility function to incorporate two sites (suppressing i and k for clarity) yields:

$$V = \beta_0 + [\beta_{11}Z_{11} + \ldots + \beta_{n1}Z_{n1}] + [\beta_{12}Z_{12} + \ldots + \beta_{n2}Z_{n2}] + \beta_Y Y \tag{2}$$
$$= \beta_0 + A[\mathbf{Z}_1] + B[\mathbf{Z}_2] + \beta_Y Y$$

Where β_{st} is marginal utility of attribute s at site t and Z_{st} is the level of attribute s at site t. $\mathbf{Z}_1$ and $\mathbf{Z}_2$ are vectors of attributes at the two sites. With off-site mitigation attributes vary simultaneously at two sites. For example, attributes at site a may be degraded, resulting in a decrease in the magnitude of utility obtained from that site ($A[\mathbf{Z}_1]$), but this could be offset by an increase in the magnitude of $B[\mathbf{Z}_2]$ because of enhancements at site b.

On-site mitigation requires that degradation of an attribute at site a (say Z_{1a}) is offset by changes in another attribute or attributes at site a. Attributes (and utilities) at other sites remain constant. This situation is illustrated with a two attribute case that identifies the change in attribute 2 at site a that does not diminish utility after a change in attribute 1 at site a.

$$\mathrm{d}U = \mathrm{d}A = \beta_{1a}\mathrm{d}Z_{1a} + \beta_{2a}\mathrm{d}Z_{2a} \geq 0$$

$$\mathrm{d}Z_{2a} \geq -\frac{\beta_{1a}}{\beta_{2a}}\mathrm{d}Z_{1a}$$

Off-site mitigation entails loss of utility because of a change in an attribute at site a being offset by changes in an attribute or attributes at site b. Letting a single attribute change at each site yields:

$$\mathrm{d}U = \mathrm{d}A + \mathrm{d}B = \beta_{ma}\mathrm{d}Z_{ma} + \beta_{nb}\mathrm{d}Z_{nb} \geq 0$$

$$\mathrm{d}Z_{nb} \geq -\frac{\beta_{ma}}{\beta_{nb}}\mathrm{d}Z_{ma}$$

For multiple attribute changes at each site, mitigation that leaves utility invariant is:

$$\mathrm{d}U = \sum_i \sum_j \beta_{ij}\mathrm{d}Z_{ij} = 0 \tag{3}$$

While the two site model allows identification of off-site mitigation, an extremely useful by-product is the ability to evaluate the adequacy of on-site mitigation (or a mixture of on-site and off-site mitigation) using the same model.

Attribute part worths (or implicit prices, m_{ij}) are simply the attribute coefficients divided by the negative of the money coefficient: $m_{ij} = -\beta_{ij}\beta_Y^{-1}$. Change in monetary value (ΔM) is then:

$$\Delta M = \sum_{i}\sum_{j} m_{ij}\mathrm{d}Z_{ij} = -\mathrm{d}U\beta_Y^{-1} \qquad (4)$$

3.2 Data collection

Auckland Regional Council personnel assisted with an initial taxonomy of stream attributes which were then presented to focus groups drawn from the case study community. Focus group participants contributed to further refinements of the attribute list as well as providing an indication of individual willingness to complete choice questions about stream attributes and perspectives on stream management. The likelihood of self-selection to focus groups on the basis of personal preferences vis-à-vis stream management was minimal because participants had no prior information on the specific purpose of the focus group meetings.

Strong views were expressed that the people creating degradation should be held responsible and should be required to pay for mitigation. However, community funding of stream improvement activities was considered to be acceptable if there was an element of 'publicness' associated with enhancement. The focus group studies indicated that stream attributes could be described in relatively simple terms that could be understood by the general public. Participants understood the idea of a choice game and were prepared and able to carefully consider the trade-offs and make meaningful choices.

The choice tasks entailed survey participants choosing between three alternatives. Each alternative was specified by four attributes at Site A and by five attributes at Site B. Sites A and B were two unnamed streams in the respondent's neighbourhood. The tenth attribute was money cost of the alternative (Table 1). Each choice included the status quo (clearly labelled as such) and two unlabelled alternatives. Attribute levels of the status quo were clearly identified and were identical for each choice task. The use of at least three alternatives provides more information from each choice event, which improves model fit and the accuracy of coefficient estimates (Rolfe and Bennett 2003). The two alternatives to the status quo entailed degradation of

Table 1   Attributes

Attribute	Attribute values: natural stream	Attribute values: degraded stream
Water clarity	Clear, Muddy	Clear, Muddy
Native fish species	1, 3, 5	2, 3, 4
Fish habitat	2 km, 3 km, 4 km	1 km, 2 km, 3 km
Native streamside vegetation*	Little or none, Moderate, Plentiful	Little or none, Moderate, Plentiful
Channel form	Natural	Straightened, Natural
Cost to household	$0/year, $20/year, $50/year	

The status quo is defined by the underlined attribute values.
*Dummy-coded, with little or none as the base.

Site A and enhancement of Site B relative to the status quo. Hence, mitigation for damage to Site A occurs at Site B. Respondent choices allow identification of whether the alternatives to the status quo provide adequate mitigation at Site B for the proposed damages to Site A plus the costs faced by the individual.

The first alternative to the status quo in each choice was developed from an orthogonal fractional factorial statistical design that permitted estimation of main effects (Hahn and Shapiro 1966). The second alternative to the status quo was the fold over of the first alternative (Louviere *et al.* 2000). Sample size limitations precluded adoption of an experimental design that included interactions between site attributes, although their potential importance is acknowledged. For example, willingness to pay for additional fish species might depend upon the amount of fish habitat available, suggesting a possible interaction between number of fish species and available fish habitat.

The experimental design required 27 different attribute profiles. Because of the relatively large number of attributes in the choice sets, the number of choice tasks faced by each individual was limited to five to reduce fatigue. This may have been an overly cautious response. Subsequent investigation of the influence of design dimensions (Caussade *et al.* 2005) confirms an increase in error variance with additional attributes, but indicates minimal error variance with about nine choice situations, albeit with fewer attributes than used here. Similarly, Arentz *et al.* (2003) found a significant increase in error variance in moving from three to five attributes, but did not find a fatigue effect when the number of choice situations was increased. Six different versions of the survey were required, with some profiles occurring in more than one version. Profiles were randomly allocated to each survey version.

The payment vehicle was regional council rates. Justification for this vehicle was provided with the following introduction, which was read out by the interviewer.

'Stream restoration and management can be expensive. Sometimes it is obvious who has caused stream changes and they can be made to pay to restore the condition of the stream. In other cases, the changes occurred a long time ago or have been caused by things done for the whole community. In these cases the condition of streams is a community responsibility. Regional Council rates could be raised to allow extra stream restoration activities to be undertaken. If this happened then costs to your household would increase through your rates bill or, if you are renting your house, through having to pay higher rent to your landlord.

While the condition of some streams continues to decline because of new and ongoing activities, other streams are getting better because of management actions. Stream managers have to decide whether it is better to try to protect streams that have not been changed much, or to restore streams that have already been degraded. Sometimes it is much easier and cheaper to restore streams that have already been degraded.

Restoring degraded streams can mean there is less money available to manage other streams, so their condition can decline.'

The statement was designed to ensure that survey participants were aware that it is not always possible to identify the people responsible for environmental degradation, yet the community may benefit from improving damaged environments. It also sought to introduce the concepts of opportunity costs through environmental trade-offs.

Data were collected in personal interviews conducted at each respondent's own home by a professional research agency. The sample was obtained by randomly drawing individual names and addresses from registered voters in North Shore postal zones. This procedure identified start point addresses. Each start point was used to obtain five interviews. From the start point interviewers approached every second house. At least two calls were made to each house where no response was obtained. The response rate was 44 per cent, with 308 interviews completed. Surveying was undertaken in January and February 2003.

4. Results

4.1 Models

A simple multinomial logit model produced significant parameter estimates, but had poor overall fit (Table 2). Latent class models account for preference heterogeneity within the survey population by allocating people to groups based on similarity of their responses to the choice questions. Boxall and Adamowicz (2002) have shown the usefulness of latent class models for improving understanding of choice experiment outcomes. Two-, three- and four-class models were tested, but four-class models failed to converge. Models were estimated with NLOGIT 4.0, which utilises both a proprietary optimisation routine and the BHHH estimator.

The number of children in the household was the only significant determinant of class assignment. However, models that included class assignment variables showed little improvement in fit. For example, the two-class model that includes class selection parameters (Table 2) has marginally better ρ^2 and Akaike Information Criterion (AIC) scores, but a marginally worse Bayesian Information Criterion (BIC) score than the two-class model that does not include class selection parameters. The three-class model, likewise, has better ρ^2 and AIC scores, but a worse BIC score than the two-class models. On the grounds of parsimony, the two-class model without class selection parameters forms the focus of the remainder of this article.

A likelihood ratio test shows that the two-class latent class model is vastly superior to the multinomial logit model ($\chi^2 = 325.54$, 14 degrees of freedom, $P = 0.000000$: McFadden's $\rho^2 = 0.173$ compared to 0.072 in the MNL), a result confirmed by improvements in both AIC and BIC.

Table 2 Latent class models

	Attribute	1 class (MNL)	2 Classes	2 Classes	3 Classes
Natural Stream Attributes	Water clarity	0.6377***	0.8239*** 0.6083***	0.8274*** 0.6097***	0.7117* 0.4694*** 1.1493***
	Fish species	1069***	0.1772*** 0.1025***	0.1749*** 0.1037***	0.2471** −0.0451 0.2866***
	Fish habitat	0.0335	−0.0282 0.0385	−0.0194 0.0359	0.1928 0.1503** −0.0805
	Moderate vegetation	0.2751**	0.2456 0.3441*	0.2875 0.3283*	0.5844 0.5616* −0.0793
	Plentiful vegetation	0.2379***	0.0216 0.2988***	0.0533 0.2929***	0.2342 0.4669*** 0.0750
Degraded Stream Attributes	Water clarity	0.4835***	0.9222*** 0.3963***	0.8934*** 0.4047***	0.7101* 0.2849*** 0.8395***
	Fish species	0.0436	−0.2188 0.1030***	−0.2292 0.1023***	−0.4737** −0.0503 0.3469***
	Fish habitat	0.1259***	0.2039 0.1114***	0.2057 0.1122***	0.1248 0.0763 0.2855***
	Moderate vegetation	0.2276*	0.0185 0.5758***	0.0169 0.5617***	−0.3921 0.0464 0.9115***
	Plentiful vegetation	0.1713*	−0.0499 0.2302***	−0.0372 0.2269***	−0.4791 −0.1601 0.6516***
	Channel	0.5164***	0.3302 0.5676***	0.3286 0.5655***	0.7098* 1.0315*** 0.0102
	Money	−0.1045***	−0.03802*** −0.005142***	−0.03891*** −0.00521***	−0.03798*** 0.00845 −0.03070***
ASC	Change	−0.2516*	−1.1845*** 0.8953***	−1.1629*** 0.8730***	−1.0835*** 1.223*** 0.1056
Class Probabilities		1.000	0.4083*** 0.5917***		0.2781*** 0.2939*** 0.4281***
Class Assignment	Constant Kids			−0.4835*** −0.00102**	
	LL$_{\text{UNRESTRICTED}}$	1493.49	1330.72	1328.33	1293.59
	McFadden's ρ^2	0.072	0.173	0.175	0.196
	AIC	2.021	1.821	1.819	1.790
	BIC	2.067	1.917	1.919	1.936

Significance levels * (10%), ** (5%), *** (1%), $N = 1491$, LL$_{\text{RESTRICTED}} = 1609.59$.

The coefficients on *Money* are highly significant and of the expected negative sign, indicating that any particular option is less likely to be selected if it costs more. The ρ^2 statistic indicates relatively good model fit (Hensher *et al.* 2005). Significance of stream attribute coefficients is generally strong for Class Two, with nine of 11 stream attribute coefficients being significant at the 1 per cent level or better. Only three stream attribute coefficients are significant for Class One.

Significant alternative specific constants (ASCs) signify that factors other than the independent variables in the model are important determinants of choice. The negative ASC for Class One indicates aversion to change from the base case, all else being equal, whereas the positive ASC for Class Two indicates an underlying preference for outcomes that diminish the quality of natural streams but enhance degraded streams.

Interaction effects allow detection of the influence of individual or household-specific characteristics (such as respondent age and household income) on the probability of selecting a particular option. Interaction effects were tested in several ways.

- First, interactions of income variables with the variable *Money* tested income effects. The effects were significant in all cases and supported prior beliefs that wealthier respondents would be prepared to pay more for any given environmental enhancement.
- Second, independent variables were interacted with the single *Change* ASC as well as individual option ASCs to test whether personal characteristics influenced choice between the options, particularly between the status quo and either of the two change options. None of these interactions was significant.
- Third, personal characteristics were interacted with each of the site attributes to identify whether particular groups of individuals within each class valued attributes differently. Addition of interaction terms to the multinomial logit model produced a very minor improvement in fit, but was far inferior to the two-class latent class model. Similarly, while some interactions were significant in the two-class LCM (for example, older members of Class Two valued natural stream fish habitat more highly), the interactions failed to improve on AIC or BIC in the simple two-class model.

Marginal rates of substitution between any two attributes (Equation (3)) can be identified from the coefficients in Table 2. The increase in attribute i required to offset a one-unit decrease in attribute *j* is the ratio β_j/β_i. For example, for members of Class Two it is necessary to increase native fish habitat by about 1 km on a degraded stream to offset the loss of one native fish species on a natural stream [$\beta_{Fish,Natural}/\beta_{Habitat,Degraded} = 0.1025/0.1114 = 0.92$]. Marginal rates of substitution are relevant guides for policy where mitigation occurs through manipulation of the natural environment. Of course, there is an infinite combination of attribute changes that yield the initial level of

Table 3 Part worths ($ per household per year)

Part worth (95% confidence interval)		Class 1	Class 2	Weighted
Natural Stream	Water clarity	$21.67 ($9.41–$38.01)	$118.32 ($73.08–$235.52)	$78.86 ($52.58–$147.72)
	Native fish species	$4.66 ($1.18–$8.71)	$19.93 ($10.94–$41.84)	$13.69 ($8.28–$26.69)
	Fish habitat	−$0.74 (−$7.45 to $6.56)	$7.49 (−$5.77 to $25.79)	$4.13 (−$4.13 to $15.19)
	Moderate vegetation	$6.46 (−$7.87 to $22.14)	$66.93 (−$2.67 to $184.92)	$42.24 ($0.80–$115.05)
	Plentiful vegetation	$0.57 (−$13.43 to $15.65)	$58.11 ($23.19–$136.88)	$34.62 ($12.65–$81.13)
Degraded Stream	Water clarity	$24.25 ($12.92–$39.73)	$77.08 ($43.44–$158.91)	$55.51 ($34.96–$104.34)
	Native fish species	−$5.75 (−$14.00 to $2.07)	$20.04 ($4.62–$51.70)	$9.51 (−$0.24 to $27.88)
	Fish habitat	$5.36 (−$1.81 to $13.42)	$21.67 ($6.88–$48.28)	$15.01 ($5.84–$30.76)
	Moderate vegetation	$0.49 (−$15.27 to $15.59)	$111.98 ($38.64–$257.94)	$66.46 ($23.07–$154.49)
	Plentiful vegetation	−$1.31 (−$15.99 to $13.65)	$44.77 ($15.48–$102.97)	$25.95 ($7.38–$60.68)
	Channel	$8.68 (−$3.34 to $21.45)	$110.40 ($67.88–$206.28)	$68.87 ($43.98–$125.63)

utility, allowing design of alternative mitigation scenarios. Part worths (Table 3) are necessary to identify monetary mitigation (compensation) measures (Equation (4)).

Several part worths are not significantly different from zero. Notably, eight out of 11 Class One part worths are not significant, whereas only two are not significant for Class Two. The two non-significant Class Two attribute part worths (Natural stream fish habitat and Natural stream moderate vegetation) also were not significant for Class One. The large number of non-significant Class One part-worths indicates that Class One respondents are less environmentally concerned than people in Class Two.

4.2 Understanding

Application of choice modelling to evaluate mitigation of incomplete site loss is novel. Because the large number of attributes involved places a significant cognitive burden on participants, and the existence of two streams in the one model is conceptually more difficult to grasp than comparing two different types of sites (such as a forest and a wetland), the question of respondent understanding arises. Related is the ease or difficulty of making choices between three alternatives with 10 attributes each. These potential concerns have been addressed by inclusion of two self-evaluation questions and one interviewer evaluation question, each measured on a 1–10 scale, with 1 being extremely easy to make choices and extremely understandable. Interviewers rated respondent understanding moderately highly, concurring with respondent self-evaluations. Participants typically found choices moderately easy to make, with median scores of four and modal scores of two. In general, most respondents appear to have understood the choice task quite well.

In order to detect any potential biases because of differences in understanding, part worths have been estimated using separate multinomial logit models for two groups of respondents. The groups comprised respondents who evaluated their own understanding with a score of four or less (high understanding, 153 individuals), and respondents who evaluated their own understanding with a score of five or more (low understanding, 147 individuals). There are no significant differences between estimated part worths for the two groups, although standard deviations are 1.6 to 3.5 times bigger for the low understanding group. When a single model permitting inter-group scale differences was applied to both groups the difference in scale was not significant ($t = 1.045$). This finding concurs with Arentze et al. (2003), who did not find an increase in error variance for the less literate. There is no evidence to suggest that use of information from respondents with lower levels of understanding has systematically biased results. While it is acknowledged that the choice tasks presented to survey respondents were relatively difficult, most respondents appear to have understood what was requested of them and have been able to make well-reasoned choices.

5. Application to mitigation

On-site mitigation entails making enhancements to a damaged stream to off-set the damage that is done on that same stream. The welfare implications of enhancements and degradations on the same stream may be evaluated by comparison of attribute coefficients for that stream type. For example, loss of water clarity on a degraded stream (Class Two, $\beta_i = 0.3963$) could be mitigated by channel enhancement on the same stream (Class Two, $\beta_j = 0.5676$) (Table 2). The difference in parameters ($\beta_j - \beta_i = 0.1713$) has a 95 per cent confidence interval of 0.0054–0.3525, indicating the strong likelihood of a net gain from the proposed mitigation. The money estimate of the value of this change is a net annual gain of $33 per Class Two household (95 per cent confidence interval = $1.43–$87, Table 3).

An alternative to on-site mitigation is to offset damage to one stream by making enhancements to another stream. Table 4 illustrates how such proposals can be evaluated in money and/or utility terms. First, consider a proposal for development in a relatively natural stream. The development would entail reduced water clarity, loss of one fish species, and loss of 2 km of fish habitat. The monetary loss incurred by the average Class Two household as a result of these changes is $152 per year (95 per cent confidence interval $94–$313).[1]

A separate proposal entails enhancement for a degraded stream, involving an improvement in water clarity, reintroduction of two fish species, an additional 1 km of available fish habitat, and an increase in streamside vegetation. The average household would benefit by $251 per year (95 per cent confidence interval $136–$516) from the proposed stream enhancement. Putting the development and enhancement projects together yields positive net benefits of $98 per Class Two household (95 per cent confidence interval $4–$247), signalling that the proposed mitigation package would be acceptable to Class Two citizens for offsetting the proposed damage to the natural stream.

Monetisation is unnecessary within a class, but is important when evaluating impacts on several classes of people because of differences in scale across classes in the latent class model, which precludes direct comparison of marginal utilities. Monetisation is necessary for valid utility comparisons with more than one class because the marginal utility of money is not uniform across classes. Because the monetary valuation of the proposed package of changes is nothing more than the utility change divided by the negative of the money coefficient within a class, the money valuation and the utility change will have the same sign for a single class as long as the marginal utility of money is unambiguously positive. However, monetisation can increase uncertainty because of the introduction of the money coefficient, which has non-zero variance.

---

[1] Part worth confidence intervals have been estimated using the Krinsky and Robb (1986) procedure, accounting for correlations between parameters.

Table 4 Off-site mitigation scenario effectiveness (Class 2)

Attribute	Initial attribute levels	Final attribute levels	Attribute change (ΔZ_i)	Part worth (m_i)	Change in value ($\Delta Z_i \times m_i$)	Change in utility ($\Delta Z_i \times \beta_i$)
Natural stream						
Water clarity	Clear	Muddy	–1	$118	–$118	–0.608
Fish species	5	4	–1	$20	–$20	–0.102
Fish habitat	4 km	2 km	–2	$7	–$14	–0.077
Vegetation	Plentiful	Plentiful	0	$0	0	0
Sub-total					–$152	–0.79
(95% confidence intervals)					(–$313 to –$94)	(–0.98 to –0.62)
Degraded stream						
Water clarity	Muddy	Clear	+1	$77	$77	0.396
Fish species	2	4	+2	$20	$40	0.206
Fish habitat	1 km	2 km	+1	$22	$22	0.111
Vegetation	Little	Moderate	+1	$112	$112	0.576
Channel	Straight	Straight	0	$0	0	0
Sub-total					$251	1.29
(95% confidence intervals)					($136–$516)	(0.86–1.71)
Net environmental benefit [per household per year]					$98	0.50
(95% confidence intervals)					($4–$247)	(0.03–0.93)

Determination of the adequacy of a specific mitigation proposal is not the only use for this type of model. There may be several different potential environmental mitigation scenarios for a specific project and each is likely to entail different (known) costs for the developer. Equation (1) can be used in association with mitigation cost information to identify the lowest cost scenario that maintains current utility, potentially decreasing development costs. Further, it may be worth questioning whether environmental mitigation passes a broader cost-benefit test. Social benefit-cost analysis entails consideration of developer costs in addition to the impacts on the local community considered to this point. If mitigation costs exceed mitigation benefits then monetary compensation, if permitted by local law, may be a more efficient alternative. Aggregation of part worth changes contingent upon a development proposal indicates the extent of monetary compensation required to maintain community welfare.

6. Mitigation ratios

Mitigation ratios in excess of one are promoted partly on the grounds that restored and created sites do not provide equivalent ecological services to natural sites (Howorth 1991; Granger *et al.* 2005). The same reasoning can be applied to economic service delivery. People may perceive lower economic benefits from a rehabilitated mitigation site than from a pristine site, despite the sites having similar attributes. This proposition is tested for individual attributes by constructing site attribute differences (W_i) for the two types of site (natural and degraded) and testing whether the differences between attribute coefficients are significantly different from zero. A positive difference is equivalent to a mitigation ratio in excess of one.

Define

$$W_j = \beta_{iN} - \beta_{iD} \qquad N \equiv \text{natural}, \ D \equiv \text{degraded}$$

$$\sigma^2(W_i) = \sigma^2(\beta_{iN}) + \sigma^2(\beta_{iD}) - 2\rho\sigma(\beta_{iN})\sigma(\beta_{iD})$$

$$\sigma^2 \equiv \text{variance}, \ \rho \equiv \text{correlation}$$

$$H_0 : W_i = 0 \qquad H_1 : W_i \neq 0$$

Estimates of W_i are reported in Table 5.

The null hypothesis is rejected for only two attribute coefficient differences. Class One assigned significantly higher utility to marginal fish species in the natural stream than they did in the degraded stream. The only significant difference for Class Two was for water clarity, also valued more highly in the natural stream than in the degraded stream. These positive differences support mitigation ratios greater than unity, consistent with the hypothesis that changes at mitigated sites are less valuable than the same changes at natural sites.

Table 5 Attribute differences (standard errors)

$W_i = \beta_{iN} - \beta_{iD}$	Class 1	Class 2
Water clarity	−0.098	0.212***
	(0.323)	(0.082)
Fish species	0.396***	−0.001
	(0.151)	(0.044)
Fish habitat	−0.232	−0.073
	(0.188)	(0.053)
Moderate vegetation	0.227	−0.232
	(0.386)	(0.255)
Plentiful vegetation	0.072	0.069
	(0.365)	(0.107)

Significance levels * (10%), ** (5%), *** (1%).

7. Conclusions

This study has used choice modelling to identify community willingness to trade-off stream attributes that change concurrently at two sites. Participants understood the tasks asked of them and provided consistent responses that have allowed estimation of utility functions, marginal rates of substitution and stream attribute part worths. The estimated values permit the evaluation and design of mitigation to offset stream damages and provide the information necessary for the assessment of mitigation alternatives. Thus, community values associated with a range of mitigation scenarios can be evaluated, provided cost data are available, to identify the cheapest mitigation option available to offset project impacts.

Adequacy of mitigation may be explored using choice experiments, whether or not they include monetary attributes. Comparison of attribute parameters (marginal utilities) or of money value estimates of welfare change both permit tests of the welfare impacts of mitigation at one site for environmental degradation at another. Exclusion of a money attribute has much to commend it. Many of the problems encountered in applying stated preference techniques arise because of the need to include monetary payments, including the need to identify a suitable payment vehicle, payment vehicle biases, lack of trust in agencies using money for the activities for which it is pledged, and anchoring biases, among others (Mitchell and Carson 1989; Bateman *et al*. 2002). Reducing the number of model attributes is also important because the necessary duplication of attributes across sites limits available statistical designs and results in increased error variance (Arentze *et al*. 2003; Caussade *et al*. 2005). If single class models[2] provide satisfactory fit, then the necessity to compare utilities across classes of the latent class model does not arise and welfare change can be signed without monetisation. However, inability

---

[2] Such as multinomial, nested and mixed logit and multinomial probit models.

to monetise values precludes the possibility of undertaking social benefit–cost analysis or assessing monetary compensation. The nature of this trade-off may be elaborated by research, but for now risk-aversion dictates inclusion of a money cost attribute.

An unexpected result was non-significance of fish habitat in the natural stream, except for one group in the three-class model. This result may be related to confounding because of lack of attribute interactions in the statistical design, or because of excluded (undesirable) site attributes that survey participants believe are associated with increased fish habitat in the natural stream. Alternatively, the marginal value of habitat may be relatively low in natural streams where it is more abundant (range 2–4 km, initial level 4 km) than in degraded streams (range 1–3 km, initial level 1 km). None of these propositions is testable with the data used here, suggesting the need for further research.

Significant differences were found in coefficients for the same attribute on different stream types, but only for two of 10 attributes tested. In both cases where differences were significant the attribute was more valuable on a natural stream. This result indicates that there may be some cases where mitigation ratios greater than one are justified on economic grounds, but that the practise is not universally endorsed, even in cases for which temporal and risk elements are eliminated (say in the case of a fully functioning mitigation bank).

Better statistical design and a larger sample size may have been helpful for resolving some of these inconclusive findings. Design efficiency tests on the simple multinomial logit model (Scarpa and Rose 2008) suggest that any major increases in statistical power would need to come from a larger sample. S-efficiency scores for the worst performing parameters are 66.9 per cent for degraded stream fish species, and 71.6 per cent for natural stream fish habitat. S-efficiency of the money parameter is 81.3 per cent. Required sample sizes for significance at the 95 per cent confidence level in the simple MNL model are 1313 (degraded stream fish species) and 2009 (natural stream fish habitat), but these decline to 878 and 1439, respectively, after S-optimisation. All other parameters required considerably fewer than the 301 people in the final sample, except for degraded stream vegetation attributes that required a sample of about 350 people.

A likely limitation of the existing study is the use of a linear utility function without interactions between site attributes. The identity between willingness to pay and willingness to accept compensation measures imposed by the linear utility function is not consistent with theoretical or empirical results (Horowitz and McConnell 2002). However, errors introduced by this restriction are likely to be small when part worths are small relative to income, as happens here. They are also likely to be avoided to a certain extent by the design of the study. By definition, natural stream attributes could only get worse when moving from the status quo, while degraded stream attributes could only improve. Consequently, the framing of the study predisposes it to estimate willingness to accept measures for damages to the natural stream,

and willingness to pay measures for enhancements to the degraded stream. This is consistent with the policy question frame. However, investigation of non-linear utility functions, although constrained by design implications with the large number of attributes in these models, provides a worthwhile avenue for future development of this approach.

The application of choice modelling to evaluate the adequacy of off-site mitigation is novel. The initial test of the approach reported here indicates that members of the public understood the relatively complex tasks they were asked to complete, and gave serious consideration to the choices offered to them. Fitted models conform to prior economic expectations and explain a high level of variance in responses, consistent with the hypothesised underlying utility function. While there are many unanswered questions, the positive results obtained here indicate this approach provides quality policy-relevant information and would benefit from further development.

References

Arentze, T., Borgers, A., Timmermans, H. and DelMistro, R. (2003). Transport stated choice responses: effects of task complexity, presentation format and literacy, *Transportation Research Part E* 39, 229–244.

Bateman, I.J., Carson, R.T., Day, B., Hanemann, M., Hanley, N., Hett, T., Jones-Lee, M., Loomes, G., Mourato, S., Özdemiroglu, E., Pearce, D.W., Sugden, R. and Swanson, J. (2002). *Economic Valuation with Stated Preference Techniques: a Manual*. Edward Elgar, Cheltenham, UK.

Bauer, D.M., Cyr, N.E. and Swallow, S.K. (2004). Public preferences for compensatory mitigation of salt marsh losses: a contingent choice of alternatives, *Conservation Biology* 18, 401–411.

Ben-Akiva, M.E. and Lerman, S.R. (1985). *Discrete Choice Analysis*. MIT Press, Cambridge, MA.

Boxall, P.C. and Adamowicz, W.L. (2002). Understanding heterogeneous preferences in random utility models: a latent class approach, *Environmental and Resource Economics* 23, 421–426.

Caussade, S., Ortúzar, J., Rizzi, L.I. and Hensher, D.A. (2005). Assessing the influence of design dimensions on stated choice experiment estimates, *Transportation Research Part B* 39, 621–640.

Granger, T., Hruby, T., McMillan, A., Peters, D., Rubey, J., Sheldon, D., Stanley, S. and Stockdale, E. (2005). *Wetlands in Washington State - Volume 2: Guidance for Protecting and Managing Wetlands*. Washington State Department of Ecology, Publication #05-06-008, Olympia, WA.

Hahn, G.J. and Shapiro, S.S. (1966). *A catalogue and computer programme for the design and analysis of symmetric and asymmetric fractional factorial experiments*. Report No. 66-0-165, General Electric Research and Development Centre, New York.

Hensher, D.A., Rose, J.M. and Greene, W.H. (2005). *Applied Choice Analysis: a Primer*. Cambridge University Press, Cambridge, UK.

Horowitz, J.K. and McConnell, K.E. (2002). A Review of WTA/WTP Studies, *Journal of Environmental Economics and Management* 44, 426–447.

Howorth, L.S. (1991). Highway construction and wetland loss: mitigation banking programs in the southeastern United States. *The Environmental Professional* 13, 139–144.

Johnston, R.J., Magnusson, G., Mazzotta, M.J. and Opaluch, J.J. (2002). Combining economic and ecological indicators to prioritize salt marsh restoration actions. *American Journal of Agricultural Economics* 84, 1362–1370.

Krinsky, I. and Robb, A.L. (1986). On approximating the statistical properties of elasticities, *Review of Economics and Statistics* 68, 715–719.

Louviere, J.J., Hensher, D.A. and Swait, J.D. (2000). *Stated Choice Methods: Analysis and Application*. Cambridge University Press, Cambridge, UK.

Lupi, F., Kaplowitz, M.D. and Hoehn, J.P. (2002). The economic equivalency of drained and restored wetlands in Michigan, *American Journal of Agricultural Economics* 84, 1355–1361.

Mitchell, R.C. and Carson, R.T. (1989). *Using Surveys to Value Public Goods: the Contingent Valuation Method*. Resources for the Future, Washington, DC.

Rolfe, J. and Bennett, J.W. (2003). *WTP and WTA in relation to irrigation development in the Fitzroy Basin, Queensland*. Paper presented to the 47th annual conference of the Australian Agricultural and Resource Economics Society, Fremantle, Western Australia. 12–14 February.

Scarpa, R. and Rose, J.M. (2008). Design efficiency for non-market valuation with choice modelling: how to measure it, what to report and why. *Australian Journal of Agricultural and Resource Economics* 52, 253–282.

Stone, H.L. (2002). Graphically modelling stakeholder values, *Journal of the American Water Resources Association* 38, 1019–1026.

USACE (2003). *The U.S. Army Corps of Engineers' Guidance for Wetland and Stream Mitigation and Mitigation Banking in the Omaha District*. The U.S. Army Corps of Engineers.

USEPA (2001). *Compensatory Mitigation Policy*. Wetlands Section, Water Management Division, United States Environmental Protection Agency (EPA) Region 4, Atlanta, GA.

Woodward, R.T. and Wui, Y.-S. (2001). The economic value of wetland services: a meta-analysis, *Ecological Economics* 37, 257–270.

Zinn, J. (1997). *Wetland Mitigation Banking: Status and Prospects*. Congressional Research Service, Report 97-849, National Council for Science and the Environment, Washington, DC.