ASSESSING THE EFFECTS OF URBAN STORMWATER MANAGEMENT ON THE SOCIAL WELLBEING COMMUNITIES DERIVE FROM COASTAL WATER BODIES.

Chris Batstone *, Sharleen Yalden **, Jonathan Moores**, and Mark Newton *

**** Cawthron Institute, Private Bag 2,Nelson 7010, New Zealand.***

***** National Institute of Water and Atmospheric Research (NIWA), Auckland, NZ***

ABSTRACT

This paper addresses the conference topic of interest "Sustaining and valuing the environment". In the course of our research we have encountered comments from stormwater professionals around the theme "It would be good to be able to value the difference that stormwater management makes to the condition of receiving water bodies". We discuss a new approach that contributes to the indicator suite of the stormwater decision support tool "Urban Planning that Sustains Waterbodies" (UPSW). Many professionals are uncomfortable with monetary methods to value the environment. Acknowledging the need for alternative ways of assessing environmental costs and benefits, we have developed a complementary method linking the concepts of experienced utility and ecosystem service provision. Experienced utility is the satisfaction that arises - as opposed to that anticipated - from an experience or decision. Ecosystem services are the benefits humankind derives from ecosystems. Examples are provisioning, regulation, supporting and cultural ecosystem services. This implementation of experienced utility modelling addresses the question, "How are a community's freedoms and capacities to undertake the things they value in and around urban coastal water bodies impacted by alternate catchment stormwater management approaches?" In the context of provisioning ecosystem service delivery by urban coastal water bodies we focus on experienced utility as an assessment tool, and the development of a method to collect and validate the data that informs the UPSW social wellbeing indicator. We describe the outcomes of recent research that supports the use of experienced utility data to identify vulnerabilities in cultural (amenity) and provisioning (food gathering) ecosystem service delivery by coastal waterbodies.

KEYWORDS

Urban stormwater, resilience, provisioning ecosystem services, experienced utility, wellbeing, thresholds.

PRESENTER PROFILE

Dr Chris Batstone is a Senior Resource and Environmental Economist with the Cawthron Institute. His research includes the valuation of community preferences for coastal water quality and the development of structured decision making processes to evaluate catchment scale urban stormwater effects on receiving water bodies.

1 INTRODUCTION

A pilot decision support system (DSS) has been developed which assesses urban impacts on ecosystem services provided by freshwater and coastal waterbodies, reported as indicators of environmental, economic, social and cultural wellbeing (Moores et al 2013). Further development of the DSS incorporates indicators of the resilience of those urban water bodies. While sustainability assessments deliver appraisals of a system at a given point in time, they do not provide information as to the potential for change. In contrast resilience analysis considers the potential for decline or improvement in future state of the system (Milman and Short, 2008), and the potential for discontinuities in system response variables that may occur through systemic change. Our objective is to develop an indicator metric that reflects (1) how social wellbeing derived from coastal waterbodies changes with varying contaminant loads, and (2) the resilience of wellbeing derived from provisioning ecosystem service supply by coastal water bodies to the effects of urban stormwater.

In this research urban development and stormwater management are conceived of as occurring within the setting of an urban aquatic social ecological system (SES). System resilience is influenced by both the capacity of natural elements of the system, i.e. receiving water bodies to provide ecosystem services, and the capacity of society to manage, adapt and potentially transform stormwater management to support the provision of ecosystem services. This paper focuses on the former: natural capacity as assessed through the trajectories of key biophysical variables and their proximity to critical ecological thresholds (Moores et al 2013). The location of the system relative to those thresholds in turn influences the quantity and quality of the benefits humans derive from receiving waterbodies, understood as ecosystem services. As those services are impacted, so too is social wellbeing, understood as the capacity of individuals to undertake the things they value, and in turn achieve and maintain wellbeing (Sen, 2008). Changes in wellbeing are assessed through a subjective wellbeing metric (Welsch and Ferreira, 2014a; 2014b) based in the experienced utility (E) concept (Kahneman et al., 1997).

The paper is structured as follows. We outline the research problem and develop a model of the response in E to changes in influential environmental parameters which includes specification of thresholds of concern. We then describe a research process to discover estimates of the location of those thresholds in social wellbeing experienced by expert consumers of coastal provisioning ecosystem services. A parallel enquiry process based in elicitation from expert scientists is described that seeks definition of those thresholds in ecological and biophysical data. We report convergence between ecological and expert consumer knowledge as to the location of these thresholds of potential concern in the coastal waters of urban Auckland, New Zealand, and conclude with a discussion of the outcomes of the research.

2 MAPPING WELLBEING TO COASTAL CONTAMINANT LOAD

In order to understand how the social wellbeing associated with the receiving waterbodies is affected by the combination of stormwater effects and their management it is necessary to establish the connection between stormwater contaminants and wellbeing, and develop a method to assess that connection. The aim of this research is to evaluate how well information held by consumers of coastal provisioning ecosystem services corresponds with that held by experts in the field of coastal ecology. In particular, whether those consumers' experiences of ecosystem service delivery reflect changes in the ecological branch of the social-ecological system (SES) so that their assessments can be employed in coastal planning and management processes to inform decision making and monitor system performance.

An SES is an integrated system of ecosystems and human society with reciprocal feedback and interdependence (Folke et al., 2010). This definition reflects a recognition that the scale and impact of human activities in modern times make it "difficult and even irrational to continue to separate the ecological and social and to try to explain them independently" (Folke et al., 2010). Resilience theory recognizes that SESs are subject to continuous disturbance. Walker et al., (2004) defined the resilience of an SES as the "capacity of a system to absorb disturbance and reorganize while undergoing change so as to still retain essentially the same function, structure, identity, and feedbacks."

Assessments of resilience have generally been qualitative in nature (Birkmann et al., 2012). In its guidelines for assessing resilience in SESs, the Resilience Alliance (2007) recognized that progressing from concepts to measurement is the "difficult part of the process." The crucial first step is to define what resilience means in any given context. Folke et al., (2010) distinguished between 'general' and 'specified' resilience, the latter term referring to the response of particular attributes of a system to a specific set of disturbances, or as Carpenter et al., (2001) put it, the "resilience of what to what?" The specification considered in this paper is one of specified resilience: the resilience of provisioning ecosystem services by urban coastal waterbodies to the effects of urban development.

An SES is in a state of flux, moving around within a 'basin of attraction' (Walker et al., 2004; Folke et al., 2010). Externally or internally-driven changes force movements within these basins or to alternate basins. Where a system lies close to the boundary between basins it is said to be characterized by high precariousness, and where a disturbance prompts a large response in the state of the system, it is said to have a low resistance. Walker et al., (2012) described the behavior of SESs in terms of the response of 'fast' variables to changes in the state of 'slow' variables. Fast variables are generally those that are of interest to humans, in other words the ecosystem services provided by an SES. As a slow variable approaches a threshold between stability domains (or 'regimes'), disturbances result in increasing fluctuation in fast variables, eventually pushing the system over a threshold and resulting in a change (usually reduction) in ecosystem service provision (Walker et al., 2012).

Figure 1: Scheffer Alternate Regime Model (Scheffer et al., 2001: 413: 591-596)

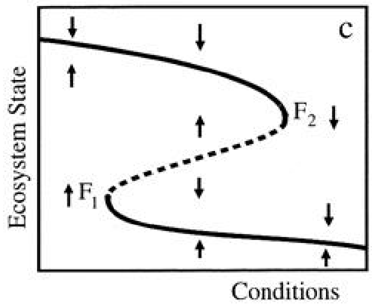
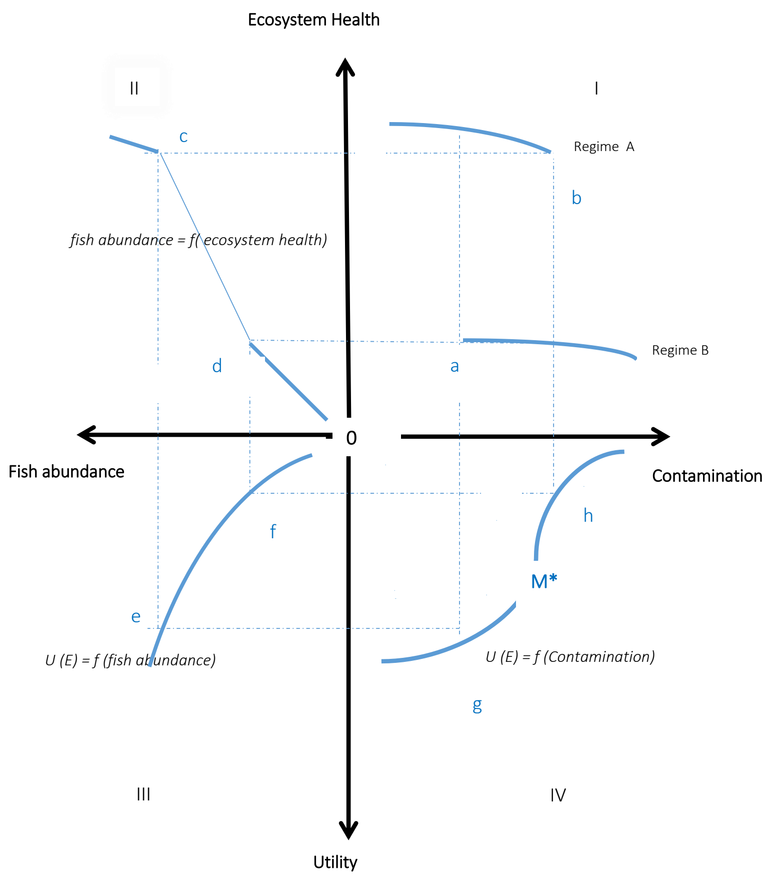


Figure1 depicts an ecological system which has two potential stable states or regimes. Ecosystem state (the "fast" variable) is modelled as function of ecosystem conditions (the "slow" variable). Between the points F1 and F2 the system has the potential to lie in one of two regimes. Outside of those points only one regime is possible: F2-F1 represents the transition pathway between the two states. In systems where there is no possible return pathway from F1 to F2 hysteresis results with subsequent permanent loss of ecosystem service provision. Environmental conditions corresponding to F1 and F2 are thresholds for the transition.

Scheffer et al., (2001) emphasize the multi-equilibria nature of ecological systems as driving variables force the system between stable (and unstable) regimes. Key considerations lie in the location of thresholds between regimes and the potential for hysteresis where the system is unable to reverse a transition from one regime to another by reversing the system conditions that motivate the change. As conditions proceed from F1 to F2 so vulnerability of transition increases, resilience of the system decreases.

The four quadrant diagram portrayed in Figure 2 is a stylized depiction of the flow effects from ecosystem to social world in an urban SES as experienced by a provisioning ecosystem services consumer. The chain of cause and effect is: Increase in contaminant (e.g. sediment) delivered to a coastal waterbody leads to changed environmental conditions (e.g. muddier bed sediments), which in turn lead to reduced water clarity. Combined, these effects result in decline in ecosystem health, which in turn leads to decline in fish and shellfish abundance, ultimately experienced by consumers as less satisfactory harvesting experiences.

Figure 2: The coastal SES: mapping contamination to utility



Quadrant I corresponds to Schaffer's regime shift model in Figure 1. It describes the regime based relationship between ecosystem health and increasing contamination, in this case sediment accumulation expressed as %mud in the coastal bed sediments (which not only influences ecosystem health directly, through changes to physical habitat characteristics, but also indirectly, through factors such as reduced water clarity). The points a and b in this quadrant represent the zone of increasing vulnerability and decreasing resilience, and correspond to F1 and F2 in Figure 1.

Quadrant II models the response of fish abundance to regime shifts in ecosystem health. Levels of ecosystem health associated with regime shift are defined in terms of c and d in this quadrant. Quadrant III proposes regimes in community provisioning ecosystem service based utility at e and f that result from declining fish abundance induced by increased contamination modelled in quadrant I. The points at e and f define the zone of increasing vulnerability / decreasing resilience of provisioning ecosystem service to declining fish abundance that results from regime changes in the coastal ecosystem. Quadrant IV extends the utility relationship from quadrant III to link with the contamination axis. The points g and h identify the zone of decreasing resilience of provisioning ecosystem service based utility to increasing contamination.

We define the point M* on the utility curve in quadrant IV as the critical point at which fishers' satisfaction changes in a significant way. M* defines the point of equivalent response (PER): the point at which declines in fish abundance produce commensurate changes in satisfaction. Up to that point increasing contamination produces less than proportionate change in satisfaction from harvest, after that increasing contamination produces ever increasing, greater than proportionate, declines in satisfaction. The point has significance from a social point of view in that with further declines in environmental quality society experiences losses more markedly after this point. Part of the research challenge is to define M* mathematically in terms of the forcing variable, contamination (expressed through changes in %mud in the coastal bed sediments).

In the following section we develop a normalized derivative approach to the mathematical derivation of M* and its ecosystem health equivalent B*.

3 MODEL

There is a precedent in the behavioral economics literature that the functional relationship between utility and environmental condition described in Figure 2 may follow a power-law model (Kahneman et al., 1997). Batstone et al (2013) have explored the collection and statistical modelling of experienced utility (Welsch and Ferreira, 2014a) data in the context of urban SES and demonstrated that the power law model proposed by Kahneman et al., (1997) holds as a model of satisfaction response to changing environmental quality. Focus group research in support of the development of a choice experiment design and its subsequent estimation (Batstone et al., 2010; Batstone and Sinner, 2010) has shown that three leading stormwater mediated influences on the quality of coastal users' experiences are underfoot condition (U), water clarity (W) and ecological health (H). The power-law model for the utility (E) experienced by expert provisioning ecosystem services consumers as a function of the three influences (Batstone et al 2013) is:

$$E = K U^{a} W^{b} H^{c}, \tag{1}$$

Where, the coefficient K and exponents a, b, and c are all dimensionless and constant.

Preliminary investigations reported in Batstone et al (2013) show that *E* is a monotonically increasing function of each of the three influences, with the parameters *a*, *b*, and *c* taking values in the range (0….1), delivering a functional relationship consistent with the notion of diminishing marginal returns.

Each of the three leading influences can be related to specific measures of a relevant underlying environmental variable. In this analysis percentage mud in the coastal bed sediments, denoted *x* (%), is adopted as a measure of underfoot condition. Turbidity, denoted *y* (NTU), is adopted as a measure of water clarity (inversely related to percentage mud) and a benthic health index (Anderson et al., 2006), denoted *z* (dimensionless), is adopted as a measure of ecological health.

Each of the categories of the variables *U*, *W*, and *H* in the choice experiment format is associated with a specific range in an underlying environmental variable as described in Table 1. The categories are assigned scores of 1 (low) through to 3 (high). Linear regression is then sufficient to describe the relationship between the category scores and the mid-points of the underlying variable ranges. As a result, the leading influences can be expressed in terms of their underlying environmental variables as:

$$\begin{aligned} U &= \alpha x + \beta, \\ W &= \gamma y + \eta, \\ H &= \lambda z + \mu. \end{aligned} \qquad (2)$$

Table 1: Category scores and underlying variables

Underlying variable	Category				
	Low	Low/Med	Med	Med/High	High
Percentage mud (%)	70 – 100	50 – 70	35 – 50	10 – 35	0 – 10
Turbidity (NTU)	> 21	16 – 21	11 – 16	6 – 11	0 – 6
Benthic Health (-)	> 1.203	0.289 – 1.203	-0.808 – 0.289	-1.897 – -0.808	< -1.897

Estimates for the parameters in equation (2) have been obtained by ordinary least squares regression (OLS) and are described in the Appendix section of the paper. Substituting these relationships in Equation (1) allows the provisioning ecosystem services experienced utility (*E*) to be expressed as a multiplicative power function of the underlying environmental variables:

$$E = K(\alpha x + \beta)^a (\gamma y + \eta)^b (\lambda z + \mu)^c. \qquad (3)$$

Equation (3) describes the way the satisfaction experienced by marine fishers (the consumers of provisioning ecosystem services) changes as the underlying environmental variables change.

The remainder of this section focuses on the derivation of a critical point on the experienced utility response function we have called the PER: the point of equivalent response. That point anticipates the rapid change in satisfaction that will follow changes in the underlying ecology generating the provisioning ecosystem services. Increasing contamination modifies ecosystem health, which in turn influences the abundance of target species, which in turn impacts the level of satisfaction experienced by fishers.

Consider the rate of change in experienced utility E that relates to underfoot condition. This is defined in terms of the first-order partial derivative of Equation (3) with respect to the underlying variable x (% mud). Similarly, the rate of change in experienced utility that relates to ecological health is defined in terms of the partial derivative with respect to z (benthic health). Equations (4) and (5) describe these partial derivatives:

$$\frac{\partial E}{\partial x} = \alpha a K(\alpha x + \beta)^{a-1}(\gamma y + \eta)^{b}(\lambda z + \mu)^{c}. \tag{4}$$

And,

$$\frac{\partial E}{\partial z} = \lambda c K(\alpha x + \beta)^{a}(\gamma y + \eta)^{b}(\lambda z + \mu)^{c-1}. \tag{5}$$

Both partial derivatives are functions of all three underlying variables, each with disparate scales and dimensions. Accordingly, we normalize as follows. Consider Equation (3). For fixed turbidity and benthic health, there is a range of experienced utility scores that may be expected based on varying % mud. The minimum score, E_{min}, is found by substituting x = 100, and the maximum score, E_{max}, is found by substituting x = 0. We let $\overline{E}$ denote the experienced utility normalized along this range of scores. As such,

$$\overline{E}(x \mid y, z) = \frac{E - E_{\min}}{E_{\max} - E_{\min}}. \tag{6}$$

Similarly, we let $\overline{x}$ denote the normalized percentage mud:

$$\overline{x} = \frac{x}{100}. \tag{7}$$

The rate of change in the normalized experienced utility with respect to the normalized percentage mud is then given by:

$$\frac{\partial \overline{E}}{\partial \overline{x}} = \left(\frac{100}{E_{\max} - E_{\min}}\right)\frac{\partial E}{\partial x}. \tag{8}$$

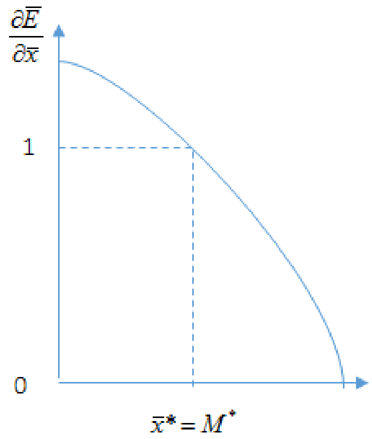
Substituting Equation (4) and the expressions for E_{min} and E_{max}, the rate of change in the normalized experienced utility with respect to the normalized percentage mud becomes:

$$\frac{\partial \overline{E}}{\partial \overline{x}} = \frac{100\alpha a(\alpha x + \beta)^{a=1}}{\beta^{a} - (100\alpha + \beta)^{a}}. \tag{9}$$

We call the derivative in Equation (9) the normalized mud gradient. Note that it depends only on percentage mud, and is independent of turbidity and benthic health. When the normalized mud gradient takes a value equal to 1 changes in % mud are associated with commensurate changes in experienced utility, i.e. a 1% change in % mud is associated with a 1% change in experienced utility. For values of the normalized mud gradient greater than 1 changes in % mud produce a larger than proportionate response to the scale of the decline, and for values of the normalized mud gradient less than 1 changes in % mud produce a less than proportionate response to the scale of the decline. Figure 3 describes the relationship between the normalized mud gradient and normalized % mud.

A similar argument may be applied to benthic health. For fixed % mud and turbidity, the minimum experienced utility score, E_{min}, is found by substituting z_{max} = 2.198 in Equation (3), and the maximum score, E_{max}, is found by substituting z_{min} = -2.781. (Maximum and minimum PC1.500 scores for classification according to pollution group (Anderson et al., 2006, p 29)).

Figure 3: Normalized % Mud Gradient



$$\hat{E}(z \mid x, y) = \frac{E - E_{\min}}{E_{\max} - E_{\min}}, \tag{10}$$

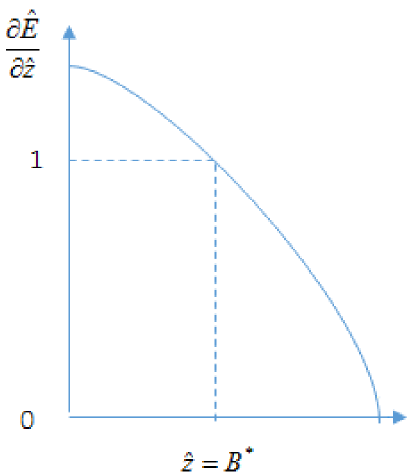
With normalized benthic health given by:

$$\hat{z} = \frac{z - z_{\min}}{z_{\max} - z_{\min}}. \tag{11}$$

The rate of change in the normalized experienced utility with respect to the normalized benthic health is given by

$$\frac{\partial \hat{E}}{\partial \hat{z}} = \left(\frac{z_{\max} - z_{\min}}{E_{\max} - E_{\min}} \right) \frac{\partial E}{\partial z}. \tag{12}$$

Figure 4: Normalized Benthic Health Gradient.



Substituting Equation (5) and the expressions for E_{max} and E_{min}, the rate of change in the normalized experienced utility with respect to the normalized benthic health becomes:

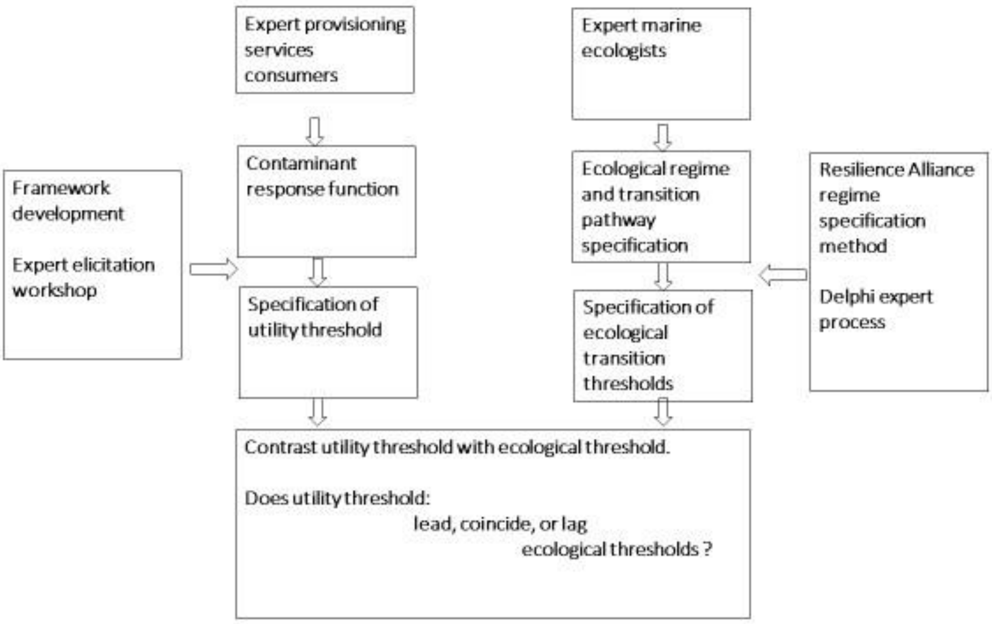
$$\frac{\partial \hat{E}}{\partial \hat{z}} = \frac{4.9790\lambda c(\lambda z + \mu)^{c-1}}{(-2.781\lambda + \mu)^c - (2.198\lambda + \mu)^c}. \qquad (13)$$

We call the derivative in Equation (13) the normalized benthic health gradient. It depends only on benthic health, and is independent of percentage mud and turbidity. Figure 4 describes the normalized benthic health gradient.

4 METHODS

The research method consists of two separate, parallel processes. First, specification of possible alternate ecological regimes in Auckland estuaries, the drivers of change, and the location of thresholds between them in terms of x and z specified as: x^*_e and z^{*e}. The second process is used to identify the parameters a, b, and c in Equation (1).This enables identification of the levels of underlying environmental variables that define the experienced utility thresholds where the normalized mud gradient and the normalized benthic health gradient equal 1. The aim of these methods is to contrast values obtained for $\bar{x}^* = M^*$ with x^*_e and $\hat{z}^* = B^*$ with z^*_e to identify in this system whether M* and B* are leading, lagged, or coincident indicators for x*e and z*e respectively. Figure 5 summarizes the overall research process and shows the links between the two strands: marine ecologists' and expert consumer workshops.

Figure 5: Research Process



4.1 MARINE ECOLOGISTS' WORKSHOPS

The right hand columns of Figure 5 summarizes the series of two Delphi expert elicitation workshops undertaken to derive an expert assessment of the ecological and biophysical specification of regimes and their associated thresholds. Four scientists with deep expertise in the ecology and physical processes of Auckland estuaries took part in the elicitation process formulated by The Resilience Alliance (2007).

The process consisted of five linked tasks:

- Task 1: Create a map of the biophysical components of a typical Auckland estuary system impacted by urban stormwater including the important independent and dependent variables and the links and feedbacks between them. Their brief was to make the output as simple or as complex as needed to identify alternate regimes and thresholds in a subsequent task.

- Task 2: Characterize the alternate regimes. Define the variables that enable discrimination between the system in alternate states /regimes, define the possible alternative states and create a table capturing this depiction of the systems possible states.

- Task 3: Describe the transition pathways in the system. Identify the key variables driving the transitions. Capture this in a summary chart.

- Task 4: Identify appropriate indicators for the state of the system that can be quantified and are appropriate for locating thresholds. These should integrate information in the system, including acting as surrogates for other variables in the system.

- Task 5: Quantify the thresholds: Identify the levels of the indicators from Task 4 that mark the points of transition between system states, or that define ranges of the levels of those indicators that define zones of vulnerability.

4.2 EXPERT CONSUMER BASED EXPERIENCED DATA COLLECTION

This part of the research has two distinct phases described by the left column of Figure 5. First, a web based survey of the Auckland Council's Peoples' Panel to identify "expert consumers" of ecosystem services delivered by Auckland's marine environments (Newton and Batstone, 2014). Prior work in this area (Batstone et al., 2013) has identified a SWB approach (Welsch and Ferreira, 2014a,2014b) using experienced utility data (Kahneman and Sugden, 2005) collected at the level of the individual to assess changes in social wellbeing reflected in use satisfaction that follow modification of receiving waterbodies of urban stormwater. To address concerns around shifting baselines (Pauly, 1995; Papworth et al., 2009) a recreation specialization method (Bryan, 1977) was employed to identify Aucklanders with deep experience in their interaction with coastal environments.

Survey respondents were asked to rate their degree of involvement in four ecosystem services categories (MEA, 2005) from little or none, to deeply involved (Needham et al., 2009). To identify expert participants in those areas the survey asked "how involved would you say you were in those activities?" Participants responded on a five point scale where 1 identified "non-experts" with "little or no involvement, there are other things I'd rather be doing; I don't spend a lot of money on these sorts of things", and 5 identified "experts": "I'm heavily involved; I take part in these activities whenever I get the chance; I'm happy to spend money on these activities." "Coastal recreation experts" were identified from respondents and asked to attend a series of data collection workshops in Auckland.

In the second phase, expert utility data was collected in workshops at various locations around Auckland. The process is described in Batstone et al., (2013). Data collection involved respondents in a six step procedure to populate a three by nine cell matrix of coastal water body quality scenarios with experienced utility scores that relate their degree of satisfaction with changing water quality defined in terms of three levels of three key stormwater mediated influences W, U, and H in equation (1.). Table 1 and equation (3) describe the categorical attributes and their relationships to the corresponding biophysical variables.

Step One: Task definition, and context specification.

Step Two: Training

Step Three: Respondents were asked to locate the cells representing scenarios that corresponded to their best, worst, and most frequently encountered experiences using "happy face" symbols or equivalent, depending on their experience.

Step Four: Respondents were asked to score these key locations based on the degree of satisfaction they recalled experiencing using an interval of {1 … 10}

Step Five: Respondents were asked to score the remaining cells relative to their best, worst, and most frequently encountered scores using the same interval of {1 … 10}

Step Six: Respondents were asked to score the reliability of the information they had provided using an interval of {1 … 10}.

5 RESULTS

5.1 MARINE ECOLOGISTS' WORKSHOPS' DATA

Figure 6 and Table 2 describe the outcomes of the marine ecology expert workshops. In Figure 6, two initial "healthy" regimes are identified, with transition pathways between them via the compromised intermediate regime corresponding to F1-F2 in Figure 1. The ultimate degraded regime corresponds to the system state where environmental conditions lie beyond F1-F2. The forcing variables are heavy metals and sediment (mud). This system diagram recognizes that not all muddy states are unhealthy:

Figure 6: Marine ecologists' workshops estuary regime and thresholds system.

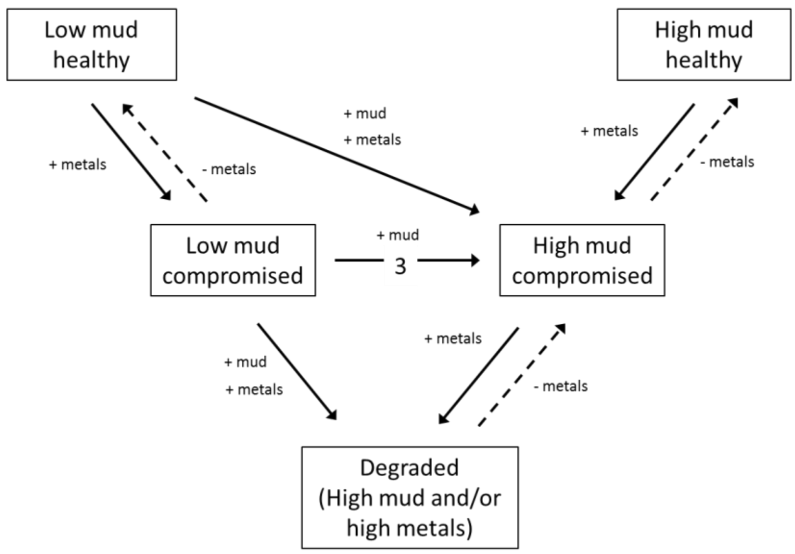


Table 2 presents the ecologists’ assessments of the levels of the significant environmental variables that define the alternative regimes presented in Figure 6 and their thresholds. Increasing levels of sediment accumulation rate (SAR) measured in millimeters per year, and total copper (Cu), lead (Pb) and zinc (Zn) measured in milligrams per kg of bed sediments indicate the progress of an estuarine location from healthy, through the compromise transitions to the degraded state.

Table 2: Estuary regime and threshold location using contaminant and benthic health indicators

Initial State		Indicator	Healthy	Threshold #1		Compromised	Threshold #2		Degraded
Mud	Metals			Location	Confidence		Location	Confidence	
Low	Low	SAR (mm/yr)	<1	1 – 2[a]	Low	2 – 10	10 – 20[b]	Low	>20
		Total Cu (mg/kg)	<10	10 – 18[c]	Medium	18 – 108	108 – 270[d]	Low	>270
		Total Pb (mg/kg)	<19	19 – 30[c]	Medium	30 – 112	112 – 218[d]	Low	>218
		Total Zn (mg/kg)	<70	70 – 124[c]	Medium	124 – 271	271 – 410[d]	Low	>410
High	Low	SAR (mm/yr)	<2	2 – 5[e]	Low	5 – 10	10 – 20[b]	Low	>20
		Total Cu (mg/kg)	<10	10 – 18[c]	Medium	18 – 108	108 – 270[d]	Medium	>270
		Total Pb (mg/kg)	<19	19 – 30[c]	Medium	30 – 112	112 – 218[d]	Medium	>218
		Total Zn (mg/kg)	<70	70 – 124[c]	Medium	124 – 271	271 – 410[d]	Medium	>410
Comparative indicators									
		% mud[f]	<10	10 – 25	High	25 – 60	60 – 80	High	>80
		BHI_{metals}	1	3 – 4	High	5	Not represented in BHI		

Comparative indicators are also described that are the underlying environmental variables described in equation (3): %mud, x^*_e and Benthic Health Index–metals (BHI_{metals}), z^*_e.

5.2 EXPERT CONSUMERS’ EXPERIENCED UTILITY DATA

5.2.1 IDENTIFYING EXPERT CONSUMERS

A total of 2817 people participated in the Auckland Council Peoples’ Panel on-line survey. The composition of the responding sample is not representative of the census demographics of the Auckland region in terms of age (older), and ethnicity (New Zealand Europeans over-represented, Asian, Pacific and Maori peoples under-represented). For the purposes of this survey, provisioning activities are defined as food gathering activities at coastal waterbodies in the Auckland region (e.g. at beaches, the sea, or lakes and streams). Vessel fishing is the most common provisioning activity, undertaken by 24% of survey respondents, followed by shore fishing (15%), and shell fishing (13%). Sixty four percent of survey respondents do not undertake activities targeting provisioning ecosystem services at Auckland coastal waterbodies. Only 30% of survey respondents had engaged in provisioning activities in the last two years: 4% of respondents self-categorized as expert (level 5), 7% as level 4, 10% as level 3, 11% as level 2, 4% as level 1 non experts, and 64% did not know.

The process identified 694 recreation specialists (i.e. Experts, level 5 on the recreation specialization scale) who were then invited by email to participate in one of five evening workshops in the months following the survey. In total, 79 experts agreed to participate – a response rate of 11%. Of these, 59 people actually attended a workshop (8.5%). The complete survey results are reported in Newton and Batstone (2014).

5.2.2 EXPERT CONSUMERS' EXPERIENCED UTILITY DATA

The 79 "experts" were offered a place at one of five workshops held mid-week, early in the evening at five locations offering close proximity to the Auckland transportation networks. Of the 59 people who attended a workshop, 27 complete responses were obtained where the self-assessed level of confidence in their responses was 8 out of 10 or better. Table 3 describes the GLS estimation outcomes for equation (1) from these data. The model Rsq = 0.97; the Anova F statistic <0.01.

Table 3: GLS estimation $E = K(U)^a (W)^b (H)^c$, *in log-log format.*

Variable	Co-efficient estimate	P -value
K	0.5190	< 0.01
a	0.5035	< 0.01
b	0.3242	< 0.01
c	0.8327	< 0.01

5.2.3 ESTIMATION OF EXPERT CONSUMER POINTS OF EQUIVALENT RESPONSE

The point estimates for the parameters a, b, and c in equation (1) described in Table 3, and the estimates of α, β, λ and μ (see Appendix) were applied to solve Equations (14) and (15). Figures 9 – 10 show the relationships between the normalized mud and normalized benthic health gradients respectively. Solutions to Equations (14) and (15) lie where $\bar{x}* = M^*$ approximately equals 60%, and $\hat{z}* = B^*$ equals 0.0336.

Figure 7 shows that the reference point where the normalized mud gradient equals one is associated with percentage mud level of 60%. Figure 8 shows that the reference point where the normalized benthic health gradient equals one is associated with a benthic health index score located in Group 3.

Figure 7: Chart of normalized mud gradient and %mud

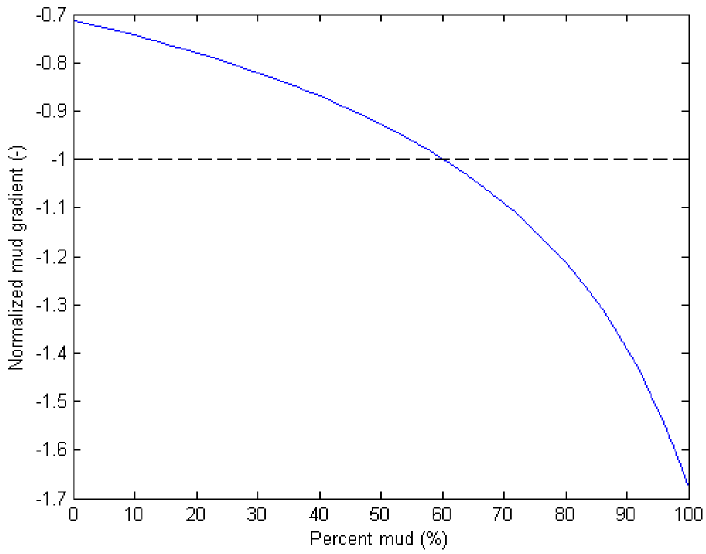
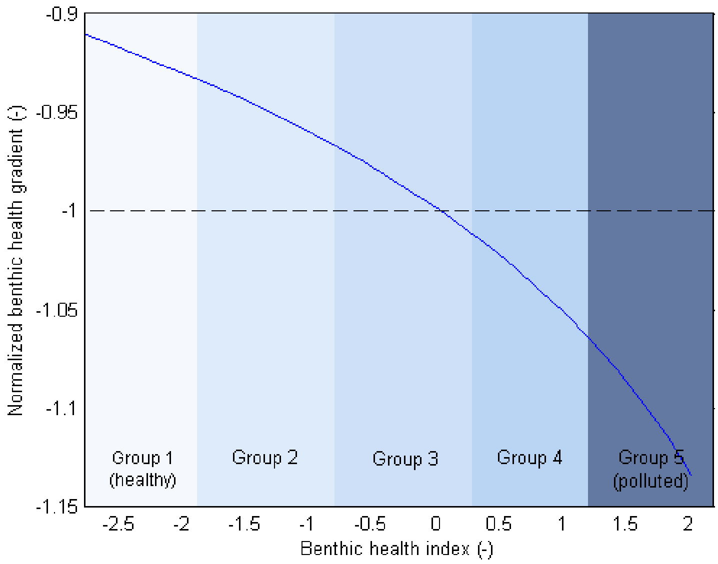


Figure 8: Chart of normalized benthic health gradient and benthic health index.



6 DISCUSSION

6.1 RESEARCH CONTEXT

The research reported in this paper contributes to the development of the UPSW decision support (DSS) software that discriminates between alternate urban development scenarios in terms of their effects on water bodies that receive urban storm water. Contrasts between scenarios are established in terms of economic, social, cultural, environmental, and resilience indicators to provide planners, engineers and other stakeholders to urban development processes information as to the whole system, life cycle value of engineering design for stormwater management.

Monetary valuation of the benefits of modern stormwater engineering practices such as water sensitive design is difficult to achieve given access to coastal ecologies in many jurisdictions is not managed through systems of rights and permits, so that relevant prices are not established in markets. While non-market valuation techniques with strong theoretical and statistical precedents such as choice modelling are available to establish prices in implicit markets, many decision makers are uncomfortable to rely on this kind of information. Key criticisms of these methods lie in areas such as framing of survey instruments, the psychological basis for the responses, and the scaling of survey outcomes. They motivate the development of alternate metrics to understand the trade-offs between human wellbeing and urban development in its various configurations.

In the urban coastal system urban stormwater is the transmission vector for development-intensified influences such as sediment and heavy metals that modify the coastal environment. In turn, changes to the coastal ecology influence its capacity to provide amenity provisioning ecosystem services – non-commercial benefits that mankind derives from fishing, shellfish gathering and the like. To complement monetized measures of wellbeing we have developed an indicator metric that reflects (1) how social wellbeing derived from coastal waterbodies changes with varying contaminant loads, and (2) the resilience of wellbeing derived from provisioning ecosystem service supply by coastal water bodies to the effects of urban stormwater. The key issue is that changes in wellbeing in SESs are likely to be non-linear and potentially irreversible because of the regime based dynamics of ecosystems. Wellbeing resilience is the capacity to absorb chronic and acute shocks, and may be understood in terms of the distance from transition zones between distinct marine ecological regimes: thresholds of potential concern.

6.2 SOCIAL WELLBEING

Social wellbeing is understood as the freedom and capacity of communities to engage in activities that they value. This definition relies heavily on the work of Harvard economist Amartya Sen. Application to wellbeing based in provisioning ecosystem services focuses on two aspects: the objects of value in the coastal ecosystem and their relative value as their delivery increases or decreases. How much more or less valuable are the provisioning services associated with alternate coastal ecological regimes in terms of their influence on the capacity to achieve wellbeing? A functional approach to the contrasting value to humans of differing states of an ecosystem's health lies in developing a metric that assesses the differing capacities for people to engage in activities / relationships with ecosystems that in turn enable wellbeing achievement (Sen, 2008).

As marine ecosystems move between regimes under the influence of driving variables, the abundance of provisioning ecosystem services consumers' target also changes. Addressing the first aspect requires identifying distinctive ecological regimes – including the derived ecosystem services - as objects of differing value. It requires an evaluative regime to address the second aspect: what is the relative value of the respective regimes and contaminant related declines in service delivery within regimes? In the UPSW DSS, the ecosystem changes that contribute to changes in their wellbeing are assessed in terms of the satisfaction or utility experienced by consumers (in this research 'expert harvesters") that has been expressed as experienced utility: preferences that have a basis in experience. Recent literature (Welsch and Ferreira, 2014a) introduced the term "experienced preference" to capture a number of approaches that use experienced satisfaction or experienced utility approaches to assessing subjective wellbeing.

6.3 INTEGRATION

Prior research (Batstone et al 2013) has established that experienced utility data meets theoretical precedents for utility data in that it reflects diminishing marginal returns. Before adopting this metric it is necessary to explore whether the information available in experienced utility data is consistent with that held by scientists. To achieve this a process has been designed to determine whether critical points derived from fisher satisfaction scores can be used as leading, lagged, or coincident indicators for the location of thresholds between regimes in coastal ecosystems defined by expert coastal ecologists. We have used values obtained for $\bar{x}* = M^*$ with $x*_e$ and $\hat{z}* = B^*$ with $z*_e$ to identify in this system whether $M*$ and $B*$ are leading, lagged, or coincident indicators for $x*_e$ and $z*_e$ respectively.

Indicator	Threshold #1		Threshold #2	
	Consumer	Ecologist	Consumer	Ecologist
%mud, ($x*_e$); %	N/A	10-25	60%	60-80
BHI ($z*_e$); (-)	Group 3 (0.03)	3 - 4	NA	NA

Table 3: Integration: Contrasting estimates for $x*_e$ and $z*_e$ with $\bar{x}* = M^*$ and $\bar{x}* = M^*$

There is close correspondence evident between the estimates for the indicators of thresholds of concern between expert consumers of coastal provisioning ecosystem services and expert coastal ecologists derived in this research process. Marine ecologists with strong experience in Auckland coastal waters have consciously draw on their knowledge of their field to identify potential alternate regimes in that coastal system, and to identify, with high confidence, comparative indicators that the thresholds of concern between those regimes. A normalized derivative approach that actions the economics discipline's *ceteris paribus* concept has been used to identify the point of equivalent response (PER) in experienced utility to changes in environmental quality. Derivation of those points for two key stormwater mediated variables has identified values that correspond to two of the thresholds identified by ecologists.

The PER to changes in bed sediment composition defined by the point where the normalized mud gradient equals one corresponds with the threshold between the transition zone F1-F2 and higher levels of contaminant in Figure 1, where a threshold has been crossed and hysteresis is possible. The PER to changes in ecological health defined by the point where the normalized benthic health gradient equals one corresponds to the transition between the state of contamination prior to the transition zone F1-F2 in Figure 1, located while vulnerability is low and resilience high. We conclude that this information may constitute a precedent for further research to confirm the application of the PER in harvester data in coastal management processes. The close correspondence between fisher data and expert scientists supports the use of experienced utility data as a social wellbeing metric.

In the UPSW DSS biophysical and probabilistic models of stormwater mediated variables (Moores et al., 2013) produce forecasted time series of the levels of the contaminants introduced into coastal processes. The points of equivalent response are located in anticipation of important changes in coastal systems. Their location in terms of the corresponding biophysical variables may be useful information in terms of the detection and communication of important limits to contamination of coastal ecosystems. Further, being able to identify key points on the trajectories may contribute to assessment and communication of declines in the relative value of varying ecosystem services and the location of thresholds of potential concern (TPC) (Biggs et al., 2011). TPCs are upper and lower levels of key biotic and abiotic variables that act as indicators to managers of the acceptability of environmental conditions (Rogers and Biggs, 2009). This research is partly cued by Biggs et al., (2011) recommendation for expansion of TPC analysis from a purely biophysical definition of ecological thresholds to an SES view of the system involved, the key challenge being the employment of preferences and other social constructs in understanding TPC.

We have selected experienced preference data as the metric in the UPSW decision support system (Moores et al., 2013) to assess changes in the capacity for people to undertake the things they value. Evident in the outcomes of this research is an inverse relationship existing between contaminant levels in coastal systems and people's capacity to access provisioning ecosystem services. We have employed "expert" services consumers to provide data for the derivation of critical points where service delivery becomes impacted to the point that wellbeing, reflected in "expert" assessment of the utility of differing environmental quality regimes in undertaking the things they value. For this reason the sample recruited to the expert consumer workshops was selected on the basis of expertise, rather census representativeness.

Specification of PER as the locations of thresholds between social-ecological regimes provides reference points denominated in terms of variables that form the attribute base in stated preference non-market valuation processes. Using these reference points it is possible to develop monetized estimates of the value the resilience of provisioning ecosystem services. This can be achieved by identifying the difference between the present value of the flows of services prior to, and subsequent to, PER that are based in dramatic changes in the coastal ecology that accompany increasing levels of key contaminants at thresholds of potential concern. This valuation of ecosystem service resilience may contribute to assessment of the value of water sensitive approaches to stormwater management.

7 CONCLUSIONS

In this paper we have reported research that describes a novel approach to understanding how community wellbeing changes as coastal ecosystems are impacted by urban stormwater. Defining wellbeing in terms of the capacity of the community to derive functioning they value from coastal ecosystems, the approach uses experienced utility data as the basis for a metric to map changes in wellbeing to changes in stormwater contaminant load. The approach extends beyond assessment of incremental changes in wellbeing that accompany differing urban development and stormwater management strategies to identification of thresholds of potential concern in coastal social-ecological systems. Those thresholds are associated with contaminant loads that have the potential to induce large and potentially irreversible reductions in wellbeing derived from coastal waters that follow regime change in coastal ecosystems.

A normalized derivative approach has been employed to establish the mathematical specification for points of equivalent response (PER) on experienced utility response functions derived from expert consumers of coastal provisioning ecosystem services. Those PER are wellbeing thresholds that correspond to expert assessments of ecological thresholds derived through a Resilience Alliance method to develop a systems map of coastal ecology ecosystem service delivery. Identification of key thresholds may enable valuation of the resilience achieved by specific stormwater management regimes such as water sensitive design that limit coastal ecological effects of urban development.

The convergence demonstrated between expert provisioning ecosystems services consumers and coastal scientists supports the use of the experienced preference metric as an indicator of the effects of urban stormwater management on the social wellbeing communities derive from coastal water bodies.

ACKNOWLEDGEMENTS

We wish to acknowledge (1) Funding by the Ministry for Business, Employment and Innovation (MBIE) funding through the Resilient Urban Futures program of research; (2) the participants in the coastal ecology workshops: Dr Malcom Green, Dr. Drew Lohrer, Dr. Judi Hewitt, from NIWA and Marcus Cameron from Auckland Council; (3) Auckland Council's Research, Investigation and Monitoring Unit (RIMU) as end-user and partner in the research team; (4) Auckland Council's Research, Consultation and Engagement, Communications and Public Affairs team, in particular Kenneth Aiolupotea, Allanah Bates, and Kenneth Chan, and (5) The Auckland Council's Peoples' Panel members that took part in the on-line survey and the expert consumers' workshops.

REFERENCES

Anderson, M.J., Hewitt, J.E., Ford, R.B., Thrush, S.F. (2006). Regional models of benthic ecosystem health: predicting pollution gradients from biological data. Prepared by Auckland UniServices Ltd for Auckland Regional Council. Auckland Regional Council Technical Publication 317.

Batstone, C., Stewart-Carbines, M., Kerr, G and Sharp, B. and Meister, A. 2010, Understanding values associated with stormwater remediation options in marine coastal ecosystems: A case study from Auckland, New Zealand, Paper presented to Australian Agricultural and Resource Economics Society National Conference, Adelaide, 2010.

Batstone, C., and Sinner, J. 2010. Techniques for evaluating community preferences for managing coastal ecosystems. Auckland region stormwater case study, discrete choice model estimation. Prepared by Cawthron Institute for Auckland Regional Council. Auckland Regional Council, Technical Report 2010/012.

Batstone, CJ, Moores, J., Baines J., Harper, S. 2013, An investigation into the use of experienced utility scores to assess multi-attribute changes in environmental quality. On-line Proceedings of the 57th annual Australian Agricultural and Resource Economists Society Conference, Sydney, NSW, Australia Feb 5-8th 2013

Biggs, H., Ferreira, S., Freitag-Ronaldson, S. and Grant-Biggs, R., 2011,'Taking stock after a decade: Does the 'thresholds of potential concern' concept need a socio-ecological revamp?', Koedoe 53(2), Art. #1002, 9 pages. doi:10.4102/koedoe.v53i2.1002

Birkmann, J.; Seng, D.; Abeling, T.; Huq, N.; Wolfertz, J.; Karanci, N.; İkizer, G., Kuhlicke, C.; Pelling, M.; Forrester, J.; Fordham, M.; Deeming, H.; Kruse, S.; Jülich, S. (2012) Systematization of Different Concepts, Quality Criteria, and Indicators. Deliverable 1.2. 7FP emBRACE project: Building resilience amongst communities in Europe.

Bryan, H. "Recreation specialization revisited." Journal of Leisure Research 32.1 (2000): 18.

Carpenter, S.; Walker, B.; Anderies, M.; Abel, N. (2001). From Metaphor to Measurement: Resilience of What to What? Ecosystems 4: 765-781.

Folke, C.; Carpenter, S.; Walker, B.; Scheffer, M.; Chapin, T.; Rockström, J. (2010).Resilience thinking: integrating resilience, adaptability and transformability. Ecol. Soc. 15(4):20.

George, D., and Mallery, P. (2003), SPSS for Windows Step by Step: A Simple Guide and Reference, 11.0 Update (4th ed), Boston, Allyn and Bacon.

Hanemann, W. (1991). "Willingness To Pay and Willingness To Accept: How Much Can They Differ? Reply". American Economic Review 81 (3): 635–647.

Kahneman, D., Wakker, P., and Sarin, R. (1997) Back to Bentham? Explorations of Experienced Utility Quaterly Journal of Economics 112(2): 376-405

Kahneman, Daniel, and Sugden ,R. (2005) "Experienced utility as a standard of policy evaluation." Environmental and Resource economics 32.1 (2005): 161-181.

Kahneman, D., and Tversky, A. (1979). "Prospect Theory: An Analysis of Decision under Risk". Econometrica 47 (2): 263. doi:10.2307/1914185.

MEA Board, 2005, Natural Assets and Human Wellbeing: Statement from the Board, downloaded from MAweb.org, 16 August, 2012

Milman, A., and Short, A. (2008). Incorporating resilience into sustainability indicators: An example for the urban water sector. Global Enviro. Chang. 16: 758-767.

Moores, J., Batstone, C., Gadd, J., Green, M., Harper, S., Semadeni-Davies, A. and Storey, R. (2013). A Tool for Evaluating Stormwater Management Across the Four Wellbeings. Water NZ 8th South Pacific Stormwater Conference, Auckland, 8-10 May 2013.

Moores, J., Batstone, C., Gadd, J., Green, M., Harper, S., Semadeni-Davies, A. and Storey, R. 2014, Evaluating the Sustainability of Urban Development in New Zealand in Relation to Effects on Water Bodies. The International Journal of Sustainability. Volume Issue 4 pp31-47

Needham, Mark D., Larry J. Sprouse, and Kerry E. (2009) "Testing a self-classification measure of recreation specialization among anglers." Human Dimensions of Wildlife 14.6 :448-455.

Newton M., and Batstone, C. (2014) Recreational Coastal Activities of Auckland Residents, Cawthron Report 2608 prepared for Auckland Council 11p

Pauly, D. (1995) "Anecdotes and the shifting baseline syndrome of fisheries." Trends in ecology and evolution 10.10 (1995): 430.

Papworth, S. K., Rist, J., Coad, L., & Milner‐Gulland, E. J. (2009). Evidence for shifting baseline syndrome in conservation. Conservation Letters, 2(2), 93-100.

Pushpam Kumar (ed) (2010). The Economics of Ecosystems and Biodiversity Ecological and Economic Foundations. Earthscan, London and Washington. http://www.teebweb.org/our-publications/teeb-study-reports/ecological-and-economic-foundations/#.Ujr1xH9mOG8 (Accessed 10 July 2014)

Resilience Alliance (2007). Assessing Resilience in Social-Ecological Systems – A Workbook for Scientists. Version 1.1 Draft for Testing and Evaluation.

Roeckelein, J. E. (2006). Elsevier's Dictionary of Psychological Theories. Elsevier. p. 147. ISBN 0-08-046064-X.

Rogers, K. & Biggs, H. (1999) 'Integrating indicators, endpoints and value systems in strategic management of the Kruger National Park', Freshwater Biology 41, 439−451. doi:10.1046/j.1365-2427.1999.00441.x

Scheffer, M., Carpenter, S., Foley, J., Folke, C., and Walker,B. (2001) "Catastrophic shifts in ecosystems." Nature 413, no. 6856 (2001): 591-596.

Sen, A. 2008, Capability and Wellbeing, Chapter Fifteen in Hausmann D. (Ed), 2008 ed). The Philosophy of Economics: An Anthology. Third Edition University of Wisconsin–Madison, Cambridge University Press.

Shogren, J., Shin,S. Hayes, D., and Kliebenstein,J . (1994). "Resolving Differences in Willingness to Pay and Willingness to Accept". American Economic Review 84 (1): 255–270.

Walker, B., Holling, C., Carpenter, S., and Kinzig, A. (2004). Resilience, adaptability and transformability in social–ecological systems. Ecol. Soc. 9(2): 5.

Walker, B., Carpenter, S., Rockstrom, J., Crépin, A.-S., and Peterson, G. (2012). Drivers, "slow" variables, "fast" variables, shocks, and resilience. Ecol. Soc. 17(3): 30.

Welsch, H., and Ferreira, S. (2014a) Environment, Well-Being, and Experienced Preference. Vol. 367. No. 14. Oldenburg Discussion Papers in Economics.

Welsch, H., and Ferreira, S. (2014b) Environment, Well-Being, and Experienced Preference International Review of Environmental and Resource Economics, 2013, 7: 205–239.

APPENDIX

Data contained in this Appendix relate to the estimates obtained by OLS regression to estimate the coefficients α, $\boldsymbol{\beta}$, λ and μ that capture the relationships between U, and H, and the underlying environmental variables x, and z from equation (2). Figures A1-A2 show the relationship between the underlying environmental variables and the categorical variables used in the expert consumer workshops (the relationship for water clarity is not reported here as it does not feature in the list of comparative indicators offered by the marine ecologists in Table 2). The functional relationships reported were estimated using R software, and are all statistically significant at the 5% level of confidence.

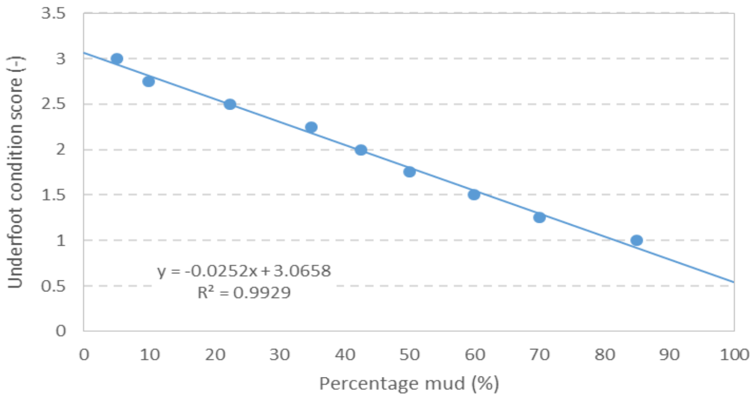


Figure A1: Underfoot condition: $U = \alpha x + \beta$, *where* $\alpha = -0.0252$ *and* $\beta = 3.0658$.

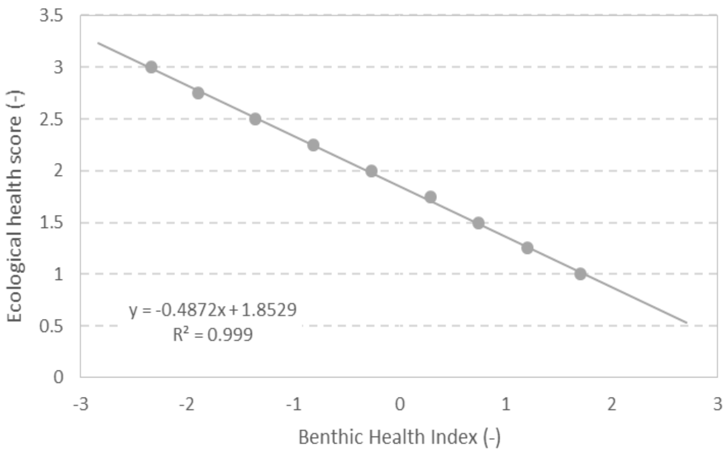


Figure A2: Ecological health: $H = \lambda z + \mu$, *where* $\lambda = -0.4872$ *and* $\mu = 1.8529$.