Non-market valuation in the economic analysis of natural hazards

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Abstract

Natural hazards have a wide range of impacts, including on factors that are normally unpriced because they are not bought and sold in markets. Key examples include impacts on human health, the environment, ecosystem services and other outcomes relevant to social welfare. Economists seek to quantify these impacts in financial-equivalent terms in order to be able to compare them with market impacts and include them in Benefit: Cost Analysis (BCA) of policies and strategies to mitigate risks. Estimating these so-called non-market values can be difficult. This paper reviews the methods available for doing so, presents a comprehensive list of the non-market values that might be affected by natural hazards and reviews the existing literature that estimates non-market values relevant to natural hazards. We find that there are few applications specifically in a natural hazard context. We conclude with a discussion on the limitations of non-market valuation in the natural hazard context.

Key words: natural hazard, mitigation, non-market valuation, intangibles

JEL classifications: Q51

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1. Introduction

To achieve maximum return from investments in management of natural hazards (including mitigation, emergency response and clean up) economists advocate the use of tools like Benefit: Cost Analysis (BCA) to evaluate actions or policies (e.g. Milne et al. 2015). Many Governments worldwide encourage the use of BCA for policy evaluation. For example, according to its Best Practice Regulation Handbook, 'The Australian Government is committed to the use of cost–benefit analysis to assess regulatory proposals to encourage better decision making' (Australian Government 2010, p. 61). In the United States, the Office of Management and Budget (OMB) provides specific guidance and requirements (OMB 2003) for BCA, Executive Orders (e.g., Executive Order 12866), and mandate this type of analysis for major federal actions. Individual agencies supply their own specific BCA guidelines (e.g., US EPA 2010).

Some of the relevant benefits and costs related to natural hazard management are relatively difficult to quantify, particularly in financial-equivalent terms. These values should be assessed as the marginal value of natural hazard management, or, in other words, the difference in value with and without the management. Economists have developed a range of techniques to assess impacts that fall outside of market responses, known as 'non-market valuation', but these methods remain underutilized in the natural hazard sector. Subsequently, there are few examples that explicitly include non-market values in BCA for natural hazard decision making. One such example is Whitehead and Rose (2008) who use benefit transfer in a BCA of the environmental and/or historical value of actions funded to mitigate against earthquake, wind and flood events.

There are a number of advantages from expressing non-financial impacts in financial-equivalent terms (Hanley 2002). It allows us to compare the benefits and costs of policy or management actions in order to evaluate whether they are worthwhile policies or actions, to rank alternative investments in terms of value for money, and to make rigorous business cases for investment. For example, many fire managers recognise that values are something of a knowledge gap for the sector. Clayton et al. (2014) found that 39% of the surveyed fire managers in Australia were familiar with Benefit: Cost Analysis.

It is challenging to assess the needs and prospects for non-market valuation in the assessment of natural hazards, for four key reasons. First, there are a range of different hazards and impacts, as well as the potential benefits from reducing the risks of occurrence. Second, there are several approaches to non-market valuation, generating some complexities about the methodology to be applied. Third, the development of the literature and case study examples varies across the types of hazards and the impacts/benefits involved. Fourth, is that BCA for natural hazard mitigation requires the analysis to account for the uncertain nature of hazard events, and non-market valuation methods often struggle to accommodate uncertainty. The purpose of this working paper is to identify:

- the methods available to quantify the marginal value of non-market impacts in financial-equivalent terms;
- the non-market values that might be affected by natural hazards; and
- the existing literature that estimates non-market marginal values relevant to natural hazards.

2. Methods for estimating non-market values

Non-market valuation is a group of techniques that use empirical evidence about observed or intended human behaviour or statements in surveys to quantify preferences for the provision of a public good or service and resulting economic values. Significant research effort has been invested in developing and testing a range of techniques, which are broadly grouped into two main categories (Adamowicz 2004; Carson 2012). Techniques that draw conclusions based on actual behaviour are known as 'revealed preference' techniques, while those that rely on statements related to behavioural intentions in surveys are called 'stated preference' techniques.

The method applied to estimate the value of a non-market good depends on the type of value the non-market good provides to the community. "Use" values cover non-consumptive uses such as recreation and amenity. "Non-use" values cover those unconnected to a "use value". They include existence value (knowing a good, like a national park, exists), bequest value (maintaining a good for future generations) and option value (protecting a good for a future, undiscovered use option). Use and non-use value are conceptually distinct, but not mutually exclusive; they can both co-exist within the same individual or good (Carson and Hanemann 2005).

2.1 Revealed-preference methods

The three types of revealed preference approaches most relevant to estimating impacts from natural hazards are hedonic pricing, defensive behaviour and travel cost. Revealed preference methods can only measure use values. Hedonic pricing estimates the relationship between the value of a marketed good and its characteristics, which could be non-market, allows inferences to be drawn about the values of those characteristics. For example, Dehring (2006) found that land prices decreased by up to 30% for flood affected properties following changes in the building regulations, indicating a negative value connected to the new regulations for flood protection. Of course they may have offsetting benefits that are not reflected in land prices, and so would need to be measured differently.

The hedonic model can also be applied to job wages, to disentangle the various factors that affect the wages that people require to undertake particular work. These models control for differences in worker productivity as well as different quality components of the job, such as personal safety, physical discomfort and timing of the work (Viscusi, 2003).

A limitation of the hedonic wage model is that it may not capture preferences of some groups within society. For example, an examination of wages to determine the value employees place on a reduced risk of work-place injury or death is only relevant to groups who are in paid work. Jones-Lee and Spackman (2013) also comment that the hedonic wage approach operates under the assumption that workers are well-informed about job risks and that wage rates are mainly determined by market forces rather than by regulations or by the market power of employers or unions in the labour market. In addition, wages can be insensitive to market conditions.

The second revealed preference technique, defensive behaviour, estimates the value of avoiding damage to a person's health by using the amount that person invests in actions that prevent health damage. For example, Richardson et al. (2013) apply the defensive behaviour method to estimate the value of reduced wildfire smoke exposure to California residents. They collected data through a community survey, asking individuals the length of time that they experienced smoke-related health symptoms, the defensive actions they took to prevent smoke exposure and how much tis action cost. The study found the mean WTP for a reduction in symptom days was US $87.

The defensive behaviour has predominately been used to estimate the value of health damages from reduction in exposure to air and water pollutants. The weakness of the defensive behaviour method is it is a lower-bound estimate of health damage: reduced pain and suffering from the health damage are not accounted for. There is also the difficulty of determining if the defensive behaviour affects more than one health outcome.

The third revealed preference technique, the travel cost method, is used to estimate the economic values associated with places that are used for recreation. The simplest form of the travel cost method uses the number of trips taken to a site per year and amount of money a person spends to get to a recreation site on each trip as a proxy measure of its value to that person. For example, a travel cost survey by Ferrini et al. (2014) found that people attach a high value to improving poor quality river sites, as compared to medium quality river sites in the United Kingdom. More complicated modelling approaches now incorporate site substitution patterns across a recreation activity in a region through the use of a random utility model applied to discrete visitation choices (reference).

There are a number of weaknesses with the travel cost method, such as how to allocate expenditure on sites in multi-purpose trips, the availability of substitute sites, and the value of travel time.

2.2 Stated-preference methods

Stated preference methods ask survey respondents about the choice they would make over health, environmental or social outcomes that come with a price. The choices and outcomes are hypothetical, as real markets for these goods do not exist. There are two common stated-preference approaches: contingent valuation (CV) and choice experiments (CE). In contingent valuation, respondents are asked questions about their willingness to pay (WTP) for a change in the provision of a good (or policy) as a whole. There are various forms that these questions can take. For example, Loomis et al. (2002) estimate the deer hunting benefits from prescribed burning using an open-ended contingent valuation question, "What is the maximum increase you would pay per trip to hunt this specific area if you knew you would be virtually certain to harvest a deer this season?" Other major alternatives include the referendum approach ("Would you support the proposed improvements if they cost you $xx, yes or no"), and the payment ladder approach ("Please indicate the minimum and maximum amounts you would be prepared to pay from the list of payment levels").

The second stated preference approach, choice experiments, instructs people to choose between options that are described by different levels of attributes and any costs they would have to pay. These attributes can be of many different types: environmental, social, technical or financial. Usually the cost associated with each scenario is included as an attribute, and this allows the researcher to estimate the trade-offs that respondents are willing to make between cost and changes in the other attributes. In other words, the trade-offs that respondents reveal in their answers indicates the financial WTP for the other attributes. In some cases it is inappropriate to include a policy cost, in which case marginal value cannot be estimated in dollars. The trade-offs the respondents make between attributes can be estimated, which may still be valuable information for the decision maker.

The key weakness of stated preference approaches continues to be hypothetical bias. For example the validity of the CV method is questioned by a growing evidence showing anomalies in the individual's answers to CV questions, and disparities between hypothetical and real WTP (Paradiso and Trisorio 2001). Hypothetical bias is likely to be exacerbated when people have a low understanding of, or familiarity with, the good being valued.

2.3 Benefit transfer

In many cases funding and time constraints make it difficult to conduct primary valuation studies to estimate benefits or impacts associated with natural hazard events. Furthermore, interest in the values seems to occur after an event has taken place which can "colour" responses to questions or can affect changes in behaviour that occurred prior to the event taking place. Benefit transfer is the use of research results from pre-existing primary studies at one or more sites or policy contexts (often called study sites) to predict welfare estimates or related information for other, typically unstudied, sites or policy contexts (often called policy sites) (Rolfe et al. 2015). Benefit transfer is advocated for use in policy making, particularly for non-market values, because it is usually cheaper, takes less time and is more straightforward than conducting primary studies.

There are two ways by which estimates can be transferred: unit value transfer and benefit function transfer. The unit value transfer is where a single WTP estimate is used for the new policy site. The simplest form of unit transfer is to assume that the per-person or per-household WTP at the study site is the same for the policy site. An aggregate WTP is generated for the number of persons or households who obtain the benefits generated by the policy. This is often the least accurate form of benefit transfer, but it is the simplest, and hence is often used by government agencies (Ferrini et al. 2014).

The unit value can be scaled or adjusted to match with, for example, the policy site population characteristics, the local currency or the quantity of the good provided through the new policy. The adjustment is made *ex post*, using objective or subjective rules. For example, Rosenberger and Loomis (2003, p 456) observed that values used within government agencies could be adjusted based on "empirical evidence from the literature, expert judgement and political screening".

Some non-market valuation studies generate a benefits function, which specifies the relationship between non-market values and a number of relevant variables (similar to independent variables in a statistical analysis). Benefit function transfer uses the benefit function from the original study site and takes values of the independent variables judged to be applicable to the new policy site, to estimate a new non-market value for the policy site. This approach requires information on at least a subset of the independent variables for the new policy site. This allows the adjustment of the benefit function from study site to policy site (Johnston et al. 2015). In principal, the benefit function transfer allows the analyst to adjust the parameter values to match characteristics of the new policy site, potentially including socio-demographic characteristics, and information about the quantity or quality of the public good.

A single-study benefit function transfer works off the assumption that the benefit function at the study site is the same as at the policy site, which will be more or less realistic in different cases. Multiple-site benefit function transfer accounts for the likely variation in benefit functions between sites, by using benefit functions from multiple sites. Values are generated for each benefit function and the results are used for sensitivity analysis of the economic analysis.

A meta-analysis benefit function is where data from several studies are combined statistically to create a single benefit function (Johnston et al. 2015). A median or mean WTP estimate can be generated from the function and used in analysis at the new policy site.

When undertaking benefit transfer, original values can be obtained from stated and/or revealed preference studies. Ferrini et al. (2014) provided one of the few comparisons of the performance of stated and revealed preference data for unit and benefit function transfer methods. Out of the three elicitation formats tested, CV payment card, CV discrete choice and travel cost, the CV data produced better benefit transfer results than did the travel cost data, with transfer errors lower than 20% for both unit value transfer and benefits function transfer. Larger errors were found for the travel cost data.

Rolfe et al. (2015) listed three main issues with the validity of the benefit transfer method: the theoretical validity of the original NMV study; structural theoretical foundation for the benefit transfer method; and the statistical validity of the original estimates. For example, there could be measurement errors from the primary study due to lack of 'incentive compatibility'[1] of the valuation question (Carson and Groves, 2007) or bias induced by the administration method.

2.4 Framework to estimate values for natural hazards

Natural hazard events generate a number of impacts that are not directly effected in market prices. Figure 1 provides a simple framework showing how the non-market values for a natural hazard event could be derived and aggregated. It is relevant to all of the estimation methods described in sections 2.1, 2.2 and 2.3.

This version of the framework is for a situation where the desired output is the aggregate non-market value for a natural hazard event, measuring and aggregating the full impacts for each type of non-market value. This could be relevant, for example to a study that estimates the total impacts of an event.

An alternative study context would be where the aim is to evaluate a particular policy or project. In that case, the relevant value would not be the total aggregate value (as illustrated in Figure 1). Instead it would be the difference in value resulting from the policy or project. This would require additional steps to estimate the difference in physical and social impacts resulting from the policy or project, rather than the total impacts due to the event.

[1] Incentive compatibility of a survey question is where the truthful response to the question asked reflects the optimal strategy of the respondent.

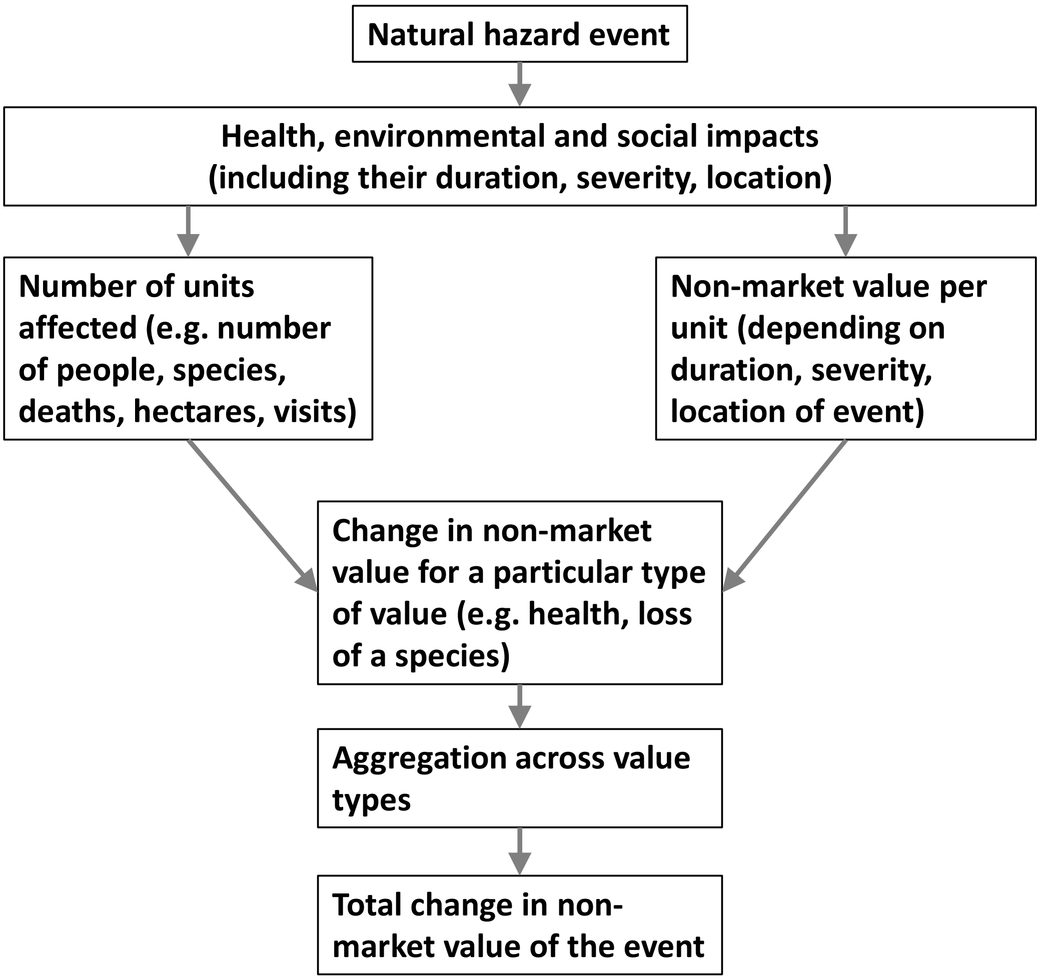

Figure 1. Simple framework for estimating non-market values affected by a natural hazard.

3. Non-market values affected by natural hazards

Table 1 provides a list of the types of non-market values that could be affected by a natural hazard. They include values related to human health, the environment, and social issues. The values of these things to society could be improved or, in some cases, diminished by the implementation of mitigation actions. In the following sub-sections we discuss the non-market valuation literature available for each value type. There are thousands of non-market valuation studies. The literature we discuss for each value type is prioritised by available studies in the following order: studies that provide a meta-analysis of valuation estimates in the context of a natural hazard; original non-market valuation studies conducted in context of a natural hazard; studies that provide a meta-analysis of non-market valuation estimates in other contexts; and original non-market valuation studies conducted in other contexts. There have been no non-market valuation studies for grief or memorabilia, so these are not discussed in the subsequent sections.

Table 1 Non-market values impacted by natural hazards		
Health	Environment	Social
Non-market good or service		
Mortality	Threatened species	Recreation
Morbidity	Ecosystem degradation	Amenity
Injury	Water quality	Safety
Stress/ anxiety	Invasive species affecting natural systems	Social disruption
Pain	Carbon storage and climate change	Cultural heritage
Grief		Animal welfare
		Memorabilia
Source: Adapted from Venn and Calkin (2013), and Milne et al (2015).		

3.1 Health-related values

3.1.1 Mortality

In relation to mortality, the non-market valuation literature has predominately focused on estimating the value of a statistical life (VSL). The VSL essentially measures the rate of substitution between wealth (or income) and the risk of dying (Cropper et al. 2011). Two approaches have been applied: the hedonic wage model and stated-preference methods. The hedonic wage model infers the trade-off individuals make between wages and job-related risks. Stated preference methods use surveys to estimate individual WTP to reduce their risk of death.

Stated preference studies are available on reducing the risk of death within the natural disaster context. Viscusi (2009) find that lives saved by reducing traffic safety deaths are valued almost twice as highly as lives saved by preventing natural disaster deaths. Carlsson et al. (2010) estimate the average VSL as AU$2.2 million (converted from SEK$13.2 million at 0.1685) for fire accidents and AU$2.1 million (converted from SEK$12.6 million) for drowning accidents. The VSL values for fire and drowning accidents are a third that of VSL for road related accidents. The most recent meta-analysis of stated-preference studies in developed and developing countries uses 862 VSL estimates (Milligan et al. 2014). The most recent meta-analysis of hedonic wage model studies uses 39 VSL estimates (Bellavance et al. 2009).

Although explanatory factors, such as education, can be controlled for in the hedonic wage equation, data representativeness, particularly with regards to age and gender, cannot. Kluve and Schaffner (2008) show that VSL estimates from the hedonic wage model are potentially more than 200% larger than those from stated preference studies The use of stated preference studies allows for the investigation of risk-context effects and the inclusion of population-representative samples (Milligan et al. 2014). But, variation in VSL estimates still occurs: Lindhjem et al. (2011), in their meta-analysis of 856 useable WTP estimates from studies in environment, health and transport disciplines, report a variation of up to $1.5 million between VSL estimates. They cite the likely causes for this variation as the variation in GDP per capita, the causes of mortality risk, and whether the risk affects others.

3.1.2 Morbidity, injury, stress or anxiety, pain and grief

Morbidity describes a poor health state, and can be broadly separated into conditions arising from disease, and those from injury. Both can be caused by natural disasters. The effects may be transitory (minor injuries) or long term, the latter potentially including chronic conditions that arise as complications of earlier acute conditions (Briere and Elliott 2000). The morbidity effects from natural disasters range from water-borne disease, respiratory disease, cancer and stress related disease, amongst others (Nohara 2011; Dohrenwend et al. 2013; Kochi et al. 2010).

The welfare impact from morbidity varies depending on the severity and duration of adverse health outcomes (Kochi et al. 2010). The cost of morbidity impacts from wildfire smoke is the only natural disaster application within the literature. The majority of studies that estimate the health cost from exposure to wildfire smoke use the cost of illness approach (Kochi et al. 2010). A shortcoming of this approach is that suffering, pain and other non-market values are not accounted for, and these can be important. For example, Richardson et al. (2013) found that a WTP approach valuing one less symptom day per person for a wildfire-smoke-related illness can be up to 30 times larger than a cost-of-illness approach.

Since the Kochi et al. (2010) review, a small number of WTP studies have investigated the non-market value of health effects from wildfire smoke (Richardson et al. 2013; Richardson et al. 2012; Moeltner et al. 2013; Kochi et al. 2012). Moeltner et al. (2013) estimated the impact of bushfire smoke on health outcomes in northern Nevada. They accounted for distance to the fire, fuel load and a four year lag in health effects, finding that aggregate treatment cost averaged US$2.2 million for 350,000 residents over a 4 year period. Some of the affected residents were 300 miles from the bushfire impact zone.

An alternative approach to valuing the change in risk of morbidity is to take advantage of the extensive research into the evaluation of generic changes in health status, and link those to social preferences and hence dollar values. This could be considered as a form of extended benefit transfer.

A common metric for evaluating health consequences is the EQ5D framework (EuroQol Group, 2005), which requires respondents to report their current health status on 5 measures (mobility, self-care, usual activities, pain/discomfort, anxiety/depression, each of which can take one of three responses). Changes in these measures are then converted to a value set based on the valuation technique Time Trade Off, which provides a utility score ranging from 1 (full health) to zero (dead) (Dolan 1997). These utility scores can then be used as the basis of a Quality Adjusted Life Year (QALY) measure: the change in the utility score multiplied by the amount of time spent in that changed state indicates the loss in quality of life, on a standard measurement scale. It is then possible to use estimated values for the loss of a QALY to estimate the monetary burden of the disease or injury (e.g. in the UK, the National Institute for Health and Care Excellence use a value of 20-30,000 pounds per QALY).

There have been attempts to quantify the physical and mental consequences of disaster-related injury, but often these use metrics other than EQ5Ds (e.g. Marres et al. 2011). In these cases it is possible to develop a mapping function to link these alternative values to the EQ5D (e.g. see Longworth et al. 2014).

An alternative is to apply benefit transfer, using estimates of the WTP to avoid generic pain and injury, and associate them with natural disasters. Carlsson et al. (2010) used a choice experiment to estimate WTP to avoid severe injury or risk of fatality, finding that one avoided fatality is equivalent to around 3.5 avoided severe injuries; Chuck et al (2009) found that the marginal WTP for a treatment that reduced both disability and pain intensity from chronic to mild severity was $1428 per person per month, and that reducing pain intensity produced a higher WTP ($1067 per month) than reducing disability $361 per month.

Although the literature contains elements of what is required to go from measurement of the extent of morbidity and injury through to placing an economic value on that change, there are few studies to do this. There is not a full set of value maps for the EQ5D worldwide. EuroQol covers 14 countries and regions, although the recommendation is to use the UK values in cases where data is absent. Other countries evaluation of the health state may not be based on EQ5D but some other index, and thus there is a requirement for a mapping process between these other indexes and the EQ5D.

3.2 Social

3.2.1 Recreation

Recreation values are typically assessed by analysing changes in behaviour following some environmental quality change. This can take one of two forms: a) changes in trip frequency holding the actual sites visited constant; and/or b) changes in the sites visited in response to quality changes at sites of interest. The revealed-preference approach uses travel costs as proxies for prices and either focuses on trip frequencies or on site choices using random utility models[2]. Both approaches utilize knowledge of substitute sites in modelling efforts, but the latter explicitly incorporates substitutes through construction of choice sets of various alternatives. Early stated preference approaches utilised contingent valuation, which uses notions of WTP largely through an increase in trip costs or entrance fees, but more recently contingent behaviour approaches[3] have been used (e.g. Englin and Cameron 1996) using changes in travel costs through changes in sites visited or changes in entrance fees.

There is a vast literature on recreation values. Studies typically focus in on particular forms of recreation, including camping, hunting and fishing. Available data typically consists of visits to various sites that arise from entry permits or registrations (e.g. Boxall et al. 1996). On-site surveys of recreationists can also be employed, although they have difficult sampling properties that must be accounted for in the statistical procedures used to analyse visitation data (e.g. Englin and Shonkwiler 1995). The difficulty with the former is the lack of associated information on the recreationist (e.g. demographic information), while the limitations of the latter are site and time specific (e.g. trips at time X to site Y).

A limitation of the literature is that it is not equally comprehensive in terms of covering all recreation activities for all types of natural hazards. For example, there are few valuation studies of off-highway vehicle use in the literature, while there are many studies of recreational fishing. Perhaps the best source of data on recreation impacts resulting from natural disasters involves wildfires. These include Canadian studies of camping (see Rausch et al. 2010) and wilderness recreation (see Boxall et al. 1996; Boxall and Englin 2008) where the intertemporal impacts of fire were assessed on recreation values. For example, Brown et al. (2008) found visitation rates to the Mount Jefferson Wilderness area did not change significantly after a major wildfire incident. They also found that 70% of the recreationalists did not change to a nearby substitute site after the fire.

3.2.2 Amenity and safety

The value placed on visual amenity and the reduced risk to an individual's life and property from a natural disaster event are inherently linked. Some people are attracted to live in areas that are more at risk from natural hazards, such as in forested areas, within flood plains and on the coast, because of their high amenity values.

[2] Random utility models specify an agent's preferences on alternatives by drawing a real-valued score on each alternative (typically independently) from a parameterized distribution, and then ranking the alternatives according to scores.

[3] Contingent behaviour models, combine elements of revealed preference and stated preference methods. The respondent is asked their actual behaviour with regards to site trips, then their preferences for a range of hypothetical variations to their usual site trip.

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The value of the reduced risk of a natural disaster affecting life and property can be estimated using hedonic pricing and stated preference methods. The hedonic price method is the most common approach to estimating the values of amenity and risk related to a natural hazard. The hedonic price model estimates the value of reduced risk as the implicit price differential associated with the location of a property in a natural disaster at-risk zone. Spatial mapping has been used to determine a property's view by using the surrounding topography, elevation of the property, proximity to parks and forested areas and obstructing built or vegetation features (Bin et al. 2008). A stated preference approach uses a survey to ask individuals about their WTP for natural disaster mitigation programs or their choices involving trade-offs between natural hazards and other factors.

Amenity and safety in relation to bushfires, floods and severe storms are the most common natural disaster applications within the literature, with over 100 studies investigating these issues. There are a handful of studies on the value of reducing the risk of earthquake damage to life and property (Hidano et al. 2015; Naoi et al. 2009; Beron et al. 1997; Keskin 2008).

Daniel et al. (2009) provide the only meta-analysis related to reduced risk from a natural hazard. The meta-analysis uses 19 studies that estimate welfare values for the reduced risk from flooding. They found that estimates of the implicit price of flood risk varied considerably. After controlling for observable and unobservable differences across studies, the marginal effect of an increase in the probability of flood risk by 1% in a year amounts to a difference in price of an otherwise similar house of -0.6%.

Stelter et al. (2010) estimate value of reducing bushfires in terms of reduced risk and increased amenity. The authors use the hedonic price method to identify the effects of 256 wildfires and environmental amenities on property values in northwest Montana between June 1996 and January 2007. Large positive effects on property values were contributed to by environmental amenities, including proximity to lakes, national forests, Glacier National Park and golf courses. Conversely, proximity to and view of wildfire burned areas had large and persistent negative effects on property values.

A limitation of this set of literature is that it is largely focused on applications to regions within the Unites States. Also, some older hedonic modelling approaches were unable to differentiate between the effects of amenity and proximity of the property to risk.

3.2.3 Cultural heritage

Impacts of natural disasters on cultural heritage can be assessed using revealed or stated preference methods if the values are recreational in nature (i.e. visits to see cultural heritage); or stated preference approaches can be utilised if the issue is the protection of heritage features from disasters.

For recreation values associated with cultural heritage the reader is advised to inspect the recreation valuation section. For this set of values the research approach would consist of assessing the impacts of changes in "quality" of the cultural heritage asset resulting from natural hazards on visitation. However, the more complex assessments on impacts of hazards on cultural heritage would include non-use values which would involve the values of knowing that the features exist in some particular condition. Here stated preference methods would be employed to examine citizens or potential visitors' WTP to avoid the damages by protecting the cultural assets from damages.

There is not a large literature on the values of cultural heritage, and few studies on non-use values of cultural heritage. There is one journal in the area the *Journal of Cultural Economics*, but most of the research reported there involves *ex situ* heritage, such as museums and art. There are few studies that examine impacts relevant to natural hazards.

We are aware of two studies relevant to the value of protecting aboriginal cultural heritage. One is by Rolfe and Windle (2003), who found that Indigenous and non-Indigenous values for the protection of Aboriginal cultural heritage sites in Australia differed significantly. In the other, Boxall et al. (2003) studied recreation values associated with aboriginal cultural heritage and vandalism, finding that "pristine" paintings along two canoe route in Manitoba, Canada, were worth about $61 and $77 per trip, whereas vandalised paintings were worth substantially less.

Most of the recent literature in this area values the presence of cultural heritage rather than its protection from damages of any form. However, with the development of anthropogenic sources of damage from climate (e.g. acid rain and air pollution) there has been interest in assessing losses in values (see Morey and Rossman 2003).

3.2.4 Social disruption

The disruption of services that are important to the functioning of communities, such as electricity, schools and government services, can cause a welfare loss to society. Paveglio et al. (2015) reviewed the social impact from wildfire and note that social disruption is a complex issue for which there is a lack of accessible, comprehensive and uniform metrics for assessing social impact. A further complication is that social impacts are likely to vary by population characteristics. An example of non-market valuation applied to the reliability of electricity supply is given by Hensher et al. (2014) in an exploration of consumer preference for electricity supply reliability in Canberra, Australia. They found residential customers' average WTP to avoid a 24 hour electricity outage was AUS$75. The length of the outage is in log form, meaning that an outage that lasts two hours is less than twice as inconvenient as an extra outage that lasts one hour. This is important, as power outages from natural hazards can be lengthy. Willingness to pay could be substantially larger for residents within a flood or bushfire prone area, as electricity is often needed for running water pumps, charging mobile phones and radio batteries.

Landry et al. (2007) estimated WTP of New Orleans residents to return home following Hurricane Katrina, by using the relationship between the economic benefit of returning home and the cost implied by the wage differential. For an individual employed full time, this implies an annual WTP to return home of US$3,954.

3.2.5 Animal welfare

There are significant community concerns about the welfare of animals in natural disasters. In the natural disaster context, these concerns are difficult to assess with revealed preference methods. As preferences for animal welfare include both non-use and use components, stated preference methods are required to assess the depth of concerns.

Studies valuing community preferences about animal welfare are often nested within broader topics such as sustainable agriculture, food production, farming systems and food labelling, and these usually relate to agricultural management practices rather than natural hazards (examples below). Concerns about animal welfare can be narrowed to farm animal welfare (FAW), as concerns about native animals tend to be incorporated into values for biodiversity and ecosystems. There is not a large literature on the values for farm animal welfare.

There have been several studies valuing improved production methods (focused on cage production with chickens and pigs), and a number of studies focused on food label attributes that include information about farm animal welfare. Most studies have been conducted in Europe, with a smaller number in the US. There are two meta-analyses available: Cicia and Colantauoni (2010 – reported in Viegas et al. 2014) analysed consumer WTP from 23 studies; Lagerkvist and Hess (2011) analysed 24 studies reporting 106 estimates of consumer WTP for farm animal welfare.

Some limitations of research in this field arise from the interweaving of public good and private good aspects of farm animal welfare. While some private good aspects for humane treatment of animals can be ascertained from consumption behaviour, these do not capture all preferences (e.g. vegetarians). It is challenging to distinguish and calibrate the private values for animal welfare transmitted through consumption purchases from the public demands for ethical treatment of animals. Other challenges are to distinguish animal welfare concerns from food safety and environmental protection as these are often treated as joint products by consumers and the public.

3.3 Environment

3.3.1 Threatened species

The International Union for Conservation of Nature (IUCN) groups threatened species into three categories, depending on the on the degree to which they are threatened:

1. Vulnerable
2. Endangered
3. Critically endangered

There exist numerous studies that aim to estimate the nonmarket values of threatened flora and/or fauna. However, despite an extensive literature search[4], only one study was found on threatened species values in a natural hazard context. Loomis and Gonzalez-Caban (1998) identify a WTP function for the protection of spotted owl habitat in California and Oregon by implementing a fire management plan.

Because of the non-use, non-market nature of species' values, nearly all studies estimate these values using stated-preference methods. Many of these studies estimated values for *native* plant or animal species, rather than *vulnerable*, *rare*, or *endangered* (i.e. threatened) species.

Of the studies found, some looked at preventing a loss of endangered species (Aldrich et al., 2007; Blamey et al., 2000; Campbell, 2008; Kotchen and Reiling, 2000), while others looked at the presence of rare species (Choi and Fielding, 2013; Kragt and Bennett, 2011). The way in which threatened species were described varies from the quantitative number of individuals (Carlsson et al., 2010; Morrison et al., 1998) to qualitative descriptors of species' protection (e.g. conserved versus extinct in Campbell, 2008). Each study was conducted in a different context, which makes it impossible to readily compare results.

Meta-analyses of the economic value of rare and endangered species showed that households' WTP varies greatly depending on what type of species is being valued. For example, WTP estimates for marine mammals and birds are significantly greater than WTP for other species such as land mammals and reptiles (Loomis and White, 1996; Richardson and Loomis, 2009). Other factors that influence the value attached to a species are the size of the species population, the frequency of the payments, the 'charisma' of a species, and whether a species has non-use value only or both use and non-use values (Richardson and Loomis, 2009).

Thus, the existing literature does not provide a conclusive answer to 'the value' of threatened species. In general, however, protecting or enhancing the abundance of a threatened species is valued by the general public, even when those who support these initiatives do not necessarily directly experience the outcomes (Meyerhoff et al., 2009).

[4] Using combinations of the following search terms "nonmarket/non market/non-market valuation/values", "valuation" and "rare/threatened/endangered/vulnerable species" and "natural hazard/disaster".

3.3.2 Ecosystem degradation

The application of non-market valuation for estimating the benefits of ecosystems has primarily focussed on the values of the services they provide to communities. In many cases, the ecosystem services valued includes the reduced risk of flood or storm damage to local communities as a result of maintaining healthy ecosystems such as wetlands, mangroves and coral reefs (e.g. Brander et al. 2013; Brander et al. 2006; Everard et al. 2014; van Zanten et al. 2014). A variety of approaches have been used to estimate the values of these services, including choice experiments (Barkmann et al. 2007; Drake et al. 2013), contingent valuation (Kim & Petrolia 2013; Li et al. 2015), meta-analysis (Brander et al. 2013; Brander et al. 2006) and benefit transfer (Everard et al. 2014).

In fewer cases, non-market valuation efforts have estimated the value of protecting ecosystems directly, as opposed to the services they provide. In a meta-analysis of conservation strategies for forest and freshwater ecosystems in Europe, Canada and the US, WTP was greatest for the preservation of intact environments relative to restoration of degraded freshwater and forest ecosystems (Hjerpe et al. 2015). This study included 127 data points from 22 original stated-preference studies conducted between 1987 and 2013.

In the context of a choice experiment on climate change mitigation, German residents were willing to pay €27.54 each per year to improve the resilience of the Hainich National Park against insect pests and storms (Rajmis et al. 2009). Petrolia et al. (2014) used choice experiments and contingent valuation to estimate WTP to restore Louisiana wetlands in the US, as well as the specific ecosystem services they provide, including wildlife habitat and storm surge protection. Mean household WTP (a one-time tax) was found to be US$909. The mean aggregate WTP was $105 billion.

A limitation of the literature is the lack of data available on non-market values related to the environmental impacts of ecosystem degradation, as opposed to the social impacts, in the context of natural hazards.

3.3.3 Invasive species affecting natural systems

Natural disasters, specifically floods, cyclones and bushfires, contribute to the spread of invasive species. Floods can spread weeds along watercourses into areas that were previously free of weeds. Cyclones can create new opportunities for weed invasion through associated flooding, soil movement and damage to native vegetation communities. For some weeds, fire can kill or suppress growth. Other weeds can benefit from fire as fire reduces competition and produces an environment in which the weed can spread rapidly (Department of the Environment 2014).

Most attention on valuing the impacts of invasive species have focused on market values, such as production losses, but some more recent work assesses holistic WTP measures to avoid or control incursions. Invasive species can have substantial impacts, through effects on agricultural production, biodiversity, ecosystem services, infrastructure, human health and communities (Pimentel et al. 2005; Lovell et al., 2006). Assessing these benefits is challenging because most involve direct use, indirect use and non-use components, especially those involving reduced impacts on human health and the protection of environmental assets and ecological processes (Born et al. 2005; Lovell et al. 2006). Estimates of value can be further complicated by the extent of invasive species control measures, which are often categorised into three broad strategies: prevention, eradication and containment (Born et al. 2005).

There have been many efforts to assess the market values of losses associated with invasive species, with particular emphasis on control costs and lost production in agriculture (e.g. Olson 2006), but work on assessing non-use values and WTP for different control strategies is much more limited. There are no WTP studies that estimate the benefits of invasive species control related to natural disaster mitigation options.

Studies that assess the benefits of control, through estimating WTP measures, are limited to a number of disparate case studies. Examples of discrete choice experiments include the work of Carlsson and Kataria (2008) to assess the benefits from weed-control programs in both Sweden and the USA, and Rolfe and Windle (2014) to assess the benefits of controlling red imported fire ants in Australia.

Most economic studies focus on estimating the losses caused by an invasive species rather than evaluating the costs and benefits of avoiding further damage to natural and managed systems; i.e. most studies are *ex poste* when the policy requirements are for *ex ante* evaluations (Pimentel et al. 2005). Moving to an ex ante framework involves risk and uncertainty about rates of invasion and their consequences; the treatment and impacts of risk and uncertainty on WTP measures for prevention and control are emerging areas in the economic literature on invasive species.

3.3.4 Water quality

There is an extensive literature evaluating the non-market values of water-quality improvements (Bergstrom et al. 2001; Young and Loomis 2014), with thousands of publications to date. This literature is highly heterogeneous, reflecting the many ways in which different types of water-quality improvements, in different areas and water bodies, benefit different user and nonuser groups. Most approaches provide estimates of WTP for water-quality improvements, quantified either directly via stated preference methods or indirectly via revealed preference methods. Johnston et al. (2003, 2005, 2015), Van Houtven et al. (2007), Johnston and Thomassin (2010) and Boyle et al. (1994) illustrate meta-analyses that evaluate patterns in stated-preference WTP estimates for water-quality improvements, including estimates derived via contingent valuation and discrete choice experiments. These values reflect both use and non-use values. Revealed preference methods, in contrast, estimate WTP for quality changes that enhance the value of a direct or indirect use of affected waters, including recreational (e.g., Adamowicz et al. 1994; Bockstael et al. 1989; Lipton 2004; Murray et al.

2001; Whitehead et al. 2000) and aesthetic uses. For example, hedonic analyses have found impacts on property values associated with multiple indicators of water quality, including levels of chlorophyll-a, nitrogen (N), phosphorus, cyanobacteria, and Escherichia coli, as well as sensory indicators such as measures of water clarity (Egan et al. 2009; Netusil et al. 2014; Poor et al. 2001). These can be used to infer marginal WTP for these changes.

The current literature provides a large body of data and meta-data that can be used to characterize the economic value of different types of water-quality improvements within different valuation contexts. A challenge, however, is reconciling data and observations from different studies across the literature, so that valid inferences may be drawn (e.g., using meta-analysis; Johnston et al. 2005; Smith and Pattanayak 2002). There are many different indicators which may be used to quantify water quality changes, and no clear consensus over which indicators are best for quantifying (and valuing) different types of water-quality improvements (Boyd et al. 2015; Van Houtven et al. 2007). Nonetheless, the literature provides significant insight into factors associated with systematic differences in water quality values across primary studies. These include affected uses (e.g., drinking, swimming, boating); user characteristics (e.g., income), the availability and proximity of substitutes, baselines and magnitudes of quality change, the type of water body affected (e.g., rivers, lakes, estuaries), whether estimates include non-use (or only use) values, spatial relationships between beneficiaries and affected water bodies, and many other factors (Johnston et al. 2003; 2005; 2015; Van Houtven et al. 2007; Johnston and Thomassin 2010; and Boyle et al. 1994).

Unlike values for other types of environmental changes, WTP for water quality improvements are not commonly elicited within the context of purely *natural* hazards (although natural hazards such as floods and fires can clearly affect water quality). More common are evaluations of water quality related to contaminants generated by human activity (e.g., agricultural runoff, mine drainage, oil spills, point source pollution, etc.). Exceptions include estimates of WTP to reduce quality reductions caused by natural hazard events such as algal blooms (Roberts et al. 2008), stormwater runoff (Londoño Cadavid and Ando), and flooding-related contamination of drinking and surface waters (e.g., related to wastewater overflows; Veronesi et al. 2014).

3.3.5 Carbon storage and climate change

Carbon storage has a value because it is related to climate change. Similar to other values described above, there exist no studies that have estimated how much economic value would be lost if natural disasters reduced carbon storage. There are, however, many studies that assessed the value of carbon currently stored in ecosystems – predominantly in forests, agricultural soils, wetlands, or oceans.

Carbon storage generally refers to capturing CO_2 in a carbon sink, such as oceans or a terrestrial sink such as forests or soils, so as to keep the carbon out of the atmosphere. Practices such as the introduction of cover crops on fallow land, retirement of land from active production to a grass cover or trees, protecting native vegetation, or preventing wetlands from being drained can all increase carbon storage. The main direct benefits of capturing and storing carbon in biomass or soils are climate-change mitigation and improvements in soil fertility. There are also many indirect (co-) benefits of sequestration activities, such as afforestation reducing soil erosion or native vegetation conservation that provides wildlife habitat (Matthews et al., 2002; McCarl and Schneider, 2001; Plantinga and Wu, 2003). Because of these auxiliary benefits, policies focussing on increasing carbon storage (or reducing carbon losses) will need to consider the impacts on multiple environmental benefits.

Once the amount of carbon sequestration and storage is estimated (through bio-physical models), one needs to estimate its value. Different methods are used to estimate carbon values, including existing carbon prices or the international spot price of Carbon under the Clean Development Mechanism (Ibarra et al., 2013). This would give the *market* value of carbon sequestration. In theory, the economic value of carbon should be equal to the marginal social cost of damage, i.e. the economic value of the damage caused by emitting an additional tonne of carbon into the atmosphere (Ninan and Inoue, 2013).

It is relatively straightforward to estimate the economic value of carbon sequestration once the amount of sequestration and marginal damage costs are known, as (Chambers et al., nd):

C-storage value ($/ha.yr) = amount of C sequestered (t C/ha.yr) x marginal damage costs ($/t C)

There exist many studies that have aimed to quantify the damage costs of climate change. A study by Tol (2005) collected cost estimates from 28 separate studies to estimate a marginal damage cost function of CO_2. His review shows a wide range in marginal cost estimates (from -US$0.5 to US$1667 per tC). The mean of estimates is US$97/tC, with a standard deviation of US$203/tC (Tol, 2005). Pearce (2003) argues that many studies over-estimate damages because they are based upon models in which there is no adaptation to climate change. Using equity weighting and a time-varying discount rate, Pearce suggests that the marginal social costs of carbon should range between £4-27/tC.

An important relevant issue here is the discount rate used to convert values in the future to present values. There is no clear consensus in economics about this. This is an important debate because use of higher discount rates results in lower present values, especially for values from the distant future.

The literature uses a range of social costs of carbon emissions, which can be considered as non-market. Fankhauser (1994) estimated the social costs of CO_2 emissions to be around US$20/tC for emissions between 1991 and 2000, rising to about US$28/tC in 2021-2030. Polasky et al. (2011) used US$42.32 per Mg of C for the social cost of carbon in a study of land use change in Minnesota. Chambers et al. (nd) used values of US$2 per ton of carbon to US$50 per ton of carbon to estimate carbon sequestration values for the Appalachian forests in the USA. They show a wide range of values, both within and across forest types. Values range from a low of $11.64 for a hectare of cove forest to $1,289.31 per hectare for a spruce-fir forest. Adger et al. (1995) use a value of US$20 per tC to estimate the value of avoiding forest conversion in Mexico (ranging from US$20 to US$100 per ha.yr). Willis et al. (2003) estimated the total value of carbon sequestration of Britain's public and private forests to be some £1.1 billion, based on a social cost of carbon value of £6.67 t/C.

Most studies account only for the carbon accumulated in the above-ground biomass, and not for the carbon stored in the forest soil. Old-growth forest accumulates significant amounts of carbon in the soil, which should be included if one wishes to account for carbon losses from a natural disaster. Exceptions are Santhakumar and Chakraborty (2003) and Croitoru (2007).

An alternative to using marginal damage costs is to estimate people's WTP for carbon dioxide emissions reductions as a means to get the social value of carbon sequestration. In a choice experiment of improved soil carbon management in Scotland, Glenk and Colombo (2011) estimated the value of a ton of CO_2-equivalent/year sequestered in Scottish soils over a period of 20 years at £38 (95% confidence interval £28.8–£47.5). A choice experiment of emissions reductions in Australia yielded a much lower annual WTP for emissions reduction: between AU$2 and AU$3 per metric tonne CO_2-e, depending on the type of model estimated (Landstra and Kragt, under review). Balderas Torres et al. (2013) used a choice experiment to estimate households' WTP for developing carbon sequestration afforestation projects in Mexico, and found a mean implicit carbon price between US$6.79-15.67/tCO2e.

4. Discussion

There are a wide range of impacts and associated non-market values to consider in the decisions about mitigation actions for natural hazards. A range of methods are available within the economics discipline to estimate the financial-equivalent value, to society, for each non-market impact. Despite a large body of literature, our review reveals gaps in the availability of WTP estimates for the value types we identify as being relevant to natural disasters. Amenity and safety values from floods, earthquake and bushfires have the most comprehensive information available. The majority of studies employ the hedonic price method to infer the value of amenity and safety from variations in property prices. Morbidity and recreation also have a handful of studies that are relevant to the bushfire mitigation context.

For the other value types, there are few estimates specific to a natural disaster context. Meta-analysis functions are available for water quality, mortality, ecosystem degradation and threatened species in contexts other than natural hazards. For stored carbon there are multiple estimates of the market value of stored carbon, and a handful that estimate the non-market aspect of the social cost from lost soil carbon. For animal welfare, cultural heritage, invasive species, social disruption and injury, stress or anxiety, pain and grief, there are few studies available.

The challenge for analysts and policy makers is to use the values information within a decision framework for prioritising mitigation actions. New studies could be conducted, if budgets and time permit, to provide accurate estimates for the specific policy question. New studies are required for those value types where no (or few) existing WTP data is available.

Benefit transfer is advocated as a suitable approach for value types for which estimates are well documented within the literature. However there are some potential issues with applying benefit transfer to a natural disaster context. The first is whether the influence of disaster context (cause, severity) significantly affects the WTP estimate. Jones-Lee and Spackmann (2013) provide some insight into the likely difference in value estimates for fatalities within the UK transport sector:

> "*…the prevailing view* [previous studies] *appears to be that the prevention of a statistical fatality in a large-scale multiple fatality accident does not warrant a higher value than is applied in the small-scale single fatality case*";

The second issue with the transfer accuracy of a WTP estimate is the target population to be considered. Natural disasters often impact large geographical areas. For example, the 2010/2011 Queensland floods affected more than 78% of the State and over 2.5 million people, killed 33 people, inundated 29,000 homes and businesses and cost in excess of $5 billion (Queensland Flood Commission of Inquiry, 2012). In this case the socio-demographic profile of the target population is variable, meaning that a single fixed unit could not be transferred to all sites. For example, age and health status have been reported to affect the VSL estimate (Krupnic et al. 2002). This is likely to be important when evaluating mitigation strategies for natural disasters. In an analysis of fatalities in Victoria Black Saturday fires, O'Neill and Handmer (2012) found

> *"…fatality dataset highlighted how many of the fatalities (44%) were particularly vulnerable due to age (either 70 or over, or under 12) and/or had a chronic and/or acute disability. Note that these vulnerabilities were sometimes compounded—2% of fatalities had both a chronic and an acute disability; and a further 9% had a chronic disability and were 70 or over.*"

The third issue is the potential influence of the context for a non-market value. For example, there is evidence emerging that the cause of death matters in people's valuation of reducing risk of death (e.g., Viscusi 2009). If one were to transfer a VSL derived from traffic accidents surveys, this may not reflect the VSL from a bushfire or drowning incident.

In conclusion, there is scope to use existing WTP studies, through benefit transfer, for some of the values affected by natural disasters. For some types of impacts, existing evidence is likely to be sufficient to support benefit transfer, while for others, additional studies are needed to fill information gaps.

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