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Economic evaluation of health risks in a developing country:

The case of arsenic contaminated drinking water in Cambodia.

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Abstract

Arsenic contamination of drinking water is a serious public health issue in many areas of South and South East Asia. One study estimates that in Cambodia over 100,000 people are exposed, with the majority of those living in Kandal Province.

In this thesis we present 3 original empirical studies focused on estimating nonmarket values for reduced arsenic risk water, based on primary data collected in May 2013. We also present a review paper which discusses the various economic techniques which have typically been used to estimate welfare values for cost-benefit analysis of mitigation startagies or appraisal of drinking water standards.

The first empirical paper presents the results of a discrete choice experiment (DCE) to estimate willingness to pay (WTP) values for reduced arsenic water. We discuss the results of scale-extended latent class choice models and underlying differences in preferences and choice consistency. We find that a reduction in the permissable limit on arsenic in drinking water may best represent underlying household preferences for risk.The second empirical paper presents the results of a split sample choice experiment focusing on differences between money (WTP) and labour contributions (WTWork) as payment vehicles in terms of choice behaviour and attribute non-attendance. We find that the results from the two experiments are relatively consistent which reinforces our results from the previous chapter that focuses on WTP measures alone and adds credibility to the large numbers of DCEs conducted in rural areas of developing countries. The final empirical paper examines actual household behaviour relative to an arsenic testing and education campaign run by a local NGO. We find that the vast majority of households change their drinking water source upon being informed that it is unsafe. On average households that switch increase their expenditures. In doing so however they also reduce the amount of time spent collecting water which limits the use of expenditure changes as an approximation of welfare values.

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1. Introduction

1.1. Background and Motivation

Arsenic contamination of drinking water is a serious public health issue in many areas of South and South East Asia. The contamination of groundwater by arsenic in Bangladesh alone has been labelled "the largest poisoning of a population in history" *Smith et al.* [2000] with around 20% of all-causes-mortality in arsenic impacted groundwater regions of Bangladesh attributed to arsenic exposure [*Argos et al.*, 2010]. However the hazard is found across the world [*Smedley and Kinniburgh*, 2002] and it has been estimated that globally nearly 50 million people have drunk arsenic contaminated water above 50 micrograms per litre (μg/l) and well over 100 million have drunk water with high concentrations of geogenic (i.e. non-anthropogenic) arsenic (defined here as >10 μg/l, the current (2014) provisional WHO guideline) [*Ravenscroft et al.*, 2009].

In Cambodia, many studies over the last 10 years have found high arsenic hazard in groundwater drinking water sources [*Polya et al.*, 2003, 2005; *Feldman et al.*, 2007; *Berg et al.*, 2007; *Buschmann et al.*, 2007; *Polya et al.*, 2008; *Sampson et al.*, 2008; *Kocar et al.*, 2008; *Quicksall et al.*, 2008; *Rowland et al.*, 2008; *Benner et al.*, 2008; *Sthiannopkao et al.*, 2008; *Polizzotto et al.*, 2008]. One study estimates that over 100,000 people are exposed in Cambodia, with the majority of those living in Kandal Province [*Sampson et al.*, 2008]. Human exposure to arsenic in Cambodia has been demonstrated through studies of various biomarkers [*Kubota et al.*, 2006; *Gault et al.*, 2008] and cases of arsenicosis have been recorded by *Mazumder et al.* [2009], amongst others.

Chronic arsenic exposure can lead to a wide range of health consequences such as lung, bladder, liver and skin cancers, skin hyperpigmentation and keratosis [*NRC*, 1999, 2001; *IARC*, 2004]. Other health outcomes include increased risks of ischaemic heart disease and immune system disorders [*Polya et al.*, 2010, and references therein]. Many arsenic attributable health outcomes are not contemporaneous with exposure; where data are readily available, such as in Chile, childhood exposures in particular have been linked through detailed epidemiological studies to peaks in arsenic attributable deaths occurring decades after the exposure [*Steinmaus et al.*, 2013]. Exposure to arsenic contaminated drinking water is thus a serious public health concern in Cambodia, both now and for the future [*Fredericks*, 2004].

There have been many studies which examine the economic welfare impacts of arsenic consumption in Bangladesh [see for instance *Maddison et al.*, 2005; *Aftab et al.*, 2006; *Aziz et al.*, 2006; *Madajewicz et al.*, 2007; *Opar et al.*, 2007] however there is relatively little research focused on Cambodia. There is thus a gap in the literature related to the household preferences towards arsenic risk in Cambodia, which is a serious public health issue. Such preference measures, for instance willingness to pay values, are often used for policy analysis such as cost benefit analysis. Although the objective of this thesis is to examine household preferences, rather than to conduct policy analysis, it is envisaged that the results from this work could be utilised in future policy appraisal.

1.1.1. Methods

The objective of this research is to examine preferences for cleaner water with reduced associated health risks, for potential future policy analysis. Given this objective economic valuation methodologies are appropriate. Water source decisions in rural Cambodia, however, are highly complex and heterogeneous and furthermore direct markets for improved water are often missing which complicates analysis. This thesis thus utilises economic nonmarket valuation techniques to examine the economic welfare impact of consumption of arsenic contaminated drinking water in Kandal Province, Cambodia using primary data collected through a period of fieldwork. Specifically choice modelling and averting/defensive expenditure methodologies are used to explore the preferences of households for reduced or arsenic-free water and thus preferences for reduced health risks. These welfare values and a thorough knowledge of household preferences are vital for commenting on current drinking water guidelines/standards and for governmental or non-governmental organisation's (NGO) mitigation strategies aimed at reducing exposure to arsenic related health risks.

The complex nature of water source choice decisions in rural Cambodia are further amplified by the property rights of the different water sources. Some water sources, such as water from a water vendor, are by their nature private goods whereas others, such as drawing water from a shared community groundwater pump or a local lake, are common or public goods. These property right aspects make the valuation of improved water sources challenging. We abstract away from these issues in the first two empirical chapters through the use of hypothetical markets which forces respondents to consider only communal goods. These issues are discussed further in the individual chapters but would need to be carefully considered when applying the values estimated in this thesis for policy appraisal.

Nonmarket valuation tools are extremely difficult to employ in rural developing areas. For revealed preference studies there is often a lack of detailed and high quality pre-existing data on which to base analysis. The conduct of a successful and meaningful stated preference study in such contexts poses many challenges ranging from educational, language, cultural and ethical issues to basic logistical and practical matters. The challenges are such that the validity of some stated preferences studies in developing countries have been called into question *Whittington* [2002]. Furthermore some of these difficulties are amplified when the scenario under analysis involves relatively difficult concepts to communicate, such as health risks. As such we add to the relatively limited, but growing, literature on nonmarket valuation in rural areas of developing countries. As well as focusing on the specific research goal of understanding preferences for reduced arsenic we also make a novel methodological contribution related to the role of payment vehicles in choice experiments in rural areas of developing countries.

The thesis comprises of 4 main chapters: 1 review chapter and 3 original empirical contributions. We end the thesis with a chapter of conclusions and discussions. In the remainder of this present introductory chapter we provide a brief outline of the main chapters, identifying the gaps in the literature to be addressed, motivations, the main research questions and hypotheses.

1.2. Review Paper (Chapter 2): “Valuing the Damage of Arsenic Consumption: Economic Non-Market Valuation Methods”

The first paper within the thesis is a review article written specifically for a forthcoming book entitled “Best Practice Guide on the Control of Arsenic in Drinking Water” edited by Prosun Bhattacharya, Dragana Jovanovic and David Polya [1]. The paper offers a review of economic nonmarket valuation methodologies which could be used to examine the welfare impact of arsenic consumption required for cost-benefit analysis or an analysis of water standards.

The paper is written with the water industry and regulators as the intended target audience. As such the paper offers a non-technical overview of the economic arguments and techniques as well as a detailed, up-to-date list of academic studies which have examined arsenic health risks using economic methodologies. This paper fills a niche

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[1] This book is in press and not yet published. As such it does not appear in the reference list.

required by industry and although it does not constitute a piece of original research it has been peer reviewed and is included in this thesis to provide a literature review and to frame the arsenic health risk issue within the economics discipline. The chapter has drawn on the research conducted during the main empirical chapters of the thesis and suggests how values estimated using methodologies utilised in this thesis could be used for policy evaluation.

1.3. Research Paper 1 (Chapter 3): "Arsenic in Drinking Water: Willingness To Pay, Preference Heterogeneity and Drinking Water Standards in Cambodia."

In this paper we estimate the value of reduced arsenic concentrations in drinking water, using a discrete choice experiment (DCE), amongst the arsenic-informed rural poor in Kandal Province, Cambodia, a province whose rural population is heavily exposed to arsenic contaminated groundwater. The estimates from this choice experiment are used to comment on the appropriateness, or otherwise, of the current arsenic drinking water standard in Cambodia.

Given the growing arsenic problem in South-East Asia, a potential public health reaction is to set and enforce a permissible limit on arsenic concentrations in drinking water. Such a limit would act as a benchmark for safe drinking water within the country and would guide remediation and education efforts. The current permissible limit in many developing countries is 50 μg/l whilst the WHO guideline and the standard used in much of the developed world is 10 μg/l. *Smith and Smith* [2004] argue that, in developing countries, lowering the permissible limit from 50 μg/l to 10μg/l could be detrimental to public health as it may postpone short-term solutions (that could lower arsenic levels to at least 50 μg/l from much higher levels), in favour of more radical solutions which may take many years to fund, organise and implement. Relatively little is known however of the values to Cambodian households of arsenic risk reductions. These values are needed to effectively assess permissible limits as well improving the design of associated education and remediation efforts.

Many of the economics studies related to arsenic contamination have focused on the effectiveness of arsenic information campaigns in promoting households switching to "arsenic-safe" water sources. These studies identify 'safe' water sources as sources that contain arsenic concentrations lower than 50 μg/l, the drinking water standard in Bangladesh and Cambodia. Switching behaviour is thus observed in relation to a statement that a source is either safe or unsafe based on the 50 μg/l threshold. Such a safe/unsafe dichotomy does not take into account that concentrations of arsenic less than 50 μg/l may not be totally risk free, nor that within the safe and unsafe categories there is a large spectrum of risk, where a source with lower concentration of arsenic is safer than a source with a higher concentration, ceteris paribus.

The DCE study which we present fills this gap in the literature by focusing on a hypothetical range of arsenic risks, some of which are lower than the risks associated with the current drinking water standard. As such we can comment on the appropriateness of the current water standard relative to household preferences for water with reduced arsenic concentrations. We also focus our study on the arsenic issue in Cambodia, whereas much of the previous literature has focused on Bangladesh. Structural and cultural differences may effect water switching behaviour, perceptions of risk or alternative mitigating strategies.

We account for preference heterogeneity through the estimation of scale extended latent class models which simultaneously allows identification of differences in choice consistency within the sample. Failing to account for these differences, i.e. by estimation of a homoscedastic latent class model that is typical within the literature, is likely to provide biased preference estimates, since the scale term is confounded with those preference estimates. We use the parameters from these models to estimate willingness to pay (WTP) values which could be used in policy analysis.

1.4. Research Paper 2 (Chapter 4): "Valuation in Developing Countries: Willingness to Work vs. Willingness to Pay."

A concern major when conducting stated preference valuation studies in rural developing or very low income contexts, such as in Chapter 4, is the use of monetary willingness to pay (WTP) estimates. In circumstances where cash incomes are extremely low, a significant proportion of the population are not engaged in waged labour and the exchange of goods or services is achieved through barter or work exchange, the role of money is likely to be different from that within a developed urban setting. As such, ability to pay using money may be impaired when compared with other mediums of exchange, and economic values using WTP measures may be downwardly biased.

This bias is particularly pernicious given that it specifically affects rural, low-income, households who are perhaps most in need of assistance programs which are sometimes evaluated using these techniques.

In attempting to circumvent this bias researchers have used a variety of alternative payment vehicles (other than money) in stated preference valuation studies. For instance *Shyamsundar and Kramer* [1996] use baskets of rice and *Asquith et al.* [2008] used in-kind payments such as beehives. The most common substitute for money in stated preference studies is labour contributions [*Hardner*, 1996; *Kamuanga et al.*, 2001; *O'Garra*, 2009; *Casiwan-Launio et al.*, 2011; *Rai and Scarborough*, 2013, 2014].

There has however been relatively little focus on the role of payment vehicles in DCEs nor specifically on the impact of payment vehicles on choice behaviour and protest votes. In this paper we present the results of a split sample DCE using money and labour contributions as payment vehicles for improved drinking water quality in Kandal Province, Cambodia.

Several other papers have used labour and money contributions as payment vehicles in DCEs in rural developing areas [*Abramson et al.*, 2011; *Rai and Scarborough*, 2013, 2014]. These studies however focus on the role of payment in project cost recovery [*Abramson et al.*, 2011], the trade-offs between money and time [*Rai and Scarborough*, 2013] by including both as attributes in the experiment concurrently or the demand for the two payment vehicles by allowing respondents to choose which payment vehicle (money or labour) they are shown [*Rai and Scarborough*, 2014].

We build on this pre-existing literature by using a randomised split sample approach where half of the respondents are shown a DCE with money as the payment vehicle whilst the other half are shown a DCE with labour as the payment vehicle. This prevents self-selection problems that act as barrier to rigorous comparison of choice behaviour.

We focus on three distinct areas of analysis. Firstly by comparing WTP and willingness to work (WTW) estimates from the two sub-samples we are able to estimate a shadow cost of time or shadow wage rate. This enable us to comment on the relationship between shadow prices and markets prices and thus the potential efficiency of the labour market.

Secondly we test for parameter (or marginal utility) parity between the samples by pooling the sub-samples using market wage rates. This enables us to comment on marginal utility differences or differences between choice consistency related to the payment vehicle shown to the respondent.

Finally we focus on differences between attribute non-attendance (ANA) towards the payment vehicle. The hypothesis here being that if the money payment vehicle is inappropriate we may see a rise in protest votes which may be channelled into ignoring the money attribute and basing choice only on the remaining attributes.

The paper thus focuses on a area of methodological importance for future stated preference studies in rural areas of developing countries. It builds on previous literature but offers a more rigorous approach, by using a split-sample, and focuses on more general areas of choice behaviour rather than narrowly on the end products (i.e. economic values).

1.5. Research Paper 3 (Chapter 5): "Arsenic Testing and Household Drinking Water: The Determinants of Water Source Switching Behaviour in Kandal Province, Cambodia."

Papers 1 and 2 focus on estimating economic values for reduced arsenic risk using stated DCEs with focus on choice heterogeneity and different payment vehicles. In this chapter we examine preferences for reduced arsenic risk through a revealed preference approach, specifically the averting behaviour framework.

Arsenic contamination of drinking water is tasteless and undetectable upon consumption. Furthermore the illnesses contracted from consuming arsenic have long latency periods and so the impact of consuming arsenic are not easily observed. Water tests are available to detect the presence or absence of arsenic but householders need a degree of trust in the testing process if they are to contemplate mitigating strategies.

In this paper we present the results of a household survey in Kandal Province, Cambodia, investigating the effectiveness of a large-scale arsenic water testing program aimed at encouraging households to switch their drinking water source to one with low arsenic risk. We position our analysis within the "averting behaviours" theoretical framework which is commonly used to examine welfare values in cases of drinking water contamination [see for instance *Abdalla*, 1990; *Laughland et al.*, 1993; *Abrahams et al.*, 2000; *Madajewicz et al.*, 2007; *Nauges and Van Den Berg*, 2009].

Although the NGO implementing the education and testing program keeps records of how many households switch their water source we conduct a more thorough analysis focusing on the new sources which households utilise. There has been some concern that mitigation strategies, aimed at promoting water source switching, could in fact be detrimental to overall health outcomes [*MacDonald*, 2001; *Lokuge et al.*, 2004; *Field et al.*, 2011], if new water sources increase the incidence of diarrheal disease. One of the factors which has led to such high degrees of exposure to arsenic risk has been the promotion of groundwater sources by governmental and non-governmental organisations. This strategy was driven by the goal of encouraging people to move away from surface water sources which were seen as high risk in terms of diarrheal diseases. For instance, *Field et al.* [2011] find that, in Bangladesh, arsenic education and testing programs may have led to a doubling of child and infant mortality from diarrheal diseases emanating from consumption of water from new sources.

In this paper we contribute to the literature on averting expenditures by examining the determinants of switching behaviour, especially aesthetic attributes of water such as taste, appearance and smell. These factors are often overlooked in the previous literature. We comment on how the burdens of expenditures and time spent collecting water have changed upon switching water sources, as a means to assessing preferences for low arsenic water. We further examine in detail the new water sources that households have switched to and any changes in water treatment behaviour which may lead to changes in the rate of diarrheal disease.

1.6. Authorship

All chapters were written by Jonathan Gibson with comments and suggestions by co-authors. All data collection was organised, managed and implemented by Jonathan

Gibson with students from Phnom Penh universities acting as interviewers. Further details of those individuals who helped with data collection are provided in the Acknowledgements. Data analysis and econometric modelling was conducted by Jonathan Gibson with comments and suggestion by co-authors. For the purposes of future publication, co-authorship will appear as:

Chapter 2 (Review Paper): Jonathan Gibson, David Polya, Noel Russell and Johannes Sauer

Chapter 3 (Paper 1): Jonathan Gibson, Dan Rigby, David Polya and Noel Russell.

Chapter 4 (Paper 2): Jonathan Gibson, Dan Rigby, David Polya and Noel Russell.

Chapter 5 (Paper 3): Jonathan Gibson, David Polya and Noel Russell.

1.7. Alternative Thesis Format

This thesis is structured in the alternative format, following the guidelines defined by the University of Manchester. The main thesis chapters (2 - 5) are written as distinct academic papers in the format that they were either published (Chapters 2), submitted for publishing (Chapters 3 and 4) or that they are intended to be submitted in (Chapter 5). Each chapter thus includes a separate section outlining the motivations and research questions, a literature review and individual conclusions, along with the main empirical analysis and arguments. Given the nature of this thesis format there is some repetition and overlap in terms of the description of the arsenic problem and data collection issues. However, the research questions, empirical analysis and conclusions are distinct. The thesis ends with a conclusions section (Chapter 6), which draws together the findings from the individual chapters to offer a series of cohesive conclusions and policy recommendations.

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2. Valuing the Damage of Arsenic Consumption: Economic Non-Market Valuation Methods

2.1. Introduction

Human consumption of arsenic, either through contaminated drinking water or rice, is a huge public health concern in many countries across the world. Resulting illnesses, such as the many different cancers attributable to arsenic consumption, lead to pain, suffering and death. The economic impacts of these illnesses can be large; for instance, research by *Maddison et al.* [2005] indicates that the willingness to pay (WTP) to avoid these arsenic related illnesses in Bangladesh amounts to US$ 2.7 billion annually. Even where the problem is not as prevalent, large economic welfare effects can be felt by those individuals affected.

In this chapter, we review the motivations, methodologies and previous empirical studies related to estimating the economic damage caused by arsenic consumption. There are two primary motives for studying this damage. Firstly, such analysis will highlight the magnitude of the problem in a unit of measurement with which policy makers are perhaps most accustomed to, namely money. Secondly, once action is being considered to help mitigate the problem, such as setting an arsenic standard for drinking water or a government funded remediation project, economic values and frameworks can help guide a cost-effective and economically efficient response, helping to allocate resources where they can be most beneficial in minimising health risks.

Although the illnesses caused by arsenic consumption will cause economic damage, along with human pain and suffering, any remediation or mitigation efforts will incur costs. These costs must either be experienced by the end water users through higher bills or, if funded by the government, by tax payers. In either case, the investment of scarce resources in remediation signifies a trade-off with other potential uses of the funds. As *Harrington and Portney* [1987] point out, for regulatory programs which aim at protecting human health to be examined and rigorously appraised, the proposed benefits must be valued. It is thus important to know whether specific projects will produce sufficient benefits to justify the required expenditures or whether it would be more efficient to spend the money on alternative arsenic mitigation projects or indeed on alternative health improvement projects entirely.

The two objectives of this chapter are to, firstly, outline and discuss the methods for valuing arsenic-related health risks and, secondly, to discuss the standard economic framework for decision making: cost-benefit analysis (CBA). It should be noted at the outset however that an economic appraisal forms only a portion of a larger overall assessment, particularly for decisions with wide-ranging implications such as altering an official arsenic concentration limit in drinking water. The values and data created by an economic analysis should be seen as an informative step in the decision making process. As the US Environmental Protection Agency (EPA) guidelines for economic analysis *US Environmental Protection Agency* [2010] point out: ethical, political and technical concerns, amongst others, will also be significant in a wider evaluation process.

The structure of this chapter is as follows: Section 2.2 briefly outlines cost benefit analysis (CBA) framework for decision making or retrospective appraisal of regulatory decisions as well as common alternatives to CBA. Section 2.3 highlights the individual valuation tools which can be used to analyse the economic damage of arsenic consumption, whilst section 2.4 discusses techniques for utilising values from previous studies, known as benefit transfer (BT) methods. Section 2.5, as a case study, discusses the US EPA's economic analysis leading to the reduction of the arsenic Maximum Contaminant Level (MCL), from 50 μg/L to 10 μg/L, along with the subsequent debate surrounding it as an indication of the main contentious issues involved in conducting an economic analysis. Section 2.6 further highlights some of these contentious issues as well as practical considerations when conducting an economic analysis. Finally, section 2.7 presents some overall conclusions from the preceding analysis.

2.2. Cost-Benefit Analysis, Willingness to Pay, Economic Value and QALYs.

The key underpinning of the economic methods presented here is that the value to society of anything that benefits individuals, be that good health or the experience of a nice environment, can be expressed in monetary units. This monetary conversion requires estimating an individual's willingness to pay (WTP) for these benefits or willingness to accept (WTA) if benefits are diminished. These economic value measures are monetary amounts which leave the individuals welfare unchanged. i.e. the measure of interest is the amount of money which exactly compensates the individual for the change in circumstance.

Given that there are no direct markets for good health within which these WTP values can be observed, non-market valuation techniques are required to estimate the damages caused by arsenic consumption and thus the benefits that could be experienced by remediation. These valuation methods will be discussed in the next section of this chapter.

While the economic values estimated using these methodologies are useful in their own right, to highlight the economic damage caused by arsenic consumption, they have wider use as part of a cost-benefit analysis (CBA). CBA is a framework used to assess a new policy or to retrospectively evaluate the effectiveness of an existing policy. Although using CBA is controversial it is also widely used, particularly in governmental decision-making . Indeed, the US EPA's decision to lower the permitted arsenic concentration level from 50 μg/L to 10 μg/L was partly informed by CBA.

CBA is an application of neoclassical welfare economics. When the net benefits outweigh the net costs for a project, there is the potential for those who gain to compensate those who are adversely affected. The potential for compensation (known as the Kaldor-Hicks potential compensation test) is one of the more controversial aspects of CBA. Given that there will inevitably be some who gain and some who lose from the implementation of a project, there is not a large role for the distribution of these differences within a CBA analysis. All that is required to pass the Kaldor-Hicks test is there to be the potential for compensation, not that it actually occurs.

The conversion of consumers' preferences to monetary units allows comparison between the costs and benefits of a project, often expressed as a benefit-cost ratio. Maximising this ratio can then be seen as an objective for government, when choosing between different regulatory options.

There are several viable alternatives to CBA. These include cost effectiveness analysis (CEA), cost utility analysis (CUA) and multi-criteria analysis (MCA). CEA examines the costs of a program in relation to its expected aims and outputs. For instance with arsenic regulation, the cost effectiveness of project's forecasted lives saved, or life years saved, might be compared. Quality adjusted life years (QALYs) are common measures of outcomes in this type of analysis and are frequently used in the analysis of health regulation. In this analysis the years of life which are saved are adjusted by the quality of life. Due to the weighting of life years, this approach, and similar approaches, is known as cost utility analysis. Finally, MCA is a method which sets out several key criteria against which the outcomes of a proposed project can be measured. For instance for exposure to contaminated water, criteria that can be used to compare projects might include lives saved, number of people exposed, costs, convenience of the water source etc.

The main purpose of this chapter is to discuss the different non-market valuation methods available to calculate arsenic damage and so a full discussion of the relative merits of these analysis frameworks is beyond scope here. We will, however, briefly justify the focus of this chapter on WTP based CBA through a discussion of its main alternative, CUA. For further details see, for example, *Drummond et al.* [2005].

CUA, through the use of QALYs, is the most common alternative to CBA in order to analyse the policy response to health risks. QALYs are a tool for measuring the performance of a regulatory intervention, such as the reduction of an arsenic limit in water, which takes into consideration not only life years saved, but also the quality of those lives. This allows, for instance, a project which prolongs many lives but at a very low standard of living to be compared with a project which results in less lives saved but with higher quality of life.

QALYS measure a human life year in terms of its quality, with 1 representing a year of perfect health and a 0 representing death [*Freeman,* 2006]. If a policy was estimated to save 100 lives in its first year but at a diminished health quality of 0.5, the benefits would be 50 QALYs in that time span. Dividing the increase in QALYs generated by a remediation policy by the policy's costs, a cost per QALY statistic can be estimated. All projects under consideration can be compared to examine which are the most cost-effective at increasing expected QALYs.

The main drawback of the CEA approach is that it cannot be used to address the fundamental question of what is the appropriate scale of arsenic regulation. It simply allows different projects to be compared on the basis of cost per QALY. Given the goal of maximising net benefits, this issue can be important when the available budget is not the only potentially binding constraint. This broader question can, however, be addressed with CBA, within which valuation techniques are employed to estimate the benefits of a potential policy.

2.3. Valuation Methods

The different valuation methods utilised for estimating benefits can be broadly divided between two categories: revealed and stated preference approaches. Revealed preference valuation methods attempt to recover WTP measures from actual behaviour in related markets. In hedonic wage models for instance, researchers attempt to decompose wages to elucidate the extra pay workers are required to be paid to undertake work in risky industries. This risk premium could then be used to estimate a value of a statistical life (VSL) to be used to estimate arsenic related damages.

Stated preference valuation methods on the other hand seek to directly elicit WTP measures through survey instruments. Contingent valuation, for example, asks respondents about potential payments in a hypothetical market constructed by the researcher. In the case of arsenic this might take the form of asking how much a person might be willing to pay in order for arsenic to be reduced in their water supply. Although such survey-based techniques require careful planning and construction due to the hypothetical nature of the questions being asked, this hypothetical basis allows great flexibility. For instance, in situations where there are no suitable related markets from which to judge consumer values, stated preference methods provide a means of analysis.

For a more in depth and technical overview of the theoretical economic framework and associated methods for health benefits valuation, see, *Cropper and Freeman* [1991], *Freeman* [2003], *Tolley et al.* [1994] and *Whitehead and Van Houtven* [1997] amongst others.

Before moving on to discuss the individual valuation methodologies that are within the revealed and stated preference categories, we first review two of the more controversial and least understood topics related to valuing human health reduction benefits: human capital and the value of a statistical life (VSL).

2.3.1. Human Capital Approach

As detailed in *Freeman* [2003], an alternative framework to measuring individuals WTP as a basis of health valuation is the human capital approach. In this framework the damage caused by arsenic would be measured by the productivity losses to society, rather than the impact on individuals, from deaths or illness. This is typically measured as the lost wages that this individual would have received, had they been alive to work. Or, for serious illness not resulting in death, the value to society would be measured by the reduction to wages which they receive. These reductions would be calculated for the potential future lost years until retirement and appropriately discounted.

This framework, however, has some controversial outcomes concerning children and the elderly. For children, full wage earnings are only likely to be generated in late teens or early twenties. Due to discounting of these distant future earnings the actual benefits assigned to risk reductions for children will be relatively low. Similarly the elderly may have stopped earning a wage income which may therefore lead to the benefits assigned to elderly health risk reductions being extremely low.

More fundamentally, however, the human capital approach is not consistent with economic welfare theory which posits that it should be individual preferences which form the basis of cost-benefit analysis. These preferences can be captured by WTP but not by the human capital measures. For instance preferences over your own health or that of friends or family are not captured by future earnings alone and so undervalue the economic damages caused by ill health. The human capital approach is often misconstrued by non-economists as the economic approach to valuing health risks. In reality WTP methods are favoured and are widely used in practice.

2.3.2. Value of a Statistical Life (VSL)

In understanding and communicating the benefits of risk reduction, VSL estimates often play a significant role. VSL is the trade-off that an individual is willing to make between wealth and a marginal change in their mortality risk. For example, if an individual is willing to pay $200 for a reduction in their mortality risk of 1 in 10,000 then this can be aggregated up to a value per statistical life of $2 million. This figure can then be used to estimate the value or benefits of reducing risks. The VSL figure used in the US EPA's decision to lower its arsenic limit, for instance, was $6.1million (in 1999 US dollars).

This risk-money trade-off can be calculated using the valuation methods discussed in the next section. The concept of VSL estimates, and their continued use, remains controversial. The placing of monetary values on human life to act as benefit estimates for risk mitigation strategies in order to be compared with costs of risk reduction has led to some controversy [see *Ackerman and Heinzerling*, 2004]. There was a political and media outcry when the EPA lowered their baseline value in a 2008 evaluation and when they reduced the value for the over 65s in an analysis of air pollution [see *Viscusi*, 2009]. Much of the public controversy comes from a misunderstanding of the concept and of the terminology used. Some economists have called for a renaming of the concept. *Cameron* [2010], for instance, proposes that VSL should be renamed "willingness to swap (WTS) alternative goods and services for a micro risk reduction in the chance of sudden death."

The study highlighted at the start of this chapter, *Maddison et al.* [2005], utilised VSL estimates as well as dose response relationships and arsenic distribution data in order to place a monetary figure on the damage caused by arsenic consumption in Bangladesh. In this sense that study is an application of benefit transfer techniques, which will be discussed in section 2.4, and is similar to the approach taken by the EPA which is further discussed in section 2.5.

2.3.3. Revealed Preference Methods

2.3.3.1. Cost of Illness (COI)

The cost of illness approach to valuing health damages examines the real resource costs of arsenic consumption. These costs represent the potential gains to society if the illness were to be avoided. The main two areas of focus for such studies are thus:

1. Lost income through reduction in working time, either through illness or death, and
2. Expenditures on medical treatment.

The financial burden of disease provides a fairly straightforward measure which is easily understood by non-experts. It does however have several draw-backs. Firstly, as noted by *Pearce et al.* [2006], problems can arise when expenditure decisions are not taken by those affected. If for instance decisions are made by social administrators which lead to larger or smaller medical expenditures, this will cloud value estimates. The required measure should be based on the individual's personal expenditure decisions which represent their preferences.

Another issue in the current context is with the correct identification of the causal effects of arsenic related illnesses. The relationship between arsenic consumption and illness is complicated and not always clearly observed, especially with consumption at low concentration levels. It is thus likely to be extremely difficult to produce accurate estimates of the true costs of illness related to arsenic. For instance other non-arsenic related cancers may be included within the valuation study, thus, biasing the figure upwards. Likewise, arsenic related illnesses could be misidentified, thus, biasing the estimate downwards.

Finally, the main draw back with the COI approach is that it does not provide a theoretically consistent approximation to WTP. A COI approach will likely under value the damage caused by arsenic consumption as it does not take into consideration the lost utility caused by the pain and suffering of illness. These utility effects are however covered by the remaining valuation methodologies and will be discussed in more detail later. In this respect the COI approach is most similar to the human capital approach discussed earlier, however, it includes medical expenditures as well as forgone earnings.

As the COI approach will almost certainly yield a lower value than the other approaches we consider, if the benefits calculated via the COI approach suggest a project passes the cost-benefit test, a fuller study would likely also come to the same conclusion. As such the COI approach, being quicker, less costly and perhaps easier to understand, might be the preferred methodology if a quick decision is needed and if the necessary data is available.

For a thorough treatment of the stages and process of conducting a COI study, see *Dickie* [2003]. The basic steps involve i) Defining the valuation problem. Here the baseline against which the damage is assessed is defined, along with a link being established between arsenic and illnesses. ii) Both the direct costs and indirect costs are estimated. Here the direct costs will be medical expenses whilst indirect costs will be those incurred through lost wages, whether through illness or death. iii) Finally, the costs should be adjusted to a common year. This entails the discounting of any future costs considered.

Another methodological approach which could be used to estimate the costs of illness is production modelling. Applied economists often try to model production relationships, such as agricultural or industrial processes, in order to examine beside productivity and efficiency developments, the impacts of policies or the progress of technological change. In terms of arsenic however, the health of workers may diminish their effectiveness in the production process. Or, if the effected individual is a manager/owner, poor health could lead to poor resource allocation. The effect of this may be a reduction in output and revenue if worker health is adversely affected leading to reduced productivity or unnecessarily high costs if there is inefficient resource allocation.

In order to examine the direct impact of worker health on output, production functions could be estimated using regression techniques with labour time and an arsenic related labour health indicator as independent variables, whilst controlling for all other relevant production inputs. The estimated labour health parameter in the model will indicate the impact that poor health has upon output. Multiplying this factor by the market price of output and the number of affected workers will allow a valuation to be estimated for arsenic consumption. Such a methodology may perhaps only be useful in the more dramatically impacted areas where arsenic consumption is high and arsenic related health cases are identifiable. For a general review of production and efficiency methods see, for example, *Coelli et al.* [2005] or *Fried et al.* [2008].

COI studies that consider arsenic are relatively rare compared to studies utilizing other methodologies. One study which uses this method is *Khan* [2007], who studied the costs of illness related to arsenic consumption in Bangladesh. The research estimated a dose response function between arsenic consumption and work days lost to illness, as well as a medical expenditure demand function and a defensive behaviour demand function. A discussion of this approach is provided in the next section. One intriguing aspect of this study was that it found that the number of work days lost to arsenic was actually very low, so, given the relatively low wages in Bangladesh, the indirect costs of arsenic illness were low. The study attributes this to the prevalence of poverty in Bangladesh, meaning that a worker would perhaps have to be severely ill in order to miss work. A further study which analyses the indirect costs of arsenic related illness in Bangaldesh is *Carson et al.* [2010]. Here the authors analysed the impact of arsenic on total labour supply, allowing them to include the impacts of arsenic-related illness on family and friends who may be forced to allocate time towards nursing and away from work.

2.3.3.2. Averting Expenditures (AE)

In the previous section we discussed a method for estimating the damages attributable to arsenic consumption by examining the resource losses, both direct and indirect, caused by related illnesses. These are, however, not the only expenditures that can be attributable to arsenic. If people have knowledge that arsenic consumption may impact their health adversely then it would be expected that they would take steps to minimise their exposure. This could be in the form of purchasing bottled water, installing a water filter, using a friend's or neighbour's water supply, or perhaps paying for a connection to a mains water supply (if the problem is related to a private well). These choices all involve costs, either direct monetary costs such as those needed to purchase bottled water or a filter, or the time and inconvenience costs for having to travel to another water source. The fact that this choice of action has been taken reveals a value that the decision maker is willing to pay in order to minimise their exposure to arsenic risk.

Whereas the previous COI method assumes that there are no feasible defensive activities or perhaps that feasible activities are ineffective, the averting or defensive expenditure approach utilises this behaviour as a basis for valuing the damage caused by arsenic consumption. There are several approaches to take within this methodology. In order for this measure to be theoretically consistent it must be very clear from the outset what is being valued and the appropriate approach must be chosen. See, for example, *Dickie* [2003] for more details.

The COI and AE are not, however, mutually exclusive methods, given that expenditures may not be effective and are often used in parallel. For a detailed theoretical model of health valuation in the context of arsenic consumption see *Acharyya* [2009]. *Khan* [2007], for instance also collected data on defensive expenditures for arsenic exposure minimisation but found only a small number of households took such actions. An earlier study by *Aftab et al.* [2006], studied averting expenditures of Bangladeshi households to avoid arsenic exposure, particularly focusing on what factors may determine a households expenditure decisions. The study found that household's with greater knowledge of arsenic illnesses were more likely to spend more time and money on minimising exposure. This finding illustrates the need for effective information and education programs to help households to formulate effective defensive strategies.

A further averting expenditure study related to arsenic consumption in Bangladesh is *Madajewicz et al.* [2007]. The researchers examine the impact of information on households' decisions to conduct averting behaviour, with results indicating that information and education have a huge role to play in reducing arsenic exposure. This study also highlights one failing of the averting expenditures methodology at capturing the true economic value of health risks. Where people are facing risks such as arsenic exposure which they know nothing about, they will not be going out of their way to avoid the risk and an economic value calculated using the averting expenditures methodology will be inappropriate. However, if they did have correct information they would be able to represent their WTP to avoid the risk through averting behaviours as long as other constraints such as social, technical and cultural aspects do not limit such behaviours. In other words, observed averting expenditures may reflect more the level of information about the illness than the true economic costs.

Furthermore how individuals perceive the risks that they face may have a large impact on behaviour, and thus any estimated WTP values based on behaviour, rather than just the pure knowledge of risks. Health risks are ubiquitous in decisions taken in everyday life, from what food to eat and what transport to take, to larger decisions such as where to live and what employment opportunities to seek. An understanding of the responses to these risks, driven by an individual's preferences towards bearing risk, is vital when considering changes in policies which will impact the degree of risks which individuals face.

A US-based study by *Jakus et al.* [2009] found that expenditure on bottled water, averting away drinking tap water, increased with risk perception of arsenic. This risk perception was different for smokers and non-smokers and was higher than expert predictions of the risk. Both knowledge of the underlying risks of arsenic and the perceptions of the magnitude of those risks will impact the economic estimates of averting expenditures.

2.3.3.3. Hedonic Pricing

Hedonic pricing methods involve decomposing the price of an item into its constituent parts to identify the impact of health risks. For example, workers who have jobs that entail higher than average health risks, such as fishermen or construction workers are often paid more than somebody in a job requiring similar skills but without the increased health risk. If the decision to be employed in this job was based on free choice and full information then it identifies an amount of risk that the worker is willing to accept (WTA) for the higher salary.

Although this method is not as suitable for studying arsenic compared with the previous methods, it is an important technique in that it is the basis for many VSL estimates. Hedonic pricing methods are also frequently used on housing price data. Where a house might have a private water supply affected by arsenic, the negative impact of that can be analysed by statistically comparing the house price with that of other similar houses, unaffected by arsenic in its water supply, whilst appropriately accounting for the many other housing characteristics.

One such property study which addresses arsenic contaminated wells is by *Boyle et al.* [2010]. This study examines the impact of arsenic testing on house prices in two towns located in Maine, USA. The study found that the initial arsenic results and media attention led to significant reductions in house prices. The effect was found to be very short-lived however, with the decrease lasting approximately 2 years. The authors attribute this short term impact to either the installation of treatment systems or the subsequent reduction in media stories related to arsenic contamination in the area. This again highlights the important role of information in estimating economic values.

For an overview of issues see *Freeman* [2003] or *Taylor* [2003]. Specific applications of housing hedonic studies which are water related but non-arsenic focused include *Steinnes* [1992], *Leggett and Bockstael* [2000] and *Poor et al.* [2007]. For an overview of hedonic wage studies, as well as wider health valuation issues, see *Viscusi and Gayer*

[2006].

2.3.4. Stated Preference

2.3.4.1. Contingent Valuation

When there is no behaviour to examine, either due to a lack of information or lack of appropriate markets, and when COI studies are complicated to conduct due to confounding factors, stated preference methods offer a relatively straightforward way to estimate values of risk reductions. Contingent valuation (CV) is perhaps the most popular of the methods and involves asking individuals about their WTP for safe, arsenic free, drinking water. The questions are posed in the form of a potential purchase in a contingent, hypothetical, market. There are several variations of this method and for a full treatment see for example *Bateman and Willis* [1999], *Bateman et al.* [2002] or *Boyle* [2003]. There are also several well-known complications and biases which may be present if a CV survey is poorly designed. These biases include hypothetical bias if the scenario under consideration is not believable to the respondent, strategic bias if respondents are trying to deliberately misrepresent their true WTP and insensitivity to scope where respondents may not be able to distinguish between different levels of risk reduction and state a similar WTP for very different outcomes. A well designed survey and realistic hypothetical scenario is thus crucial for obtaining unbiased and high quality WTP results.

A major CV study examining arsenic damage is *Ahmad et al.* [2005]. This Bangladesh based study examined households' WTP for safe drinking water. Surprisingly the WTP values were relatively low considering the health issues involved. In fact the study found that households had much larger WTP for the convenience offered by piped water systems than for arsenic free water. The conclusions of this study are that awareness levels need to be raised in order promote defensive behaviours by households.

2.3.4.2. Choice Experiments

Although contingent valuation methods have been the predominant stated preference technique, choice experiments have become increasingly popular in the past few decades. Choice experiments, as with contingent valuation, are based on survey questionnaires. Respondents to the survey are asked to pick a preferred choice between several different alternatives. These alternatives are defined by several characteristics, where price is often a key characteristic of interest. These options have different levels for the characteristics. For instance if a study was interested in the levels of arsenic in a piped water supply, the options would include as characteristics, the arsenic concentration, price and any other variables of interest such as taste or appearance. Statistical analysis of the results of a choice experiment can then provide estimates of the sample's WTP to reduce arsenic in their water supply or the trade-offs that respondents are making between the characteristics.

An example of a choice experiment study analysing arsenic damage is *Pascual et al.* [2009]. Here the researchers were specifically interested in the latency aspect of arsenic consumption. They studied the trade-offs that respondents were willing to make between risk and the latency of illness.

A common use of choice experiments is in the analyse of the trade-offs that individuals are willing to make between small changes in risk and money. The results can then be used in a VSL calculation, as an alternative to the hedonic wage method. Some recent examples include *Adamowicz et al.* [2011] and *Cameron et al.* [2010]. Using stated preference techniques in this situation allows the researcher to focus on a specific type of risk rather than a job based risk. This would be advantageous when the source of risk matters. *Dekker et al.* [2011] for example conduct a meta-analysis of stated preference studies and find that the risk context is important in determining WTP. This would suggest that VSL estimates which are not based on arsenic or cancer related studies, may not be appropriate for analysing the impacts of arsenic poisoning, without suitable adjustment.

2.4. Benefits Transfer

The previous sections of this chapter have briefly described the methods available for conducting an original valuation study to estimate the impacts of arsenic consumption. These techniques, however, require much time, effort and expertise to obtain high quality estimates. Costs are also likely to be high, especially when a new survey is required as with stated preference techniques. Instead of conducting a novel study, specifically tailored to examine the exact issue of interest, data from previous studies could be transferred to the new setting. This use of existing data is known as benefits transfer.

Rosenberger and Loomis [2003] categorise benefits transfer studies into two groups: value transfers and function transfers. Value transfers are the simplest method of utilising previous research and involve conducting an appropriate literature review and selecting the most appropriate values for the current focus. In terms of arsenic, a researcher would examine the literature regarding health benefits and pick a study with a suitable scenario. The values from this study, such as WTP to reduce risks could then be used to assess the issue at hand. Alternatively an agency could have a pre-selected value to be used for analysis, such as VSL figures used by the EPA and other agencies. Function transfers on the other hand are more complex and involve tailoring the values in pre-existing studies to the new scenario by taking into consideration differences between the scenarios. Although more involved, this exercise has been shown to provide more accurate results than value transfers.

2.5. US Environmental Protection Agency Cost Benefit Analysis

In the course of the previous discussion regarding valuation techniques which can be utilised to assess the impact of arsenic consumption, the empirical applications were mainly based in areas of Bangladesh where the problem is catastrophic. In this section we examine the application of economic valuation in a developed country, with a discussion of the US EPA's decision to lower the maximum contamination level of arsenic in drinking water from 50 μg/L to 10 μg/L. Furthermore, the empirical studies highlighted in this chapter have mainly been stand-alone studies to investigate the damage caused by arsenic consumption or have used arsenic consumption issues to explore wider issues of how individuals behave in the face of environmental risks. Here, however, the valuations are used to assess a potential policy through a CBA.

The decision to lower the arsenic limit to 10 μg/L rather than lowering it even further or retaining it at its original level was partly based on a CBA. This decision and the role of CBA have led to much controversy, with much of the debate surrounding the use of VSL estimates, which was set at $6.1 million per life. Here we describe some of the key points of the debate. For full details of the economic analysis involved in this decision, see *US EPA* [2001].

The use of $6.1 million was based on hedonic wage studies from jobs which involve workplace risk. The risk premium for this workplace risk was then aggregated up to the value of whole life saved. This type of analysis has received much criticism. *Heinzeling* [2002], who offers many criticisms of CBA in general as well as the role of CBA in the arsenic decision, notes that workplace risks are very different from those experienced through the consumption of arsenic. Workplace risks are risks of sudden death or injury whereas arsenic consumption has impacts over a potentially large latency period. Workplace risks are also somewhat voluntary whereas risks through the consumption of drinking water could be argued to be involuntary, especially if there is a lack of information. The type of individuals employed in the risky jobs may also not be representative of the wider population who are exposed to arsenic, particularly in their level of risk aversion or tolerance. Considering these factors, the application of VSL estimates based on hedonic wage studies to a wildly different context may not be appropriate without alteration.

A further comment by Heinzerling is that the EPA analysis did not include a ‘cancer premium’. A ’cancer premium’ is an addition to the VSL to account for the fear of cancer that it is claimed that people have, over and above other illnesses. At the time of the EPA decision there was little empirical evidence related to the existence of a cancer premium. Since this time there have been many new studies examining the WTP to lower cancer risk as opposed to other non-cancer health risks. For instance the studies of *Hammitt and Liu* [2004], *Van Houtven et al.* [2008] and *Tsuge et al.* [2005] all found evidence of a cancer premium whereas *Hammitt and Haninger* [2010] found little evidence. An arsenic focused study by *Viscusi et al.* [2012] found that the cancer premium is roughly 25% above the normal VSL. Taking this into account would result in a substantial increase in the benefits calculation for any new arsenic rule.

Subsequent analysis of the arsenic decision by *Burnett and Hahn* [2001] criticised the EPA analysis for not appropriately considering the latency period of the illnesses caused by arsenic consumption. For instance many of the illnesses caused by arsenic consumption occur many years in the future. In order to account for this latency properly, future costs should be discounted. Based on their analysis, factoring in this issue by discounting, the new regulation is not cost-effective and may in fact lead to a net loss of life. This is attributed to the lowering of income levels through additional water costs and the potential resultant lowering of health care expenditures.

2.6. Critical Issues with Cost-Benefit Analysis

The description of methodologies provided here is intended as an introduction to CBA and the valuation of health, rather than an in-depth exposition. Although more detailed overviews, as well as practical examples, have been cited, there are many important issues which need to be considered when conducting such an analysis. In this section we highlight the more prevalent issues.

The first among these issues is subjective risk preferences and the potential of a cancer premium. It has been understood for some time that individuals perceive different risks in very different ways and in particular there are differences between expert and lay judgements of risk *Slovic* [1987]. As discussed in the previous section there is evidence of an increased WTP to reduce cancer risks over and above other non-cancer based risks. Other sources of risk heterogeneity which may impact an individual's preferences and WTP to reduce the risk include the latency of the risk, the level of perceived control over the risk and severity of the risk. See *Erdem and Rigby* [2013] for some discussion of this literature and an analysis of risk heterogeneity using a stated preference methodology. With respect to arsenic, *Pascual et al.* [2009] examined the role that latency plays in individual decision making regarding risks. The study involved a stated preference choice modelling experiment which asked villagers in Bangladesh to choose between wells which varied between the absolute risk of drinking water from them and the time span with which the illness would occur.

Another key issue is the discount rate used to weight future benefits. When considering a public investment which may have benefits for several generations in the future, these benefits need to be converted to a net present value (NPV) so as to be compared with costs within a CBA framework. Differing discount rates however may result in different outcomes of whether to accept or reject the project. Thus, a high discount rate assigns little weight to future benefits and so a rejection of the investment becomes more likely. A thorough analysis would justify the use of a discount rate and provide sensitivity analysis to highlight the potential impact that this parameter has on the outcome of the analysis. More importantly, and following research by *Weitzman* [2001] and *Gollier* [2001] a schedule of discount rates that decline over time, rather than a single constant rate over the whole decision horizon, may be more appropriate. This will depend upon the extent to which the decision horizon extends across generations and the degree of uncertainty about what discount rate to use.

When analysing potential future projects it is highly unlikely that outcomes will be fully known and thus some degree of uncertainty will be an issue with CBA analysis and appraisal in general. A thorough analysis would include sensitivity tests on any uncertain variables or if probabilities and outcomes can be estimated, expected outcome analysis could be utilised. The precautionary principle could be employed if uncertainty is seen as a major problem, whereby any potential action is delayed until more information is known about possible outcomes. Applying the precautionary principle too frequently however may result in inaction. Where individuals are facing serious health risks, inaction of policy makers may not be a viable option.

The monetary valuation techniques discussed in this chapter allow projects to be ranked according to their net present benefits. However, when monetary values are used as general information within a democratic debate, theoretically, it is not obvious that these values are more useful than physical units (such as lives lost or additional cancer cases) due to normative assumptions that accompany such an analysis. A CBA cannot take every citizen's or decision makers' normative views into account simultaneously. As such the information that they will receive from the valuation process will not be based on their own personal normative views. (See *Nyborg* [2000] for a theoretical analysis of the usefulness of CBA compared with physical units in policy analysis.)

A final major consideration when performing CBA is the distribution of winners and losers of any policy. As stated earlier, the CBA framework is built on the criteria that those who gain could potentially compensate those who lose out from the implementation of a policy. This, however, does not require that this compensation will take place. As such there will be variable outcomes for individuals which should be analysed before action is taken. For instance, a policy which benefits individuals who are relatively wealthy at the expense of those who are worse off may not be appropriate even if the policy results in net benefits.

2.7. Conclusions

The methods and studies reviewed here indicate ways of valuing the health impacts of arsenic consumption, both in developed and developing countries, to be used to help allocate resources to policies where they can be most cost effective and thus have the largest benefits. These methods involve either observing economic behaviour in related markets, such as the market for medical care for COI approaches, or where this is not possible conducting a thorough survey which questions respondents about their likely behaviour in hypothetical markets. Both types of methodology require careful understanding of what information and perceptions that the respondents have for the arsenic risks under consideration.

Conducting a detailed, original, health valuation study can be time-consuming and costly. Where time is of the essence and budgets or expertise are constrained, benefits transfer methods can be used to transfer data from pre-existing studies to the new context. Moreover, a pragmatic approach is sensible when considering a new remediation scheme. Where experience suggests that large risk reduction benefits can be acquired for little cost, clearly a full individual economic analysis, with associated costs, will be unnecessary and unhelpful.

2.8. Epilogue

This chapter has focused on providing a non-technical introduction to the methods of economic non-market valuation for policy makers and other stake-holders in the water industry. As such the scope for in-depth critical reflection was minimal. In this short epilogue we provide further reflection on the methods discussed in this chapter and provide specific discussion in terms of the research objectives of the empirical chapters of this thesis, i.e. to examine and value preferences for water with reduced arsenic risks in rural Cambodia.

One of the early economic analysis' of the arsenic crisis is *Maddison et al* [2005] who used value of statistical life estimates and dose response functions to place a value on the health crisis. Although the health effects of arsenic are fairly well known, the use of VSL estimates from other studies and transferring them to the arsenic context has many associated issues. Many of these issues were explored in the chapter but of particular note is the potential of a cancer premium and the latency aspects of arsenic related illnesses. VSL estimates are often based on the studies involving risks of immediate death, a far removed situation to that of cancers developing after years of contaminated water consumption. In light of this, and the fact that relatively few VSL studies exist for Cambodia, this type of analysis is ruled out for this thesis.

In order to value preferences for improved water sources in rural Cambodia one or more of the methods outlined in this chapter is required. We categorised the techniques into revealed and stated preference groups depending upon whether values are based on actual behaviours or on responses to questions in a survey, respectively. The 'cost of illness' revealed preference methodology is perhaps the quickest and easiest to understand of all methods. In developing countries however medical treatments are often hard to access. Demand for medications and ultimately expenditures, due to preferences for improved health, may thus be unfulfilled and basing welfare values on these may lead to biased results. Furthermore estimating accurate causal relationships between arsenic consumption, health impacts and medical expenditures would be challenging in rural Cambodia.

Averting expenditures, in contrast, uses cost data from market transactions that households may make in order to avoid exposure to arsenic. In rural Cambodia, and elsewhere, the causal link between the expenditures and arsenic exposure can be established through direct questioning in social surveys. However a further problem with identification in this methodology is 'joint production' whereby other factors that may affect utility directly may also play a prominent role in the decision process leading to a change of water sources. For instance the households may have preferences for improved taste of water which may also motivate a change in water source. Differences in other attributes of water sources are thus likely to play an important role in the households' water source choice decisions. Furthermore averting actions are limited not only by money but also on technical restrictions and cultural bounds. For instance a well informed household who is willing to pay substantial sums of their income on lowering their exposure to arsenic may be prevented from doing so by the ownership rights of other potential water sources. These theoretical and empirical issues are explored in detail in chapter 5 where we present the results of an averting expenditures analysis.

The other category of nonmarket valuation methodologies, stated preference methods, are relevant for the study of preferences when other attributes, in addition to arsenic risk, are assumed to play a significant role in water source choice and where details of the households actual choice options are unavailable. In the first two empirical chapters of the thesis we present the results from a choice experiment designed to analyse these important factors, in addition to controlling for these attributes when estimating WTP values.

Although choice experiments are widely used in developing countries there are many additional challenges in these settings over and above those faced in developed countries. Chief among these challenges is the issue of constrained money income. Households in developing countries are often more likely to use barter or work exchange in lieu of monetary transactions to facilitate the exchange of goods or services, than households in developed countries. We argue in Chapter 4 that this will result in biased results and present the results from a split sample choice experiment that examines this issue further.

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3. Arsenic in Drinking Water: Willingness To Pay, Preference Heterogeneity and Drinking Water Standards in Cambodia

3.1. Abstract

We estimate the value of reduced arsenic concentrations in drinking water, using a discrete choice experiment (DCE), amongst the arsenic-informed rural poor in Kandal Province, Cambodia, a province whose rural population is heavily exposed to arsenic contaminated groundwater. Controlling for scale differences, we identify 3 latent segments whose valuations of alternative arsenic related cancer risk thresholds differ markedly. Each of the 3 segments exhibit a coherent ranking of values regarding the arsenic risk levels presented. However, there exists variation in the pattern of diminishing marginal utility regarding reduced arsenic concentrations and associated risks. For a 4th segment we were unable to estimate reduced arsenic risk-cost trade-offs due to lexicographic behaviour towards the price attribute. This behaviour was evident from serial selection of the highest-priced water source. Willingness to pay (WTP) estimates indicate that a majority of households are willing to pay to mitigate their

arsenic risks beyond the current arsenic drinking water standard in Cambodia. Our results, thus, suggest that a lower permissible limit for arsenic, resulting in a higher proportion of household water sources being labelled 'unsafe', may better represent the preferences of households in Kandal Province and may help to instigate further water source switching or other mitigation behaviours by households.

3.2. Introduction

Consumption of arsenic contaminated groundwater in many countries is a serious public health concern. The contamination of groundwater by arsenic in Bangladesh alone has been labelled "the largest poisoning of a population in history" [*Smith et al.*, 2000] with around 20% of all-causes-mortality in arsenic impacted groundwater regions of Bangladesh attributed to arsenic exposure [*Argos et al.*, 2010]. However the hazard is found across the world [*Smedley and Kinniburgh*, 2002] and it has been estimated that globally nearly 50 million people have drunk arsenic contaminated water above 50 micrograms per litre (μg/l) and well over 100 million with water with high concentrations of geogenic (i.e. non-anthropogenic) arsenic (defined here as >10 μg/l, the current (2014) provisional WHO guideline) [*Ravenscroft et al.*, 2009]. In addition to Bangladesh, and in roughly decreasing order of peak exposure (ibid.), other countries with high groundwater arsenic hazard include India, China, USA, Myanmar, Pakistan, Argentina, Vietnam, Mexico and Cambodia.

In Cambodia, many studies over the last 10 years have found high arsenic hazard in groundwater drinking water sources [*Polya et al.*, 2003, 2005; *Feldman et al.*, 2007; *Berg et al.*, 2007; *Buschmann et al.*, 2007; *Polya et al.*, 2008; *Sampson et al.*, 2008; *Kocar et al.*, 2008; *Quicksall et al.*, 2008; *Rowland et al.*, 2008; *Benner et al.*, 2008; *Sthiannopkao et al.*, 2008; *Polizzotto et al.*, 2008]. A consideration of the geological/geographical factors controlling the development of high geogenic arsenic groundwater systems [*Charlet and Polya*, 2006] and more detailed geostatistical modelling

[*Lado et al.*, 2008; *Winkel et al.*, 2008; *Sovann and Polya*, 2014] indicates that these systems are to be found in many of the more flat-lying provinces of Cambodia, particularly in areas near the Mekong River. Kandal Province, immediately south of the capital Phnom Penh (see Figure 3.2) is the province most significantly impacted as a result of the coincidence of high groundwater arsenic, high population density and a high dependence on groundwater for drinking water supplies [*Sovann and Polya*, 2014].

Human exposure has been demonstrated through studies of various biomarkers [*Kubota et al.*, 2006; *Gault et al.*, 2008] and cases of arsenicosis have been recorded by *Mazumder et al.* [2009], amongst others. One study estimates that over 100,000 people are exposed in Cambodia, with the majority of those living in Kandal Province [*Sampson et al.*, 2008].

Chronic arsenic exposure can lead to a wide range of health consequences such as lung, bladder, liver and skin cancers, skin hyperpigmentation and keratosis [*NRC*, 1999, 2001; *IARC*, 2004]. Other health outcomes include increased risks of ischaemic heart disease and immune system disorders [*Polya et al.*, 2010, and references therein]. Many arsenic attributable health outcomes are not contemporaneous with exposure; where data are readily available, such as in Chile, childhood exposures in particular have been linked through detailed epidemiological studies to peaks in arsenic attributable deaths occurring decades after the exposure [*Steinmaus et al.*, 2013]. Exposure to arsenic contaminated drinking water is thus a serious public health concern in Cambodia, both now and for the future [*Fredericks*, 2004].

As well as examining the numbers of people who are at risk from this public health emergency an economic value of the problem has also been investigated to highlight the potential economic benefits of remediation. In Bangladesh for instance, *Maddison et al.* [2005] estimated the willingness to pay (WTP) to avoid the potential arsenic-related health impacts was \$2.7 billion annually. This estimation was based on dose-response functions, estimates of the population at risk and value of statistical life estimates.

One potential public health reaction to this growing problem is to set and enforce a permissible limit on arsenic concentrations in drinking water. Such a limit would act as a benchmark for safe drinking water within the country and would guide remediation and education efforts. Implementation and enforcement of such a limit, particularly in rural areas where households may have individual water sources, may however prove to be expensive and challenging. Furthermore there exists some debate on what that limit should be, notwithstanding the existence of the current (2014) World Health Organization (WHO) provisional guideline of 10 μg/l. In the US the arsenic permissible limit was lowered from 50 μg/l to 10 μg/l and came into effect in 2006. This decision was supported by evidence from a cost benefit analysis. However the details of this analysis and the subsequent change in the limit have been questioned by some [see, for instance, *Burnett and Hahn*, 2001; *Sunstein*, 2002a; *Heinzeling*, 2002; *McGarity*, 2002; *Sunstein*, 2002b; *Oates*, 2006; *Cho et al.*, 2010]. In Cambodia, and many other developing countries however, the permissible limit remains at the older provisional guideline level of 50 μg/l and many groundwater sources in Kandal Province, the focus of this study, have concentrations in excess of even this level.

Smith and Smith [2004] argue that, in developing countries, lowering the permissible limit from 50 μg/l to 10μg/l could be detrimental to public health as it may postpone short-term solutions (that could lower arsenic levels to at least 50 μg/l from much higher levels), in favour of more radical solutions which may take many years to fund, organise and implement. They conclude that the setting of water standards must be made with considerable care and that it might be advisable to maintain the current standard of 50 μg/l in developing countries. Furthermore it is critical that alternative safe sources are available so that arsenic risks are not merely substituted for other microbial risks.

However, such a logistical argument, as argued by Smith and Smith, does not consider that the impacted population may be averse to exposure at even relatively low arsenic concentrations. Given information regarding potential risks households may be willing to spend relatively large sums of their own money to change water sources or otherwise limit their exposure. The expenditures from these individual actions would also provide an indication of the potential benefits of remediation and education work. It is important that these concerns and preferences of the population be considered when setting permissible standards in order to realise the most efficient allocation of resources. Moreover, as *Hope* [2006] points out, a key lesson from project experience is that people must be at the centre of the development process and that appraising social priorities and preferences before projects commence permits a more accurate estimation of policy impacts.

Relatively little is known, however, of the values to Cambodian households of arsenic risk reductions needed to effectively assess permissible limits as well improving the design of associated education and remediation efforts. In this study we address this issue, presenting the results of a DCE aimed at understanding the strength of preferences towards risk from arsenic contaminated drinking water in Kandal Province.

The use of DCEs to estimate values for environmental stocks and services is increasingly common-place in developing countries. The successful application of stated preference valuation tools, and DCEs in particular, is challenging. Careful selection of the payment vehicle, survey sample, questionnaire design, experimental design and clear communication of the environmental good/service in question, amongst other aspects, are critical for a valid study[*Bateman et al.*, 2002]. The use of these methods in rural parts of less-developed countries, where education levels are often lower, is more challenging still. Additional complications include language, cultural, ethical and logistical barriers [*Bennett and Birol*, 2010; *Whittington*, 1998, 2002, 2004, 2010]. Moreover, these difficulties are more acute when using an attribute such as risk; a concept that is challenging for the researcher to effectively communicate [*Harrison et al.*, 2014] and for the respondent to accurately comprehend. Our results however suggest that we were able to successfully overcome these issues and thus meaningful conclusions can be drawn.

Traditional choice models such as conditional logit assume preference homogeneity. In order to assess preference heterogeneity within our sample area we present the results from latent class choice models[1]. We further extended these results by estimating scale classes which allows an examination of the degree of choice consistency within segments of the sample. Willingness to pay (WTP) values are estimated for each latent preference class and the potential benefits of remediation and a more stringent permissible arsenic limit are discussed.

This paper is organised as follows: part 3.3 provides an overview of the economics literature on water quality in general and arsenic contamination in particular; part 3.4 presents the econometric models estimated; part 3.5 details the choice experiment and data collection process; part 3.6 presents and discusses the empirical results and part 3.7 concludes.

3.3. Context and Related Literature

The number of valuation studies addressing the value of arsenic risk reductions in developing countries is comparatively small. There exists, however, a broader body of work which examines the value of water quality improvements in terms of microbial risk or convenience improvements, in both developed and developing countries [see, for example, *Whittington et al.*, 1990, 1991; *Persson*, 2002; *Casey et al.*, 2006; *Genius and Tsagarakis*, 2006; *Genius et al.*, 2008; *Vasquez et al.*, 2009; *Nauges and Whittington*, 2010; *Orgill et al.*, 2013; *Jessoe*, 2013]. However, important differences between arsenic risk and microbial risk complicate a potential benefits transfer approach. Most notable among these differences is the latency of arsenic-related illnesses [*Pascual et al.*, 2009], with skin lesions occurring 5-10 years after exposure whilst cancers can take decades to develop [*Mazumder et al.*, 1998]. Of those economic studies which do address the

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[1] Other models, such as random parameter logit, permit the analysis of preference heterogeneity. We analyse latent class models however as they allow the analysis of discrete preference groups which is useful for communicating results to policy makers.

arsenic crisis, the overwhelming majority focus on the problem either in Bangladesh [see, for example, *Ahmad et al.*, 2005; *Maddison et al.*, 2005; *Aftab et al.*, 2006; *Aziz et al.*, 2006; *Madajewicz et al.*, 2007; *Bennear et al.*, 2013] or in the US [see, for example, *Shaw et al.*, 2005; *Walker et al.*, 2006; *Jakus et al.*, 2009; *Boyle et al.*, 2010; *Cho et al.*, 2010; *Nguyen et al.*, 2010; *Konishi and Adachi*, 2011; *Shaw et al.*, 2012].

There are several studies which have examined household knowledge and preferences towards the arsenic crisis in Kandal. A knowledge, attitudes and practices (KAP) report [2009] by the Ministry for Rural Development and UNICEF found that, of those households who were aware of arsenic, 47% stated that they would not be willing to pay for an alternative source. Out of those stating an unwillingness to pay however, 75% stated that their response was due to there being enough free water sources available as alternatives. A further Cambodian-focused study, conducted for the World Bank's Water and Sanitation Program (WSP), explored the willingness of households to pay for different types of alternative, arsenic-free, water sources [*RDIC*, 2012]. The study found that, of the potential alternative sources, households were most interested in, and were most likely to express a willingness to pay for, piped water supplies and water from a vendor, ahead of sources such as rainwater and alternative wells.

Given that these reports indicate the abundance of alternative sources and the household preferences for certain types of sources, especially those with convenience properties, revealed preference approaches for examining arsenic risk reduction preferences become challenging. A stated preference methodology allows abstraction to a more constrained scenario for specifically examining the preferences of households for arsenic risk reduction rather than other attributes associated with water sources.

In Bangladesh, a large scale valuation study, found that households have a relatively low WTP for arsenic-free drinking water, compared to the WTP for convenience attributes of a water source, such as having access to a tap within the households grounds [*Ahmad et al.*, 2005]. The authors attribute this low value for arsenic risk reductions to a combination of high personal discount rates, relatively long latency periods of arsenic related illnesses and inadequate levels of information related to the dangers of arsenic.

A number of studies focus on the importance of education and awareness in determining the demand for environmental goods. *Jalan et al.* [2009], for instance, suggests that a presumption commonly made within the economics literature is that low demand or WTP for environmental quality, as with *Ahmad et al.*, is due to low incomes. A low WTP would therefore be a reasonable reflection of the actions of an agent acting in their own self-interest. The authors however point out that this presumption overlooks that a lack of education and awareness may be behind a low, and thus inefficient, level of demand.

The role of arsenic risk awareness and arsenic information campaigns has been examined in a number of water source switching studies. In Matlab, Bangladesh, *Aziz et al.* [2006] found that arsenic information campaigns had little impact on source switching behaviour relative to the convenience of available alternative sources. Conversely, *Aftab et al.* [2006], also focusing on source switching behaviours in Bangladesh, found that awareness of arsenic health consequences has a significant and positive impact on the likelihood of utilising arsenic safe water sources.

Madajewicz et al. [2007] found that after communicating arsenic test results to a household, 60% of the households who find that their well water is contaminated with unsafe levels of arsenic (> 50 μg/l) switch their water source within 6-12 months. Controlling for other factors, learning that drinking water is contaminated with arsenic increases the probability of the household changing their drinking water source by 0.37. The study also found that for those households who change their source, the average time that they spent collecting water increases 15 fold. This indicates that switching water sources is not costless and that a proportion of households are willing to use their time to reduce their arsenic exposure. In a subsequent and related study *Opar et al.* [2007], examining interventions aimed at reducing arsenic exposure, found that 65% of households with 'unsafe' levels of arsenic (> 50 μg/l) switch their drinking water source.

The results of studies discussed here illustrate the importance of information and awareness, amongst other factors, for well-switching behaviour as well as responses to stated preference questions. In light of this issue, the study presented here focuses exclusively on households who have previously been targeted by a water testing and education program and thus have a good awareness of the issues. Such testing campaigns are taking place in many areas known to be affected by arsenic contamination and so our sample is representative of this. Although the *Ahmad et al.* [2005] study suggests that the WTP for arsenic-free water in Bangladesh is low, subsequent revealed preference work related to water source switching illustrates the strong desires for informed households to mitigate their exposure to arsenic. It is important to note however that these source switching studies identify 'safe' water sources as sources that contain arsenic concentrations lower than 50 μg/l, the drinking water standard in Bangladesh and Cambodia. Switching behaviour is thus observed in relation to a statement that a source is either safe or unsafe based on the 50 μg/l threshold. Such a safe/unsafe dichotomy does not take into account that concentrations of arsenic less than 50 μg/l may not be totally risk free, nor that within the safe and unsafe categories there is a large spectrum of risk, where a source with lower concentration of arsenic is safer than a source with a higher concentration, ceteris paribus.

One important study which considers this issue is *Bennear et al.* [2013], who examine two information messages given to households after well-testing and labelling in Bangladesh. One group received a message that explained that the national drinking water standard was 50 μg/l whereas an alternative group received a more detailed message that made it clear that lower arsenic wells were safer, as well as information related to the arsenic standard. This included a message that amongst wells labelled unsafe, a well that had an arsenic concentration of 100 μg/l would be better than a well with a 200 μg/l concentration. A parallel statement was also given for wells labelled safe. The water sources were then signed as safe/unsafe and the actual arsenic concentration levels were displayed on the well. In investigating the impact of the two messages on well switching behaviour however the study was unable to find an increase in switching to lower arsenic sources due to the more detailed message. The authors caveat their results however by noting that the safe/unsafe dichotomy is the common method of communicating arsenic risk in Bangladesh and thus the extra information may have been overpowered by the simpler message. Furthermore the more detailed risk guidance was only delivered once, verbally, rather than through a lasting material guide such as a leaflet.

The principal contribution of the DCE presented here is to examine the preferences of Kandal households for water sources with a range of cancer risk levels, including levels lower than the current Cambodian permissible arsenic concentration limit, which allows us to comment on the appropriateness, or otherwise, of the current limit in relation to household preferences in Kandal Province, Cambodia. We consciously avoided linking the risk levels used in the DCE with the current permissible arsenic limit so as to examine the households choices related to lifetime cancer risk, rather than in reference to the current choice of safety threshold. This avoids the dominance of the safe/unsafe label as discussed by *Bennear et al.* [2013].

Given the focus on estimating WTP for different arsenic risk levels, so as to comment on drinking water standards, the ability of the respondents to differentiate these levels within the choice experiment is critical. A significant finding by Sunstein and Zeckhauser (2011), who surveyed WTP responses for reduced levels of arsenic in drinking waters in the US, was that the description of the risk and thus the level of fear that the risk induced, was fundamental to the ability of respondents to distinguish risk magnitudes for WTP responses. Sunstein and Zeckhauser found that an emotional or graphic description of cancer risk led to a ‘probability neglect’: i.e. comparable WTP values were expressed for risks with wildly different probabilities of occurrence. In our study the households were provided with a more graphic description and photographs of arsenic health risks/outcomes during the NGO’s arsenic testing and education program. This occurred in advance of our study, in some cases several years prior to the

DCE. Directly before the DCE was completed each household was provided an 'unemotional' description of arsenic risks, i.e. without any photographs of arsenic related skin conditions or graphic descriptions, to act as a reminder and to minimise the likelihood of 'probability neglect'.

3.4. Conceptual Framework and Choice Models

The goals of this study are to examine the impact of arsenic risk, aesthetic qualities and price on the likelihood of water source selection and to estimate WTP values for arsenic reduction to help assess the benefits of remediation work and to help determine appropriate drinking water standards. In order to address these research goals we investigate water source choice within the random utility framework [*McFadden*, 1974; *Ben-Akiva and Lerman*, 1985]. In a random utility model (RUM) respondents are assumed to make decisions which maximise utility:

$$U_{in} = V(\mathbf{x_{in}}, \mathbf{s_n}, \beta) + \varepsilon_{in} \tag{3.1}$$

where the utility that individual n derives from alternative i is comprised of a component observable to the researcher, V, and an unobservable, random, component, ε_{in}. The observable component is a function of observable alternative characteristics, x_{in}, characteristics of the individual, s_n, and a vector of parameters, β, to be estimated. Given that an individual is assumed to be a utility maximiser, the individual will choose alternative i from the set of all available alternatives, J_n, when the utility gained through the choice of alternative i is greater than the utility gained through the choice of all the possible alternatives, i.e. $U_{in} > U_{jn}, \forall j \in J_n, j \neq i$. Given the components of utility shown in 3.1 the probability of selecting alternative i thus becomes:

$$\begin{aligned} P_{in} = \Pr(V_{in} + \varepsilon_{in} > V_{jn} + \varepsilon_{jn}) = \\ \Pr(\varepsilon_{jn} - \varepsilon_{in} < V_{in} - V_{jn}) \forall j \neq i \in J_n \end{aligned} \tag{3.2}$$

The probability that an individual chooses alternative i is thus the probability that the difference between the random components of utility, a random variable, is smaller than the difference between the observable component of utility. If it is assumed that the random components are identically and independently distributed in accordance with the extreme value type 1 distribution, the probability that the decision maker will choose alternative i is:

$$P_{int} = \frac{\exp(\lambda \beta X_{int})}{\sum_{j=1}^{J} \exp(\lambda \beta X_{int})} \tag{3.3}$$

where λ (typically normalised to 1) is the scale factor and is equal to $\pi/\sqrt{6}\sigma$, (the scale parameter is thus inversely related to the error variance[2]) and where σ is the standard error of ε. t is the choice situation that respondent n faces. This is the conditional logit choice model. Allowing the scale term to vary and parametrization provides the heteroskedastic conditional logit model [*DeShazo and Fermo*, 2002; *Hensher et al.*, 1999]:

$$P_{int} = \frac{\exp(\lambda_n \beta X_{int})}{\sum_{j=1}^{J} \exp(\lambda_n \beta X_{int})} \tag{3.4}$$

The λ_n is parametrised as $\exp(\gamma Z_n)$, where $Z_n$is a vector of individual characteristics and γ is a vector of parameters to be estimated [see *Hole*, 2006 for further details].

---

[2] variance= $\frac{\pi^2}{6\lambda_i^2}$

Estimation of the heteroscedastic conditional logit model enables the examination of potential heteroscedasticity and thus the degree of choice consistency between different segments of the sample.

Although the conditional logit model is commonly used to analyse discrete choice experiments it has several significant drawbacks. Predominantly for our case it is unable to capture preference heterogeneity unless attribute and demographic variables are interacted. Accounting for preference heterogeneity allows more specific water policy recommendations to be drawn, focusing on particular portions of the population. Alternative methods which allow preference heterogeneity to be modelled include latent class and random parameter logit choice models, amongst others.

In this study we employ latent class choice models to account for this preference heterogeneity which allows differences in preferences to be examined in discrete segments. In the latent class model the choice probabilities and the marginal utilities are conditional on being in class c. The latent class choice model is thus:

$$P_{int|c} = \frac{\exp(\lambda\beta_c X_{int})}{\sum_{j=1}^{J}\exp(\lambda\beta_c X_{int})} \tag{3.5}$$

Probability of membership to a class is modelled using a multinomial logit form with a vector of individual respondent characteristics as explanatory variables. However, as with the conditional logit model, the scale term is typically normalised to 1 and any heteroskedasticity associated with differences in choice consistency will be confounded with the marginal utility parameter estimates [*Swait and Louviere*, 1993, *Louviere and Eagle*, 2006 and *Magidson and Vermunt*, 2007]. Thus choice models which fail to account for this confound are likely to be biased and misleading [*Louviere and Eagle*, 2006]. The scale adjusted latent class (SALC) model [*Magidson and Vermunt* 2007 and *Vermunt and Magidson*, 2014] allows for differences in the scale term through estimation of multiple scale classes (d) in addition to multiple preference classes (c).

In this model the probability of selecting alternative i, is conditional on the scale and preference class membership:

$$P_{int|c} = \frac{\exp(\lambda_d \beta_c X_{int})}{\sum_{j=1}^{J} \exp(\lambda_d \beta_c X_{int})} \tag{3.6}$$

Scale and preference class membership is modelled on individual characteristics using a multinomial logit form:

$$P_{nc} = \frac{\exp(Z_n \phi)}{\sum_{c'=1}^{C} \exp(Z_n \phi)} \tag{3.7}$$

where

$$P_{nd} = \frac{\exp(Z_n \varphi)}{\sum_{d'=1}^{D} \exp(Z_n \varphi)} \tag{3.8}$$

Identification is achieved by imposing that $\sum \phi = \sum \varphi = 0$. See *Burke et al* [2010] or *Rigby et al* [2014] for an application and further discussion.

To summarise, we model household hypothetical water choice probabilistically within the random utility framework. We account for preference heterogeneity through the estimation of latent preference classes and account for the confounding of the scale term with the marginal utility estimates by allowing multiple scale terms. This allows an analysis of differences in choice consistency across the sample and estimation of unbiased marginal utilities.

3.5. The Choice Experiment

The design of the choice experiment was initiated with 40 in-depth interviews conducted with households in Kandal in July 2012. The interviews focused on the level of understanding of the households with regards to arsenic contamination and the key issues which they face regarding water. The key outcome of these interviews was that water taste and appearance is a key determining factor for water source choice, along with arsenic contamination. The taste and appearance of water was therefore selected as a key attribute for the choice experiment.

Following this initial period of fieldwork and utilising data from the Cambodian Socio-Economic Survey (CSES) 2009 to select appropriate price levels, the attributes and initial levels for the choice experiments were selected. The final attributes and levels used are shown in Table 3.1 and an example of a choice card is shown in Figure 3.1.

Table 3.1.: Attributes, Levels and Coding for Choice Experiment

Attribute	Levels	Coding
Taste/Appearance	Dark with bad taste	2
	Cloudy with OK taste	1
	Clear with good taste	0
Lifetime Cancer Risk	10/500	10
Attributable to	5/500	5
Arsenic in Water	1/500	1
	0/500	0
Money (Riels Per Month)	50,000	50
	30,000	30
	18,000	18
	6,000	6

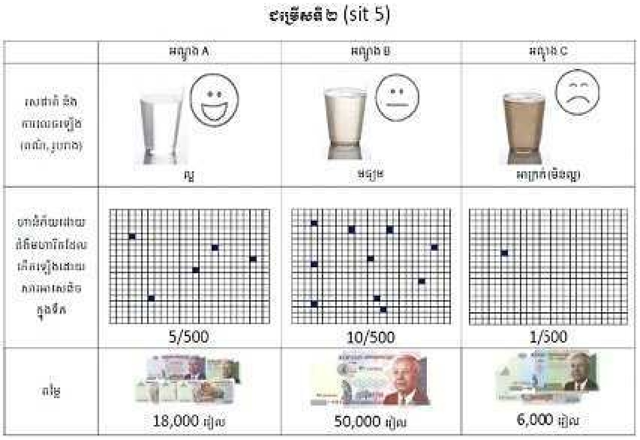

Figure 3.1.: Example of Choice Card

Only three attributes were chosen for the choice experiment so as to keep the choice task as simple as possible, given the low levels of education in the sampling area, and to focus the attention of the respondents on the three primary factors of interest. Thus price, arsenic risk and taste/appearance were selected. All other aspects of the well, such as distance from the household, water availability and microbial health risks, were described as being identical for each well. In order to further reduce the complexity of the choice task, visual aids were used for all attributes.

The choice scenario did not include a status quo or 'none' option and as such the respondents faced a forced choice scenario. The predominant reason for choosing to leave out a status quo option was that we did not have completely accurate risk information for the household's water source and so our knowledge of the choice set would be incomplete. In addition it is the households own comprehension of the water risks that would be most applicable to be included as the status quo option.

One option which might have permitted the inclusion of an appropriate status quo was utilising laptop computers or tablets for data collection so that the household's opinions could be included as a status quo and an appropriate choice set generated. We were, however, advised by local experts that these types of technology are rarely, if ever, seen in the more remote rural locations which could have potentially diverted much attention away from question answering and discussion of water collection. We thus opted for a more basic pen and paper data collection procedure and a forced choice scenario.

Forced choice scenarios have particularly been utilised in DCE studies which examine preferences for water, such as drinking water [Hensher et al., 2005] or irrigation *Rigby et al.* [2010], as water is an essential resource which households cannot go without. As such the use of a forced choice scenario in this setting is less problematic than it is more generally where an opt-out is appropriate. Furthermore tubewell hand pumps frequently break thus our scenario is not unrealistic. The choice scenario asked the respondent to imagine that their current water source had become unavailable and that the only available alternatives were the 3 tubewell sources presented on the choice cards.

Risk levels were chosen to represent conceivable arsenic concentrations in the local area; these being 100μg/l, 50μg/l, 10μg/l and 0μg/l. Additional lifetime cancer risk levels of 10/500, 5/500, 1/500 and 0/500 were presented to represent these arsenic concentration levels respectively. The risk and concentration mapping was approximated using results from *Smith et al.* [2000] and an assumption of a linear dose-response relationship with no threshold, although this is contended by some [e.g. *Lamm et al.*, 2004]. It is beyond the scope of this study to provide a detailed critique of dose-response models for arsenic-attributable cancers, however we note that the combined cancer risks chosen for various arsenic concentrations are consistent with the *NRC* [1999] reported risk of 1 in 100 (i.e. 5/500) for the then US Maximum Contaminant Level (MCL) of 50 μg/l and with their methodology assuming a linear dose-response relationship.

The increased lifetime cancer risk from drinking contaminated water was communicated via risk grids. A grid of 500 cells represented 500 people who consumed water from the well as their primary drinking water source. The shaded cells represent people who will contract cancer at some point in their life due to arsenic consumption. Risk grids are commonly used in the literature [e.g. *Krupnick et al.*, 2002]; however conveying risks within stated preference surveys is a challenging task and there are several alternative communication devices such as risk ladders, pie charts or expressing risk as a number. See *Harrison et al.* [2014] for a review of risk in the DCE healthcare literature. In order to keep the choice card as simple as possible, and to keep all attributes comparably displayed, visual aids were used to illustrate all attribute levels and, after discussion with local experts, risk grids were found to be the simplest and most effective of the alternatives.

The taste/appearance of water was communicated via pictures of coloured water in glasses to illustrate the appearance, accompanied by a face icon to express the taste. Other illustrations, such as photographs of local people making facial expressions to indicate taste, were considered and discussed with local experts. The icon depiction, however, was found to be the clearest for the respondents. The levels, attributes and choice card illustrations were discussed with local experts to ensure that all the options were sensible and realistic and that the choice cards were non-leading.

Piloting was conducted in order to inform the questionnaire, choice sets and experimental design. A Bayesian Efficient experimental design from weakly informative priors was used in the pilot study of 24 households. This allowed estimation of more informed priors for the final design and to collect general feedback on the choice experiment. All experimental designs were generated using Ngene [*Rose et al.*, 2009].

Enumerator recruitment was conducted on the basis that the selected candidates should have a good ability to speak and write English and preferably had data collection experience. Two days of classroom training was conducted, following recruitment, so that all enumerators were fully aware of how to conduct the choice experiment and complete the accompanying questionnaire. The majority of enumerators employed were students from the Royal University of Phnom Penh (RUPP) and the Royal University of Agriculture (RUA). We then conducted a further 2-day piloting period to provide the enumerators with first-hand experience of conducting the survey. The data from this period was not used for the analysis.

During the training and piloting process it became clear that attribute non-attendance towards the price of the well was an issue for many respondents. A proportion of respondents were ignoring the price and selecting wells on the basis of risk and taste/appearance alone. Following this we changed the introduction to the choice experiment to add emphasis to a 'cheap talk' section where we made it clear that although the task was hypothetical, it was important that they also consider their budgets and likely behaviour whilst making choices.

The final data collection period commenced in May 2013. The sampling strategy was to visit households who had previously been visited by a testing and education team from the NGO RDI Cambodia and so were aware of the arsenic issue. These households were targeted so as to increase engagement with our study so as to reduce attribute non-attendance and hypothetical bias.These households had at some stage used a tubewell as their drinking water source which was tested for arsenic by the NGO. Following this test, those households who received a positive test for high arsenic (>50 μg/l) faced the decision whether to change their water source. GPS co-ordinates of tested households were recorded by RDI and these were used to locate households for sampling.

Selected households were located within three communes: Banteay Daek, Samraong Thom and Kampong Phnum. These communes are shown in Figure 3.2 and are situated roughly 30-50 km south-east from Phnom Penh along the Mekong River. The communes were selected due to the high numbers of households which had used tubewells and the high proportion of households who received positive test results for high arsenic levels. The survey was conducted with one household member per dwelling who was either the household head or the person responsible for water collection. Figure 3.3 shows an interview taking place.

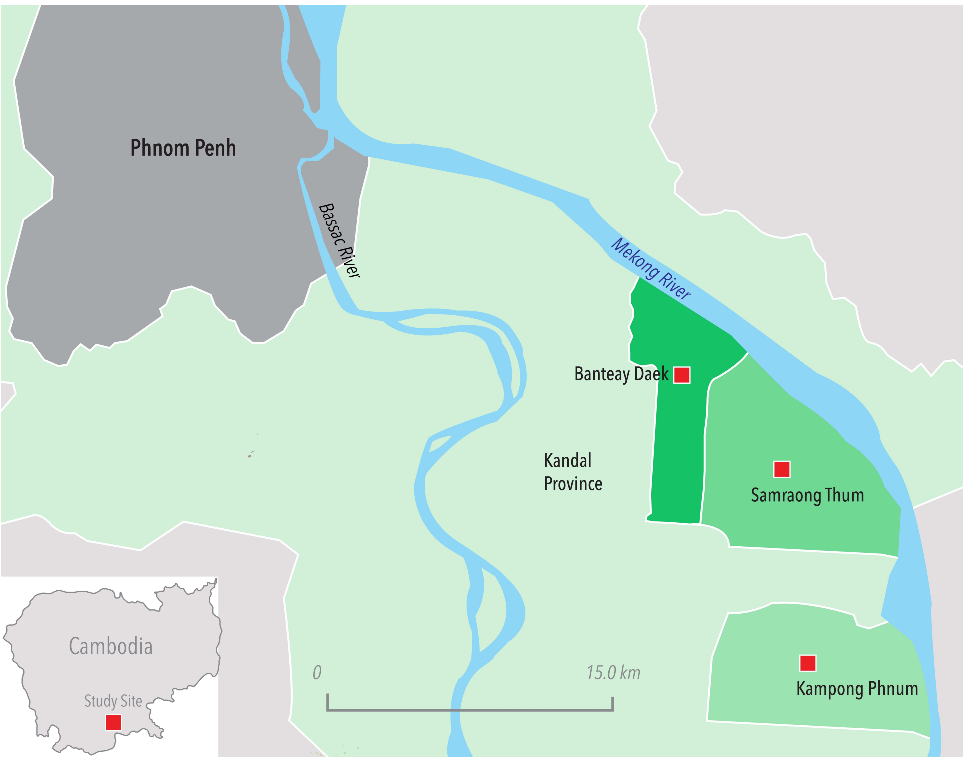

Figure 3.2.: Data Collection Sites

Descriptive statistics for the 174 sampled households are shown in Table 3.2. The descriptive statistics indicate that the average age of respondents was 49, 64.5% of respondents were female and 64% of households had received a positive test result for high arsenic in their tubewell. The Cambodian census of 2008 shows that across the 3 sampled communes the average proportion of females was 51%. The proportion of females sampled was thus relatively high, due in part to the role that females have in collecting water. The breakdown of the statistics in Table 3.2 is discussed in more detail in the next section.

Table 3.2.: Summary of Respondents by Lexicographic Preferences

	Total	HPLB	Not HPLB
Age of Respondent	48.86	47.21	49.48
	(14.81)	(14.13)	(15.01)
Gender of Respondent (0-Female, 1-Male)	0.355	0.277	0.384
	(0.478)	(0.448)	(0.486)
Years of Education	4.744	4.319	4.904
	(3.814)	(3.857)	(3.786)
Can the Respondent Read? (0-No, 1-Yes)	0.657	0.574	0.688
	(0.475)	(0.495)	(0.463)
Can the Respondent Write? (0-No, 1-Yes)	0.634	0.574	0.656
	(0.482)	(0.495)	(0.475)
Health of Respondent (5-Very Good...0-Very Bad)	3.064	2.979	3.096
	(0.815)	(0.838)	(0.804)
Lost > 2 Weeks Work to Illnes? (0-No, 1-Yes)	0.343	0.404	0.320
	(0.475)	(0.491)	(0.467)
Smokes? (0-No, 1-Yes)	0.233	0.170	0.256
	(0.423)	(0.376)	(0.437)
Household Size	5.087	5.021	5.112
	(1.898)	(2.170)	(1.786)
ln(Total Household Income)	12.85	13.06	12.77
	(3.373)	(3.255)	(3.414)
Is Arsenic Fatal? (0-No, 1-Yes)	0.913	0.936	0.904
	(0.282)	(0.245)	(0.295)
Know Someone with Arsenic Poisoning?(0-No, 1-Yes)	0.0988	0.0213	0.128
	(0.298)	(0.144)	(0.334)
Water Supply Test Positive >50ppb Arsenic?	0.640	0.489	0.696
	(0.480)	(0.500)	(0.460)
Subjective Arsenic Risk (/500)	121.1	109.6	125.4
	(193.8)	(186.7)	(196.2)
Observations	174	47	127

Note: Price lexicographic respondents identified by consistent choice of most expensive well.
HPLB refers to High Price Lexicographic Behaviour.
Mean of each variable with standard deviation in parentheses.

Figure 3.3.: Enumerators conducting the DCE

3.6. Results

Results from estimation of conditional logit and heteroscedastic conditional logit models are presented in Table 3.3. The first model shows that all attribute coefficients are statistically significant and have intuitive signs; higher risk, higher prices and worse taste/appearance all reduce utility.

Table 3.3.: Conditional and Heteroscedastic Conditional Logit Models - Full and Reduced Samples

	(1) FS	(2) FS	(3) RS	(4) RS	(5) RS	(6) RS
Variables						
Taste/Appearance of Water	-0.484***	-0.987***	-0.506***	-1.138***		
	(0.055)	(0.230)	(0.062)	(0.325)		
Lifetime Cancer Risk	-0.334***	-0.658***	-0.422***	-0.926***		
	(0.023)	(0.141)	(0.031)	(0.247)		
Price of Well	-0.039***	-0.064***	-0.081***	-0.176***	-0.095***	-0.221***
	(0.007)	(0.018)	(0.008)	(0.049)	(0.009)	(0.063)
Taste/Appearance (Good)					1.179***	2.871***
					(0.143)	(0.853)
Taste/Appearance (OK)					0.758***	1.719***
					(0.138)	(0.544)
Risk 5/500					2.100***	4.830***
					(0.252)	(1.410)
Risk 1/500					3.784***	8.730***
					(0.307)	(2.395)
Risk 0/500					4.831***	11.199***
					(0.363)	(3.061)
Scale						
High Arsenic in Tubewell		-0.433***		-0.322**		-0.328**
		(0.116)		(0.134)		(0.135)
Health of Respondent		-0.153**		-0.186**		-0.202**
		(0.065)		(0.079)		(0.079)
log-likelihood (ll)	-886.342	-876.452	-662.322	-656.877	-643.811	-637.950
N	3132	3132	2286	2286	2286	2286

Note: Standard errors in parentheses.

* p<0.10, ** p<0.05, *** p<0.01

FS - Full Sample

RS - Reduced Sample

Reduced Sample excludes households who serially select highest priced well

In Model 2 in Table 3.3 we present the results of a heteroscedastic conditional logit which allows us to examine the consistency of choices made by respondents. People with high arsenic concentrations in the tubewell that they drank from and respondents with higher health have a lower scale parameter. Given that the scale parameter is inversely related to the error variance[3] in the conditional logit model this indicates that people with high arsenic in their tubewell water and respondents with better health are less consistent in their choices. As well as allowing an examination of choice consistency the heteroscedastic model also provides a better fit as illustrated by the larger log-likelihood compared with model 1. Using the model parameters from Model 2, marginal willingness to pay (MWTP) for arsenic risk reduction is Riels 10,281[4] per month or roughly USD 2.57 per month[5].

Upon more detailed examination of the choice behaviour of the respondents, a group of respondents who consistently selected the highest price well in each of their 6 choice sets was identified. This group consists of 47 respondents or 27% of the overall sample. The breakdown of this behaviour by the arsenic test result is given in Table 3.4. We refer to those households who serially select the highest price well as exhibiting 'High Price Lexicographic Behaviour' (HPLB). One possible explanation for this behaviour is that the respondents were identifying higher prices as a proxy for higher quality. Anecdotal evidence from the experiment enumerators however suggests that there may have been some element of 'showing off' by a section of respondents by choosing the higher-priced wells. This is perhaps a costless display of affluence, whether that be true affluence or otherwise, in a hypothetical choice experiment. The enumerators were students from Phnom Penh and some of the rural respondents may have felt a benefit from expressing a preference for more expensive wells within the experiments to the city based students.

We explored the determinants of this behaviour through the estimation of a binary logit

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[3] variance= $\frac{\pi^2}{6\lambda_i^2}$

[4] MWTP is calculated as the ratio of the risk parameter to the price parameter

[5] Local exchange rate of Riels 4,000 = 1 USD used

Table 3.4.: Lexicographic Behaviour by Arsenic Test Result

	Low Arsenic	High Arsenic	Total
HPLB	24	23	47
Other Households	38	89	127
Total	62	112	174

High arsenic concentration defined as >50 micrograms per litre

model, the results of which are shown in Table 3.5. Results suggest that respondents who consistently chose the highest priced well have less education, are younger, are more likely to be women, have larger household sizes, have less income and are more likely to live in the Banteay Dek commune. Furthermore they are also more likely to have received a negative result for high arsenic in their tubewell. This suggests that those households who have not been faced with the decision to switch their water source may be less engaged with the arsenic issue and the choice experiment and thus may be more likely to use the experiment as a means of ‘showing off’.

The results of Model 2 shows that the rate of lexicographic behaviour fell during the data collection period and that there are significant interviewer effects, relative to the baseline. This perhaps suggests that the interviewers were becoming more successful at getting the respondents to engage with the choice experiment in a meaningful way rather than using the experiment as an opportunity for ’showing off’ through interacting with the interviewer. The interviewer dummy variable coefficients suggests that some interviewers were more successful than others.

Given the initial hypothesis, from anecdotal evidence, that it may have been the interaction with the interviewer that was causing this behaviour, we also tested and found evidence of significant interviewer effects; all but interviewer H have significant influence on this behaviour. It should be noted that the baseline for interviewers is a group of interviewers who did not experience any HPLB in their responses. Several earlier studies have found evidence of interviewer effects, [*Leggett et al.*, 2003; *Bateman and Mawby*, 2004; *Loureiro and Lotade*, 2005; *Snowball and Willis*, 2011]. However in our study areas, high levels of illiteracy and low levels of education precluded the use of

Table 3.5.: Logit Model: Lexicographic Price Behaviour

	(1)	(2)
Lexicographic Preferences - Higher Prices		
Water Supply Test Positive for High Arsenic (>50ppb)?	-0.995***	-0.928***
	0.087	0.112
Years of Education	-0.063***	-0.040***
	0.013	0.015
Gender of Respondent (0-Female, 1-Male)	-0.291***	-0.726***
	0.095	0.117
Age of Respondent	-0.021***	-0.027***
	0.003	0.004
Household Size	-0.104***	-0.062**
	0.024	0.027
Ln(Total Household Income)	0.036**	0.024
	0.014	0.018
Banteay Dek Commune	0.306**	2.621***
	0.137	0.232
Samrong Thom Commune	-0.099	-1.205***
	0.100	0.216
Interview Day		-0.404***
		0.053
Interviewer A		0.890***
		0.231
Interviewer B		2.545***
		0.221
Interviewer C		4.427***
		0.246
Interviewer D		1.652***
		0.238
Interviewer E		4.430***
		0.286
Interviewer F		2.061***
		0.204
Interviewer G		4.988***
		0.355
Interviewer H		0.275
		0.249
Interviewer I		3.260***
		0.303
Interviewer J		3.512***
		0.258
Constant	1.079***	1.792***
	0.320	0.527
ll	-1717.916	-1310.460
N	174	174

self-completed questionnaires which may have negated this issue.

In summary we have a high proportion of households who are serially selecting the highest priced well in each of the 6 choice sets and anecdotal evidence suggests that this may in part be attributed to some respondents attempting to appear affluent or 'show-off'. Such lexicographic behaviour precludes analysis of trade-offs between attributes. There exists a wide literature on dealing with protest votes, non-participation or lexicographic preferences in DCEs [see, for instance, *Lancsar and Louviere*, 2006; *Burton and Rigby*, 2009]. Given that the behaviour of these households precludes exploration of trade-offs and that this behaviour appears to be an artifact of the survey, we drop these respondents from the sample for the estimation of many of the subsequent choice models.

Models 3 and 4 in Table 3.3 are a repeat of Models 1 and 2 but on the now reduced sample. Again all parameter estimates are significant to the 5% level and all have intuitive signs. A MWTP of Riels 5,261 per month or USD 1.32 for arsenic risk reduction is calculated using the parameter estimates from Model 4. The MWTP is roughly half the value of that calculated using the full sample, which is expected due to the exclusion of households who select only the most expensive water sources.

In order to test for the appropriate ranking of risk levels we re-estimate the models including attribute levels for risk and taste/appearance as dummy variables, rather than a continuous variable. This is shown as Model 5 in Table 3.3. In this model, and Model 6, the risk level of 10/500 and the bad taste/appearance level are the baseline. All coefficients are statistically significant and show appropriate ranking as well as intuitive signs. Higher risk levels reduce utility as does 'worse' water taste/appearance levels. The hetereoscedastic model in attribute levels for risk and taste/appearance (model 6) shows the same pattern of choice consistency for those with high arsenic and good health and again provides a better model fit.

Although the conditional logit models presented in Table 3.3 show that the choice responses adhere to economic intuition and ex ante hypotheses, the models do not allow the analysis of choice heterogeneity. In addition conditional logit models assume independence from irrelevant alternatives (IIA) [see for example *Hensher et al.*, 2005] which is not appropriate in many choice situations. In order to address these issues we estimate latent class choice models.

Models of varying preference and scale class numbers were estimated, first on the full sample, the information criteria [6] for which are presented in Table 3.6. The statistics in this table suggests that the model with four preference classes and one scale class is the model of best fit. The parameters from this latent class choice model, along with MWTP estimates for arsenic risk reduction, are shown in Table 3.7.

Table 3.6.: Information Criteria for Alternative Class Specifications

Class Structure*	LL	BIC(LL)	AIC3(LL)	CAIC(LL)	NPar	df
1:1	-886.343	1788.163	1781.686	1791.163	3	171
1:2	-807.840	1656.952	1117.679	1664.952	8	166
1:3	-754.234	1575.536	1547.468	1588.536	13	161
1:4**	-733.809	1560.481	1521.618	1578.481	18	156
1:5	-732.918	1584.493	1534.835	1607.493	23	151
1:6	-724.237	1592.928	1532.473	1620.927	28	146
2:1	-832.240	1690.275	1679.480	1695.275	5	169
2:2	-773.877	1599.344	1577.754	1609.344	10	164
2:3	-747.788	1572.961	1540.575	1587.961	15	159
2:4	-731.602	1566.386	1523.205	1586.386	20	154
2:5	-728.013	1585.001	1531.025	1610.001	25	149
2:6	-725.824	1606.420	1541.648	1636.419	30	144

* a:b indicates 'a' scale classes and 'b' preference classes.

** Selected Model.

The variables were selected for the model on the basis of maximising parameter significance and model fit. Furthermore selection was based upon the a priori hypothesis that those households who have direct contact with arsenic, through a positive test for high arsenic in their water, might have a higher WTP for reduced arsenic water, than those households who have received a negative test.

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[6] The BIC, AIC3 and CAIC have the following form: $-2l + A_n p$, where l is the log-likelihood, p is the number of parameters and A_n is a penalty weight. For BIC $A_n = ln(n)$, for AIC3 $A_n = 3$ and for CAIC $A_n = ln(n + 1)$

Table 3.7.: Latent Class Choice Model - Full Sample

	Class 1	Class 2	Class 3	Class 4
Variables				
Taste/Appearance	-0.425***	0.002	-1.062***	-2.082***
	(0.141)	(0.193)	(0.171)	(0.526)
Risk	-0.203***	-0.629***	-0.209***	-5.072***
	(0.072)	(0.115)	(0.049)	(1.015)
Price	-0.091***	-0.067***	0.000	1.088***
	(0.023)	(0.025)	(0.013)	(0.231)
Class Membership				
Intercept	-1.749**	-0.400**	1.159***	0.990***
	(0.812)	(0.555)	(0.393)	(0.354)
Positive for High Arsenic (>50ppb)	1.679**	0.644**	-1.428***	-0.895**
	(0.377)	(0.560)	(0.417)	(0.407)
N	3132			
LL	-733.809			
BIC	1560.481			
AIC3	1521.618			
Posterior Class Membership				
Class Size(Percent)	15.66	23.73	28.97	31.64
High Arsenic HHs	25	30	21	36
Low Arsenic HHs	1	4	29	28

Note: Standard errors in parentheses.

* p<0.10, ** p<0.05, *** p<0.01

Class 1 in this model consists of individuals who have negative marginal utilities for higher risk, higher prices and worse water taste, consistent with ex ante hypotheses. Households who have received a positive test for high arsenic are more likely to be members of this class, perhaps indicating that those who have a high arsenic water source might be more engaged with a DCE regarding arsenic risk.

Class 2 consists of households who are not considering the taste/appearance attribute in their choice calculus, instead they are focused solely on the risk and price attributes. Households who have received a positive test for arsenic and have previously had to engage with the issue of water source selection.

Class 3 in this model has an insignificant coefficient on the price parameter. In this model respondents are either non-attending to the price attribute, or have negligible marginal utility, and are focusing on the risk and taste/appearance alone. This issue was initially prevalent in the piloting stages of the study. There are several reasons why respondents could be ignoring price. Firstly the range of prices could be too low for a proportion of the sample to consider them binding to their decision. An alternative hypothesis however could be that the respondents simply ignore the prices as they believe that they will never conceivably have to pay.

The most striking class is class 4 due to the positive coefficient on the price attribute. This suggests that respondents are more likely to choose a higher priced well which is surprising and perhaps not representative of true market behaviour. It might be argued that higher prices might proxy higher quality. However the respondents were told that these wells were identical in all aspects except the shown attributes in the choice cards. Given this result perhaps representing the previously identified HPLB behaviour, we re estimate the scale extended latent class choice models with these households dropped from the sample. The information criteria, shown in Table 3.8, suggests that now a 3 preference class and 2 scale class model best suits the new sample. The parameters from this model are presented in Table 3.9.

Table 3.8.: Information Criteria for Alternative Class Specifications

Class Structure*	ll	BIC(ll)	AIC3(ll)	CAIC(ll)	NPar	df
1:1	-662.322	1339.177	1333.645	1342.177	3	124
1:2	-608.755	1256.263	1241.509	1264.263	8	119
1:3	-588.792	1240.559	1216.585	1253.559	13	114
1:4	-571.780	1230.755	1197.560	1248.755	18	109
1:5	-566.700	1244.816	1202.399	1267.816	23	104
1:6	-564.879	1265.395	1213.757	1293.395	28	99
2:1	-633.232	1295.528	1284.463	1301.528	6	121
2:2	-593.979	1241.243	1220.957	1252.243	11	116
2:3**	-571.759	1221.025	1191.518	1237.025	16	111
2:4	-567.131	1235.991	1197.263	1256.991	21	106
2:5	-561.897	1249.742	1201.793	1275.742	26	101
2:6	-559.103	1268.375	1211.205	1299.375	31	96

* a:b indicates 'a' scale classes and 'b' preference classes.

** Selected Model.

The attribute parameter estimates, from the reduced sample model, are all statistically significant and have intuitive coefficient signs. Preference class 1, which consists of 53.96% of the sample, has the largest MWTP estimates. Older respondents are more likely to be a member of this class. Preference class 3 has the lowest MWTP estimate (Riels 2,307.69). Preference class 2 has a MWTP estimate of Riels 6,200.00 per month and having low arsenic increases the likelihood of membership of this class.

The scale classes show that there are two groups of individuals within the sample; those who are more consistent in their choices (scale class 2) and those who are less consistent (scale class 1). Age is a significant covariate of class membership and signifies that older individuals are less consistent in their choices. A cross-tabulation of the preference and scale class structures, shown in Table 3.10, reveals that roughly a third of the members of each preference class are members of the less consistent scale class, whilst roughly two thirds are members of the more consistent scale class.

Although these classes allow the estimation of MWTP valuees for arsenic reduction, as shown in Table 3.9, these WTP estimates are expected values for a 1 unit change in risk. Of more interest in addressing the research questions is the estimation of WTP

Table 3.9.: Latent Class Choice Model

	P Class 1	P Class 2	P Class 3
Variables			
Taste/Appearance	-0.162***	-0.125***	-0.047**
	(0.041)	(0.042)	(0.023)
Risk	-0.432***	-0.031***	-0.030**
	(0.096)	(0.011)	(0.012)
Price	-0.039***	-0.005**	-0.013***
	(0.010)	(0.002)	(0.005)
Preference Class Membership			
Intercept	1.512	1.582	-3.094
	(0.971)	(0.980)	(1.900)
Positive for High Arsenic (>50ppb)	-0.851	-2.189**	3.040
	(0.985)	(1.029)	(1.913)
Posterior Preference Class Membership			
Class Size(Percent)	53.96	26.55	19.49
High Arsenic Households	54	10	25
Low Arsenic Households	17	21	0
MWTP (Riels Per Month)	11,076.92	6,200.00	2,307.69
	Scale Class 1	Scale Class 2	
Scale Estimates			
Scale (lambda)	1.00	11.976***	
	(-)	(3.311)	
Scale Class Membership			
Intercept	-1.426***	1.426***	
	(0.543)	(0.543)	
Age of Respondent	0.218**	-0.218**	
	(0.010)	(0.010)	
Posterior Scale Class Membership			
Class Size(Percent)	34.84	65.16	
High Arsenic Households	29	60	
Low Arsenic Households	10	28	
N	2286		
log-likelihood (ll)	-571.759		
BIC(ll)	1221.025		
AIC3 (ll)	1191.518		

Note: Standard errors in parentheses.

* p<0.10, ** p<0.05, *** p<0.01

Table 3.10.: Breakdown of Preference and Scale Classes

	Preference Class 1	Preference Class 2	Preference Class 3	Total
Scale Class 1	23	7	9	39
Scale Class 2	48	18	22	88
Total	71	25	31	127

values for different levels of arsenic risk and taste/appearance from a baseline. In order to estimate these WTP values the latent class models are re-estimated in levels. The parameter estimates for this model are shown in Table 3.11.

The model estimated in attribute levels uses a risk of 10/500 and a bad taste/appearance as the baseline. As such all parameter estimates should be viewed as marginal utility changes from this baseline. Using the parameter estimates from this model we calculate successive WTP estimates for risk reductions from the 10/500 baseline to the displayed level of lifetime cancer risk. These estimates, converted into USD using a conversion rate of 1 USD = 4,000 Riels, are shown in Figure 3.4.[7]

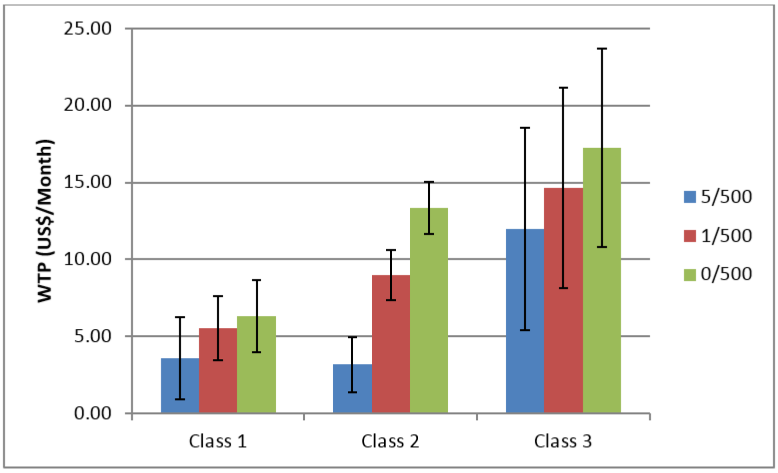

Figure 3.4.: WTP estimates, from 10/500 baseline, by preference class

The graph shows that the 3 preference classes have distinct WTP estimates, with class 3 members having the largest WTP for risk reductions of all levels and class 1 members having the lowest WTP for all risk reduction levels bar the initial reduction from

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[7] A full table of WTP estimates and accompanying choice models, from all baselines, is available from the author on request

Table 3.11.: Latent Class Choice Model in Attribute Levels

	P Class 1	P Class 2	P Class 3
Variables			
Taste/Appearance (Ok)	0.950**	0.541**	0.281**
	(0.406)	(0.233)	(0.118)
Taste/Appearance (Good)	1.337***	1.025***	0.332**
	(0.447)	(0.373)	(0.131)
Risk 5/500	8.107***	0.417*	0.550*
	(2.753)	(0.221)	(0.297)
Risk 1/500	9.919***	1.173**	0.851**
	(3.072)	(0.465)	(0.401)
Risk 0/500	11.682***	1.742***	0.970**
	(3.456)	(0.646)	(0.448)
Price	-0.169***	-0.033**	-0.038**
	(0.050)	(0.013)	(0.015)
Preference Class Membership			
Intercept	1.304	1.637*	-2.941
	(0.969)	(0.977)	(1.898)
Positive for High Arsenic (>50ppb)	-0.823	-2.110**	2.933
	(0.982)	(1.005)	(1.901)
Posterior Preference Class Membership			
Class Size(Percent)	47.51	30.81	21.67
High Arsenic Households	49	11	29
Low Arsenic Households	16	22	0
	Scale Class 1	Scale Class 2	
Scale Estimates			
Scale (lambda)	1.00	4.017***	
	(-)	(1.151)	
Scale Class Membership			
Intercept	-1.611**	1.611**	
	(0.785)	(0.785)	
Age of Respondent	0.028**	-0.028**	
	(0.014)	(0.014)	
Posterior Scale Class Membership			
Class Size(Percent)	39.57	60.43	
High Arsenic Households	27	62	
Low Arsenic Households	11	27	
N	2286		
log-likelihood (ll)	-552.957		
BIC (ll)	1227.019		
AIC3 (ll)	1180.915		

Note: Standard errors in parentheses.

* p<0.10, ** p<0.05, *** p<0.01

10/500 to 5/500. Heterogeneous risk preferences may lead to diverse responses to well testing and education programs. It should be noted however that testing and education programs are currently based upon the 50 μg/l definition of safe and unsafe. In this research we map concentration levels, which perhaps do not have great significance to people other than experts, to lifetime cancer risks. We estimate that each class has, on average, a significant WTP for arsenic concentration reductions from 50 μg/l to 10 μg/l (or arsenic risk reductions from 5/500 to 1/500).

The WTP for this change is displayed on the graph as the difference between the 1st and 2nd bars in each latent class grouping. This WTP change is largest for class 2 and is similar for classes 1 and 3. Our analysis thus indicates that a reduction in risk similar to a change in the drinking water standard from 50 μg/l to 10 μg/l would be most valued by members of class 2. The WTP differences for both of the higher levels differences (5/500 to 1/500 and 1/500 to 0/500) are largest for class 2 whilst the initial WTP change (between 10/500 and 5/500) is largest for class 3. Interestingly those households who have had a negative test for high arsenic (i.e. $< 50\mu$g/l) are statistically more likely to be members of class 2. Under a stricter testing regime these households might find that their water source becomes ‘unsafe’ whereas households who already have an ‘unsafe’ source would be unaffected by a lowering of the arsenic limit, unless they have switched to another tubewell source. It should be noted however that the respondents were purposefully not informed of the link between the arsenic risk levels in the DCE and the current drinking water standard. These households may however be able to judge that they do not have the most severe level of arsenic in their water, due to the negative high arsenic test result, which could explain the relatively high WTP for reductions in arsenic from medium to low levels compared with reductions from high to medium concentrations.

The different thresholds of risk reduction preference displayed in the results shows that each class is sensitive to the probability of cancer risk. This result differs from the finding of *Sunstein and Zeckhauser* [2011] who found an insensitivity to scope in the

WTP results for reduced concentrations of arsenic in drinking water for students in the US. The authors attribute this insensitivity to a level of fear, induced by a graphic description of arsenic risk, leading to ‘probability neglect’. In our study the households were provided with a graphic description and illustrations of arsenic risks during the NGO’s arsenic testing and education program. This occurred in advance of our study, in some cases several years prior to the DCE. Directly before the DCE was completed each household was provided with an unemotional description of arsenic risks to act as a reminder. The level of fear that might have been elicited by the initial description of arsenic may have subsided in the elapsed time. *Sunstein and Zeckhauser* [2011] note that fear cannot be experienced over a sustained period of time, which could explain our scope sensitive results for this potentially emotive and fearsome issue.

From a 2009 survey of Kandal households, *Horn* [2011] finds an average monthly cost of 40,580 Riels (USD 10.15) for those households purchasing water from a private supplier. Data from the RDIC survey suggests that households pay on average 7 USD per month for using a water vendor [*RDIC*, 2012]. Our own survey data suggests that those households purchasing from a water vendor spend on average USD 1.78 per week or USD 7.11 per month. There is however a high degree of heterogeneity in water costs owing to differences between water vendors, varying household sizes, diverse water use behaviours and different distances that the water vendors travel. These results however indicate that a range of 7-10 USD is approximately the average costs of using a water vendor as the main source of household drinking water, with some households reporting that they pay up 25 USD per month for using a water vendor. Given these costs it appears that members of preference class 1 from our latent class choice model estimates would not be willing to purchase water from a water vendor to mitigate their exposure to arsenic whereas members of the other two preference classes would be prepared to buy water from vendors. Compared with actual costs currently paid by households to have water delivered, where in many cases the water is pumped from the Mekong River and delivered untreated, our WTP results do not appear to be extreme in magnitude. The prudent WTP results, along with the coherent ranking of risk parameter magnitudes and degree of statistical significance of the choice model parameter estimates, suggest that despite i) the degree of difficulty conducting stated preference (SP) studies in a developing country settings [*Whittington*, 1998, 2002, 2004, 2010; *Bennett and Birol*, 2010] and ii) the difficulties in presenting risk information in SP studies [*Harrison et al.*, 2014], meaningful conclusions can be drawn from our results.

3.7. Conclusions

In this paper we have estimated the value of reduced arsenic concentrations in drinking water, using a discrete choice experiment (DCE), amongst the arsenic-informed rural poor in Kandal Province, Cambodia, a province whose rural population is heavily exposed to arsenic contaminated groundwater. The motivation for such a study was threefold. Firstly to examine the benefits of possible remediation work conducted by local and international NGOs. Secondly to estimate preferences for arsenic-free drinking water to help policy makers or health workers to construct drinking water standards or remediation strategies by indicating the priorities of those people who are at risk. Finally, previous stated and revealed preference studies have used the dichotomy of safe/unsafe, set by the relevant government, to evaluate household preferences for arsenic risk reductions. By conducting a detailed DCE we were able to address the more fundamental issue of risk preferences, tolerance levels and WTP.

The estimation of scale extended latent class models indicates that there are significant differences in choice consistency within the sample. Failing to account for these differences, i.e. by estimation of a homoscedastic latent class model that is typical within the literature is likely to provide biased preference estimates, since the scale term is confounded with those preference estimates. Controlling for these scale differences, we have identified 3 latent segments whose valuations of alternative thresholds of water contamination differ markedly. Each of the 3 segments exhibit scope sensitivity regarding the risk levels presented. However, there exists variation in the pattern of diminishing marginal utility regarding reduced arsenic concentrations and associated risks. A 4th segment displayed lexicographic preferences, evident from serial selection of the highest price option, which prevented the estimation of marginal risk-cost trade-offs. Anecdotal evidence suggests that this behaviour is an artifact of the experiment process rather than representing likely market behaviour.

Overall our results suggest that a lower permissible limit for arsenic in drinking water may better represent the preferences of households in Kandal province, Cambodia. A lower permissible limit of arsenic in drinking water would set the framework for NGO and government-led education and remediation programs. The lower limit would result in more households receiving a positive test for unsafe levels of arsenic which better matches the stated preferences of households for lower risk exposure. Due to the technologies of arsenic testing, a lower arsenic standard may require laboratory analysis of water samples to more accurately detect the lower level of arsenic. This would substantially increase the costs and logistical complexity of arsenic remediation strategies. We conclude, in agreement with *Smith and Smith* [2004], that substantial care must be taken when setting drinking water standards and that those households with the highest concentrations of arsenic should be targeted first by remediation and education strategies. Furthermore efforts should be made to prevent a substitution away from arsenic risk but towards sources with increased water-borne-pathogens [*Field et al.*, 2011]. Recent research suggests that end-user practices are more significant in determining end-use drinking water pathogen levels than the levels in water direct from the source [*Mondal et al.*, 2014] and that improved water sources and water treatment might be substitutes [*Jessoe*, 2013]. Thus further education related to best practice of water storage and treatment may be required in addition to arsenic testing and education programs.

We caveat our conclusions in that our results are based on household preferences in Kandal province. Although this province is the most heavily exposed to arsenic risk in

Cambodia, further research would be needed before a nationwide policy change could be recommended. Furthermore other important considerations such as cultural and technical attributes of water sources have been abstracted away from in this research to simplfy the complexity of the choice task for the respondents. These issues are, however, likely to form an important part of the decision and this should be considered by policy makers. Our results however suggest that Kandal households would be willing to pay to limit their exposure to arsenic, to concentrations lower than the current drinking water standard.

3.8. References

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4. Valuation in Developing Countries: Willingness to Work vs. Willingness to Pay

4.1. Abstract

A worry when conducting stated preference valuation studies in rural developing contexts is the use of monetary willingness to pay (WTP) estimates. In circumstances where cash incomes are low, a significant fraction of the population are not engaged in waged labour and the exchange of goods or services is concluded through barter or work exchange, the perception of money in the economy is likely to be different from that within an urban developed setting. As such ability to pay using money may be impaired, when compared with other mediums of exchange, and economic values using WTP measures may be downwardly biased. In recognition of this several studies have used hypothetical labour contributions as payment vehicles to measure economic value and a common finding is that households are more often willing to contribute labour than money. There has been however, little focus on the impact of payment vehicle on choice behaviour and protest votes. This is particularly true in relation to discrete choice experiments (DCE). In this paper we present the results of a split sample DCE using money and labour contributions as payment vehicles for improved drinking water quality in Kandal Province, Cambodia. This paper is a methodological examination of payment vehicles in rural developing countries using arsenic contamination as a case study. We find little differences between the payment vehicles in terms of the impact on choice behaviour and the derived welfare measures. Overall we present 3 novel findings: i) We find that the inferred respondents' value of time is similar to the market value (wage rates) of the labour tasks to be undertaken. ii) We find no differences in marginal utility estimates between respondents who were presented a monetary or labour payment vehicle. iii) Through the estimation of latent class models we further find that levels of attribute non-attendance (ANA) towards the payment vehicle is also stable between the 2 payment vehicles. We conclude that in the context of our study site and subject matter, the use of labour contributions, rather than money, to estimate welfare values offers little additional benefit and entails further difficulties regarding the analysis of the value of time. This provides support for the use of WTP in rural developing areas.

4.2. Introduction

The valuation of environmental stocks and services is increasingly undertaken in developing countries. Elicitation and estimation of willingness to pay (WTP) in such contexts poses many challenges ranging from lack of detailed data for revealed preference studies to language, cultural and logistical barriers for stated preference studies. The challenges are such that the validity of some stated preferences studies in developing countries have been called into question *Whittington* [2002].

One significant concern highlighted in the literature [see for instance *Hardner*, 1996; *Hung et al.*, 2007; *Casiwan-Launio et al.*, 2011] is the use of monetary WTP measures with the rural poor. Since cash incomes are extremely low, a significant proportion of the population are not engaged in waged labour and the exchange of goods or services is augmented through barter or work exchange, the role of money income in the rural developing setting is likely to be different from that of an urban developed setting.

It has been argued that, in these contexts, WTP measures, based on ability to pay using money, will not appropriately represent the true economic value of the policy/good/attribute under consideration, given that economic exchanges may occur without the use of money [see for instance *Hardner*, 1996; *Hung et al.*, 2007; *Casiwan-Launio et al.*, 2011]. Furthermore, since this downward bias would particularly affect the WTP estimates of the rural poor, those households who are perhaps most in need of assistance may be unduly discriminated against in wider project analysis that utilises WTP estimates. The problem may be further amplified through increased hypothetical biases. Given that rural poor households may not be as integrated with markets compared with households in developed countries, hypothetical contingent markets, which are key for stated preference surveys, may seem less realistic which could lead to unreliable results.

Fundamental to this issue is a lack of markets using money as the medium of exchange, the predominant reason why non-market valuation studies are undertaken in the first place. Complete labour markets would permit a household to liquidate current assets to allocate resources efficiently or to access credit for increased current consumption. In these circumstances WTP estimates based on payments using money would be an appropriate welfare measure. In rural developing settings barter and work exchange can facilitate exchanges in lieu of fully functioning monetary markets.[1]

If a household does not have the ability to utilise markets to access money, response to stated preference survey questions, which often specifically ask households to consider their money income before responding to stated preference questions, will be made in relation to these liquidity constraints and may thus provide a biased estimate of true economic value. It may therefore be pertinent to consider trade-offs with respect to other resources, rather than money or in addition to money.

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[1] However transaction costs are likely to be high in terms of matching buyers and sellers which may lead to inefficiencies.

Although researchers have used a variety of alternative payment vehicles in stated preference valuation studies to account for this issue, for instance *Shyamsundar and Kramer* [1996] use baskets of rice and *Asquith et al.* [2008] used in-kind payments such as beehives, the most common substitute for money in stated preference studies is labour contributions [*Hardner*, 1996; *Kamuanga et al.*, 2001; *O'Garra*, 2009; *Casiwan-Launio et al.*, 2011; *Rai and Scarborough*, 2013, 2014].

Labour is perhaps the most commonly used alternative to money due both to the ubiquitous nature of potential working hours, i.e. all households are faced with the decision of how best to allocate their time between productive and leisure activities, and also that labour time is arguably the most important asset at the disposal of the rural household.

The utilisation of household time for productive activities, however, may be limited by the available labour market or self-employment opportunities, potentially leading to an unsatisfied demand for work and constrained money income. Although utilising these available working hours, rather than constrained money income, as a medium of exchange in stated preference studies might ease the problem of downwardly biased WTP measures, using working time as a payment vehicle for assessing welfare effects is potentially complicated by the need to apply an opportunity cost of time to achieve monetary measures of welfare and the uncertain impact that the choice of payment vehicle may have on choice behaviour.

Further complications occur due to the fact that the nature of work task involved, for example the laboriousness of the task, will likely have an effect on the decision of the household or individual to accept the offer [*Ahlheim et al.*, 2010], thus different studies using different labour tasks would likely lead to different welfare valuations. and also that those individuals who are in full-time work may be constrained in their free time. A major concern with using labour time as a utility measure is that the value of time is highly dependent on the opportunity cost of that time. For instance during the crop-harvesting period for rural households, time is likely to be at a premium whereas in other periods time is likely to be relatively cheap. Unlike money, time cannot be saved during plentiful periods to smooth consumption. Further those individuals who are in full time work may be heavily time constrained but relatively well off in terms money income. Using either payment vehicle would lead to potential issues depending on the work and income levels of the individual respondent.

Given the potential issues related to using labour contributions as a payment vehicle in stated preference valuation studies, recent empirical literature has focused on several themes including: how do the size of welfare measures, derived using the different payment vehicles, compare and how might a welfare measure, expressed in labour hours, be monetised for use in cost-benefit analysis. What has not received much attention, however, is how the payment vehicle affects choice behaviour in SP tasks, in terms of protest votes, estimated marginal utilities and attribute non-attendance (ANA). It is these points that we address in this paper.

As noted by *Vondolia et al.* [2014] a common finding from CV studies which utilise the labour payment vehicle in rural areas of developing countries is that there is a higher rate of acceptance for the scenario, i.e. they provide a positive WTP. The use of a labour payment vehicle is conceivably seen as a more realistic scenario within these contexts and thus protest votes could be low and acceptance rates high.

In a choice experiment setting protests or low acceptance due to the payment vehicle could be channelled in several ways: i) the respondents could consistently choose the status-quo option, ii) ignoring the payment attribute, regardless of the level, when making their selection from the choice set (i.e. not attend the price attribute), iii) they could respond by using a heuristic such as choosing the lowest priced alternative, iv) they could choose randomly or v) they could refuse to participate in, or drop out of, the choice experiment.

In the choice experiment presented here there was not a status-quo option which rules out potential route (i). Additionally we encountered no households who refused to participate in the choice experiment or dropped out midway through (v). This leaves alternatives (ii)-(iv) as methods to protest the payment vehicle or display a lack of acceptance.

In this study we examine the appositeness and acceptability of the work and labour payment vehicles through examination of the marginal utility parameters in the estimated choice models to identify whether alternatives (ii) and (iii) were utilised and we estimate latent class choice models to examine the degree of attribute non-attendance towards the two payment vehicles (iv). This allows us to use the estimated models to comment on the degree of acceptance of the two payment vehicles.

In light of the increased use of labour contributions as a medium of exchange in stated preference studies and the potential deficiencies of money-based WTP measures in low income contexts, the research presented here examines the differences between rural households' willingness to contribute labour and households' willingness to contribute money for water source improvements in order to examine the validity of money-based WTP estimates in rural developing areas. This paper presents the results of a split sample discrete choice experiment in Kandal province, Cambodia, where householders have been informed of the results of geochemical surveys of arsenic content in their drinking water. Half of the respondents are asked about improved drinking water sources in relation to using money as a payment (MAP) with the other half asked in relation to using work as a payment (WAP) method. We investigate i) the implicit value of time and draw comparisons to local market wage rates, ii) attribute non-attendance stability between payment vehicles through estimation of latent class choice models, and iii) parameter stability between the samples through estimation of a pooled heteroscedastic conditional logit model.

The structure of the paper is as follows: part 4.3 presents a review of previous WTW studies; part 4.4 describes the arsenic problem on which the choice experiment focuses as well as the experimental design, sampling procedure and the data collection process; part 4.5 presents the empirical results and part 4.6 concludes with discussion.

4.3. Related literature

4.3.1. Contingent Valuation Studies

Early applications of utilising labour as a payment vehicle were used within contingent valuation studies estimating welfare measures for common goods in highly non-monetised locations.

One early application of labour as a payment vehicle in a stated preference study is *Hardner* [1996] who conducted a contingent valuation (CV) exercise to study the economic benefits of potable water provision in rural Ecuador. The study area was a rural, subsistence agriculture-based community where many economic exchanges were non-monetised. In recognition of this the study used working hours towards construction of water treatment systems as a payment vehicle instead of money. Of their sample, 72% provided a positive willingness to work value which suggests that the respondents were receptive to forms of exchange other than money. Due to the extreme lack of markets or property rights in the area questions related to money were not asked and so a direct comparison of WTW and WTP was not possible.

In later studies researchers were able to ask respondents dual contribution questions, related to both labour time and money. For example, a trio of studies [*Swallow and Woudyalew*, 1994; *Echessah et al.*, 1997; *Kamuanga et al.*, 2001] focusing on tsetse fly reduction in Africa implemented contingent valuation studies with both money and labour as numeraires, allowing comparison. *Swallow and Woudyalew* [1994], working in Ethiopia, conducted a CV study to examine the willingness to contribute labour and money towards theft prevention of tsetse fly reduction baits. The study found that more individuals were willing to contribute time rather than money although no attempt is made to compare the two measures directly.

The authors suggest that this might be due to high demands for cash and relatively constrained opportunities for generating income. The study finds that households headed by females and households headed by someone who works ‘off farm’ are willing to contribute less time than other households. They also find that households who have more cattle, as a proxy for wealth, are willing to contribute more of both labour and money.

Both *Echessah et al.* [1997] and *Kamuanga et al.* [2001] also find that a higher proportion of households were willing to contribute labour than money. Although the studies investigate the factors that influence willingness to contribute money and labour, a direct comparison in terms of economic value estimates provided by the two measures, through monetisation of labour contributions, is not made.

Hung et al. [2007] conduct a small scale CV study to examine willingness to supply labour or payments to develop firebreaks in order to limit forest fires in Vietnam. This study draws the same conclusion as the tsetse fly studies: that respondents were more willing to contribute labour than money. Only 7 out of 70 respondents were unwilling to contribute any time whereas 57 out of 70 respondents were unwilling to contribute money.

The contingent valuation studies discussed up to this point clearly illustrate the responsiveness of the respondents to labour contributions as a payment vehicle and that in these developing settings a proportion of households has been willing to give up time even when they have declined to contribute money. This suggests that valuation studies focused exclusively on WTP estimates using money as the medium of exchange would undervalue the economic benefits under investigation. These studies however do not make a direct comparison of WTP and WTW values through the monetisation of work time contributions, which would be required in order to utilise the benefit estimates for cost-benefit analysis.

A contingent valuation study which estimated monetary values of labour contributions is *O’Garra* [2009], who questioned respondents on their willingness to contribute time and money towards marine conservation in Fiji. The study estimates monetised welfare measures using two opportunity costs of time: a wage rate and a leisure rate. The leisure rate assumes that respondents would reallocate time currently being used for leisure, rather than work time, when allocating time in the contingent valuation question. As such, using a result accredited to *Cesario* [1976], one third of the wage rate was used as the opportunity cost of leisure time. This result was used to monetise labour contributions for comparison with direct WTP estimates.

The study finds that monetised WTW, using the leisure conversion rate, welfare estimates are not significantly different from direct WTP estimates using money. The use of the one third wage rate as the opportunity cost of time is a critical assumption for this result as using the full wage rate would clearly result in a higher monetised WTW estimate than the WTP estimate.

In researching the potential sustainability of marine protected areas in the Philippines, *Casiwan-Launio et al.* [2011] consider both money and labour hours as payment vehicles using a split-sample CV approach. They also utilise the assumption that the opportunity cost of time is one third the value of the wage rate. Wage data is thus used to monetise the WTW estimates for comparison with WTP and they find that WTW is between 3 and 8 times larger than WTP. They provide two potential explanations for this divergence.

Firstly that missing markets might decrease the opportunity cost of time. By not having access to a labour market with available demand to sell excess labour, or by having high transaction costs to access the market, households may have too much labour applied to self-employment, or leisure, than would otherwise be the case. This would lead to increased preferences for money, relative to labour, and could lead to the observed divergence between WTP and WTW. A second suggested hypothesis put forward by the authors is that the divergence is the result of an endowment effect. Commitment of money represents the fruits of previously applied labour whereas commitment of labour hours is the opportunity cost of potential future income derived from work not yet undertaken.

A further paper that makes use of the assumption that the opportunity cost of leisure is a third of the size of hourly wage rate is *Arbiol et al.* [2013]. This paper utilised labour contributions to examine the economic value of human leptospirosis prevention in urban Manila. They find that respondents are receptive to using labour time to reduce human leptospirosis, even when the jobs involve activities potentially seen as unpleasant such as environmental clean-up activities to reduce rat populations. The authors attribute this acceptance rate to high levels of unemployment. The study did not simultaneously collect WTP with respect to money and so a direct comparison was not made.

Tilahun et al. [2013], examining forest conservation in Ethiopia, discuss the potential issues of using wage rates to value the opportunity cost of labour contributions. Firstly they point to imperfect labour markets in rural settings. There may be more households willing to contribute labour than there are willing to employ labour. As such the slack labour market may reduce the opportunity cost of time relative to the households' preference to supply labour. The households will therefore be less likely to consider market wages as a foregone opportunity when contemplating labour contributions in the valuation scenario. They further note that as in some rural contexts the households will spend little time working in employed labour and the majority of the time self-employed, the opportunity cost of time in slow agricultural seasons, such as after crops have been harvested, is likely to be zero. As such the study utilises per capita daily income of the respondent rather than market wage rates to compare monetised WTW estimates with WTP estimates and finds a 99 percent overlap in the confidence intervals of the two estimates.

Seemingly many of the problems associated with using money as a payment vehicles in these contexts is associated with the experience that the individual will have had with using money as means of exchanging goods or services. If a household has little experience with money, the hypothetical gap and limited access to money may lead to biases. Investigating this issue*Vondolia et al.* [2014] examines the influence of experience with using either money or labour as a means of payment on protest votes and mean willingness to pay. The study finds that experience with the payment vehicles reduces asymmetries in acceptance rates. This suggests that respondent experience should be carefully considered when selecting a payment vehicle for a stated preference valuation study for the rural poor in a developing country.

These CV studies focus on the valuation of a public good, for a local area, such as firebreaks, tsetse fly traps or conservation. As such, discernible labour contributions are perhaps an obvious way to sustain these communal goods where concerns such as free riding might impact the effectiveness of less visible monetary contributions. There has been little focus however on willingness to contribute labour in order to obtain a private good, which is what we examine in this paper.

4.3.2. DCE Studies

Abramson et al. [2011] conducted one of the first DCEs which uses WTW as a payment vehicle, as part of an examination of financing alternatives for improved water services which also included a repayable loan. The financing option was included as an attribute in the choice experiment which enables the estimation of an internal opportunity cost of time, as the ratios of WTP to WTW. The study finds that WTP using money is low compared with potential project costs although WTP significantly increases when using loans or labour contributions. In addition to cost recovery for beneficial development programs, this is also an important result for valuation studies in general. In cash constrained economies WTP using money may return a relatively low economic value compared with other payment vehicles.

In a further DCE which addressed the mode of payment, *Rai and Scarborough* [2013] included both labour and monetary contributions as attributes in a study of invasive plant species in Nepal. Having both attributes for each option in the choice set enabled the examination of individual time-money trade-offs, or shadow wage rates. The shadow wage rate estimated was found to be different from the local wage rates. This shadow wage rate or the shadow value of labour is an alternative to other studies which use local wages either in there entirety or weighted.

In a related study *Rai and Scarborough* [2014] report the findings of a similar DCE, however, on this occasion a split sample approach was used where the respondent was permitted to choose their preferred payment option, either money or labour, for the DCE. Roughly 35% chose the money option whilst the remainder (65%) chose the labour option. In doing so the authors were able to examine the determinants of payment vehicle choice as well as comparing the WTP or implicit prices. Self-selection of the payment method by the respondent however diminishes the applicability of comparisons between welfare estimates from each payment vehicle, given that households who are relatively better educated and hold more land are observed to choose the monetary option more often. As such, in the study presented here we use a random assignment of respondents with half of the sample being assigned a choice task involving money as the payment vehicle and half being assigned a labour based choice task.

In summary the literature in this field, to date, addresses the concern that WTP using money potentially underestimates economic values. The central conclusion is that respondents are expressing a greater willingness to contribute labour than cash in rural developing environments and in areas with high levels of unemployment. This poses serious questions for the use of WTP with money as payment vehicle in these settings. The vast majority of literature is based on CV studies, thus, there is a gap in the current literature about how choice behaviour is affected by different payment vehicles in a DCE study. Further examination of the use of labour as an alternative payment vehicle is warranted within the DCE methodology given the relatively little attention paid to choice behaviour, choice consistency, protests or attribute non-attendance. In the research presented here we test for differences in marginal utilities, ANA and choice consistency, due to the choice of labour as a payment vehicle, by conducting a split sample DCE where respondents are randomly assigned to a DCE with either a labour or money payment vehicle.

4.3.3. Labour, Markets and Shadow Wage Rates

The literature on agricultural household models (AHMs) focuses on the role of markets in allowing separation of production and consumption decisions [*Singh et al.*, 1986; *Skoufias*, 1994; *Taylor and Adelman*, 2003] and elucidates the potential effects of markets on rural agricultural household labour demand and supply. In a situation of complete markets household's agricultural production decisions are separate from their labour supply decisions and appear only as a budget to fund consumption and leisure. In these circumstances and when households spend some time in market based work the wage rate that they achieve represents the opportunity cost of their time. In these circumstances WTP would provide an unbiased estimate of economic value.

In circumstances where labour markets are missing household members may be constrained to farm or household work. Labour applied to the farm is thus dependent on household preferences for leisure and the amount of leisure taken is dependent on the technical capabilities of the farm. The preference for money will thus rise relative to leisure (time) for those households with limited access to labour markets. This lack of markets is a potential factor leading to the finding of higher willingness to contribute labour than money.

4.4. Experimental Design and Data

The choice experiment conducted here focuses on rural households' willingness to contribute either money or labour towards improved drinking water quality. The study was conducted in Kandal province, Cambodia, where high levels of non-anthropogenic arsenic have been found in many household tubewells.

In Cambodia, many studies over the last 10 years have found high arsenic hazard in groundwater drinking water sources [*Polya et al.*, 2003, 2005; *Feldman et al.*, 2007; *Berg et al.*, 2007; *Buschmann et al.*, 2007; *Polya et al.*, 2008; *Sampson et al.*, 2008; *Kocar et al.*, 2008; *Quicksall et al.*, 2008; *Rowland et al.*, 2008; *Benner et al.*, 2008; *Sthiannopkao et al.*, 2008; *Polizzotto et al.*, 2008]. Human exposure has been demonstrated through studies of various biomarkers [*Kubota et al.*, 2006; *Gault et al.*, 2008] and cases of arsenicosis have been recorded by *Mazumder et al.* [2009], amongst others. One study estimates that over 100,000 people are exposed in Cambodia, with the majority of those living in Kandal province [*Sampson et al.*, 2008].

Chronic arsenic exposure can lead to a wide range of health consequences such as lung, bladder, liver and skin cancers, skin hyperpigmentation and keratosis [*NRC*, 1999, 2001; *IARC*, 2004]. Other health outcomes include increased risks of ischaemic heart disease and immune system disorders [*Polya et al.*, 2010, and references therein]. Many arsenic attributable health outcomes are not contemporaneous with exposure; where data are readily available, such as in Chile, childhood exposures in particular have been linked through detailed epidemiological studies, to peaks in arsenic attributable deaths occurring decades after the exposure [*Steinmaus et al.*, 2013]. Exposure to arsenic contaminated drinking water is thus a serious public health concern in Cambodia, both now and for the future [*Fredericks*, 2004]. Due to the long latency period of arsenic poisoning as well as the lack of information possessed by the rural villagers most likely to be effected, the full scope of arsenicosis (arsenic poisoning) is not fully known.

Arsenic is predominately found in deeper tubewell sources (where a narrow, deep hole is drilled into the ground and connected to a hand pump) rather than shallower dug wells. The distribution of arsenic in the groundwater of Kandal is highly heterogeneous [*Lado et al.*, 2008; *Winkel et al.*, 2008; *Sovann and Polya*, 2014] which, when combined with the fact that arsenic is tasteless, has led to some confusion amongst households concerning the safety of drinking water.

An extensive testing program conducted by the local organisation Resource Development International (RDI) has been operating a testing and education project in the region to provide households with information on the safety of their drinking water. In addition, the pumps of wells which have tested positive for high levels of arsenic (>50 μg/l) have been painted red so as to alert others to the dangers posed by drinking the water. The NGO also provides education for the household regarding the dangers of arsenic poisoning and on alternative drinking sources. Figure 4.1 shows a painted tubewell which has been found to contain high levels of arsenic as well as a photograph of a Cambodian suffering from arsenicosis, which is used as a teaching aid by the NGO to show to households during the education process.

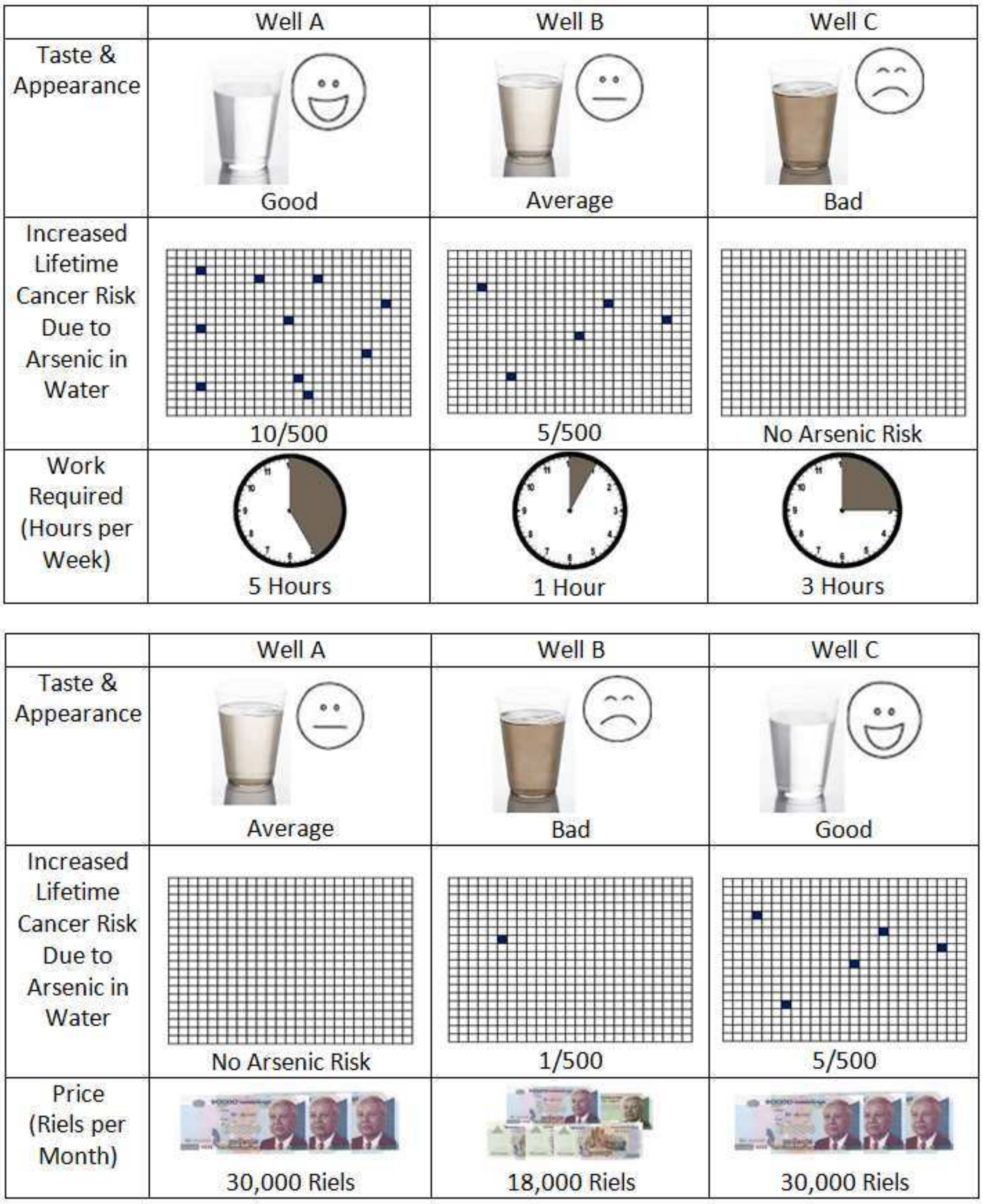
Figure 4.1.: Tubewell containing high levels of arsenic and Cambodian suffering from arsenicosis. (Photos a) Jonathan Gibson, b) RDI)

The sampling strategy for this study was to target households which had been tested for arsenic by RDI. Households in the sample therefore include those with water sources which have tested positive for high levels of arsenic (>50 μg/l) and have changed their water source, those whose drinking water has tested positive and have not changed, as well as those whose water has tested negative for high levels of arsenic.

The households sampled for the study were asked to imagine a hypothetical situation where their current drinking water source had become unusable. They were then presented with 6 choice cards each containing three different tubewells which were available to the household as a drinking water source. Each tubewell was described as being on communal land, with each source being approximately the same distance from the house. The water sources were defined in terms of the taste/appearance, the risk of arsenic from the source and the payment that they were required to make to access the source. Attribute levels are shown in Table 4.1 and examples of choice cards are illustrated in Figure 4.2.

Table 4.1.: Attributes, Levels and Coding

Attribute	Levels	Coding
Taste/Appearance	Dark with bad taste, Cloudy with OK taste, Clear with good taste.	2, 1, 0
Lifetime Cancer Risk Attributable to Arsenic in Water	10/500, 5/500, 1/500 0/500.	10, 5, 1, 0
A) Work Hours (Per Week) 171 Households	5 hours, 3 hours, 1 hours.	5, 3, 1
B) Money (Per Month) 174 Households	50,000 Riels, 30,000 Riels, 18,000 Riels, 6,000 Riels.	50, 30, 18, 6



	Well A	Well B	Well C
Taste & Appearance	Good	Average	Bad
Increased Lifetime Cancer Risk Due to Arsenic in Water	10/500	5/500	No Arsenic Risk
Work Required (Hours per Week)	5 Hours	1 Hour	3 Hours

	Well A	Well B	Well C
Taste & Appearance	Average	Bad	Good
Increased Lifetime Cancer Risk Due to Arsenic in Water	No Arsenic Risk	1/500	5/500
Price (Riels per Month)	30,000 Riels	18,000 Riels	30,000 Riels

Figure 4.2.: Example of Choice Card Shown to WAP and MAP Samples.

Given that the status quo was unavailable as part of the choice scenario, this was a 'forced-choice' situation for the respondent. A forced-choice scenario was chosen due to a lack of precise risk information related to the current water source that the household used. Forced choice scenarios have particularly been utilised in DCE studies which examine preferences for water, such as drinking water [*Hensher et al.*, 2005] or irrigation [*Rigby et al.*, 2010], as water is an essential resource which households can not go without. The primary focus of the DCE was to obtain WTP estimates for risk reductions concerning arsenic exposure and so the household's current water source was not included as a status quo due to the uncertainty that this would create in the choice set.

The piloting of the survey consisted of two sections. Firstly a period of formal household interviews was conducted in July 2012 to ascertain appropriate attributes and levels for the DCE and to trial survey formats. A key attribute of concern for households was the taste and appearnce of the water from the source. Thus, taste/appearance was chosen as an attribute in the choice task, along with arsenic risk and the cost. The second stage of piloting was conducted in early May 2013 and consisted of full trials of the survey design. An Bayesian Efficient experimental design was generated, using weakly informative priors, with the software Ngene [*Rose et al.*, 2009]. This enabled estimation of more informed priors, from the pilot, for the final experimental design.

171 households were presented with choice cards showing 3 wells with weekly work hours as the payment medium. The levels of this attribute were chosen to reflect time spent collecting drinking water for rural households in the dry season, using data from the Cambodian Socio-Economic Survey (CSES), 2009[2]. The work tasks were described as unskilled jobs that benefited the community such as levelling the roads, fetching drinking water or collecting firewood for households unable to do so. The tasks thus involved working towards improvements of both common goods and private goods. 174 households were presented with choice questions using money as the payment medium. Again the levels of this attribute were chosen using questionnaire data from CSES 2009 to reflect payments of rural households to water vendors.

Households were chosen at random from the database of RDI (Resource Development International), the NGO which runs the arsenic testing and education project. GPS co-ordinates were used to identify the households for interview. In order to achieve an efficient data collection period, households were selected from villages either side of Highway 1 in Kandal province, South East of Phnom Penh. The questionnaire was

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[2] See http://www.nis.gov.kh/nada/index.php/catalog/CSES

administered by students of Royal University of Phnom Penh and the Royal University of Agriculture in Cambodia over a 10-day period and the average interview lasted approximately 45 minutes. The questionnaire was composed of the following sections:

- Household composition, demographics and health questions.
- Arsenic knowledge and result of water test questions.
- Water source usage and characteristics of water sources used (smell, taste, price etc) questions.
- Choice experiment.
- Household Income and agricultural production questions.

Descriptive statistics for the households interviewed are shown in Table 4.2, broken down by which payment vehicle they were presented with in the choice task.

4.5. Empirical Analysis and Results

4.5.1. Willingness to Pay and Willingness to Work

The theoretical framework for welfare measurement is the compensating varaition measure. The compensating variation measure of welfare can be illustrated for the WTP and WTW approaches using the indirect utility function, as shown by *Vondolia et al.* [2014] and *Eom and Larson* [2006]. In the below equation, the maximum WTP for an improved water source (q_1) is the amount of money which equates the indirect utility function with the utility the household receives at the same prices with the original water source (q_0). Z represents socio-economic characteristics which influence utility.

$$V(y - WTP, \mathbf{p}, q_1; Z) = V(y, \mathbf{p}, q_0; Z) \tag{4.1}$$

Table 4.2.: Summary of Respondents by Payment Vehicle

	MAP	WAP	Total
Age of Respondent	48.86	46.68	47.78
	(14.81)	(15.45)	(15.17)
Gender of Respondent (0-Female, 1-Male)	0.355	0.331	0.343
	(0.478)	(0.471)	(0.475)
Years of Education	4.744	5.148	4.944
	(3.814)	(3.351)	(3.597)
Can the Respondent Read? (0-No, 1-Yes)	0.657	0.787	0.721
	(0.475)	(0.410)	(0.448)
Can the Respondent Write? (0-No, 1-Yes)	0.634	0.769	0.701
	(0.482)	(0.421)	(0.458)
Health of Respondent (5-Very Good, 3-Average, 0-Very Bad)	3.064	2.982	3.023
	(0.815)	(0.846)	(0.832)
Lost More Than 2 Weeks Work Due to Illnes? (0-No, 1-Yes)	0.343	0.314	0.328
	(0.475)	(0.464)	(0.470)
Smokes? (0-No, 1-Yes)	0.233	0.178	0.205
	(0.423)	(0.382)	(0.404)
Household Size	5.087	5.077	5.082
	(1.898)	(1.771)	(1.836)
Ln(Total Household Income)	12.85	13.11	12.98
	(3.373)	(3.226)	(3.303)
Is Arsenic Fatal? (0-No, 1-Yes)	0.913	0.923	0.918
	(0.282)	(0.267)	(0.275)
Know Someone with Arsenic Poisoning?(0-No, 1-Yes)	0.0988	0.0947	0.0968
	(0.298)	(0.293)	(0.296)
Water Supply Test Positive for High Arsenic (>50ppb)?	0.640	0.769	0.704
	(0.480)	(0.421)	(0.457)
Subjective Arsenic Risk (/500)	121.1	132.0	126.5
	(193.8)	(184.9)	(189.5)
Observations	174	171	345

Mean of each variable with standard deviation in parentheses.

Maximum WTW, on the other hand, is the amount of time a household would contribute to receive the improved source that equates utility with utility from the original water source without the labour payment. In the two equations y and l represent the full income of the households. i.e. the value of the time endowment and the household income. Similarly the prices (p) represent 'full prices'. In the WTP statement the full income is presented in money units whereas in the WTW statement full income is represented in labour units. See *Eom and Larson* [2006] for further discussion and details.

$$V(l - WTW, \mathbf{p}, q_1; Z) = V(l, \mathbf{p}, q_0; Z) \tag{4.2}$$

Analysis of respondent choices is based on Random Utility Theory (RUT). We begin the analysis of choice with the following attributes-based utility specifications for the MAP and WAP samples.

$$U_{nj} = \beta_1 Taste_{nj} + \beta_2 Risk_{nj} + \beta_3 Money_{nj} + \varepsilon_i \tag{4.3}$$

$$U_{nj} = \beta_1 Taste_{nj} + \beta_2 Risk_{nj} + \beta_3 Work_{nj} + \varepsilon_i \tag{4.4}$$

Here the utility gained from option j by person n is a function of the three attributes which define the wells in addition to an error term. The choice between alternatives is therefore probabilistic and the option leading to the highest utility is selected by the respondent. This is illustrated in (4.5). These utility functions, being linear and additively separable can be used, along with the assumption that the error terms follow a type I extreme value distribution, to specify the conditional logit model used in estimation (4.6).

$$P_{ni} = P(\varepsilon_{nj} - \varepsilon_{ni} < X_{ni}\beta - X_{nj}\beta) \forall j \neq i \in J \tag{4.5}$$

$$P_{ni} = \frac{\exp(\mu X_{ni}\beta)}{\sum_{j=1}^{J} \exp(\mu X_{nj}\beta)} \tag{4.6}$$

The conditional logit model shown in equation (4.6) assumes constant error variance across all individuals in the sample where the scale term is inversely proportional to the error term ($\mu = \frac{\pi}{\sqrt{6\sigma_\varepsilon^2}}$). An alternative model allows for heteroscedasticity, i.e. unequal error variance between individuals [*Hensher et al.*, 1999; *DeShazo and Fermo*, 2002]. This is illustrated in (4.7) with the scale term (μ) now varies between individuals (μ_n).

$$P_{ni} = \frac{\exp(\mu_n X_{ni}\beta)}{\sum_{j=1}^{J} \exp(\mu_n X_{nj}\beta)} \tag{4.7}$$

The scale term is parameterised with individual level variables (Z).

$$\mu_n = \exp(Z_n\gamma) \tag{4.8}$$

There are now β and γ parameters to be estimated via maximising the log likelihood function ($LL = \sum_{n=1}^{N} \sum_{j=1}^{J} y_{nj} \ln P_{ni}$ where y_{nj} is an indicator variable which is a 1 if option j is chosen or zero otherwise). Estimation of the heteroscedastic conditional logit model allows the analysis of choice consistency between different groups of individuals, where an equal consistency suggests homoscedasticity.

The parameter estimates for the heteroscedastic conditional logit models for the WAP and MAP samples are presented in Table 4.3. All attribute coefficients (the upper portion of the table) are statistically significant at the 99% level and have signs consistent with a priori hypotheses. Improved levels of taste/appearance, lower risk and lower payment, either in terms of money or labour hours, lead to increased probability of well selection. The variables included in the scale term, presence of arsenic, the ability of the respondent to read and the health of the respondent, were selected as proxies for the closeness that the respondent will likely have with the issue in the case of health and arsenic test results and as a proxy for ability to comprehend the choice task in the case of ability to read. The parameter estimates indicate that for the MAP sample, the scale term is lower (and the error variance higher) for those households who have been told that they high levels of arsenic ($>$50 μg/l) in their water source and if the respondent has good health. This indicates a lower choice consistency relative to households who have not received a positive test for high arsenic and where the respondent has lower health levels.

With the WAP sample the scale parameters indicate that health has the same influence on choice consistency, better health is associated with lower consistency, if the respondent is able to read and if high levels of arsenic are found in the household's water supply, are associated with higher levels of choice consistency.

In order to check the models for a coherent ranking of parameter magnitudes, the models are re-estimated using the dummy variables for the levels of the attributes. The results of these are presented in Table 4.4. The parameter estimates for the choice attributes are all statistically significant and have an appropriate ranking. Wald tests reject parameter equivalence which indicates that the choice behaviour of both samples are sensitive to the different arsenic risk levels.

The parameter estimates from the choice models can be used to estimate marginal

Table 4.3.: Heteroscedastic Conditional Logit Models

	(1)	(2)
Taste/Appearance	-0.961***	-0.535***
	(0.281)	(0.139)
Risk	-0.642***	-0.318***
	(0.173)	(0.079)
Payment(money)	-0.062***	
	(0.021)	
Payment (work)		-0.164***
		(0.043)
Scale		
Arsenic in water	-0.434***	0.236*
	(0.116)	(0.140)
Read	0.019	0.325**
	(0.123)	(0.149)
Health	-0.149**	-0.130**
	(0.071)	(0.064)
ll	-876.441	-883.259
n	3132	3078

Note: Standard errors in parentheses.

* p<0.10, ** p<0.05, *** p<0.01

Model (1) estimated using MAP sample

Model (2) estimated using WAP sample

Table 4.4.: Heteroscedastic Conditional Logit Models (Levels)

	(3)	(4)
Taste/Appearance (OK)	-0.727***	-0.491***
	(0.262)	(0.160)
Taste/Appearance (Bad)	-2.520***	-1.377***
	(0.677)	(0.348)
Risk 1/500	-2.258***	-1.075***
	(0.576)	(0.276)
Risk 5/500	-5.331***	-2.010***
	(1.320)	(0.476)
Risk 10/500	-7.927***	-3.808***
	(1.958)	(0.904)
Payment (money)	-0.113***	
	(0.031)	
Payment (work)		-0.204***
		(0.051)
Scale		
Arsenic in water	-0.421***	0.084
	(0.107)	(0.129)
Read	0.040	0.317**
	(0.111)	(0.139)
Health	-0.157**	-0.113*
	(0.065)	(0.061)
ll	-827.094	-851.793
N	3132	3078

Note: Standard errors in parentheses.

* p<0.10, ** p<0.05, *** p<0.01

Model (3) estimated using MAP sample

Model (4) estimated using WAP sample

willingness to pay (MWTP) values for risk reduction or taste/appearance improvement either in terms of money or work hours for the relevant samples. For instance, the WTP for marginal risk reduction with money as numeraire is:

$$MWTP_{risk} = -\beta_{risk}/\beta_{money} \tag{4.9}$$

Utilising the Krinsky-Robb simulation method to estimate upper and lower levels we estimate MWTP shown in the lower portion of Table 4.5. The results indicate that, using the linear utility assumption, mean WTP for a 1/500 reduction in lifetime cancer risk is 10,298 riels per month or using the local USD exchange rate (1 USD = 4,000 Riels) about USD 2.57. For a 'unit' increase in taste/appearance the WTP is 15,419 riels per month or USD 3.85 per month. The MWTP estimates for the WAP sample are presented in the upper portion of Table 4.5. The MWTP for risk reduction in terms of labour is 1.94 hours a week, or 7.76 hours a month.

Table 4.5.: MWTP Estimates and 95% Confidence Intervals (95%) for MAP and WAP Samples

	Appearance	**Risk**
MWTW (hrs/week)	-3.264	-1.940
95% CI Lower Level (Work)	-4.515	-2.516
95% CI Upper Level (Work)	-2.375	-1.563
MWTP (riels/month)	-15.419	-10.298
95% CI Lower Level (Money)	-26.743	-17.609
95% CI Upper Level (Money)	-10.344	-7.579

4.5.2. Testing the equivalence of WTP and WTW

In order to compare the two payment vehicles, we must consider how the respondents are valuing their time when responding to the choice task. There are however several alternative scenarios to consider in placing a monetary value on time. One potential method of valuing the labour contributions of the respondents is to consider the market wages for the actual tasks, within the survey, required to be undertaken for access to the water. These tasks were described to the respondents as unskilled manual tasks which included helping to collect firewood, transport water and levelling roads.

Although the markets for these tasks may not be complete, meaning that actual external employment opportunities for the household members in these activities may be limited, the households may have an idea of the going rate for these activities or at least the minimum wage that they would require to perform these tasks.

To examine whether the household might be comparing their willingness to work with the actual monetary remuneration that could be expected from these tasks, wage data from the Cambodian Socio-Economic survey (2009) is utilised. As part of this survey the heads of villages were asked questions related to village wages. Presented in Table 4.6 are summary statistics for the mean per day wages for four relatively unskilled manual tasks, provided by the heads of rural villages in Kandal province.

Table 4.6.: Daily Wage Rates (in Riels) Kandal Province (CSES, 2009)

Task	Males	Females
Transplanting of paddy	12,375.02	12,394.23
Caring for Crops	12,074.07	11,000.00
Harvesting	12,915.09	13,179.25
Unskilled Construction	13,348.84	8,364.29

Using 12,500 as an approximate daily wage for the activities described to the respondents we can estimate an hourly market wage rate for the labour tasks. In order to do this we require an approximation of daily working hours. Data from the sample suggests that between 8 and 10 hours is common. Using these hours per day, hourly market wages are as in Table 4.7.

Table 4.7.: Monetised WTW Estimates Using a Riels 12,500 Daily Wage and Different Daily Working Hour Assumptions

Hours Per Day	Hourly Pay (Riels)	Monetised WTW (Per Month)
10	1,250	9,700.00
8	1,563	12,128.88

In order to further test the payment vehicles for any inherent utility differences which may lead to different marginal utilities, the data was pooled using the wage data to monetise the working hour attributes in the data set. By pooling the data we are able to estimate the parameters from utility function in the equation below:

$$\begin{aligned} U_{nj} = \beta_1 Taste_{nj} + \beta_2 Risk_{nj} + \beta_3 Payment_{nj} + \beta_4 Taste_{nj}.WAP_i \\ +\beta_5 Risk_{nj}.WAP_i + \beta_6 Payment_{nj}.WAP_i + \varepsilon_i \end{aligned} \tag{4.10}$$

The WAP variable is a dummy variable representing if the payment vehicle used for the respondent was work hours. Individual and joint insignificance of these interaction variables would indicate no differences in utility due to the payment vehicle and that the two samples can be pooled. The final attributes and levels for the pooled data set is shown in Table 4.8.

Table 4.8.: Attributes, Levels and Coding (Pooled Samples)

Attribute	**Label**	**Levels**	**Coding**
Taste and Appearance	Taste	Dark with bad taste, Cloudy with OK taste, Clear with good taste.	2, 1, 0
Lifetime Cancer Risk Attributable to Arsenic in Water	Risk	10/500, 5/500, 1/500 0/500.	10, 5, 1, 0
Payment (Money or Monetised Work)	Payment	50,000 Riels, 30,000 Riels, 18,000 Riels, 6,000 Riels, 5 Hours =25,000 Riels, 3 Hours = 15,000 Riels, 1 Hour = 5,000 Riels	50, 30, 18, 6, 25, 15, 5
Work as Payment	WAP	Work**,** Money	1, 0

By estimating the pooled model we can test that the choice of payment vehicle has an effect on the utility that a person derives from the attributes. We test the null hypothesis that there is no effect against the two-sided alternative hypothesis that the choice of payment vehicle does have a utility impact.

$$H_0 : \beta_4 = \beta_5 = \beta_6 = 0 \tag{4.11}$$

$$H_A : \beta_4, \beta_5, \beta_6 \neq 0 \tag{4.12}$$

In addition to the estimation and testing of the interaction variables we also test for differences in scale (i.e. heteroscedasticity) due to the payment vehicle. In order to test for any payment vehicle effects a parametrised heteroscedastic conditional logit model is estimated. The models are presented in Table 4.9 as models 5 and 6.

All the interaction variables and scale terms are insignificant indicating that the choice of payment vehicle does not lead to any direct differences in marginal utility or choice consistency. The two payment vehicles are equivalent in this regard. The equivalence result from these tests is perhaps not surprising given the between sample exchange rate, or shadow wage rate:

$$\frac{WTP}{WTW} = \frac{10,298 Riels/Month}{7.76 hrs/Month} = 1,327.06 Riels/hr \tag{4.13}$$

The exchange rate between the samples, 1,327.06 Riels/Hour, is extremely similar to the market wage rate for the labour tasks that we used for monetising the work time attribute. The shadow wage rate that the respondents seem to be utilising is

Table 4.9.: Pooled Models

	(5)	(6)
Taste/Appearance	-0.484***	-0.526***
	(0.055)	(0.045)
Risk	-0.334***	-0.329***
	(0.023)	(0.023)
Payment	-0.039***	-0.038***
	(0.007)	(0.006)
WAP.Payment	0.006	0.006
	(0.008)	(0.006)
WAP.Taste	-0.090	
	(0.076)	
WAP.Risk	0.003	
	(0.030)	
Scale		
WAP		0.014
		(0.088)
ll	-1776.228	-1777.187
N	6210	6210

Note: Standard errors in parentheses.
* p<0.10, ** p<0.05, *** p<0.01

highly similar to the market value of the labour tasks in question, in addition to the consistencies in choice behaviour examplified by the parameter estimate parities from the pooled model.

An alternative method for monetising the WTW estimates is to use the approach taken by *O'Garra* [2009] and *Arbiol et al.* [2013] which is based on work by *Cesario* [1976]. In these studies the authors use one third of the working wage rates as the opportunity cost of time as they argue that households will be substituting current leisure time for the required work hours, rather than time already spent in work. This low wage rate would lead to a monetised WTW estimate substantially lower than the money based WTP estimate. The assumption that households would be substituting leisure time rather than productive time such as agricultural or household work time is doubtful in situations of full markets.

In full markets the literature on agricultural household models suggests that household members will consume leisure up to the point where the monetised marginal utility of leisure is equal to the market wage rate. The opportunity cost of time is therefore the full market wage rate, rather than some fraction. In situations where there are missing labour markets the literature suggests that households will consumer leisure up to the point where the monetised marginal utility of leisure is equal to the marginal revenue product of labour on the farm or the off-farm wage rate they can receive. The shadow wage rate in this situation is therefore dependent on the specific technology of the farm.

4.5.3. Attribute Non-Attendance

The use of different mediums of exchange in the choice experiment raises the question of how the payment vehicle used in the DCE might impact choice behaviour. Given the arguments about the inappropriatness of money as a payment vehicle in rural developing contexts, in relation to labour contributions, one way in particular that the change in payment attribute may affect choice behaviour is through protest votes or non-engagement with the choice tasks. As we discussed previously, given the set up of this DCE with no status quo option (i) and the high response rates and zero drop outs (v), non-engagement or protests, due to the payment vehicle, could be channelled in 3 ways: ii) ignoring the payment attribute, regardless of the level, when making their selection from the choice set (i.e. not attend the price attribute), iii) they could respond by using a heuristic such as choosing the lowest priced alternative, iv) they could choose randomly. The significant and coherent parameter estimates from the previous section suggest that protests or non-engagement was not channelled through (ii) or (iii). We now test whether attribute non-attendance (ANA) towards the payment vehicle is different between the labour and work choice experiments.

In this section we present the results of latent class choice models and examine ANA differences or similarities between the MAP and WAP samples. Latent class models further allows the extension of the analysis to incorporate preference heterogeneity.

In the latent class model the choice probabilities and the marginal utilities are conditional on being in Class c . The latent class choice model is thus:

$$P_{int|c} = \frac{\exp(\lambda\beta_c X_{in})}{\sum_{j=1}^{J}\exp(\lambda\beta_c X_{in})} \tag{4.14}$$

Class membership is modeled using a multinomial logit form with a vector of individual respondent characteristics (Z) as explanatory variables

$$P_{nc} = \frac{\exp(Z_n\phi)}{\sum_{c'=1}^{C}\exp(Z_n\phi)} \tag{4.15}$$

Identification is achieved by imposing that $\sum\phi = 0$.

The fit statistics[3] for latent class choice models with varying preference class numbers are presented in Table 3.10.

Table 4.10.: Information Criteria for Alternative Class Specifications

Number of Classes	LL	BIC(LL)	AIC3(LL)	CAIC(LL)	NPar	df
MAP						
1	-886.343	1788.163	1781.686	1791.163	3	171
2	-807.840	1656.952	1117.679	1664.952	8	166
3	-754.234	1575.536	1547.468	1588.536	13	161
4*	-733.809	1560.481	1521.618	1578.481	18	156
5	-732.918	1584.493	1534.835	1607.493	23	151
6	-724.237	1592.928	1532.473	1620.927	28	146
WAP						
1	-889.887	1795.198	1788.773	1798.198	3	168
2	-789.878	1620.889	1603.756	1628.889	8	163
3	-757.953	1582.747	1554.905	1595.747	13	158
4*	-743.073	1578.696	1540.146	1596.696	18	153
5	-735.189	1588.636	1539.378	1611.636	23	148
6	-724.413	1592.793	1532.826	1620.793	28	143

* Selected Models.

The first similarity between the two sub-samples is that the information criteria, used to aid model selection for non-nested models, suggests that there are 4 underlying preference classes in both the money and labour sub samples. The parameters from the 4 preference class choice models are presented in Table 4.11.[4]

Focusing first on the MAP sample, the Class 1 attribute parameter estimates have expected signs on quality and risk, however the cost term has a positive sign which indicates that respondenst who are members of this class have a positive marginal utility for higher prices on wells.[5] Households who have received a positive test for high levels of arsenic in their water source are less likely than those who have negative results to be members of this class.

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[3] The BIC, AIC3 and CAIC have the following form: $-2l + A_n p$, where l is the log-likelihood, p is the number of parameters and A_n is a penalty weight. For BIC $A_n = ln(n)$, for AIC3 $A_n = 3$ and for CAIC $A_n = ln(n+1)$

[4] The latent class model for the MAP sub-sample shown in Table 4.11 is a repeat of the model shown in Table 3.7, with preference class number ordering inverted.

[5] See Chapter 2 for a detailed analysis of this issue

Class 2, which accounts for 28.97% of the sample has an insignificant parameter estimate for the money variable indicating that these respondents were ignoring, or non-attending, this attribute or may have negligible utility from this attribute [see for example *Scarpa et al.*, 2009; *Lagarde*, 2013]. Those respondents' whose households have received a negative arsenic test are more likely to be a member of Class 2 than those with a positive result.

Class 3 has an insignificant parameter estimate on quality which indicates that members of this class are either ignoring this attribute, or gain negligible utility from it, and that choice is mainly based on risk and cost. Respondents with poor health are significantly more likely to be a member of Class 3.

Class 4 attribute parameter estimates are all significant and have expected signs. It accounts for 15.66% of the sample and household who have a tubewell source which has tested positive for arsenic are more likely to be memebrs of this class.

Turning now to the WAP sample household, health and water source characteristics were hypothesised to influence class membership. However the only significant covariate was agricultural income (see Table 4.11).

Class 1 has significant main attribute parameters, with expected signs. Class 3 has a significant parameter estimate for water quality only. Class 4 has low parameter significance for all attributes and a positive parameter estimate for the cost (work). Class 2 in this model represents 27.76% of the sample and has an insignificant parameter value on the payment vehicle (work). This suggests that the members of this class are non-attending or have low preference for the payment vehicle and are basing their choices on risk and quality. Moreover this class mirrors Class 2 from the money sub sample in both having very similar percentages of the respondents' where the payment vehicle parameter estimate was insignificnat whilst all other attribute parameters are significant and intuitively signed.

One potential empirical issue is a confounding of tastes/preferene and non-attendance.

Table 4.11.: Latent Class Choice Models - A) MAP and B) WAP

MAP	Class 1	Class 2	Class 3	Class 4
Taste/Appearance	-2.082***	-1.062***	0.002	-0.425***
	(0.526)	(0.171)	(0.193)	(0.141)
Risk	-5.072***	-0.209***	-0.629***	-0.203***
	(1.015)	(0.049)	(0.115)	(0.072)
Price	1.088***	0.000	-0.067***	-0.091***
	(0.231)	(0.013)	(0.025)	(0.023)
Class Membership				
Intercept	0.990***	1.159***	-0.400	-1.749**
	(0.354)	(0.393)	(0.555)	(0.812)
Positive for High Arsenic (>50ppb)	-0.895**	-1.428***	0.644	1.679**
	(0.407)	(0.417)	(0.560)	(0.377)
LL	-733.809			
BIC	1560.481			
AIC3	1521.618			
Class Size(Percent)	31.64	28.97	23.73	15.66

WAP	Class 1	Class 2	Class 3	Class 4
Taste/Appearance	-0.551***	-1.798***	-1.891***	-7.802*
	(0.0712)	(0.558)	(0.4831)	(4.509)
Risk	-0.3184***	-4.638***	-0.044	-1.226*
	(0.028)	(0.934)	(0.097)	(0.707)
Work	-0.306***	-0.124	-0.220	2.076*
	(0.029)	(0.126)	(0.160)	(1.116)
Class Membership				
Intercept	0.818***	0.246	-0.274	-0.791**
	(0.227)	(0.249)	(0.305)	(0.359)
Ln Agricultural Income	0.071**	0.051*	-0.083	-0.040
	(0.028)	(0.029)	(0.057)	(0.051)
LL	-743.073			
BIC	1578.696			
AIC3	1540.146			
Class Size(Percent)	59.80	27.76	7.3	5.15

Note: Standard errors in parentheses.

* p<0.10, ** p<0.05, *** p<0.01

This potential confounding may lead to a misinterpretation of the insignificant paramater estimate as non-attendance when it could be related to low or insignificant preferences. A commonly used method in the literature is to impose a parameter restriction to account for non-attendance of an attribute in the latent class model [see for instance *Burton and Rigby*, 2009; *Scarpa et al.*, 2009; *Campbell et al.*, 2011; *Lagarde*, 2013]. To investigate this issue further we re-estimate the latent class models of Table 4.11 with a restricted parameter for the money and work attributes in the Class 2s in the 2 models set equal to zero. The results from these models are shown in Table 4.12. A likelihood ratio test reveals that the restricted Models in Table 4.12, with the marginal utility of the payment vehicle in a single class for both MAP and WAP samples set equal to zero, are preferred[6].

The prefered models of Table 4.12 show very little differences from the unrestricted models of Table 4.11 in terms of parameter significance, parameter signs, fit statistics and class sizes. The key result, that ANA towards the payment vehicle is highly similar between the two DCE versions, is retained in the restricted model. This suggests that the choice of payment vehicle, money or work, does not impact the rate of non-attendance towards the payment vehicle for members of preference Class 2. A stable proportion of the sample does not consider the payment attribute, regardless of the medium of exchange. This result indicates that ignoring the payment vehicle as a form of protest or disengagement is independent of the choice of payment vehicle.

4.6. Discussion and Conclusions

The motivation for this research is the potential downward biad of WTP estimates that use money as a payment vehicle in rural developing economies where ability to pay is limited by a lack of markets, high transaction costs, low incomes and transactions

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[6] LR test of the hypothesis that the parameter in class 2 is equal to zero. LRχ^2=1.158, df =1, p-value=0.2818

Table 4.12.: Constrained Latent Class Choice Models - A) MAP and B) WAP

MAP	Class 1	Class 2	Class 3	Class 4
Taste/Appearance	-2.083***	-1.063***	0.003	-0.425***
	(0.526)	(0.169)	(0.191)	(0.140)
Risk	-5.072***	-0.210***	-0.629***	-0.203***
	(1.015)	(0.039)	(0.115)	(0.072)
Price	1.088***	0.000	-0.067***	-0.091***
	(0.231)	(-)	(0.025)	(0.023)
Class Membership				
Intercept	0.992***	1.162***	-0.402	-1.752**
	(0.346)	(0.380)	(0.553)	(0.805)
Positive for High Arsenic (>50ppb)	-0.897**	-1.429***	0.645	1.681**
	(0.399)	(0.415)	(0.559)	(0.800)
LL	-733.807			
BIC	1555.319			
AIC3	1518.615			
Class Size(Percent)	31.64	29.00	23.71	15.65

WAP	Class 1	Class 2	Class 3	Class 4
Taste/Appearance	-0.549***	-1.596***	-1.887**	-7.525*
	(0.071)	(0.460)	(0.483)	(4.486)
Risk	-0.319***	-4.529***	-0.044	-1.186**
	(0.029)	(0.873)	(0.097)	(0.701)
Work	-0.307***	0.000	-0.221	2.003**
	(0.030)	(-)	(0.160)	(1.117)
Class Membership				
Intercept	0.818***	0.246	-0.272	-0.790**
	(0.226)	(0.249)	(0.304)	(0.359)
Ln Agricultural Income	0.070**	0.050*	-0.083	-0.038
	(0.028)	(0.029)	(0.057)	(0.052)
LL	-743.652			
BIC	1574.712			
AIC3	1538.304			
Class Size(Percent)	59.84	27.61	7.33	5.23

Note: Standard errors in parentheses.

* p<0.10, ** p<0.05, *** p<0.01

which occur through barter or work exchange. In light of this researchers have begun to explore using labour contributions as a payment vehicle for valuing environmental goods or services. However little attention has been given to choice behaviour differences owing to changes in the units of the payment attribute. In this paper we have presented the results of a randomly assigned split-sample discrete choice experiment using both money and labour contributions as payment vehicles.

We find three novel results. Firstly the internal opportunity cost of time is found to be very similar to the market wage rates used in the local area for similar labour tasks. The agricultural household literature indicates that in situations of functioning labour markets the households' shadow wage rate will equal the market wage rate. Our result thus provides evidence of a functioning labour market.

Our second novel finding is that non-attendance of the payment vehicle is consistent between the monetary and labour payment vehicles. Roughly 28-29% of each sample ignores the payment attribute whilst considering the other two attributes. This result indicates that the choice of payment vehicle has less of an impact on valuation estimates than if the payment vehicle itself was leading to non-attendance or low engagement.

Our third novel finding is that we find no significant marginal utility differences or differences in the consistency of choice due to the payment vehicle. The payment vehicle itself is not leading to significantly different levels of noise and choice inconsistency.

The results of these tests indicate that there is little difference between the money or work payment vehicles for estimating welfare values in our study. This provides support for the use of WTP using money in rural developing contexts with functioning labour markets. It suggests that respondents have a valuation of their time similar to the market and the choice of payment vehicle itself does not impact the estimated models in terms of choice probabilities or choice consistency. Given the additional complications and uncertainties interpreting the monetary costs of time in order to monetise labour contributions, WTP would seem the most straightforward economic measure in this circumstance.

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5. Arsenic Testing and Household Drinking Water: The Determinants of Water Source Switching Behaviour in Kandal Province, Cambodia.

5.1. Abstract

Arsenic contamination of drinking water is a serious public health issue in many areas of South and South East Asia. Arsenic contamination of water is undetectable by taste or appearance and the illnesses associated with arsenic poisoning have long latency periods. Water tests are available to detect the presence or absence of arsenic but householders need a degree of trust in the testing process if they are to contemplate mitigating strategies. In this paper we present the results of a household survey in Kandal Province, Cambodia, investigating the impact of a large-scale arsenic water testing program. We position our analysis within the "averting behaviours" theoretical framework. We find that 86% of households switching away from arsenic contaminated sources indicating a high degree of trust in the testing program and a strong preference

for arsenic free water. We find that aesthetic qualities such as the taste of the original water source have an important role in determining the switching behaviour, with those households with unpleasant tasting water being more likely to switch given a positive test result for high arsenic for their original source. This suggests that in order to reduce exposure to arsenic risk, mitigation policies should focus on providing alternative sources which are both arsenic free and have improved aesthetic qualities, in addition to testing and education programs. Further we examine the new water sources chosen by the households who switch and the household's decision to treat drinking water before consumption, given concerns that some households might inadvertently trade arsenic risk for increased risk of diarrheal illnesses. We find that water purchased from a vendor, who collects the water from local rivers, is the most common alternative source and we further find no significant changes in water treatment behaviour, for those who switch, which raises questions of risk substitution, given the increased trend towards the use of surface water. We find that on average costs roughly double for households who switch sources; however in doing so there is a drop in the time they spend collecting water.

5.2. Introduction and Research Goals

Consumption of unsafe drinking water is a serious public health issue in many developing countries. As noted by *Field et al.* [2011] the primary contaminants of concern are water-borne pathogens leading to diarrheal disease, which is a leading factor contributing to high rates of child mortality. In parts of East and South East Asia however, contamination of drinking water by arsenic is a particular concern with the consumption of arsenic in Bangladesh being labelled the "the largest poisoning of a population in history" [*Smith et al.*, 2000], with around 20% of all-causes-mortality in arsenic impacted groundwater regions of Bangladesh attributed to arsenic exposure [*Argos et al.*, 2010].

The seriousness of this public health emergency has also been measured in economic welfare terms. In Bangladesh for instance, *Maddison et al.* [2005], estimated the willingness to pay (WTP) to avoid the potential arsenic-related health impacts was \$2.7 billion annually.

In Cambodia and many affected countries the predominant mitigation strategy is to implement a water source testing and education program. Well testing programs typically test for arsenic concentration levels higher than the current drinking water standard, which in most developing countries is 50 μg/L[1]. Tubewells that have water with arsenic concentrations in excess of this level are labelled unsafe and the household is advised to find an alternative source for their drinking water.

Much attention has focused on the determinants of successful information and education campaigns in Bangladesh, in terms of promoting water source switching behaviour [*Madajewicz et al.*, 2007; *Opar et al.*, 2007; *Bennear et al.*, 2013]. Relatively little work however has been conducted in Cambodia, where cultural and water source availability differences may impact the effectiveness of water testing and education programs relative to Bangladesh. In this paper we examine the impacts of one such program in Kandal Province, Cambodia and analyse the determinants of water source switching behaviour. The program in question is on-going and is conducted by Resources Development International (RDI) Cambodia. We focus on households which have received testing and education within three communes in Kandal Province.

Although these mitigation strategies, aimed at promoting water source switching, are intended to prevent arsenic-related illnesses there has been some concern that switching away from arsenic contaminated groundwater sources could in fact be detrimental to overall health outcomes [*MacDonald*, 2001; *Lokuge et al.*, 2004; *Field et al.*, 2011], if the new sources increase the incidence of diarrhea disease. One of the factors which has led to such high degrees of exposure to arsenic risk has been the promotion of groundwater

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[1] The WHO provisional guideline is 10 μg/L as a maximum arsenic contamination level, which is the standard employed by many developed countries.

sources by governmental and non-governmental organisations. This strategy was driven by the goal of encouraging people to move away from surface water sources which were seen as high risk in terms of diarrhoeal diseases.

Field et al. [2011] find that, in Bangladesh, arsenic education and testing programs may have led to a doubling of child and infant mortality from diarrhoeal diseases emanating from consumption of water from new sources. The hypothesised reasoning behind this increase is twofold. Firstly surface water sources, which many households switch to using, are often heavily contaminated with pathogens which can lead to diarrhea diseases. Secondly tubewells are often located in close proximity to a family's residence. Water can thus be accessed 'on demand' within a short period of time. This reduces the need for water to be collected in batches and stored for long periods of time whereupon contamination can occur.

Lokuge et al. [2004], also evaluating the impact of mitigation strategies in Bangladesh, finds that increased burden from water-borne infectious diseases could potentially outweigh the benefits of switching away from arsenic contaminated groundwater sources. Mitigation strategies for low arsenic risk sources could thus be deleterious to overall household health. [2]

Other studies however have found that improvements to hygiene practices and sanitation are more critical for reducing rates of diarrheal disease than improved water sources [*Esrey*, 1996; *Clasen and Cairncross*, 2004]. This suggests that water source switching behaviour might not be the key driver of infant mortality. *Mondal et al.* [2014], studying water quality in West Bengal, finds that the microbiological quality of end-use water is only weakly dependent on the microbiological quality of the water direct from the source.

This suggests that household practices are critical in determining the final (at con-

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[2] *Mondal et al.* [2014] question the assumptions made by the *Lokuge et al.* [2004] study. They suggest that a key result used in the analysis, that substitution of water supplies could result in 20% increases in diarrheal disease incidence, is originally from as study by *Esrey* [1996] where the result in question is not statistically distinguishable from zero.

sumption) microbiological quality of the water, and thus risks of contracting diarrheal illnesses. Programs which target hygiene and sanitation along with arsenic avoidance are likely to have larger public health improvements than focusing solely on the microbiological quality of household water sources or the arsenic concentrations within the water.

Although this potential risk substitution, away from arsenic risk but towards diarrheal risks, is a contested issue but it is a subject of great importance for public health improvements. An estimated 1.8 million people die every year from diarrhoeal diseases with an estimated 90% of these deaths being children under 5 and mostly in developing countries (WHO, 2004)[3]. What is clear however is that the type of source that the household switches to is of great importance due to the microbiological quality of that water but also characteristics of the source which necessitate transportation and long storage times of the water prior to consumption. Furthermore household behaviours related to treatment, to reduce risks of water-borne diseases, are also vital in achieving public health improvements.

A potential issue to consider in relation to the arsenic mitigation strategies and diarrheal risks is household behavioural change. In rural India, *Jessoe* [2013] found that the introduction of improved water sources led to a decrease in household expenditure on water treatments such as boiling. Improved water sources were seen to be a substitute for the households own mitigating activities as the improvement to source quality meant that household time and money spent on treatment could be reduced whilst keeping risk levels at or below the original levels. Thus much of expected gains from new sources are lost through behavioural changes. Although the example of water source switching due to arsenic is different to households using an improved source, behavioural changes related to treatment is still an important issue related to the arsenic mitigation strategy and overall health improvements.

In order to address these issues, in this paper we document the new sources which

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[3] http://www.who.int/water_sanitation_health/diseases/burden/en/

are used by households switching away from an arsenic contaminated source. There are many water source options available to Kandal Province households which vary in quality and risks involved and the water use pattern is complex. In addition we examine the household's treatment decisions across the various water source types.

The households surveyed for this research had all received the same information regarding risks of arsenic poisoning and had all been informed whether their water source contains 'high' ($> 50\mu$g/L) levels of arsenic. However, the subjective perception of the risks involved from drinking water contaminated with arsenic are likely to be different than the objective scientific risks [*Slovic*, 1987]. As such these perceptions matter greatly in relation to the subsequent actions that the households make. The arsenic risk perceptions that the household have are likely to be partly determined by unobserved variables such as previous health events and levels of optimism. These unobserved variables may also, in part, help to determine the households decision to treat their water and to change their water source. As such we examine this issue through the estimation of seemingly unrelated bivariate probit models [*Nauges and Van Den Berg*, 2009; *Onjala et al.*, 2013].

The final issue that this paper addresses is predominantly an economic one. A household, in switching from a source that they have been told is unsafe due to arsenic contamination to an alternative source, will necessarily have to incur some form of cost. This cost may be a monetary cost if, for example, they choose to purchase water from a vendor, it may be a time and effort cost if they need to travel to collect water from a shop or from a surface water source or the household may have to accept a water source which they believe offers a lower quality of water in terms of aesthetic qualities of water such as the smell, taste and appearance of the water. Even if these trade-offs are not encountered the household will still have to change their daily water habits which itself may prove an inconvenience.

All of these factors are trade-offs made to access arsenic free water and thus express the risk preferences of the households towards arsenic. Under certain conditions these averting expenditures form a lower bound on theoretically consistent welfare measures. In this paper we examine these trade-offs in order to comment on the preferences of households for arsenic reduction in the averting expenditures framework.

The paper thus addresses a number of inter-related research questions, using primary data from a household survey conducted in Kandal Province, Cambodia. These research questions are thus:

1. What proportion of households with high arsenic have changed to a new water source? (RQ1)
2. What determines the household's decision to switch away from the arsenic contaminated tubewell source? (RQ2)
3. What are the new water sources used by those households who switch their source? (RQ3)
4. What determines the type of new water source chosen? (RQ4)
5. What are the characteristics of the new water sources and what do they imply about the trade-offs that households have made to access arsenic free water in terms of money, time and the aesthetics of water? (RQ5)

The structure of this paper is: part 5.3 discusses the arsenic problem in Cambodia; part 5.4 discusses the related literature on arsenic mitigation strategies and averting expenditures; part 5.5 provides further detail on the RDI testing and education program; part 5.6 presents the sampling strategy and data collection; part 5.7 presents the econometric switching model along with the analysis of water treatment behaviour and new water sources and part 5.8 concludes with discussion of the findings.

5.3. The state of arsenic risks in Cambodia

Consumption of arsenic contaminated groundwater in many countries is a serious public health concern. The contamination of groundwater by arsenic in Bangladesh alone has been labelled “the largest poisoning of a population in history” [*Smith et al.*, 2000] with around 20% of all-causes-mortality in arsenic impacted groundwater regions of Bangladesh attributed to arsenic exposure [*Argos et al.*, 2010]. However the hazard is found across the world [*Smedley and Kinniburgh*, 2002] and it has been estimated that globally nearly 50 million people have drunk arsenic contaminated water above 50 micrograms per liter (μg/l) and well over 100 million with water with high concentrations of geogenic (i.e. non-anthropogenic) arsenic (defined here as >10 μg/l, the current (2014) provisional WHO guideline) [*Ravenscroft et al.*, 2009]. In addition to Bangladesh, and in roughly decreasing order of peak exposure (ibid.), other countries with high groundwater arsenic hazard include India, China, USA, Myanmar, Pakistan, Argentina, Vietnam, Mexico and Cambodia.

In Cambodia, many studies over the last 10 years have found high arsenic hazard in groundwater drinking water sources [*Polya et al.*, 2003, 2005; *Feldman et al.*, 2007; *Berg et al.*, 2007; *Buschmann et al.*, 2007; *Polya et al.*, 2008; *Sampson et al.*, 2008; *Kocar et al.*, 2008; *Quicksall et al.*, 2008; *Rowland et al.*, 2008; *Benner et al.*, 2008; *Sthiannopkao et al.*, 2008; *Polizzotto et al.*, 2008]. A consideration of the geological/geographical factors controlling the development of high geogenic arsenic groundwater systems [*Charlet and Polya*, 2006] and more detailed geostatistical modelling [*Lado et al.*, 2008; *Winkel et al.*, 2008; *Sovann and Polya*, 2014] indicates that these systems are to be found in many of the more flat-lying provinces of Cambodia, particularly in areas near the Mekong River. Kandal Province, immediately south of the capital Phnom Penh (see Fig. 2) is the province most significantly impacted as a result of the coincidence of high groundwater arsenic, high population density and a high dependence on groundwater for drinking water supplies [*Sovann and Polya*, 2014].

Human exposure has been demonstrated through studies of various biomarkers [*Kubota et al.*, 2006; *Gault et al.*, 2008] and cases of arsenicosis have been recorded by [*Mazumder et al.*, 2009], amongst others. One study estimates that over 100,000 people are exposed in Cambodia, with the majority of those living in Kandal Province [*Sampson et al.*, 2008].

Chronic arsenic exposure can lead to a wide range of health consequences such as lung, bladder, liver and skin cancers, skin hyperpigmentation and keratosis [*NRC*, 1999, 2001; *IARC*, 2004]. Other health outcomes include increased risks of ischaemic heart disease and immune system disorders [*Polya et al.*, 2010, and references therein]. Many arsenic attributable health outcomes are not contemporaneous with exposure; where data are readily available, such as in Chile, childhood exposures in particular have been linked through detailed epidemiological studies to peaks in arsenic attributable deaths occurring decades after the exposure [*Steinmaus et al.*, 2013]. Exposure to arsenic contaminated drinking water is thus a serious public health concern in Cambodia, both now and for the future [*Fredericks*, 2004].

5.4. The Economics of Water Source Choice, The Averting Expenditures framework and WTP.

5.4.1. Arsenic and Water Source Switching

The role of arsenic risk awareness and arsenic information campaigns has been examined in a number of water source switching studies in Bangladesh. In Matlab, Bangladesh, *Aziz et al.* [2006] found that arsenic information campaigns had little impact on source switching behaviour whereas the convenience of available alternative sources and household health had a significant impact. Conversely, *Aftab et al.* [2006], also focusing on source switching behaviours in Bangladesh, found that awareness of arsenic health consequences has a significant and positive impact on the likelihood of utilising arsenic safe water sources. They also find that higher incomes are associated with a greater likelihood of finding arsenic-free water sources.

Madajewicz et al. [2007] found that after communicating arsenic test results to households, 60% of the households who find that their well water is contaminated with unsafe levels of arsenic (> 50 μg/l) switch their water source within 6-12 months. Controlling for other factors, learning that their drinking water is contaminated with arsenic increases the probability of the household changing their drinking water source by 37%. The study also found that for those households who change their source, the average time that they spent collecting water increases 15 fold. This indicates that switching water sources is not costless but that a proportion of households are willing to use their time to reduce their arsenic exposure. In a subsequent and related study *Opar et al.* [2007], examining interventions aimed at reducing arsenic exposure, found that 65% of households with 'unsafe' levels of arsenic (> 50 μg/l) switch their drinking water source.

The way in which information is conveyed to households is potentially of prime importance in achieving a successful and cost effective outcome, though the relationship between information, risk perceptions and averting behaviours is complex. *Bennear et al.* [2013] examines two information messages given to households after well-testing and labelling in Bangladesh. One group received a message that explained that the national drinking water standard was 50 μg/l whereas an alternative group received a more detailed message that made it clear that lower arsenic wells were safer, as well as information related to the arsenic standard. This included a message that amongst wells labelled unsafe, a well that had an arsenic concentration of 100 μg/l would be better than a well with a 200 μg/l concentration. A parallel statement was also given for wells labelled safe. The water sources were then signed as safe/unsafe and the actual arsenic concentration levels were displayed on the well. In investigating the impact of the two messages on well switching behaviour however the study was unable to find a difference between rates of source switching between the two samples. This suggests that households were focusing on the dichotomous safe/unsafe element of the message rather than the extra detail. The information campaign that we study provided a binary message of safe vs. unsafe.

5.4.2. The Averting Expenditures theoretical framework

Access to clean, uncontaminated water is, in many cases within Kandal Province, a nonmarket good with households frequently harvesting rainwater, collecting surface water or travelling increased distances to access arsenic free water. These activities do not involve direct ongoing market transactions. As such this complicates the estimation of household preferences for uncontaminated water, which is required to formerly assess potential welfare value of competing mitigation policies.

In many studies, inferences about WTP for improved environmental quality and reduced health risks are derived from an individual's choice of mitigating or protective actions. The basis of this framework, often labelled the averting or defensive expenditures approach, comes from the household production theory of consumer behaviour [*Becker*, 1965; *Lancaster*, 1966]. This framework of consumer behaviour posits that individuals do not gain utility directly from goods, but instead gain utility from the combination of the attributes of those goods. This framework thus implies that observed behaviour is a combination of both individual preferences as well as household production technologies.

A simple example of the averting behaviour approach, given in *Bockstael and McConnell* [2007], is when a household/individual gains utility from a pure market good z_2 and a good such as health z_1 which itself depends upon an environmental public good (such as arsenic contamination) q and market purchased mitigation technologies such as filtration units x. If the individual faces prices p and r for the market good and the market bought mitigating devices, respectively, and has a budget y, the consumers utility maximisation problem is:

$$\max_{z_2,x}\{u(z_1,z_2)|y \geq pz_2 + rx, z_1 = f(x,q)\} \tag{5.1}$$

A more intuitive representation of this issue is given by the expenditure function, from the cost minimising approach. Utilising the expenditure function (m), i.e. the minimum expenditure required to reach a specified level of utility, the compensating variation (CV) welfare measure for a change in environmental quality (arsenic) can be shown to be:

$$CV = m(p,r,q^0,u^0) - m(p,r,q^1,u^0) \tag{5.2}$$

Here the superscripts relate to the levels of utility and environmental quality before the change (0) and after the change (1). In order to approximate this welfare measure many researchers have used changes to expenditures made to limit exposure to the environmental 'bad'. We will briefly outline in this section some of the relevant literature related to this issue and the empirical difficulties involved in estimating the difference between these two expenditure functions.

One of the initial studies which attempted to set out theoretical framework for using averting expenditures to estimate WTP for environmental goods is *Courant and Porter* [1981]. Using the notation of *Bockstael and McConnell* [2007] a defensive expenditure function, which is the minimum expenditure required to maintain the level of z_1 given the level of an environmental bad, b, is:

$$c(z_1,\mathbf{r},b) = \min_x\{\mathbf{rx}|z_1 = g(\mathbf{x},b)\} \tag{5.3}$$

The expenditure function required to reach a specified level of the broader consumer objective of utility is thus:

$$m(p_2, \mathbf{r}, b, u) = \min_{z_1, z_2} \{p_2 z_2 + c(z_1, \mathbf{r}, b) | u(z_1, z_2) \geq u^0\} \tag{5.4}$$

That is the minimum amount of money spent on purchases of the market commodity z_2 and on maintaining the composite commodity[4], c, at a required level, in order to reach a level of utility u^0. Given this function and the notion that compensating variation is the difference between the expenditure functions for 2 levels of b, a marginal WTP can be derived as:

$$-m_b(b, u) = -c_b(z_1, b) \tag{5.5}$$

That is the marginal CV is equal to the change in defensive expenditure function given a marginal change in b. Thus it was believed that the technologies of averting expenditures could be estimated in order to provide a welfare measure for changes to environmental quality. No actual preferences would need to be estimated. As noted by *Bockstael and McConnell* [2007] however this applies only for marginal changes in pollutants and under very specific conditions for non-marginal changes. It is non marginal changes however which are the most common case in the literature and indeed that is the case which we examine in this paper.

The theoretical case of non-marginal changes, such as receiving information that a household water source is contaminated with arsenic, was examined by *Bartik* [1988]. In this paper Bartik showed that defensive expenditures are a theoretical bound on

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[4] Where the composite commodity $z_1 = g(\mathbf{x}, b)$ is a commodity produced through inputs of x and the level of environmental quality.

the compensating variation welfare measures for non-marginal pollution changes. This condition required 2 assumptions. Firstly that defensive behaviours do not in themselves provide direct utility[5] and secondly that changes to environmental quality can be completely mitigated by private expenditures.

Defensive expenditures in the Bartik framework however are not the same as the actual defensive expenditures made by households which are observable and form the basis of most averting expenditure studies. In reality the individual will end up with a different level of environmental quality, rather than maintain it at its original level, after an exogenous change in water quality and a subsequent change in behaviours.

Actual defensive behaviours respond to changes in environmental quality in line with consumer preferences and the change in the marginal cost of defensive expenditures brought about by the change in environmental quality (b). Defensive or averting expenditures on the other hand do not involve preferences and are a purely technological relationship between exogeneous environmental quality and the costs required to maintain consumption of the composite good at the initial level.

In this study we focus on the water source characteristic trade-offs that the household has accepted in order to mitigate their exposure to arsenic. As such these expenditures and trade-offs form a component of the households' WTP for reduced arsenic exposure. However clearly these expenditures/trade-offs are not a precise measure of WTP but instead provide some indication of preferences in light of the arsenic testing and education program. We assume here that the changes in expenditures would form a lower bound on the true welfare measure [*Harrington and Portney*, 1987; *Abdalla et al.*, 1992], provided that aspects such as joint production do not play a significant role in water source choice decisions. For an account of the theoretical aspects of averting expenditures see *Bockstael and McConnell* [2007] or *Dickie* [2003].

Alternative revealed preference models of the demand for environmental quality of-

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[5] When defensive/averting expenditures directly impact utility this is known as joint production

ten include the costs of contracting an illness, such as medical expenses or foregone wage income. *Harrington and Portney* [1987] develop a theoretical valuation model which incorporates costs of illness as well as behavioural changes intended to mitigate risks. From this framework WTP equals the sum of medical costs, lost wages, defensive expenditures and monetised utility loses. This formulation has been used by many empirical studies as the starting point for examining WTP for environmental quality improvements [see for instance *Pattanayak et al.*, 2005; *Madajewicz et al.*, 2007]. *Bockstael and McConnell* [2007] however show that this result is incorrect and in fact defensive expenditures should not appear in the WTP expression of this model which includes costs of illness. The averting behaviours will be implicit within the framework of an individual 'choosing' an optimal level of health.

The examination of expenditures related to changes in water quality as a means to placing a monetary welfare value on these changes has received much attention. Recent work however has called into question the theoretical framework of many of these studies. In the study we present here we limit our examination to averting behaviours which are more easily attributed to the change in water quality. Identifying the causal link between arsenic consumption and ill health and thus medical expenses on the other hand is an empirically complicated exercise and theoretically unconvincing. This exercise is not attempted here.

5.4.2.1. Empirical applications of averting expenditures

The averting expenditure and cost of illness models have been used as the framework for many studies which examine the welfare impact of water contamination incidents, predominantly in the US. *Harrington et al.* [1989] estimate a welfare loss value for a waterborne giardiasis outbreak in Pennsylvania, US. They estimated direct illness costs such as doctor visits, time and travel losses and costs of laboratory tests. They then combine these figures with losses due to averting actions such as time costs needed to treat and transport water.

Other studies focus solely on the costs of averting activities such as treating water, purchasing bottled water or transportation/time costs [*Abdalla*, 1990; *Laughland et al.*, 1993] or instead focus on the determinants of households/individuals deciding to commence averting behaviours [*Abdalla et al.*, 1992; *Smith and Desvousges*, 1986].

In studies focused on WTP for air quality improvements several different variations on the revealed preference framework have been used. *Bresnahan et al.* [1997] examine the determinants of individual actions to mitigate their exposure to air pollution. *Dickie and Gerking* [1991] utilise a cost of illness approach, deriving WTP estimates for avoiding air pollution from a health treatments demand function. *Gerking and Stanley* [1986] conversely use the parameters from an estimated health production function with respect to exogenous air pollution and individual health care purchases, to derive WTP estimates. In a study of nuisance pest mitigation *Jakus* [1994] examined the determinants of averting behaviours and also the determinants of expenditure levels.

One significant potential impediment to empirical estimation of averting expenditure functions using the Bartik framework is the influence of joint production [see for instance *Bartik*, 1988; *Dickie and Gerking*, 1996; *Abrahams et al.*, 2000; *Dickie*, 2003]. Joint production is where averting expenditures actually appear in the utility function. For instance *Abrahams et al.* [2000] found that household perceptions of their water quality (smell/taste/appearance), in addition to risk perceptions, were significant in the choice of averting behaviours. Thus the expenditures allocated to water filtration or bottled water could not be fully attributed to risk reductions alone. We thus test for the impact of taste, appearance and odour factors in determining a household's decision to switch their water source.

A further empirical hurdle inherent in the theoretical frameworks of averting expenditures is the issue of marginal or non-marginal changes in exogenous 'pollution' or arsenic in the empirical case presented here. The theoretical models show the relationship between a marginal WTP value, a marginal change in pollution and averting expenditures. In our case however it is a non-marginal change. A household has gone from a position of believing that there was no cancer risk from their water to a position of being informed that their drinking water is potentially unsafe. In these situations *Bartik* [1988] suggests using a discrete choice methodology, rather than estimating a continuous function which is subject to optimal adjustments, which maps out a limited portion of the preference relationship. This methodology was utilised by *McConnell and Rosado* [2000] which examined water treatment alternatives in Brazil.

5.5. RDI Testing procedure

The on-going arsenic programme carried out by RDI in Kandal province identified tubewell users and provided a test of the water for high arsenic levels and offered advice on alternative sources. In one leaflet provided by the NGO the following educational information is provided:

- “Many tube wells have high arsenic and you should only use tube well water if an organization has tested your well and told you that it is safe.
- Arsenic is a poison, and if you drink tube well water with high arsenic for more than 3 years, you can develop a bad skin condition usually on your hands and feet and have a much higher risk of cancer than normal people.
- Boiling or filtering your water does not remove the arsenic.
- Arsenic in your well can change over time and if you use it for drinking, it is best if you test it every year.
- Remember that if you change your water source it is very important to boil or filter your water before drinking and cooking to make sure that your family is safe from diarrheal disease.
- Arsenic in the water cannot be seen, smelled, or tasted – only a special test can determine whether a well is safe or not because of arsenic.
- Tube well water can still be used for bathing and washing.

- To get more information or to find out how to have your well tested again, you can call the organization RDI."

In addition to the above details of arsenic in groundwater, the following on alternative water sources was provided:

- "Rainwater - If a household has enough water jars and only uses rainwater for drinking and cooking, rainwater can last through the dry season and wet season. Rainwater should still be boiled or filtered because of bacterial contamination from the roof of the house or in the water jars.
- Piped Water Systems- Some areas (like Kean Svay) have access to piped water systems. Piped water systems provide good water but can be expensive for some families. Consider connecting to a piped water system and use it when your rain water runs out.
- Shallow Well, Rope Pump, or Surface Water- Shallow well or surface water is usually safe from arsenic but can be very high in bacteria. This water must be boiled or filtered before drinking to make sure your family is safe. Note: If the surface water (Pond, lake, stream...), you are going to use for drinking is close by the place where they are using Pesticide or other chemicals. You need to talk to people who are experts about water before using. (sic)"

Also shown to the households were several photographs of arsenicosis sufferers (Figure 5.1), in order to educate the households of the potential symptoms of arsenic poisoning.

Figure 5.1.: Visual symptoms of arsenicosis shown to households as part of the RDI door to door well testing and education programme.

5.6. Sampling procedure and Data Collection

5.6.1. The Sampling Procedure

In order to ascertain the behaviours of the households in Kandal province after being informed of the level of arsenic in their tubewell a household survey was conducted in May 2013. The sample of households was drawn from those households who had been targeted previously by RDI and had received a test of their tubewell water and education regarding arsenic. Households were chosen at random from the RDI database and GPS co-ordinates were used to identify the households for interview. In order to achieve an efficient data collection process, households were chosen from villages either side of Highway 1 in Kandal province (see Figure 5.2).

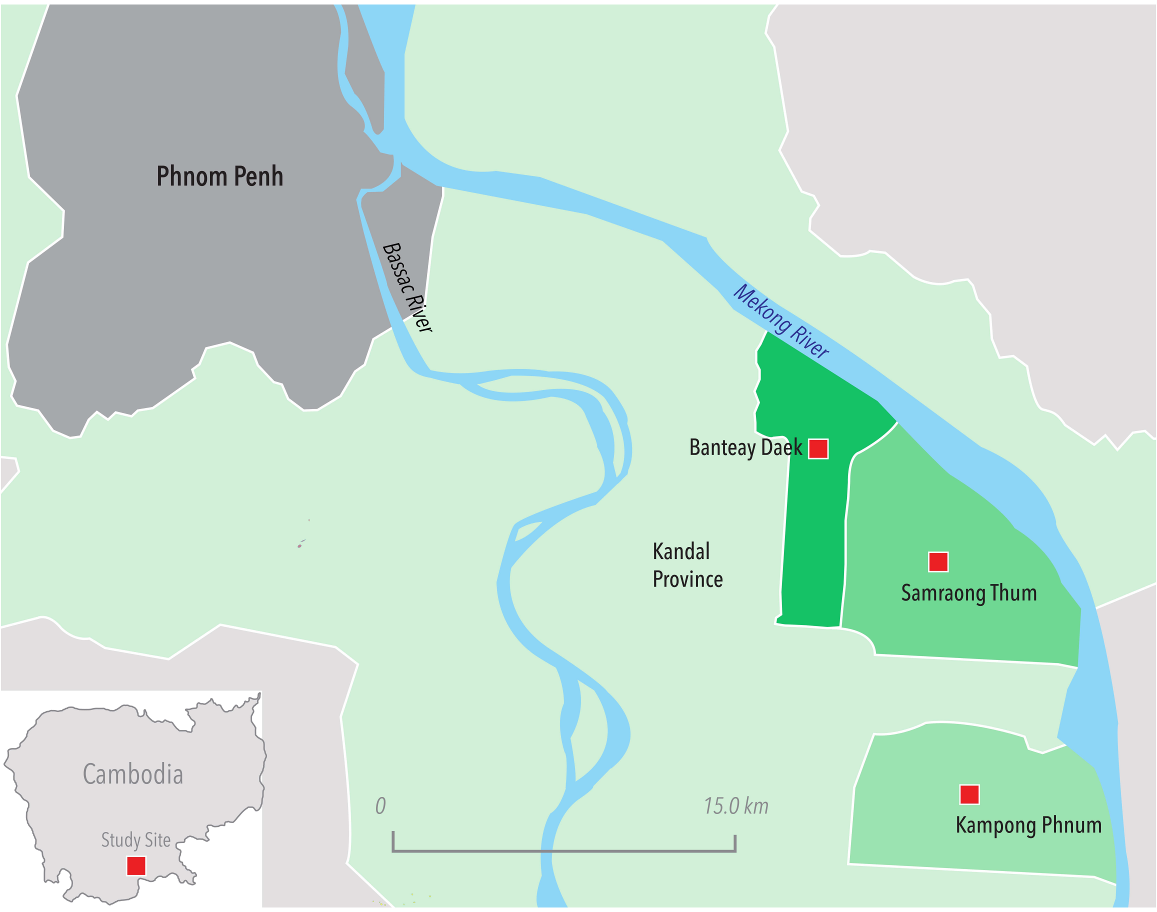

Figure 5.2.: Data Collection Sites

As shown in Figure 5.2, the sample region is along the Mekong River which allows access to water, both by the household members collecting water themselves but also water vendors collecting water from the river to distribute to the households.

The questionnaire was administered by students of Royal University of Phnom Penh and the Royal University of Agriculture in Cambodia over a 10-day period (May 21st to May 30th) and the average interview lasted approximately 45 minutes. The questionnaire used in the fieldwork is included in the appendix but questions were asked related to:

- Household composition and demographics
- Household members' health
- Arsenic knowledge

- Water source usage and characteristics of water sources used (smell, taste, price etc)
- Household Income

5.6.2. Sampled Households

Table 5.1 presents descriptive statistics for respondents and their households, divided by the three sampled communes. These variables are potentially important explanatory variables for well switching behaviour. The descriptive statistics presented in Table 5.1 are for the outlier reduced sample, described in more detail in the next section.

From Table 5.1 we can see that there is broad knowledge of arsenic between all three communes with 91% overall correctly answering that arsenic is a health hazard. Actually knowing someone who has been affected by arsenic poisoning however is rare. Only about 8% (26/303) of the sample reported knowing somebody with either a confirmed or suspected arsenic illness. The implications of this low proportion will be tested in the water source switching analysis.

The table also presents both respondent and household level descriptive statistics which show that a higher proportion of respondents were female with the average age in their 40s with roughly 5 years spent in education. Although the respondent descriptive statistics are important for the analysis of the household switching decision, as we requested a respondent to be either the head of the household or the person responsible for water collection, household characteristics may also play an important role. Factors such as household size, the number of people in the house using the water and the number of children in the house may influence the decision of the household to switch sources.

Table 5.1.: Summary of Respondents by Commune

	BD	ST	KP	Total
Age of Respondent	47.54	47.02	48.89	47.57
	(13.24)	(16.36)	(13.54)	(15.31)
Gender of Respondent (0-Female, 1-Male)	0.314	0.344	0.367	0.347
	(0.471)	(0.476)	(0.485)	(0.477)
Years of Education	5.114	5.296	4.392	5.040
	(3.428)	(3.505)	(3.550)	(3.518)
Can the Respondent Read? (0-No, 1-Yes)	0.800	0.762	0.620	0.729
	(0.406)	(0.427)	(0.488)	(0.445)
Can the Respondent Write? (0-No, 1-Yes)	0.800	0.730	0.620	0.710
	(0.406)	(0.445)	(0.488)	(0.455)
Health of Respondent (5-Very Good, 0-Very Bad)	3.171	2.984	3.013	3.013
	(0.785)	(0.847)	(0.824)	(0.834)
Lost > 2 Weeks Work to Illness? (0-No, 1-Yes)	0.314	0.296	0.342	0.310
	(0.471)	(0.458)	(0.477)	(0.463)
Smokes? (0-No, 1-Yes)	0.171	0.185	0.266	0.205
	(0.382)	(0.389)	(0.445)	(0.404)
Household Size	5.600	5.074	4.924	5.096
	(2.003)	(1.791)	(1.852)	(1.836)
Number of children (<16) in house	4.371	3.810	3.506	3.795
	(1.573)	(1.623)	(1.492)	(1.598)
Ln(Total Household Income)	13.90	12.49	13.42	12.90
	(1.514)	(4.065)	(2.040)	(3.451)
Is Arsenic Fatal? (0-No, 1-Yes)	0.971	0.889	0.949	0.914
	(0.169)	(0.315)	(0.221)	(0.281)
Know Someone with Arsenic Poisoning?(0-No, 1-Yes)	0	0.106	0.0759	0.0858
	(0)	(0.308)	(0.267)	(0.281)
Observations	35	189	79	303
	(0)	(0)	(0)	(61.22)

Mean of each variable with standard deviation in parentheses.

BD -Banteay Dek, ST -Samrong Thom, KP -Kampong Phnum

5.6.3. Original Tubewell Sources (Dry Season)

In this section we examine in detail the attributes of the tubewells used by the households prior to the arsenic test being conducted. The descriptive statistics for these attributes are shown in Table 5.2.

5.6.3.1. Distance from household

The distance statistics show that on average the tubewells were located within 11-19m of the household. The median distance from the household is only 6m, with the mean values being inflated by households who travel large distances to the tubewell for water collection. In order to make the sample more representative and to prevent outlier problems, households who utilised tube wells more than 1km away from their house were dropped from the analysis. This constituted 6 dropped households from a total original sample size of 345.

5.6.3.2. Cost of using tubewell

The mean costs for using a tubewell range approximately between 1,300 Riels and 2,000 Riels per week (or between 0.325 and 0.5 USD). The median costs per week for using the tubewell is however 0 Riels, reflecting that once a tubewell has been installed, ongoing costs are usually zero, with some occasional maintenance costs for the hand-pump or other equipment used to pump or transport water. The positive values on the tube well cost variables perhaps reflect petrol costs for those households using a motorised pump to extract the water, or fees paid to tube well owners if the well is not family owned. To reduce outlier impacts, those households who report paying more than 10,000 Riels per week (USD 2.5) for using the tube well have been dropped from the data set. This constitutes 16 households.

Table 5.2.: Summary of Tubewell Water Sources Tested for Arsenic

	BD	ST	KP	Total
Distance from Source to House (m)	11.71	19.05	10.05	15.85
	(16.87)	(76.78)	(12.87)	(61.33)
Cost of using source (riels/week)	1500	1945.8	1293.7	1724.3
	(2167.1)	(2690.8)	(2333.2)	(2554.8)
Daily Mins collecting water	37.29	23.54	21.90	24.70
	(28.88)	(22.65)	(16.78)	(22.52)
Reliability Source?*	1.114	1.111	1.089	1.106
	(0.404)	(0.390)	(0.286)	(0.367)
Is raw water safe to drink?**	0.200	0.386	0.392	0.366
	(0.406)	(0.488)	(0.491)	(0.483)
Arsenic in water?*	0.457	0.577	0.696	0.594
	(0.505)	(0.495)	(0.463)	(0.492)
Total Risk (500/500)*	0	0.190	0.139	0.155
	(0)	(0.394)	(0.348)	(0.363)
Water not clear?*	0.314	0.418	0.203	0.350
	(0.471)	(0.495)	(0.404)	(0.478)
Water tastes bad?*	0.143	0.222	0.165	0.198
	(0.355)	(0.417)	(0.373)	(0.399)
Water smells bad?*	0.257	0.116	0.253	0.168
	(0.443)	(0.322)	(0.438)	(0.375)
Source used for irrigations?*	0.943	0.889	0.911	0.901
	(0.236)	(0.315)	(0.286)	(0.299)
Used for bathing/washing/cleaning*	0.971	0.958	0.962	0.960
	(0.169)	(0.202)	(0.192)	(0.195)
Always treat water before drinking?*	0.743	0.767	0.709	0.749
	(0.443)	(0.424)	(0.457)	(0.434)
Change source?*	0.629	0.624	0.759	0.660
	(0.490)	(0.486)	(0.430)	(0.474)
Observations	35	189	79	303

Mean of each variable with standard deviation in parentheses.

Commune names: BD -Banteay Dek, ST -Samrong Thom, KP -Kampong Phnum

* 0-No, 1-Yes

** 0-No/Don't Know, 1-Yes

5.6.3.3. Time costs

Daily time spent collecting water depends upon many factors, including the distance from the household to the source, the use of that water (for example cooking, cleaning or watering crops) and the number of people in the household. Average time spent per day ranges from 20 to 38 mins. This time includes: traveling to the water source, pumping the water from the ground, transporting the water to the household and transferring the water to the main storage containers. Median time spent collecting water is 20 minutes per day which indicates some outliers.

5.6.3.4. Reliability of source

The reliability variable has three levels: 1 indicates that the source is always available, 2 indicates that the source is normally available (>50% of the time) and 3 indicates that the source is often not available (<50% of the time). Availability might indicate that no water can be drawn or that the household believes that the water should not be consumed. A higher number thus indicates a lower reliability. The median value for this variable is 1 which shows that tubewells are predominately reliable. There are no observable reliability differences between the sampled communes.

5.6.3.5. Safety of water

The households were questioned regarding whether they believed their specific tubewell source to be safe to drink without treatment (i.e. the raw water). This safety included microbial and arsenic risks. The households were given the option of choosing a) yes it is safe (coded as 1), b) no it is not safe (coded as 0) or c) don't know. There are 5 households who answered 'don't know' to this question. We transform the safety variable to be a 1 if the respondents felt confident in the safety of the raw water (124 households from the sample excluding outliers) or a 0 otherwise[6].

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[6] This includes the no, the water is unsafe' and the 'don't know' responses.

5.6.3.6. Arsenic in water?

Households were included in the sample conditional upon them having had their water source tested for arsenic. This variable records the household's response to whether the source tested positive for high arsenic (1) or negative (0). Out of the outlier excluded sample 181/306 households reported receiving a positive high arsenic test result. Interestingly a cross-tabulation of the safety and arsenic variables reveals that a similar proportion of respondents believe that their tubewell water is safe to drink without treatment for those who have received a positive or negative arsenic test result (35% and 38% respectively). This perhaps reflects the low proportion of households who consider their raw water safe to drink.

5.6.3.7. Subjective arsenic risk

Each respondent was asked to give their quantified opinion on the safety of the tubewell water in relation to arsenic risk. The RDI education strategy previously informed the households whether their water was safe to drink or not and had provided information of the potential impact of arsenic poisoning.

Each household however may have a different recollection of the education material or a different interpretation of the risks involved, which will likely impact the household's decision to switch water sources. In order to account for this we asked each respondent who had a tubewell which tested positive for high arsenic 'if 500 people used the tubewell as their main drinking water source, how many would get sick from arsenic poisoning over their lifetime'. The response to this question was thus a number between 0, which indicates that the respondent felt that the water was safe and nobody would get sick, and 500, where the respondent believed that everyone would suffer from arsenic poisoning. Three households did not provide answers to this question and have been dropped from the sample. Figure 5.3 presents a histogram of the subjective arsenic risk responses for those households who received a high arsenic test result for their tubewell water.

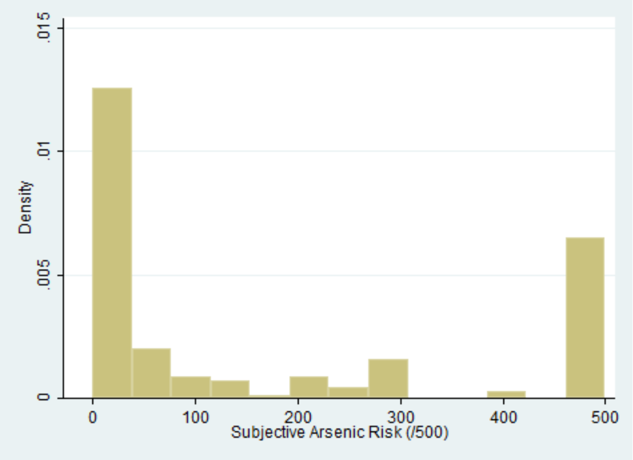

Figure 5.3.: Subjective arsenic risk for households with high arsenic tubewell

Both a standard OLS regression and a Tobit regression, truncated at 0 and 500, of the subjective arsenic risk on a set of water source characteristics, respondent characteristics and commune indicator variables failed to find any significant determinants of the households risk response. 45 of the 181 respondents with high arsenic reported that everyone who drank the tubewell water would become sick (500/500). Descriptive statistics for this variable (500/500) are included in Table 5.2.

5.6.3.8. Appearance, taste and smell

In addition to the risk and costs variables, household reported aesthetic properties of the water sources were also recorded. The appearance property of the water source was recorded in 3 levels. Clear (1), cloudy (2) and dirty/visible particulates (3). Only 19 of the outlier reduced sample stated that their tubewell water was dirty. We construct a new variable to denote whether the household viewed the water had any form of visual impairment (1) or whether the water was clear (0). Overall 35% of households reported that the tubewell water was cloudy or dirty, with households in Samrong Thom commune reporting higher rates of visual impairment.

Taste was recorded in 5 levels: very good (1), good (2), average (3), bad (4) and very bad (5). We note that 223 find the taste of the tubewell water to be good or average, whilst 21 find the water to taste very good. 60 households reported the water to taste bad and 2 reported a very bad taste. For further analysis we construct a new variable which takes a value 0 if the water is reported as average or above and a 1 if the water is described as bad or very bad. Table 5.2 shows that roughly 20% of respondents report their tubewell water having a bad taste.

Smell was recorded in 3 levels: no smell (1), smells bad (2) and smells OK (3). We create a new variable to record whether the respondent stated that the source had a bad smell (1) or otherwise. A new variable was then created which indicated whether the respondent stated that the tubewell water had a 'bad' smell. Overall roughly 17% of respondents stated that the water had a bad smell. Households in Samrong Thom reported a higher rate of visual problems to their water, however there is a lower proportion of water from tubewells which have a bad smell, relative to households from the other 2 communes.

5.6.3.9. Use for water

The vast majority of households used the tubewell water source for irrigation (90%) and cleaning (69%), in addition to using the water as their primary drinking water source.

5.6.3.10. Treatment Decision

Perhaps of greatest interest is the household's decision to treat their water prior to consumption. Respondents were asked, when they drank the water how regularly the applied a treatment: always (1), sometimes (>50%), sometimes (<50%) or never. Figure 5.4 illustrates that the majority of respondents (229 households) report that they always treat their tubewell water prior to drinking. The remaining households are more evenly distributed between the other treatment frequencies. We create a new variable which records whether the household always treats their water, or otherwise. This variable is shown in Table 5.2. Nearly 75% of households in the sample report that they always treat the tubewell water prior to drinking it.

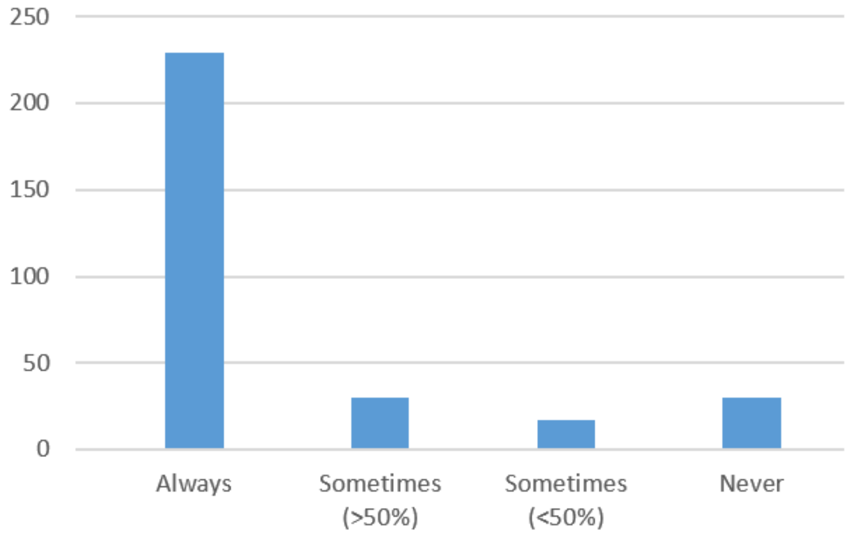

Figure 5.4.: Frequency of treatment decision for tubewell prior to arsenic test

A logistic regression analysis shows that respondents who are female, older and who have reported that they were not confident in the safety of the raw water were more likely to report that they always treat the tubewell water. These issues are explored in more detail in the next section.

5.6.3.11. Summary

Overall we see only small differences between the three provinces for the characteristics of the tubewell sources. Of note the households in Samrong Thom on average have to travel further to the tubewell and have to pay more per week for water which tastes and smells worse than the water from tubewells in the 2 other communes.

The self-reported tubewell and tubewell water attributes presented here show that there is some dissatisfaction with the taste, smell and appearance of the water and that roughly 60% of the households had tubewells with high arsenic concentrations. In the next section we analyse the impact of these original water source characteristics on the decision to switch away from this source.

5.7. Empirical Analysis

5.7.1. Water Source Switching (RQ1&RQ2)

Table 5.3 breaks down the new dry season water sources used by the surveyed households in relation to the arsenic test result they received. The sampling strategy focused on households which had previously used a tubewell as their primary drinking water source which had been tested for high arsenic concentrations. 27 of the sampled households had switched away from using the tubewell source prior to the arsenic test being conducted. Although the arsenic test result itself did not directly result in a switch, concerns about arsenic in general may have had a role in their decision. Of those 27 households 12 subsequently received a negative test result whilst 15 received a positive test result for high arsenic.

Table 5.3.: New Water Source by Arsenic Test Result

	Water Supply Test Positive for High Arsenic (>50ppb)?								
newsourced2	**< 50 ppb Arsenic**			**> 50 ppb Arsenic**			**Total**		
	No.	Col %	Cum %	No.	Col %	Cum %	No.	Col %	Cum %
No Change	70	75.3	75.3	27	12.9	12.9	97	32.0	32.0
Changed Prior to Test	12	12.9	88.2	15	7.1	20.0	27	8.9	40.9
Groundwater	0	0.0	88.2	23	11.0	31.0	23	7.6	48.5
Surface Water	4	4.3	92.5	15	7.1	38.1	19	6.3	54.8
Rain Water	0	0.0	92.5	22	10.5	48.6	22	7.3	62.0
Water Vendor	7	7.5	100.0	92	43.8	92.4	99	32.7	94.7
Piped Connection	0	0.0	100.0	16	7.6	100.0	16	5.3	100.0
Total	93	100.0		210	100.0		303	100.0	

Source: revealed.dta

There are 93 households in total who received a negative high arsenic test result.

Interestingly however 11 of these households report that they had switched away from using the tubewell since receiving the test result, even though the test result indicated low arsenic content. These households switched to either a surface water source or a supply delivered by a water vendor. [7]

210 households received a positive test result for arsenic on the tubewell water they used as their primary drinking water. Of those households 27 remained using their arsenic contaminated tubewell source as their primary drinking water source. For the remaining 183 households who have changed their source, over half have switched to paid water sources, i.e. water vendors (92 households) and piped connections (16 households). The remaining households utilise surface water (15), rain water (22) or alternative groundwater sources (23). The key result here however is that 86% (168/195) of households who found that they had arsenic in their water subsequently changed their drinking water source.

In order to examine water switching behaviour, which is represented by a binary variable (1 = change water source, 0 = no change), in more detail we estimate logit models, focusing on household socio-economic factors and tubewell specific characteristics. In Model 1 in Table 5.4 we present the results of a logit model with many of the socio-demographic and original well specific factors included as explanatory variables.

In common with much of the previous literature [e.g. *Abdalla et al.*, 1992], many of these factors are not statistically significant in explaining averting behaviours. What is statistically significant is the subjective appraisal of arsenic risk, the taste of the original tubewell and the appearance of the water. Households with higher subjective risk appraisal of their water source and which believes that their tubewell water tastes bad or looks bad is more likely to switch away from that source. Interestingly we

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[7] It is somewhat surprising that a household would switch away from using a groundwater source and towards a surface water source given the extra effort often required to collect the water and the impaired water quality given that the groundwater has tested negative for high arsenic. Potential explanations for such behaviour may perhaps include a general lack of trust in groundwater sources due to friends and neighbours receiving a high arsenic test result or aesthetic qualities of the tubewell water influencing a switch.

also find that there is a small negative but significant estimate on the costs parameter. Households who pay for their tubewell water are less likely to switch which may indicate that, given the household already has to pay, there may be limited alternative sources.

Given the large number of insignificant parameter estimates we re-estimate the model with reduced parameters numbers (Model 2). This model retains the significance and parameter signs from the previous model. We also include in this model a parameter to indicate the commune in which the household is located. This parameter indicates that households in Samrong Thom are less likely to switch their water source than households in other communes, which is seemingly not explained by differences in household socio-economic factors [8].

In addition to the factors that impact choice, we also examine the factors which might influence the consistency of that choice. To examine this we utilise a heteroscedastic probit model (Model 3) where the error term of the model is parameterised [*DeShazo and Fermo*, 2002]. The parameters in the bottom section of the table shows that the higher the subjective risk of arsenic poisoning, the lower the variance of the error term.[9] Households with male respondents, who were the head of the household or in charge of water decisions, were found to have a larger variance in terms of switching behaviour.

In terms of policy issues the results of these models clearly indicate the strong household preferences for the more aesthetic aspects of water quality such as taste and appearance. Remediation policies might therefore focus on where households may acquire water sources which are not only arsenic free but also have improved aesthetic qualities.

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[8] We estimated multi-level models to account for potential differences between the communes, however test statistics indicate that the models including fixed effects are better at accounting for heterogeneity between communes.

[9] The scale term, which is parameterised and shown in the bottom section of the table, is inversely related to the error variance.

Table 5.4.: Logit and Het Probit Models: Change Water Source

	(1)	(2)	(3)
Subjective arsenic risk (x/500)	0.007***	0.007***	0.026***
	0.002	0.002	0.006
Cost of using source (riels/week)	-0.000**	-0.000*	-0.000
	0.000	0.000	0.000
Daily time spent collecting water (mins)	0.002		
	0.006		
Water not clear? (0-No, 1-Yes)	1.095***	1.287***	0.629***
	0.378	0.369	0.219
Water tastes bad?(0-No, 1-Yes)?	2.468***	2.463***	1.281***
	0.679	0.652	0.376
Water smells bad? (0-No, 1-Yes)	0.444		
	0.430		
Age of Respondent	-0.014		
	0.013		
Gender of Respondent (0-Female, 1-Male)	0.573	0.545*	0.312*
	0.398	0.314	0.163
Years of Education	0.028		
	0.047		
Health of Respondent (5-Very Good, 0-Very Bad)	0.378		
	0.230		
Lost > 2 Weeks Work to Illness? (0-No, 1-Yes)	-0.082		
	0.379		
Smokes? (0-No, 1-Yes)	0.008		
	0.450		
Household Size	-0.135	-0.095	-0.026
	0.125	0.083	0.041
Number of children (<16) in house	0.018		
	0.139		
Ln(Total Household Income)	0.058		
	0.047		
Is Arsenic Fatal? (0-No, 1-Yes)	0.673		
	0.555		
Know Someone with Arsenic Poisoning?(0-No, 1-Yes)	0.534		
	0.578		
Samrong Thom Commune		-0.825***	-0.191
		0.311	0.169
Constant	-1.823	0.562	-0.285
	1.254	0.527	0.293
Scale			
Subjective arsenic risk (illnesses/500)			0.005***
			0.001
Gender of Respondent (0-Female, 1-Male)			-0.737***
			0.254
log-likelihood	-138.279	-139.174	-130.667
N	303	303	303

* p<0.10, ** p<0.05, *** p<0.01

1- Over parameterised model 2- Preferred logit model 3- Het Probit

5.7.2. New Water Sources (RQ3)

Given the significant commune parameter in Model 2 we investigate these commune differences in more detail. There are perhaps differences between opportunities for switching and other commune specific aspects that may influence the final water source decision. To examine these issues Table 5.5 shows the breakdown of post arsenic test water sources, used in the dry season, for those households who have received a positive test for high arsenic. The 15 households who had already changed water sources, prior to testing but received a positive test result, are included in this table for a more general analysis of new water sources.[10]

Table 5.5.: New Water Source by Commune

	Commune											
newsourced2	**Banteay Dek**			**Samrong Thom**			**Kampong Phnum**			**Total**		
	No.	Col %	Cum %	No.	Col %	Cum %	No.	Col %	Cum %	No.	Col %	Cum %
No Change	6	25.0	25.0	18	14.3	14.3	3	5.0	5.0	27	12.9	12.9
Groundwater	5	20.8	45.8	17	13.5	27.8	3	5.0	10.0	25	11.9	24.8
Surface Water	1	4.2	50.0	16	12.7	40.5	0	0.0	10.0	17	8.1	32.9
Rain Water	2	8.3	58.3	16	12.7	53.2	6	10.0	20.0	24	11.4	44.3
Water Vendor	10	41.7	100.0	59	46.8	100.0	28	46.7	66.7	97	46.2	90.5
Piped Connection	0	0.0	100.0	0	0.0	100.0	20	33.3	100.0	20	9.5	100.0
Total	24	100.0		126	100.0		60	100.0		210	100.0	

Source: revealed.dta

From this table a number of themes appear. Firstly, the highest proportion of households switch to using water vendors as their primary water source, in all communes. Secondly the proportion of households that change their water source is higher in Kampong Phnum where piped water connections appear to be a viable alternative. There are no households in the other communes which utilise piped water connections, which are able to provide the same 'on demand' access to water as a tubewell, without requiring storage (e.g. such as water from surface water or vendor water). In Kampong

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[10] Comparison of Table 4.3 and Table 4.5 shows the breakdown of new sources for these 15 households is: 2-New Groundwater, 2-Surface Water, 2-Rain Water, 5-Water Vendor and 4-Piped connection.

Phnum, where piped connections are in use, we also see that there are no households who use surface water, with the vast majority of households utilising the sources requiring ongoing payments.

5.7.3. New Water Source Choice (RQ4)

In order to examine the determinants of water source choice in more detail a multinomial logit model is estimated, the results of which are shown in Table 5.6. The model presented in this table has the current water source as the dependent variable with 'no change' as the base category. As such the parameters should be considered as probability of selection of the alternative water sources, compared with staying with the tubewell source, tested by RDI. Given the assumption within the averting expenditures framework that household decisions are the result of utility maximising behaviour the newly selected sources can be viewed as the second most preferred source compared with the tubewell source prior to knowledge that it contained arsenic. This new information to some households has prompted some to use the second best option whilst for others, who decide not to switch, the disutility from the arsenic contamination is not enough to prompt a switch.

The model presented in Table 5.6 is the model of best fit given the log-likelihood result and includes parameters which are significant for at least one water source option. [11] The dependent variable for the model presented in Table 5.6 is a nominal variable, as follows: 0- no change to water source, A- groundwater source, B- surface water, C- rain water, D- water vendor, E- piped connection. The results from this model shows that the impact of subjective arsenic risk increases the probability of selection for all of the

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[11] Hausman-McFadden (HM) test results provides evidence that for this model, independence of irrelevant alternatives (IIA) has not been violated. The HM test for this model consists of 7 tests after sequentially excluding independent variable categories. This produces 7 restricted models with which to compare to the unrestricted model. The test produces the following statistics which suggest that IIA has not been violated. No Change: Chi2-0.015 , df-7. Piped Water: Chi2- (-0.000), df-3. Tube Well: Chi2- 0.049, df- 6. Dug Well: Chi2- 0.257, df-10. Water Vendor: Chi2- 1.9, df-5. Pond/River: Chi2- 0.867, df-8. Rain Water: Chi2- (-0.000), df- 4.

Table 5.6.: Multinomial Logit Model: New Water Source

	A	B	C	D	E
Subjective arsenic risk (/500)	0.008***	0.006***	0.008***	0.006***	0.012***
	(0.002)	(0.002)	(0.002)	(0.002)	(0.002)
Cost (riels/week)	0.000	-0.000	-0.000	-0.000***	0.000
	(0.000)	(0.000)	(0.000)	(0.000)	(0.000)
Water not clear?+	1.920***	0.864	1.053*	1.342***	0.874
	(0.584)	(0.554)	(0.574)	(0.397)	(0.827)
Water tastes bad?+	2.804***	2.586***	2.804***	2.443***	1.621
	(0.784)	(0.796)	(0.794)	(0.671)	(1.143)
Respondent Male?	0.907*	1.088**	1.300**	0.181	0.235
	(0.527)	(0.500)	(0.505)	(0.352)	(0.666)
Samrong Thom Commune	-0.970*	2.109**	-0.693	-0.867**	-17.787
	(0.574)	(1.066)	(0.537)	(0.349)	(456.906)
Household Size	-0.098	-0.397***	0.063	0.003	-0.475**
	(0.144)	(0.153)	(0.138)	(0.092)	(0.204)
Constant	-2.578***	-2.518**	-2.791***	-0.228	0.084
	(0.958)	(1.267)	(0.935)	(0.586)	(1.040)
log-likelihood	-351.175				
Observations	303				

* p<0.10, ** p<0.05, *** p<0.01

Standard error in parentheses.

A -Groundwater, B-Surface Water, C-Rain Water, D-Water Vendor, E-Piped Connection.

Base Category- No Change

+ 0-No, 1-Yes

alternative water sources. The next 3 parameters all refer to attributes of the original tubewell utilised by the household. The cost of using the original tubewell has a small yet significant negative effect on converting to water supplied by a water vendor.

An unclear appearance of the tubewell water increases the probability of change towards all sources, bar surface water and piped water. This perhaps indicates that households are motivated to move towards a more visually appealing source and have ruled out surface (river/lake/pond) water which can contain debris and piped water which is the most expensive option and thus appearance is not a key motivating factor. Bad taste of the original tubewell source increases the probability of source switching to all sources bar piped supply, which again suggests that aesthetic attributes were not motivating switching to the premium option.

Households where the respondent was male (and given the sampling strategy this was the head of the household or the person in charge of water collection) were significantly more likely to choose one of the ’free’ options rather than the options involving ongoing fees (water vendor and piped connection). Households in Samrong Thom Commune are more likely to select surface water (free apart from the expenses involved in collecting the water) and less likely to purchase water from a vendor (a payment is required but the water is delivered).

A larger household size reduces the probability of switching to a surface water source or a piped connection. This is perhaps indicates that larger households would have to pay more per household for water through a piped connection and may have to make more trips to a surface source to collect enough water for a larger family. The burden of those trips and the expense would however be shared between a larger number of people.

From a policy perspective all sources are being utilised, to some degree and by different households, as an option to avert their exposure to arsenic risk. As with the binary choice analysis, the aesthetic attributes of the original water source appear to be key determinants of switching behaviours, for all sources bar the piped supply which is the most expensive option and which is not available in many areas. Household size, the gender of the decision maker and the subjective view on arsenic risk are also important influences on water source selection.

5.7.4. New Water Source Characteristics (RQ5)

Given that many of the households have changed their water source, largely driven by arsenic and aesthetic factors, it is important to examine in more detail the characteristics of those new sources. For instance have households had to increase their expenditure of drinking water or have they had to increase the amount of time spent collecting water. Table 5.7 presents the mean values for water source characteristics, separated by the type of water source.

From this table several significant relationships are shown. Firstly proportions of households that treat their drinking water (e.g. by boiling) are higher than sources that require ongoing payments i.e. water vendors (D) at 80% and piped connections (E) at 81%. This perhaps reflects that if a household has already made an investment in purchasing the water, the extra time and expense required to boil the water is relatively insignificant.

The treatment difference variable is calculated as the difference between dummy variables which indicate if the household always treated their water when using the original water source and a dummy variable which indicates if the household treats their water from the new source. As such the variable can only take 3 values: -1 if the household now treats their water when they previously did not, 0 if there was no change and 1 if the household no longer treats their water. The positive statistics in Table 5.7 for this variable show that households who have switched to surface water, rain water and piped sources now treat their drinking water less regularly. Those households who now use water from a vendor treat their water more often prior to consumption.

Table 5.7.: Summary of New Water Source Characteristics

	A	B	C	D	E
Always Treat Water?+	0.652	0.526	0.636	0.798	0.813
	(0.487)	(0.513)	(0.492)	(0.404)	(0.403)
Treatment Difference	0	0.105	0.0455	-0.0101	0.0625
	(0)	(0.459)	(0.486)	(0.463)	(0.443)
Cost of using source (riels/week)	3282.6	11547.4	1818.2	9373.9	9987.5
	(4136.4)	(14642.7)	(8528.0)	(14445.7)	(24221.1)
Mins per day collecting water	25.30	50	5.227	18.45	5.625
	(23.57)	(60.85)	(10.74)	(31.21)	(15.80)
Reliability of Source	1	0.737	0.545	0.818	0.938
	(0)	(0.452)	(0.510)	(0.388)	(0.250)
Safe to drink?++	0.522	0.684	0.409	0.394	0.250
	(0.511)	(0.478)	(0.503)	(0.491)	(0.447)
Water not clear?+	0.304	0.421	0.136	0.212	0
	(0.470)	(0.507)	(0.351)	(0.411)	(0)
Water tastes bad?+	0.174	0	0	0	0
	(0.388)	(0)	(0)	(0)	(0)
Water smells bad?+	0.217	0.158	0.0455	0.101	0.188
	(0.422)	(0.375)	(0.213)	(0.303)	(0.403)
Source used for irrigations?*	0.652	0.263	0.0909	0.0909	0.313
	(0.487)	(0.452)	(0.294)	(0.289)	(0.479)
Used for bathing?+	0.826	0.316	0.273	0.283	0.438
	(0.388)	(0.478)	(0.456)	(0.453)	(0.512)
Observations	25	17	24	97	20

Mean of each variable with standard deviation in parentheses.

A -Groundwater, B-Surface Water, C-Rain Water, D-Water Vendor, E-Piped Connection.

+ 0-No, 1-Yes

++ 0-No/Don't Know, 1-Yes

As we indicated in the introduction the decisions to treat drinking water prior to consumption is likely to be in some way determined by unobservables such as optimism or social networks. These unobservables are also likely to play a key determining role in the households' formation of risk beliefs and the decision to switch water source. These common unobservable variables may lead to biased parameter estimates when these decisions are jointly determined. The error terms for models estimating changing water sources, subjective arsenic risk and the decision to treat their water are likely to be correlated leading to biased parameter estimates. To account for this bias seemingly unrelated bivariate probit models are estimated where the error terms for these models are linked. The estimation of these models showed that the covariance of the error term was not statistically different from zero. As such we can estimate separate models without bias [*Nauges and Van Den Berg*, 2009; *Onjala et al.*, 2013][12].

Costs of using water is least for households harvesting rain water and using new tubewells. Interestingly costs of using surface water are relatively high suggesting high transportation costs. Reliability of sources is lowest for rain water which relies upon weather and household storage jars. The self reported subjective safety of the water is highest for surface water which indicates that those households using this water believe it to be safe. The lowest belief that the water is safe to drink without treatment are users of piped water supplies and water vendors. This proportion matches closely with the proportion of people who treat water being highest for these sources.

In order to examine these issues in more details we estimate OLS regression models and Logit models of the cost, time, taste and treatment of the water used by the household against household demographic variables, original water source attributes and whether the household changed their source. This is done for those households who have received a positive test for high arsenic. The parameters from these models are presented in Table 5.8.

The vast majority of the parameters in these models are insignificant at the 10% level.

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[12] Results available from author upon request

Table 5.8.: OLS and Logit Models: New Source Characteristics

	Riels/Week	Mins/Day	Taste	Treat
Cost of using source (riels/week)	0.499*	0.002	0.000	0.000
	(0.267)	(0.002)	(0.000)	(0.000)
Mins per day collecting water	7.369	0.352**	-0.001	-0.000
	(29.979)	(0.179)	(0.000)	(0.001)
Reliability of Source	998.062	-10.089	0.018	-0.028
	(1838.272)	(10.960)	(0.022)	(0.055)
Safe to drink?++	1435.156	0.284	0.016	-0.109**
	(1446.428)	(8.623)	(0.017)	(0.043)
Water not clear?+	1859.966	7.209	-0.033*	-0.004
	(1520.074)	(9.062)	(0.018)	(0.045)
Water tastes bad?+	-2754.025	14.516	0.104***	-0.101*
	(1854.073)	(11.054)	(0.022)	(0.055)
Water smells bad?+	2445.192	-4.041	-0.013	-0.000
	(1789.196)	(10.667)	(0.021)	(0.053)
Used for irrigations?+	4446.557	8.096	0.019	-0.052
	(2811.006)	(16.759)	(0.033)	(0.084)
Used for bathing?+	-859.389	-0.064	-0.017	0.015
	(4320.311)	(25.757)	(0.051)	(0.129)
Always Treat Water?+	2355.796	10.821	-0.034*	0.631***
	(1654.318)	(9.863)	(0.020)	(0.049)
Age of Respondent	-38.783	0.432	-0.001	0.000
	(56.038)	(0.334)	(0.001)	(0.002)
Respondent Male?+	-1633.222	6.636	0.006	0.013
	(1812.687)	(10.807)	(0.021)	(0.054)
Years of Education?	-27.685	-0.194	0.000	0.000
	(280.327)	(1.671)	(0.003)	(0.008)
Can Respondent Read?+	509.911	2.353	-0.022	-0.029
	(2109.497)	(12.577)	(0.025)	(0.063)
Health of Respondent	1112.120	0.787	0.012	0.002
	(963.676)	(5.745)	(0.011)	(0.029)
Smokes?+	-680.571	-16.429	-0.027	-0.009
	(2040.457)	(12.165)	(0.024)	(0.061)
Money on medicine	-0.001	0.000	-0.000	-0.000
	(0.001)	(0.000)	(0.000)	(0.000)
Changed Source?+	5750.502***	-21.972**	0.001	0.044
	(1530.698)	(9.126)	(0.018)	(0.046)
Constant	-6864.206	-2.319	0.029	0.346**
	(5591.338)	(33.335)	(0.066)	(0.167)
log-likelihood	-3248.692	-1696.613	189.120	-91.700
N	303	303	303	303

* p<0.10, ** p<0.05, *** p<0.01

OLS Models: Money and Time, Logit Models: Taste and Treatment

A-Costs (Riels/week), B-Time (Mins/day), C-Tastes Bad(1-Yes, 0-No), D-Always Treat? (1-Yes, 0-No)

\+ 0-No, 1-Yes

++ 0-No/Don't Know, 1-Yes

The main parameter of interest is for the dummy variable which indicates whether the household has changed their water source. We use this parameter to calculate predictive margins, which show the mean values of the dependent variable at the two levels of the independent variable (change or not changed), controlling for the other variables in the model. These mean values are shown in Table 5.9 for the models of weekly costs and minutes per day collecting water. We are unable to calculate these values for the models of taste and treatment as the parameters for change water source are insignificant. Table 5.9 shows that mean weekly costs go up from 2,131 Riels to 7,882 Riels for those households who switch water sources. The second column in Table 5.9 shows that on average daily time collecting water actually falls from roughly 40 to 20 minutes. This is due to the predominance of water vendors as an alternative water source. Water vendors deliver water direct to the household. Household may have to organise their water between storage jars but they do not have to actually collect the water.

We also re-estimate the models with water sources as dummy variables. This allows the estimation of predictive margins for each water source type. These are shown in the final two columns of Table 5.9. The third column shows mean riels per week for each water source, controlling for the other variables in the regression. The figures show that costs are somewhat similar (around 2,500 to 5,000 riels) for households who do not change, have already changed, use new groundwater sources and for households that harvest rainwater. Households which choose surface water (either collecting themselves or through a vendor) or piped water pay roughly double (11,387.21 riels).

The final column in Table 5.9 shows the predictive margins of time spent collecting water from the individual water sources. These figures show that households that switch to collecting their own surface water spend the most time (49 mins per day) and households who now harvest rainwater, utilise a water vendor or a piped connection spend the least amount of time.

Table 5.9.: Predictive Margins: Time and Money Spent on New Sources

	(Riels/Week)	(Mins/Day)	(Riels/Week)	(Mins/Day)
Change Water Source				
Yes	2131.157*	41.57960***		
	(1201.865)	(7.165)		
No	7881.659***	19.607***		
	(833.3056)	(4.968)		
Water Source				
No Change			2365.314**	42.59***
			(1206.941)	(7.3)
Already Change			5200.059**	22.193*
			(2229.354)	(13.484)
Groundwater			2085.858	22.639
			(2440.759)	(14.763)
Surface Water			11387.21***	48.915***
			(2627.324)	(15.891)
Rain Water			2642.952	2.98
			(2458.09)	(14.867)
Water Vendor			9555.284***	18.416***
			(1160.939)	(7.022)
Piped Connection			(9847.132)	8.418
			(2900.337)	(17.542)
n	303	303	303	303

* p<0.10, ** p<0.05, *** p<0.01

5.8. Discussion and Conclusions

This study has provided new evidence of household behaviour relative to a risk information program in a developing country, using primary household data collected in Kandal Province, Cambodia. Our results provide specific varification of a testing and education program related to arsenic risk but also allows significant broader conclusions to drawn which are important for further risk mitigation programs in developing countries. .

The headline result of this study is that 86% of households who have been informed that their drinking water had high levels of arsenic have switched to a new source. This is substantially higher than reports of testing campaigns in Bangladesh where previous studies have found average switching rates of 60% [*Madajewicz et al.*, 2007]. We caveat this result however given that we did not have a control group so as to assign this full proportion of change to the information and testing campaign, which the *Madajewicz et al.* [2007] study used. We note that there was a proportion of households who had already switched prior to being informed of the arsenic issue.

Furthermore taste and appearance of the original source appear to be a critical factor for switching which indicates that arsenic risk is not the only pertinent issue to a household's water source choice decision. That 86% have changed their drinking water source however is encouraging and one policy suggestion given our results is that a focus on taste and appearance of alternative water supplies may help encourage further switching behaviour and limit exposure to arsenic risk further.

The overall water situation in Cambodia is complex with different sources used for different uses and in different seasons. We have focused on drinking water and some of the key aspects of the decision to switch and the attributes of the water sources used.

Water is being used from a variety of different sources with paid sources (piped water and water vendors) forming a significant proportion of new sources which households have switched to. This reveals that people are willing to allocate a proportion of their income to mitigating their exposure to arsenic. Households which have switched to using water from a vendor, on average, have increased their treatment of water which illustrates the general view that groundwater is safer than surface water, because water from a vendor is generally surface water.

Through the estimation of a multinomial logit model we can see that the original water source characteristics play a significant role in switching to water vendor supply. Specifically a poor taste and appearance of the original water source increases the probability of the household switching their drinking water source, relative to the base category of not switching. Moreover larger households are less likely to use either piped water or surface water perhaps due to the increased cost, in terms of time, money or effort that collecting water for a large number of people entails.

Estimating OLS regression and logit models of the key water characteristics allows analysis of how these characteristics are affected by the household's decision to change their source. Overall we find that expenditure on water roughly doubles for those households who decide to change their source. Time spent collecting water however reduces by roughly half due to the switch to sources such as water deliveries or rain water harvesting.

In conclusion the education and testing program conducted by RDI has been largely successful in convincing households to switch away from tubewell sources if they test positive for high arsenic and in switching the households' have revealed their preferences fore reduced arsenic water.

Our results, although they relate to a specific project can be used to provide more general results for risk mitigation and drinking water improvement programs in developing countries. Importantly the attributes of the water source and aesthetic attributes in particular are significant in promoting use of improved water sources. Households may be reluctant to switch if they perceive that the taste of the water from the improved source is inferior relative to their current source.

Households are able to reduce their exposure to risks through increased expenditures. However these expenditures allows the household to gain other utility increasing attributes, such as reducing the amount of time they spend collecting water. This ’joint production’ issue limits the applicability of expenditure estimates to be utilised as welfare measures however it illustrates the complex nature of water source choices in rural developing countries. These issues are often simplified and abstracted in many empirical studies which biases the results and limits the generality of conclusions which they draw.

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6. Conclusions and Discussion

6.1. Conclusions

This thesis has examined the economic impact of arsenic contaminated drinking water in Kandal Province, Cambodia. Stated and revealed preference techniques have been utilised to estimate household WTP for water with reduced arsenic concentrations. We have presented the results from discrete choice experiments (DCEs) along with results from a household survey conducted to record household behavioural responses to an arsenic testing and education program. We have contributed to the literature in several distinct areas: the role of economic values in assessing drinking water standards, the role of payment vehicles in DCEs and the choice of drinking water sources. This research has been based upon primary data collected by the author following a period of fieldwork in May 2013.

In this section we highlight the main results from each section and provide an overarching discussion with conclusions. As such this section is as follows: Sections 6.2-6.4 provides invidual chapter conclusions. Section 6.5 provides a short discussion of overarching concepts and conclusions.

6.2. Paper 1: “Arsenic in Drinking Water: Willingness To Pay, Preference Heterogeneity and Drinking Water Standards in Cambodia.”

In this paper we have estimated economic values of reduced arsenic concentrations in drinking water, using a DCE, amongst the arsenic-informed rural poor in Kandal Province, Cambodia, a province whose rural population is heavily exposed to arsenic contaminated groundwater. Our purpose for doing so was due to 3 motivations: Firstly to estimate the economic benefits of possible remediation work conducted by local and international NGOs. Secondly to use these estimates to comment on the appropriatness, or otherwise, of the current drinking water standards. Finally, previous stated and revealed preference studies have used the dichotomy of safe/unsafe, set by the relevant government, to evaluate household preferences for arsenic risk reductions. By conducting a detailed DCE we were able to address the more fundamental issue of risk preferences, tolerance levels and WTP. This type of analysis is under-represented in the current literature.

The estimation of scale extended latent class models indicates that there are significant differences in choice consistency within the sample. Failing to account for these differences, i.e. by estimation of a homoscedastic latent class model that is typical within the literature, is likely to provide biased preference estimates, since the scale term is confounded with those preference estimates. Controlling for these scale differences, we have identified 3 latent preference segments whose valuations of alternative thresholds of water contamination differ markedly.

Each of the 3 segments exhibit scope sensitivity regarding the risk levels presented. However, there exists variation in the pattern of diminishing marginal utility regarding reduced arsenic concentrations and associated risks. A 4th segment displayed lexicographic preferences, evident from serial selection of the highest-price option, which prevented the estimation of marginal risk-cost trade-offs. Anecdotal evidence suggests that this behaviour is an artifact of the experimental process rather than representing likely market behaviour. Two scale segments were identified which constituted those who were more and those who were less consistent in their choice behaviour.

Our results suggest that a lower permissible limit for arsenic in drinking water may better represent the preferences of households in Kandal province, Cambodia. A lower permissible limit of arsenic in drinking water would set a more appropriate framework for NGO and government-led education and remediation programs. The lower limit would result in more households receiving a positive test for unsafe levels of arsenic which better matches the stated preferences of households for lower risk exposure. Preferences were found to be heterogeneous which makes application of a representative standard more complicated.

Due to the technologies of arsenic testing, a lower arsenic standard may require laboratory analysis of water samples to more accurately detect the lower level of arsenic. Any decrease in the current standard would substantially increase the costs and the logistical complexity of arsenic remediation strategies. We conclud the paper, in agreement with *Smith and Smith* [2004], that substantial care must be taken when setting drinking water standards and that those households with the highest concentrations of arsenic should be targeted first by remediation and education strategies.

6.3. Paper 2: "Valuation in Developing Countries: Willingness to Work vs. Willingness to Pay."

Whilst planning and conducting the DCE analysis of arsenic risk preferences in Cambodia it became clear that there was a significant gap in the literature related to using money payment vehicles in rural areas of developing countries. Specifically there was a lack of research focused on the effect of payment vehicles on choice behaviour and protest votes. In order to address this gap we conducted a split sample DCE, with half the respondents receiving a DCE with money as the payment vehicle whilst the other half were presented with a DCE with labour contributions as the payment vehicle.

From this study we present 3 novel findings: Firstly we find that the respondents' inferred value of time is similar to the market value (wage rates) of the labour tasks to be undertaken. This result is different from all other previous studies which find a shadow value of time which is significantly lower than the market wage rate. We suggest that our finding is due to a knowledge and acceptance of the market value of time by the respondents and an indication of functioning labour markets. Although our study site was a rural area it was situated near a highway which allows access to the capital city and thus access to markets. We find that willingness to pay is 10,298 riels/month and willingness to work is 7.76hrs/month for a marginal reduction (1/500) in lifetime cancer risk from arsenic consumption.

Second we find no differences in marginal utility estimates (parameters from the choice models) between respondents who were presented with a monetary or labour payment vehicle. We test this through pooling the data sets by using local market wage rates. We then construct interaction variables to account for any utility differences and test the statistical significance of the parameters. We find no difference in terms of choice consistency or marginal utility between those individuals offered either a money or labour payment vehicle. This interesting finding suggests that households are rational in their allocation of money and time.

Finally, through the estimation of latent class models we find that levels of attribute non-attendance (ANA) towards the payment vehicle is also stable between the 2 payment vehicles.

We conclude that in the context of our study site and subject matter, the use of labour contributions, rather than money, to estimate welfare values offers little additional benefit and entails further difficulties regarding the analysis of the value of time. This provides some support for the use of WTP in rural developing areas.

6.4. Paper 3: "Arsenic Testing and Household Drinking Water: The Determinants of Water Source Switching Behaviour in Kandal Province, Cambodia."

This paper provides new evidence of household behaviour relative to a risk information program in a developing country, using primary household data collected in Kandal Province, Cambodia. Our results provide specific verification of a testing and education program related to arsenic risk but also allow significant broader conclusions to drawn which are important for further risk mitigation programs in developing countries. .

We find that 86% of households who have been informed that their drinking water had high levels of arsenic have switched to a new source. This is substantially higher than reports of testing campaigns in Bangladesh where previous studies have found average switching rates of 60% [*Madajewicz et al.*, 2007].

Taste and appearance of the original source appear to be critical factors for switching which indicates that arsenic risk is not the only pertinent issue related to a household's water source choice decision. These factors were identified earlier in a piloting study which allowed their inclusion in the DCE studies. That 86% have changed their drinking water source could suggest that there is a high degree of trust in the NGO carrying out the testing campaign, given that arsenic is undetectable through consumption or it could be due to other factors such as the relative abundance of alternative water sources. A key policy implication given our results is that a focus on taste and appearance of alternative water supplies may facilitate further switching behaviour and limit exposure to arsenic risk further.

Water use patterns in Cambodia are complex, with different sources used for different purposes and in different seasons. Water is being used from a variety of different sources, with paid sources (piped water and water vendors) forming a significant proportion of new sources to which households have switched. This reveals that people are willing to allocate a proportion of their income to mitigating their exposure to arsenic.

Through the estimation of a multinomial logit model we can see that the original water source characteristics play a significant role in switching to water vendor supply. Specifically a poor taste and appearance of the original water source increases the probability of a household switching their drinking water source, relative to the base category of not switching. Moreover larger households are less likely to use either piped water or surface water perhaps due to the increased cost, in terms of time, money or effort that collecting water for a large number of people entails.

Estimating OLS regression and logit models of the key water characteristics allows analysis of how these characteristics are affected by the household's decision to change their source. Overall we find that expenditure on water increases by a factor of 4 (2,000 riels/week to 8,000 riels/week) for those households who decide to change their source. Time spent collecting water however reduces by roughly half (40 mins/day down to 20 mins/day) due to the switch to sources such as water deliveries or rain water harvesting.

In conclusion the education and testing program conducted by RDI has been largely successful in convincing households to switch away from tubewell sources if they test positive for high arsenic and in switching the households' have revealed their preferences for reduced arsenic water. However, in doing so the household may gain other utility increasing attributes, such as reducing the amount of time they spend collecting water. This 'joint production' issue limits the applicability of expenditure estimates to be utilised as welfare measures however it illustrates the complex nature of water source choices in rural developing countries. These issues are often over-simplified and abstracted in many empirical studies which could bias the results and limits the generality of conclusions which they draw.

6.5. Discussion

Overall this thesis has shown that there are strong preferences for reduced arsenic drinking water sources in Kandal Province, Camboida. These are exhibited in both the responses to hypothetical choice questions as well as actual revealed behaviours with the vast majority of households switching their water source, upon a positive test for “high” arsenic, sometimes at great expense or inconvenience.

The predominant sub-theme that runs throughout the thesis is the role of both time and money in expressing the preferences of rural households. As we have argued, time is perhaps the most important asset of an agricultural household given the nature of agricultural production and self-employment on the farm. Working time is reasonably flexible which allows the household to spend time collecting water or potentially engaging in labour activities to acquire improved water rather than needing to pay for the water from a vendor. Upon being informed that their water source is dangerous to drink many households have switched to water provided by a vendor which simultaneously increases the expenditures needed to access water whilst reducing their time requirements.

This is particularly interesting given our analysis of payment vehicles in DCEs. In Chapter 4 we estimated an average shadow wage rate of 1,327 riels/hour. In Chapter 5 however we note that on average expenditure increases by 6,000 riels/week whilst time spent collecting water decreases by 2.3 hrs/week. Thus potentially some of this expenditure, rather than being spent solely on risk reductions, can be seen as a payment for reductions in time spent collecting water. Our results suggests that 3052.10 Riels/week (1,327x2.3) or roughly half of the total expenditure can be accounted in this way.

The first contribution made by the thesis is the estimation of WTP value for arsenic risk reduction which allowed us to suggest that current drinking water limits are too low. We further extend our results by accounting for preference heteregeneity and choice consistency. These results suggests that preferences are complex, yet for two of the three estimated latent segments WTP estimates suggests that a lower permitted arsenic level, and associated mitigation strategies, would most appropriately represent their preferences for reduced arsenic water.

Through the conduct of a split sample study we are able to comment on the appropriatness of WTP measures, based on money income, in rural areas of developing countries, against a labour contibutions alternative (WTW). We find that both vehicles offer stable results in terms of choice behaviour (marginal utilities), attribute non-attendance and the ultimate conclusions drawn from both studies. This important result provides some grounds for conducting stated preference studies based on either WTP or WTW measures.

Finally we are able to offer a range of policy recommendations for helping to mitigate arsenic risk. Firstly the permissible limit for arsenic in drinking water could be lowered to represent preferences for reduced arsenic water. Secondly water switching behaviour, although complex, seems to be driven by aesthetic attributes of water and time issues. Thus these factors should be considered by future mitigation programs. For instances alternative sources which offer time savings and pleasant tasting water will likely lead to a rise in switching rates.

The values estimated in the first empirical paper could be used for future economic evaluation of potential arsenic mitigation policies. The values represent the WTP of households to reduce their exposure to arsenic related illnesses. Before utilising these values however it is important to understand the limitations of the methods used and the study conducted. Firstly the study was based in one province of Cambodia, whilst the arsenic problem exists in other parts of the country too. As such further work would be warranted to examine preferences for reduced arsenic risk in other parts of

Cambodia. Alternatively a benefits transfer approach could be taken to adjust the values estimated in Kandal to other parts of country, by the different socio-democratic characteristics.

Secondly further work is warranted to investigate household decision making and water source choice. Our choice experiment targeted the member of the household which was in charge of the water source choice decision. In reality, decision making is likely to be a deliberative process between members of the family. Furthermore peer effects emanating from neighbours' water decisions are also likely to have an influence decision making. These factors were not explicitly taken into consideration in this study.

Despite these issues we believe that the results reported in this thesis are of high quality. The topic which we considered was challenging given the need for overseas fieldwork with the inherent cultural, logistical and language barriers that we faced. Further the use of nonmarket valuation tools in rural areas of developing countries is potentially biased given low money incomes and other methods for facilitating the exchange of goods and services. We explicitly examined this issue in the second empirical chapter and achieved novel results.

6.6. References

Madajewicz, M., A. Pfaff, A. van Geen, J. Graziano, I. Hussein, H. Momotaj, R. Sylvi, and H. Ahsan, Can information alone change behavior? Response to arsenic contamination of groundwater in Bangladesh, Journal of Development Economics, 84 (2), 731–754.

Smith, A. H., and M. M. H. Smith, Arsenic drinking water regulations in developing countries with extensive exposure, Toxicology, 198 (1-3), 39–44.

A. Questionnaire used for Fieldwork

Cambodian Household Survey

Part A. Household and Household Members

1. Village Name		6. Date of Interview	
2. Commune Name		7. Time of Interview	
3. Province		8. Interviewer Name	
4. Household Co-ordinates		9. Previously answered questionnaire?	
5. Number of people in house		10. If yes, roughly what date?	

	Household member	(A) Age	(B) Gender (M/F)	(C) Years of Education	(D) Can read a newspaper? (Y/N)	(E) Can write? (Y/N)	(F) Respondent? (x)
11.	1						
12	2						
13.	3						
14.	4						
15.	5						
16.	6						
17.	7						
18.	8						
19.	9						
20.	10						

Part B. Health of Household Members

	Household member (same as previous)	(A) How would you say their health is? 1 = Very Good 2 = Good 3 = Average 4 = Bad 5 = Very Bad 6 = Don't Know	(B) Compared with others the same age, would you say that their health is…. 1 = Much Better 2 = Somewhat Better 3 = About the Same 4 = Somewhat Worse 5 = Much Worse 6 = Don't Know	(C) In the past year has (name) suffered from a period of illness greater than two weeks? (Yes/No)	(D) Type of illness, if known? (See Code Book)	(E) Roughly how many work days or school has (name) lost due to illness in the past year? Or looking after sick relative? (If not sure of year please ask about last month and make note)	(F) Roughly how much is spent on medicine or treatment per month for (name)? (Riels)	(G) On average how many cigarettes does (name) smoke a day?	(H) Does (name) use a mosquito net at night? (Yes/No)
21.	1								
22.	2								
23.	3								
24.	4								
25.	5								
26.	6								
27.	7								
28.	8								
29.	9								
30.	10								

PART C. Drinking Water, Firewood and Sanitation

31. Have you heard of arsenic? (Yes/No)	
32. Have you heard about arsenic before RDI education?	
33. What kinds of water might have arsenic?	
34. Can arsenic cause potentially fatal health	

problems? (Yes/No)				
35. Do you know anyone who is ill from arsenic poisoning? (Yes/No)				
36. Have you had your water tested for arsenic? (Yes/No)	(If No, go to Q41)			
37. If, yes, when was the test?				
38. Was arsenic found in your water? (Yes/No)				
39. Have you changed water source as a result of test result? (Yes/No)				
40. If not, why?	(A) Can't afford	(B) Don't trust test results	(C) Don't like large water containers	(D) Other: Specify

	Before Arsenic Test						After Arsenic Test					
	Wet Season			**Dry Season**			**Wet Season**			**Dry Season**		
	41	42	43	44	45	46	47	48	49	50	51	52
A) Source (See Code Book)	1)	2)	3)	1)	2)	3)	1)	2)	3)	1)	2)	3)
B) Coordinates												
C) Distance from house? (m)												
D) Money spent on water per week? (Riel)												
E) Daily time spent collecting water from source? (mins)												
F) Reliability of Source? 1 - Always Available, 2 - Normally Available, 3 - Often not Available												
F) Do you believe the water is safe to drink without treatment? (Yes/No)												
K) Has water been tested for arsenic? (If yes, add a tick if tested positive or a cross if tested negative)												

G) Appearance of water? 1-Clear, 2-Cloudy, 3-Dirty/Visible particulate												
H) Taste of water? 1-Very Good 2-Good 3-Average 4-Bad 5-Very Bad												
I) Smell of water? 1-Doesn't smell 2-Smells bad 3-Smells OK												
J) Use water source for irrigation of crops? (Yes/No)												
K) Use water for bathing/washing/cleaning? (Yes/No)												
L) When you have used this water, do you treat it before drinking? 1-Yes, always, 2 - Sometimes, 3-Never												
M) Have you purchased a new large container to store this water? If yes how much did it cost? (Riels) and when purchased?												
N) If you have installed a piped connection, what was the initial cost? (Riels) and when purchased?												

53. If you ever treat you drinking water, which treatment method do you use most often?	(A) Boiling	(B) Filtration	(C) Chemicals	(D) Other: Specify	
54. If filtration is used, which type?	(A) Clay Pot	(B) Korean	(C) Other: Specify		
55. Have you ever owned a filter?	(A) Yes, still have	(B) Yes, but not now	(C) Never		
56. If yes, roughly how much did you pay?	(Riels/USD)				
57. If you have owned a filter but currently do not use one, why?	(A) Broken	(B) Takes too long	(C) Water doesn't taste nice	(D) Other: Specify	
58. If you have never owned a water filter, what best describes why?	(A) Never heard about filters	(B) Don't trust	(C) Cannot afford	(D) Prefer other methods of treating water	(E) Other: Specify
59. How do you store water?	(A) Pieng Jar	(B) Cement Rings	(C) Earth (Sphere) Tank	(D) Plastic	(E) Other: Specify
60. If jar, how often do you keep it covered?	(A) Always	(B) Mostly (>50%)	(C) Sometimes (<50%)	(D) Never	
61. How often do you clean the jar?	Month/Year				
62. On average, how much money do you spend per week on firewood? (Riels)	(A) Dry Season	(B) Wet Season			
63. On average, how much time do you spend	(A) Dry Season	(B) Wet Season			

collecting firewood per week? (mins)		
64. What trees do you collect wood from?		
65. Which is your preferred type of wood used for firewood?		
66. How much of firewood is used for boiling drinking water? (e.g. 50%)		

67		
68		
69		

PART D. Choice Experiment (USE BOOKLET)

70. Booklet						
71. Answers	**Choice 1:**	**Choice 2:**	**Choice 3:**	**Choice 4:**	**Choice 5:**	**Choice 6:**

Part E. Wage and Self-Employment Activities

72. In the past month has any family members been in employed labour?	(A) Yes	(B) No (Proceed to question 82)

	(A) What trade or Industry?	(B) On average how many hours per week is worked in this industry?	(C) How much Riels per week is income in this industry?	(D) How many months have you been employed in this industry?	(E) Have you received payments in food or goods? (Yes/No)
73					
74					
75					
76					
77					
78					
79					
80					
81					

Part F. Agricultural Production

82. Does the household engage in crop production?	(A) Yes	(B) No (Proceed to Q117)

	(A) Crop Name (Code Book)	(B) Unit of Quantity (Code Book)	(C) Amount harvested in past year?	(D) Sales price (Riels per unit)	(E) Amount Sold?
83.					
84.					
85.					
86.					
87.					

88.					
89.					
90.					

91. On Average how many hours a week do family members work tending crops?	(A) Dry Season	(B) Wet Season	
92. On average, how many household members are involved?	(A) Dry Season	(B) Wet Season	
93. On average, how many non-household members work tending your crops?	(A) Dry Season	(B) Wet Season	
94. How are these people paid?	(A) Cash	(B) In Kind/For Goods	(C) Work Exchange

95. Number of fields used in crop production?

	(A) Unit of Area (Code Book)	**(B) Area**	**(C) Is plot irrigated?**	**(D) What crop is grown on plot? (Code Book)**
96.				
97.				
98.				
99.				
100.				
101.				
102.				

	Asset	**Number Owned**
103	**Tractor**	
104	**Bulldozer**	
105	**Plough**	
106	**Threshing Machine**	

107	**Harrow/Rake/Spade/Axe/Hoe**	
108	**Semi-Tractor (Kou Yon)**	
109	**Cart (Pulled by animal)**	

	How much was pent (in Riels) on the following items in the past year?	**(A) Dry Season**	**(B) Wet Season**
110	**Seeds**		
111	**Fertiliser**		
112	**Pesticides**		
113	**Crop Transport**		
114	**Irrigation Charges**		

115. Do you own any livestock?	**(A) Yes**	**(B) No (End of Questionnaire)**
116. How many hours per week does your family spend working with livestock?		

		(A) Quantity Owned?	**(B) Estimated current value of stock (sales value)? (in Riels)**
117.	**Cattle**		
118.	**Buffalo**		
119.	**Horses**		
120.	**Pig**		
121.	**Sheep**		
122.	**Goats**		
123.	**Chickens**		
124.	**Ducks**		
125.	**Quail**		
126.	**Other**		

	How much have you spent on the following in the past 12 months?	**Riels**
127.	**Feed**	
128.	**Transport**	
129.	**Veterinary or Medicine Costs**	

End of Questionnaire