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*Original Contribution*

# Economic Feasibility Study for Improving Drinking Water Quality: A Case Study of Arsenic Contamination in Rural Argentina

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**Abstract:** Economic studies are essential in evaluating the potential external investment support and/or internal tariffs available to improve drinking water quality. Cost–benefit analysis (CBA) is a useful tool to assess the economic feasibility of such interventions, i.e. to take some form of action to improve the drinking water quality. CBA should involve the market and non-market effects associated with the intervention. An economic framework was proposed in this study, which estimated the health avoided costs and the environmental benefits for the net present value of reducing the pollutant concentrations in drinking water. We conducted an empirical application to assess the economic feasibility of removing arsenic from water in a rural area of Argentina. Four small-scale methods were evaluated in our study. The results indicated that the inclusion of non-market benefits was integral to supporting investment projects. In addition, the application of the proposed framework will provide water authorities with more complete information for the decision-making process.

**Keywords:** arsenic, drinking water, cost–benefit analysis, environmental benefits, health effects, non-market benefits

## INTRODUCTION

Water is a key resource in sustainable development and poverty eradication, and it is essential for human health. Therefore, access to safe water is a human right (UNDP 2006). Globally, improved[1] drinking water sources have reached high levels, with 89% of the world population and 86% of people in developing regions obtaining drinking water from safe sources. Even so, 780 million people

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[1]According to the UNDP (2012), improved sources are a piped household water connection located inside the user's dwelling, plot, or yard, public taps or standpipes, tube wells or boreholes, protected dug wells, protected springs, or rainwater collection. Unimproved drinking water sources are unprotected dug wells, unprotected springs, carts with small tanks/drums, surface water (rivers, dams, lakes, ponds, streams, canals, irrigation channels), and bottled water.

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remain without the right to safe water; for example, in 2010, the UNDP (2012) found that more than one-tenth of the global population relied on unimproved drinking water sources.

Developing countries are focusing their efforts on increasing the access to safe drinking water, and in developed countries the primary challenge is to improve the quality of drinking water. Consequently, most countries have regulations to protect human health from the adverse effects of contaminated water intended for human consumption. For example, the European Directive 98/83/EC sets the drinking quality standards at the tap and maintains a general obligation that drinking water must be wholesome and clean. In the United States (US), the Safe Drinking Water Act (SDWA) places strict limits on approximately 90 chemicals and contaminants in community drinking water systems.

Rational decision making involves the adoption of criteria based on the policy, project and/or intervention, which are judged appropriate for the region, and subsequently performing an appraisal of how alternative options compare (Pearce et al. 2012). Under the current global economic crisis, an economic investment criterion has become essential to support improvements in drinking water quality. While politics typically determines which programmes are implemented, a need for consistent economic standards to review and establish competitive water interventions should be considered in the current economic climate (Ward 2012). Although several tools are available to evaluate the economics of water programmes, cost–benefit analysis (CBA) is most widely applied to evaluate the economic feasibility of water programmes.

CBA is a useful tool in the decision-making process, since it proposes various decision rules: (i) an intervention is only feasible if the benefits are greater than the costs; (ii) if alternative options are available, the best option is the one with the highest net present value and (iii) time can be incorporated into the assessment through the use of discount rates.

Undertaking CBA of actions with environmental impacts is complex because many environmental resources (including most water resources) are public goods, and therefore, do not have a market that sets the price. However, improvements contribute to human well-being, and therefore, result in economic value. If these improvements are omitted from appraisals, the economic assessment will underestimate the intervention's benefits. Unlike financial analyses, CBA measures the overall welfare impact of interventions, including non-market benefits. Therefore, the use of this approach in government decisions is strongly recommended (Hutton et al. 2007).

Despite the significant amount of resources allocated to improving the access to and quality of drinking water, information about the economics of such interventions remains scarce. Most published studies have evaluated the economic feasibility of increased access to safe water in developing countries, but not water quality issues. Hutton et al. (2007) carried out a CBA of water supply and sanitation for the Millennium Development Goal to be achieved in 2015. Haller et al. (2007) conducted a cost-effectiveness analysis to assess the increased access to improved water supplies and sanitation facilities in 10 WHO sub-regions. Hunter et al. (2009) completed CBAs in rural populations of developing countries to provide non-market benefits for water interventions. However, the studies applied empirical data restricted to the prevention of infectious diarrhoea. Based on legal regulations, not only microbiological but also chemical contaminants must be monitored. As a consequence, other health effects resulting from water contaminants should be evaluated in an economic assessment, e.g. the increased risks of various cancers (EPA 2013). Furthermore, improving the water quality provides environmental benefits that should be integrated into feasibility studies. Otherwise, a systematic underestimation of the benefits can result.

Previous studies have modelled different intervention types to achieve specific objectives, but none have reported a systematic framework to assess the economic feasibility of interventions. Furthermore, most studies have focused on evaluating an increase in the access to drinking water, but not on an improvement in the drinking water quality. The primary objective of this study was to propose an economic feasibility framework to assess drinking water quality improvement, including market and non-market effects. An empirical application was conducted to evaluate the feasibility of reducing arsenic (As) contamination in a rural area of Argentina. Because several water treatment technologies are available to remove As from water, a scenario analysis based on the assessment of four technologies was performed. Moreover, and taking into account the fact that uncertainty and variability exist with all the variables used in CBA (Narrod et al. 2012), a sensitivity analysis was carried out by considering two discount rates, three correction factors for environmental benefits and a tolerance of ±10% in benefit values.

The proposed economic framework provides an enhanced understanding of improving the economics of drinking water quality. Therefore, stakeholders and planners are now equipped with knowledge for planning and decision-making processes. These results will also be valuable in selecting the most feasible water treatment when several alternatives are available.

## The Proposed Economic Framework

The proposed economic framework is an adaptation of the traditional CBA to assess specifically the economic feasibility of interventions aiming to improve the drinking water quality. The objective of CBA as a tool is to compare the economic feasibility of several scenarios, including the 'do-nothing' scenario, i.e. maintaining the current conditions. It is important to note that this approach considers externalities[2] associated with policy, i.e. the benefits and costs without a market value.

CBA's main premise considers that projects should only be commissioned when the benefits exceed the aggregate costs. Such an analysis methodology is based on the net present value (NPV) calculation for each of the available options or scenarios, as shown in Eq. (1):

$$\mathrm{NPV} = \sum_{t=1}^{T} \frac{\sum B_i - \sum C_i}{(1+r)^t} \tag{1}$$

where NPV is the net present value; $B_i$ is the value of the benefit item $i$; $C_i$ is the value of the cost item $i$; $r$ is the discount rate; $T$ is the time horizon of the intervention and $t$ is the time (in our case, years).

An intervention is economically feasible only if NPV $> 0$, i.e. the intervention benefits outweigh the costs. If NPV $< 0$, the costs exceed the benefits and the intervention is not economically feasible. Implementation may be justified by other factors, including social ones, but not from an economic point of view. The best economic options offer the greatest NPV. The second choice for selecting the best alternative within a group is to produce a benefit–cost ratio; the preferred option is the one with the highest benefit-to-cost ratio (Molinos-Senante et al. 2012).

The proposed economic framework is similar to a traditional CBA, since both are based on comparing the benefits and the costs of several alternatives to select the most suitable one from an economic point of view. A modification is that both the benefits are associated with the reduction of adverse human health effects and the positive environmental benefits of improving drinking water quality. Therefore, the proposed economic framework integrates market and non-market benefits.

The proposed framework to assess the economic feasibility of improving water quality is based on five main steps: (i) define the study area and the objective to be achieved; (ii) define the different scenarios to be evaluated; (iii) estimate the cost of each scenario; (iv) valuate the intervention benefits and (v) calculate the net present value of each scenario (Eq. 1), including uncertainty assessment. Figure 1 shows a scheme for the proposed framework. Subsequently, the main methodological issues of each step are described and the results for the case study are presented.

## Proposed Steps

### Step 1: Definition of the Study Area and the Objective to be Achieved

*The Study Area*

The case study focused on assessing the economic feasibility of reducing drinking water As concentrations in the Department of Union, Province of Cordoba, Argentina. The study area covers 11,182 km$^2$, and the total population in 2010 was 105,727 (INDEC 2012) (Fig. 2). The As groundwater concentration exceeds the WHO recommended drinking water guidelines and the Argentine national drinking water standard (Francisca and Carro Pérez 2009; Gomez et al. 2009). Pérez (2006) reported mean As concentrations of 1,200 μg l$^{-1}$ in groundwater samples from the study area. Efforts to resolve the problem on the medium and large scales have been successful, but very few interventions have been carried out in rural areas (Litter et al. 2012). In rural settings, the As concentrations are even higher in shallow aquifers, which are the only affordable drinking water resources available for most of the rural population (Bundschuh et al. 2004). The regional rural population still affected by As contamination consists of approximately 25,000 people (INDEC 2012).

*The Objective of the Intervention*

Water contamination by As is a worldwide problem, which is particularly prevalent in developing countries. Generally,

[2]Externality refers to any consequence (positive or negative) of a purchase or use decision by one set of parties on others who did not have a choice and whose interests were not taken into account.

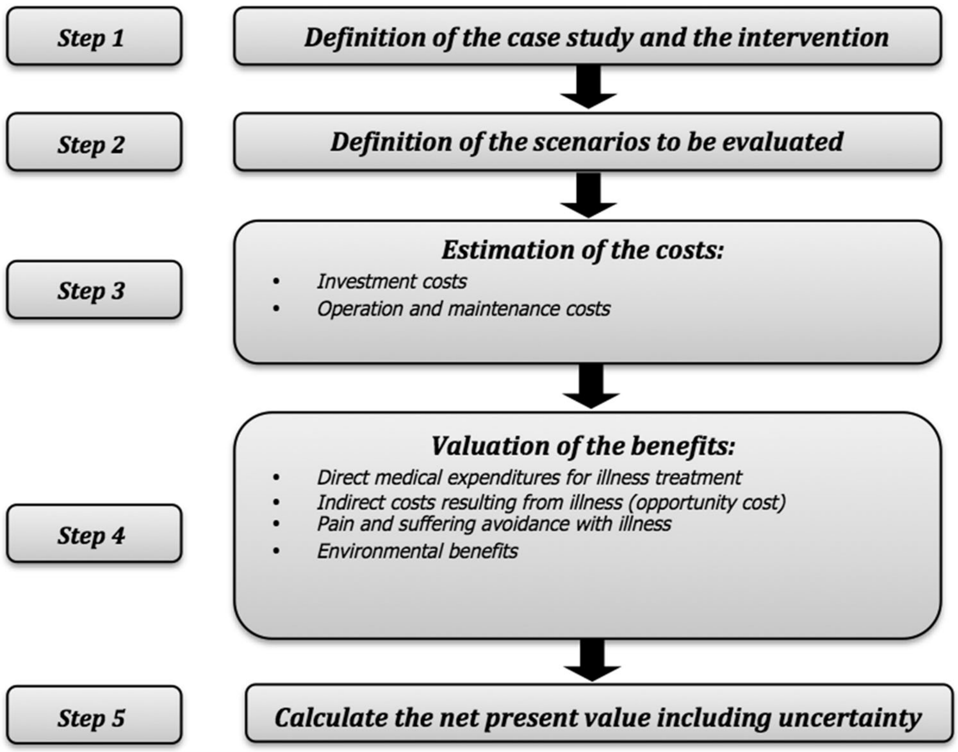


**Figure 1.** Steps of the proposed economic framework.

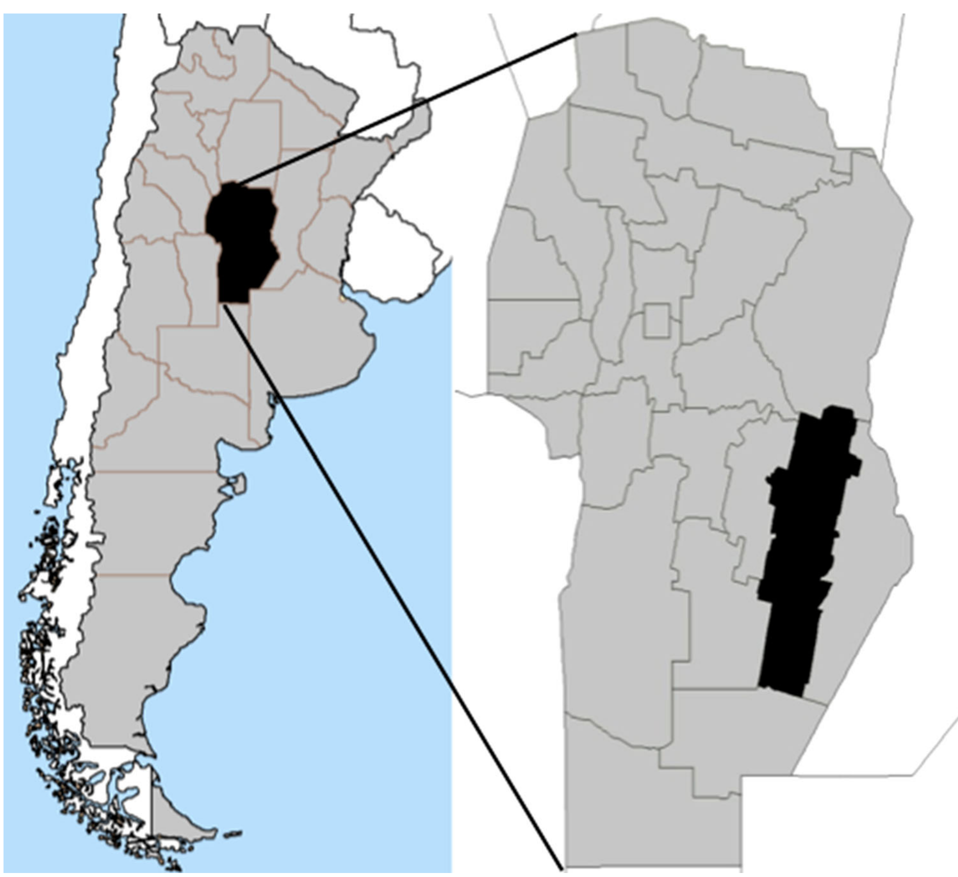

**Figure 2.** Location of the area of study: Argentina, Province of Cordoba and Department of Union.

the concentration of As in surface water is below 10 μg $l^{-1}$, which is the WHO recommended limit in drinking water. However, near various forms of anthropogenic disturbance, individual samples may range up to 5,000 μg $l^{-1}$. Regarding groundwater, the average As concentration is approximately 1–2 μg $l^{-1}$, although in areas with volcanic rock and sulphide mineral deposits, the concentration of As in groundwater can be as high as 3,000 μg $l^{-1}$ (WHO 2001). It is estimated that in Latin America alone, around 14 million people are exposed to the effects of drinking water contaminated by As. In Argentina, this value reaches approximately 4 million people (Litter and Bundschuh 2010). The affected region in Argentina is one of the largest in the world and covers approximately 1 million $km^2$ (Pérez 2006).

In this context, the main objective of the intervention proposed is to reduce the concentration of As in the drinking water to 50 μg $l^{-1}$ or less, since this is the maximum threshold permitted by the Argentine national drinking water standard.

## Step 2: Definition of the Scenarios to be Evaluated

Arsenic removal from water is challenging. Several treatment trains can be used to achieve the quality requirements; consequently, various scenarios should be assessed. The selection of the most suitable processes involves many possible options; however, the economic aspects are perhaps the most important factors (Litter et al. 2010). Because our case study focused on a rural region, conventional technologies, such as membrane processes or ion exchange resins that involve high costs, were not considered. However, emergent technologies suitable for implementation in rural populations were assessed. Litter et al. (2012) reported small-scale and household low-cost technologies to remove As from drinking water in isolated rural areas in Latin America, which served as scenarios for our study.

The basics of each technology (scenario) evaluated are as follows:

(1) Combined oxidation, adsorption and coagulation/flocculation methods
ALUFLOC is a sachet that contains an oxidant (chlorine), activated clays (acting as As adsorbents and/or ion exchangers) and a coagulant ($Al_2(SO_4)_3$ or $FeCl_3$) (Castro de Esparza 2006). It acts to remove As and other chemical pollutants in addition to disinfecting the water.

(2) Adsorption with low-cost materials
Geological materials have been demonstrated to be a viable solution for removing As in rural areas (Claesson and Fagerberg 2003). A basic requirement for low-cost technology is locally available materials. However, the study area does not support suitable remediation soils, i.e. with a low pH and high Fe content; therefore, materials from other regions would need to be imported. Claesson and Fagerberg (2003) performed experiments using an acidic Fe-rich laterite from Misiones, which is an Argentine province located around 1,200 km from Union Department.

(3) Biological methods
Many aquatic plants accumulate high levels of As from contaminated water. Consequently, constructed wetlands are an emerging alternative to conventional methods for removing As from water. These are an environmentally conscious and low-cost technology. Several authors have reported As removal processes in constructed wetlands (Lizama et al. 2011; Litter et al. 2012; among others). Moreover, due to land requirements, this technology is only suitable for small communities, similar to that of our case study.

(4) Zerovalent iron
As removal from water using zerovalent iron has been studied at the laboratory and bench scales. In both cases, the As concentrations in the effluent were lower than the legislation's requirements. The procedure is initially based on oxidative chlorination, followed by the use of small iron pieces for conditioning the flocs and ends with a filtration process. Several prototypes have been built and applied in different provinces of Argentina. The reactors were constructed with plastic materials (PVC and polypropylene), the operating materials were sodium hypochlorite, iron wool or filings, and the reactor does not use electric power (Litter et al. 2012).

## Step 3: Estimation of the Costs

### *Methodological Issues*

The cost of each water quality initiative must be estimated in performing CBA. The total costs involve the investment costs (ICs) and the operational and maintenance costs (OMCs) of the systems required to reduce water contamination.

The ICs include land, civil works, machinery and equipment. The OMCs involve the maintenance and

**Table 1.** The Investment Cost (IC), Operation and Maintenance Cost (OMC) and Total Equivalent Cost (TEC) for the Technologies Evaluated

| Technology | IC (€) | OMC (€ year$^{-1}$) | TEC (€) $r$ = 3% | TEC (€) $r$ = 7% |
|---|---|---|---|---|
| ALUFLOC | | 839,664 | 10,048,311 | 7,702,519 |
| Adsorption | | 930,750 | 11,138,342 | 8,538,081 |
| Biological methods | 3,528,700 | 372,190 | 7,982,721 | 6,942,923 |
| Zerovalent iron | 2,500,000 | 38,000 | 2,954,748 | 2,848,587 |

operation of the water treatment system, e.g. staff, energy and reagents. The total costs do not include any costs related to the water distribution network (either ICs or OMCs). The economic framework followed in this study addressed the monetary means to improve the drinking water quality and not increased access to safe water.

Similar to any investment project, the costs must be commensurate with the duration of the project. Hence, an appropriate discount rate and the project lifespan should be applied to discount the total costs to the present value. The total economic cost (TEC) reflects the ICs and annual OMCs discounted to the present value.

$$\mathrm{TEC} = \mathrm{IC} + \sum_{t=1}^{T} \frac{\mathrm{OMC}}{(1+r)^t} \qquad (2)$$

where TEC is the total economic cost (€); IC is the investment cost (€); OMC is the operational and maintenance cost (€ per year); $r$ is the discount rate; $T$ is the useful intervention duration and $t$ is the year.

The discount rate selection can play an important role, as a higher discount rate can result in a greater reduction of future to present values. Furthermore, it reflects the investment opportunity cost. Despite the importance of the discount rate value, economic theory has not established a 'correct value' (Kaplow and Weisbach 2011). The identification of the discount rate is challenging; therefore, the net present value should be computed for several possible discount rates. The United Kingdom Government uses a declining discount rate formula for long-term impacts (Treasury 2003).

Several approaches can be followed to assess the costs. The simplest is to carry out case-by-case analysis. Other methods, such as a Monte Carlo simulation model, will be useful if the number of agglomerations to be evaluated is high.

### *The Costs of Reducing the Arsenic Concentration in Drinking Water in the Case Study*

Table 1 summarises the total economic costs of the four scenarios evaluated. Based on previous studies (Almansa and Martínez-Paz 2011; Molinos-Senante et al. 2011b, 2012), two discount rates were applied to update the costs, and the intervention duration was assumed to last for 15 years. Substantial cost differences are illustrated. However, it should be noted that the objective of our study was not to rank the technologies based on cost. For this purpose, a cost-effectiveness analysis is more suitable. Our aim was to evaluate the economic feasibility of reducing As contamination in drinking water. However, due to the wide variety of methods available for removing As from water, we considered it necessary to assess the costs of four of the most appropriate regional technologies, and the choice of suitable technology is influenced by a combination of factors, including economic, social and technological influences.

Regarding the removal of As by coagulation/flocculation methods (scenario 1), Castro de Esparza (2006) confirmed that thirty sachets are required per family to treat the water consumed in one month. The sachet cost is approximately S/.0.916 (€0.28) (Castro de Esparza et al. 2005). The number of households in the study area is approximately 8,330; therefore, 2,998,800 sachets are required to treat the drinking water consumed annually. The total estimated costs are €839,664 per year. It is important to emphasise that this technology does not involve ICs, since the sachets are added directly to the contaminated water. Castro de Esparza et al. (2005) reported that if the ALUFLOC is sold in a solution in polyethylene containers instead of sachets, the costs could be reduced to S/.11.67–S/.16.25 per household month$^{-1}$. However, the individuals in the household should prepare the solution concentration, which may result in inaccuracy. Therefore, at least initially, the sachet option is likely to be more suitable. Based on these figures, the total costs in the study region were estimated at between €353,858 and €491,803 per year.

The second scenario evaluated was based on the removal of As by adsorption with low-cost materials. In this case, the costs for treating the water were estimated at

\$0.003 per litre (approximately €0.0023), including transport costs for 1,100 km. The water consumption was 20 l per person per day (WHO 2003), which included hand washing, food preparation, drinking and basic hygiene. Based on these figures, the total costs for our case study would be around €410,625 per year. Our estimates were derived from 2003 figures; therefore, because transportation costs have increased significantly in recent years, our figures should be updated. Montamat & Asociados (2012) reported that in the last two years, the fuel costs in Argentina have increased by nearly 70%. Therefore, assuming that the transportation costs from 2003 to the present have increased by 70%, the total cost for water treatment in our case study using this method would be €930,750 per year. This value verifies that if the material is unavailable locally, adsorption is not an affordable or competitive technology.

To estimate the IC and OMC of removing As from drinking water using constructed wetlands (scenario 3), the cost functions reported by Molinos-Senante et al. (2012) were employed. Hence, the IC was approximately €3,528,700, and the OMC was estimated at €372,190 per year.

Based on the material requirements and water consumption, we estimated that As removal by applying zerovalent iron (scenario 4) would involve approximately €2,500,000 for IC and €38,000 per year for OMC.

## Step 4: Valuation of the Benefits

Once the costs of each water treatment alternative have been evaluated, the next step is to quantify the benefits. Many potential benefits are associated with improving drinking water quality. The market benefits are easily identifiable and quantifiable; however, the non-market benefits are difficult to measure and require economic valuation methods. CBA must include the tangible and intangible benefits resulting from interventions.

The benefits from improving drinking water quality can be grouped into the following two main categories: (i) a reduction in adverse human health effects and (ii) positive environmental effects.

Hutton et al. (2007) reported that the greatest benefit associated with water supply interventions was the reduction in community time expenditure from accessing water associated with closer water sources. However, because our study focused on water quality, we did not address the benefits related to time saving. Puig and Freire (2011) assessed the economic savings in drinking water quality improvement from the distribution of bottled water. In our study, this assumption would involve double counting benefits, because we considered that the health avoided costs indicated that the population was drinking tap water instead of bottled water.

Both human and environmental benefits are subject to uncertainty. To overcome this limitation instead of considering just the mean value, certain tolerance levels for each benefit considered were defined. The introduction of tolerances allows the narrowing of uncertainty and the ability to make more reliable estimations of the benefits. According to Bonilla et al. (2004), tolerances of ±5, ±10 and ±25% are considered to be representative of a normal variation, a high variation and an extraordinary variation, respectively. Based on such criteria, in our study, the medium value of tolerance (±10% of the mean value) was defined. Hence, each benefit is not just a value but an interval of values.

### *The Methodology to Estimate the Benefits of Human Health Effects*

The customers' value in reduced adverse health effects risk included the following components: (i) direct medical expenditure on illness treatment; (ii) indirect costs resulting from illness, which included the value of time lost from work, decreased productivity, potential for demotion, money spent on caregivers and premature death—these factors representing the opportunity cost of illness (Calhoun and Bennett 2003) and (iii) pain and suffering avoidance associated with illness and the reduction in premature mortality risk.

The first two effects refer to market benefits. The effects can be estimated directly, i.e. these are health avoided costs gained by reducing drinking water contamination. However, the latter is a non-market benefit that can only be quantified by economic valuation methods.

The medical costs of caring for several illnesses associated with drinking unsafe water have been widely studied (Hutton et al. 2007; Gordon et al. 2011; Kim et al. 2012). The methodologies available to analyse medical costs are diverse and dependent on the type of illness. Therefore, to quantify the benefit type, the first step is to identify the most significant diseases associated with water contamination, and cooperation between regional economic and health professionals is vital.

In general, many factors affect patients' medical costs, including geographic region, sex and age, among others. All the available variables should be considered to characterise patient medical costs.

The value of productivity lost from illness or premature mortality results in substantial losses to society. Consequently, an economic assessment of interventions to improve water quality should integrate the avoided costs. For example, the indirect costs of cancer—due to several causes—in 2009 for Poland accounted for more than 0.8% of its GDP (Macioch and Hermanowski 2011). In Germany, the indirect cost of absenteeism in the workplace due to cancer (excluding premature mortality costs) was estimated at 0.7% of the GDP (Wilking and Jonsson 2005). These figures emphasise the importance of decreased productivity associated with disease for national economies.

Despite the existence of several approaches to estimate the indirect costs associated with illness, the human capital approach is the most widely applied. It relies on earnings as a measure of productivity. The methodology calculates the expected lifetime earnings that would have been made had disease or death been avoided (Bradley et al. 2008). Consequently, earnings are used as a proxy for loss of productivity. Wage and production structure data can be obtained from the national or regional bureaus of statistics.

Non-market benefits, the methods used to estimate the economic value of the risk reductions, are based on willingness to pay (WTP). Many studies have examined the WTP for reducing different types of risk, including air pollution (Roman et al. 2012), road safety (Hakes and Viscusi 2007) and accidents in the workplace (Tsai et al. 2011). The contributions to drinking water quality have been more limited, and have served to estimate the economic value of the health benefits derived from risk reductions associated with improving drinking water quality (Adamowick et al. 2011; Maddison et al. 2005).

The avoided costs from reducing negative health effects can be expressed as follows:

$$\mathrm{HAC} = \sum_{t=1}^{T} \frac{\left(\sum_{K=1}^{K} \mathrm{ME}_{kt} * n_{kt}\right) + \left(\sum_{J=1}^{J} \mathrm{AW}_{jt} * n_{jt}\right) + \left(\mathrm{WTPR}_{t} * n_{kt}\right)}{(1+r)^{t}} \tag{3}$$

where HAC are the health avoided costs (€); $\mathrm{ME}_k$ is the direct medical expenditure for one person to treat the illness $k$ (€ person$^{-1}$ year$^{-1}$); $n_k$ is the number of avoided cases of the illness $k$; $\mathrm{AW}_j$ is the average wage of the productive sector $j$ (€ person$^{-1}$ year$^{-1}$); $n_j$ is the number of avoided cases of the illness in the productive sector $j$; WTPR is the average WTP from risk reductions (€ person$^{-1}$ year$^{-1}$); $N$ is the total population of the area affected by the intervention; $r$ is the discount rate; $t$ is the year and $T$ is the intervention time horizon.

### *The Economic Value of the Human Health Effects of Reducing Arsenic Concentration in Drinking Water in the Case Study*

There is substantial epidemiological evidence linking elevated As concentrations in drinking water to human health problems (WHO 2001; Hopenhayn 2006). Most data detailing the health effects of As-contaminated drinking water come from a series of studies conducted in Taiwan, India, and Bangladesh. However, McClintock et al. (2012) reviewed the health effects from As exposure in Latin America. In the province of Cordoba, Argentina, Hopenhayn-Rich et al. (1996) found a strong association between As exposure and bladder and lung cancers. In addition, in the same study area, kidney cancer mortality was elevated in people who were exposed to high As concentrations in drinking water (Hopenhayn-Rich et al. 1998). The few studies published in Latin America addressing liver and skin cancer were not conducted in Cordoba. Therefore, due to individual risk factors attributable to metabolic variation in different geographic regions (Steinmaus et al. 2010), we did not include the latter types of cancer in our analysis.

Other chronic non-malignant effects traditionally associated with As exposure include skin lesions, cardiovascular disease and respiratory illness (Gómez 2008). However, McClintock et al. (2012) reported that chronic exposure symptoms were inconsistent among patients. Therefore, we excluded these health effects from our study.

Based on a review of published reports from our study area, it is clear that decreased As concentrations in drinking water will reduce the number of cases of liver, lung and kidney cancers.

The standardised mortality ratios (SMRs) for all ages combined in Union Department for lung, bladder and kidney cancers (Hopenhayn-Rich et al. 1996, 1998), as well as the expected deaths due to high As concentrations in drinking water for each cancer type, are provided in Table 2. These figures do not represent the total expected deaths, but the deaths associated with high As concentrations in drinking water. The values were calculated based on the cancer mortality rates in all of Argentina (Matos et al. 2003), and the Union Department SMRs were applied to each gender for the 2010 population. We multiplied by 15 to obtain the expected deaths for the intervention duration.

The total cost savings were calculated by multiplying the health service unit cost by the number of averted cases. Due to the absence of studies determining the medical costs of treating cancer in Argentina, data from the US were extrapolated to our study area. Chang et al. (2004) estimated the direct costs of seven malignant neoplasm types, including lung cancer. The specific costs for treating patients with bladder and kidney cancer were not evaluated; therefore, the mean total direct medical expenditure was used. The total estimated medical costs avoided by reducing the As concentrations in drinking water for our study area would be approximately €385,696 per year.

The indirect costs associated with illness and premature death were estimated based on productivity loss. Hopenhayn-Rich et al. (1998) did not detect age-specific SMR heterogeneity. Therefore, we assumed that the distribution of cancer cases by age in the study area was similar to that in all of Argentina. INDEC (2012) showed that the working age of the population ranges from 14 to 65 and the employment rate is 60%. Furthermore, our study was conducted in rural areas, and due to the absence of data, we determined that 100% of the population was employed in the primary sector. Based on these assumptions, we estimated that the productivity loss in the Union Department rural areas due to high As concentrations in drinking water amounted to approximately €168,291 per year.

Regarding health effects without market value, several studies have addressed how much money people are WTP for improved drinking water quality to reduce health risks in developed and low-income countries (Um et al. 2002; Dutta et al. 2005; Dupont and Jahan 2012). Ahmad et al. (2005) determined the WTP for access to arsenic-free drinking water in rural Bangladesh. Based on the contingent valuation survey data for 2,700 households, the WTP was estimated at US$2.4 per year per household. This figure represented 0.2% of the mean household income. Given the socioeconomic differences between the population of Bangladesh and that of Argentina and taking into account the recommendations suggested by Ready and Navrud (2006) to transfer benefits at the international level, it was considered more reliable to transfer an income percentage rather than a mean WTP value. Hence, based on the household number in the study area, and its average income, the value to reduce health risks was €57,477 per year.

Similar to the assessment costs, the benefits associated with human health improvements must be updated for the intervention duration (15 years) considering the two discount rates (3 and 7%). The economic value from reducing As concentrations in drinking water, as the result of positive human health effects, is shown in Table 3. The data indicate that 63% of the total HACs were due to direct avoidance of medical expenditure on treating cancer patients. The productivity loss represented 28% of the total avoided costs, while the non-market benefits, i.e. the value of risk reductions, comprised 9% of the HACs.

### *The Methodology to Estimate the Environmental Benefits*

The improvement of drinking water quality exhibits positive environmental effects, as water is ultimately reintroduced into the environment. However, an estimate requires the application of economic valuation methods. Consequently, economic theory has developed a wide range of techniques to value non-market goods, including placing a value on environmental intervention benefits.

Traditional valuation techniques are based on the demand approach. In the field that concerns us, stated preference methods are the most common (Bateman et al. 2006). These approaches are based on simulating a hypothetical market by surveying the processes. Consequently, the WTP for a particular good or service, or the willingness to accept (WTA) payment in exchange for bearing a particular loss, is estimated. The primary categories of stated preference methods are the contingent valuation method and choice modelling techniques (Fig. 3).

**Table 2.** Standardised Mortality Ratios (SMR) (Hopenhayn-Rich et al. 1996, 1998) and Expected Deaths Due to High Concentrations of As in Drinking Water in the Department of Union (Argentina)

| | Lung | | Bladder | | Kidney | |
|---|---|---|---|---|---|---|
| | SMR | Expected deaths | SMR | Expected deaths | SMR | Expected deaths |
| Male | 1.72 | 112 | 2.60 | 23 | 1.56 | 13 |
| Female | 2.29 | 30 | 2.87 | 5 | 1.76 | 6 |
| Total | | 142 | | 28 | | 19 |

**Table 3.** Health Avoided Costs Associated with a Reduction in the Concentration of As in the Area of Study

| Human health effect | Avoided costs (€ year$^{-1}$) | Avoided costs (€) $r$ = 3% | Avoided costs (€) $r$ = 7% |
|---|---|---|---|
| Direct medical costs avoided | 347,126–424,265 | 4,154,083–5,077,213 | 3,184,306–3,891,930 |
| Loss of productivity avoided | 151,462–185,120 | 1,812,554–2,215,344 | 1,389,411–1,698,169 |
| Risk health avoided | 51,729–63,225 | 619,048–756,614 | 474,530–579,982 |
| Total health avoided costs (HAC) | 550,318–672,610 | 6,585,684–8,049,170 | 5,048,247–6,170,080 |

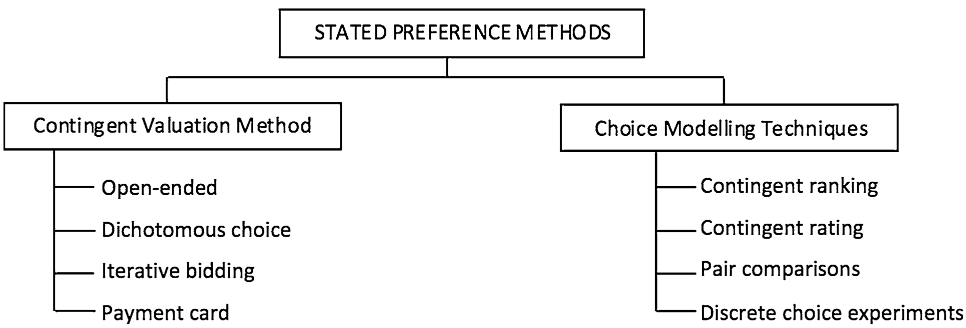


**Figure 3.** Scheme of the stated preference methods.

**Table 4.** Environmental Benefits Associated with a Reduction in the Concentration of As in the Area of Study Considering Three Correction Factors

| | Environmental benefits (€ year$^{-1}$) | Environmental benefits (€) $r$ = 3% | Environmental benefits (€) $r$ = 7% |
|---|---|---|---|
| Correction factor 0.3 | 136,633–166,995 | 1,635,092–1,998,446 | 1,253,378–1,531,906 |
| Correction factor 0.6 | 273,265–333,991 | 3,270,183–3,996,891 | 2,506,755–3,063,811 |
| Correction factor 0.9 | 409,898–500,986 | 4,905,275–5,995,337 | 3,760,133–4,595,718 |

Respondents are questioned directly regarding their WTP or WTA for increase or decrease in public goods in the contingent valuation method. Alternatively, the choice modelling techniques are based on ranking or rating a series of 'product profits' that characterise products with specific attribute levels (Pearce et al. 2002).[3]

It should be noted that most water contamination issues are due to anthropogenic sources. Therefore, improving the drinking water quality does not involve achieving the good ecological status of water bodies. Nonetheless, the WTP typically refers to this status. An overestimation of environmental benefits must, therefore, be avoided. The WTP must be multiplied by a correction factor that ranges from 0 to 1; 0 indicates that the intervention does not result in any environmental improvement and a correction factor of 1 indicates that a reduction in drinking water pollutant concentration achieves the good ecological status of the receiving water body. The correction factor is a tool to establish the environmental benefits to real improvement gained from the proposed intervention. Pulido-Velazquez et al. (2008) followed a similar approach.

Unless changes in the pollutant distribution in and out of a water body are quantitatively modelled and subsequently assessed, our analyses cannot determine the degree to which an intervention can contribute to improving the ecological conditions of a water body. Therefore, the correction factor selection is subjective and depends on the analyst's criteria.

Based on a distance function approach, Färe et al. (1993) developed an alternative methodology to value undesirable outputs resulting from activities with no market value; unlike traditional methods, it is based on cost production. Several empirical applications have been developed to estimate the shadow prices of pollutants removed from wastewater as a result of treatment (Hernández-Sancho et al. 2010; Molinos-Senante et al. 2011a). Contaminants extracted from wastewater are undesirable

[3]Many reports and books have been published describing these methods. For more information, see Alberini and Kahn (2006) or Carson (2011).

**Table 5.** Net Present Value (€) (Minimum and Maximum Value) for Each Technology Considering Two Discount Rates and Three Correction Factors for the Environmental Benefits for 15 Years

| NET PRESENT VALUE (€) | CF 0.3 | | CF 0.6 | | CF 0.9 | |
|---|---|---|---|---|---|---|
| | $r$ = 3% | $r$ = 7% | $r$ = 3% | $r$ = 7% | $r$ = 3% | $r$ = 7% |
| ALUFLOC | Min: −1,827,535 | Min: −1,400,894 | Min: −192,444 | Min: −147,517 | Min: 1,442,648 | Min: 1,105,861 |
| | Max: −695 | Max: −533 | Max: 1,997,750 | Max: 1,531,372 | Max: 3,996,196 | Max: 3,063,279 |
| Adsorption | Min: −2,917,566 | Min: −2,236,456 | Min: −1,282,475 | Min: −983,079 | Min: 352,617 | Min: 270,299 |
| | Max: −1,090,726 | Max: −836,095 | Max: 907,719 | Max: 8,538,081 | Max: 2,906,165 | Max: 2,227,717 |
| Biological methods | Min: 238,055 | Min: 641,298 | Min: 1,873,146 | Min: 692,079 | Min: 3,508,238 | Min: 1,865,457 |
| | Max: 2,064,895 | Max: 759,063 | Max: 4,063,340 | Max: 6,942,923 | Max: 6,061,786 | Max: 3,822,875 |
| Zerovalent iron | Min: 5,266,028 | Min: 3,453,038 | Min: 6,901,119 | Min: 4,706,415 | Min: 8,536,211 | Min: 5,959,793 |
| | Max: 7,092,868 | Max: 4,853,399 | Max: 9,091,313 | Max: 2,848,587 | Max: 11,089,759 | Max: 7,917,211 |

outputs, i.e. if pollutants are released into the environment, they generate a negative impact. Therefore, the shadow prices of pollutants are considered as a proxy for the environmental benefits of wastewater treatment.

The same methodology can be applied to estimate the environmental benefits gained from improving the drinking water quality, since it is based on pollutant removal from water. However, it is important to emphasise that to apply this approach, detailed data regarding the relationship between operational costs and efficiency in pollutant removal are required. To date, studies that value the environmental benefits of drinking water have not been published. Therefore, we consider it more appropriate to follow the traditional methodology, i.e. WTP.

The environmental benefits gained from improving the water quality can be expressed as follows:

$$\mathrm{EB} = \sum_{t=1}^{T} \frac{(\mathrm{WTPE}_t * \mathrm{NH}_t)}{(1+r)^t} * \mathrm{CF} \tag{4}$$

where EB are the environmental benefits (€); WTPE is the average willingness to pay for improving the quality of water bodies (€/household year); NH is the number of households in the area affected by the intervention; CF is a correction factor ranging from 0 to 1; $r$ is the discount rate; $t$ is the year and $T$ is the intervention time horizon.

*The Economic Value of the Environmental Benefits of Reducing the Arsenic Concentration in Drinking Water in the Case Study*

Quantifying the environmental benefits using the stated preference methods requires complex and expensive survey processes; therefore, benefit transfer methods are gaining interest. Benefit transfer methods are based on estimates from the value of an environmental target 'policy' site using analyses conducted from another 'study' site (Hanley et al. 2006). Despite the procedure's reliability remaining under discussion in the literature (Johnston and Thomassin 2010), from a practical viewpoint it is not feasible to carry out an economic valuation exercise for each policy intervention.

Therefore, some sophisticated approaches have been developed to improve the accuracy of benefit transfer, including the analysis of a spatial dimension in environmental valuation (Martin-Ortega et al. 2012) or the use of transferable value functions (Bateman et al. 2011). However, due to the increased complexity, in practical decision making, two simpler approaches are frequently employed (Bateman et al. 2011): (i) direct benefit transfer (DBT),

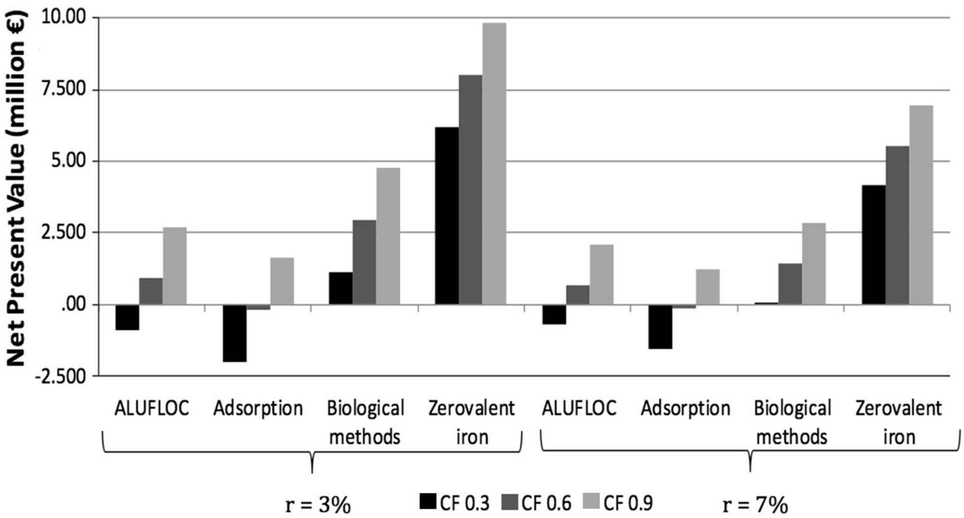


**Figure 4.** Net present value in millions of euros for the selected technologies based on mean values.

which is the simplest method, whereby the mean unit study site value is transferred to the policy site and (ii) the adjusted benefit transfer (ABT), which is based on the same concept as DBT, but makes distinctions between the two sites, e.g. income levels (Muthke and Holm-Mueller 2004).

The Water Framework Directive (Directive 2000/60/EU) emphasises the role of economic analyses in water resource management; therefore, most studies have been conducted in European countries that value environmental benefits to improve water quality. The DBT approach assumes that the attributes of the environmental good and local population preferences are identical at the two sites. Taking into account this drawback, to estimate the environmental benefits in our case study, it was considered more appropriate to use the empirical application developed by Sarmiento and Carignano (2011). The authors estimated the WTP to increase the water quality in the Dulce River in Santiago del Estero, Argentina. The mean WTP was US$6.75 per household month$^{-1}$. The transfer of this value to our case study determined an environmental benefit of €506,047 per year.

Three correction factors were considered to narrow the uncertainty associated with the economic value of environmental benefits. Moreover, as regards the health benefits, a tolerance level of ±10% of the mean value was defined. The values, including the two discount rates, are shown in Table 4.

### Step 5: Calculation of the Net Present Value

Considering all the costs and benefits associated with improving the quality of the drinking water (Eqs. 2, 3, 4), the NPV (Eq. 1) can be expressed as follows:

$$NPV = HAC + EB - TEC \qquad (5)$$

Based on Eq. (5), the NPV was calculated by comparing the total benefits derived from improving the drinking water quality with the total economic costs of each treatment evaluated. Detailed results based on interval values are gathered in Table 5, while Figure 4 shows the results based on average values.

The results indicated that zerovalent iron and biological methods, i.e. constructed wetlands, have a positive NPV in all the scenarios. Consequently, from an economic point of view, these treatments are always feasible. In fact, the zerovalent iron method is feasible considering only the market benefits, i.e. direct medical costs and loss of productivity avoided. According to the economic assessment, this treatment would be the most suitable for reducing As contamination in drinking water. On the other hand, adsorption processes might be the only feasible treatment if the correction factor is high, i.e. the economic viability of adsorption is contingent on the process resulting in high environmental quality improvement of water bodies. Finally, the ALUFLOC method is feasible if the environmental improvement is medium or high.

## Conclusions

Although access to safe drinking water is a human right, interventions to increase the quality of drinking water should be evaluated from an economic perspective to support stakeholders' investments. Since many undeveloped regions of the world can simply not afford to undertake these projects, it is essential to conduct some form of economic assessment to determine the feasibility of an intervention. CBA is a useful tool for that purpose.

It is well known that some of the benefits gained from improved water quality, such as environmental ones, have a non-market price. However, it is essential to include these benefits in economic assessments. Otherwise, the real benefits would be underestimated.

Consequently, we proposed an economic framework to assess the feasibility of interventions to increase drinking water quality, including market and non-market benefits. The costs involved in reducing water contamination included investment and operational and maintenance costs. In addition, the benefits can be classified into market benefits, such as direct medical expenditure on illness treatment and productivity loss avoided, and non-market benefits, which involve the reduction of premature mortality risk and environmental benefits. Therefore, the true intervention benefits and costs are incorporated into the net present value.

In order to illustrate the value of the proposed economic framework, an empirical application was conducted to assess the economic feasibility of reducing the As water concentrations in rural Argentina. We estimated the net present value from four technologies. To narrow the uncertainty, two discount rates, three correction factors for environmental benefits and a tolerance of ±10% in benefit values were considered. The results indicated that zerovalent iron and constructed wetlands are economically feasible systems, and investments to improve drinking water quality should support these approaches economically. On the other hand, ALUFLOC and adsorption processes are feasible depending on how the environmental benefits are incorporated into the CBA.

The application of CBA instead of financial analysis allows the incorporation of social benefits into the economic assessment. However, some caution should be exercised when employing CBA. First, economic valuation methods just provide a proxy for non-market benefits. Second, to narrow the uncertainty, several scenarios should be evaluated. Finally, it is a challenge to carry out an economic exercise for each intervention or project. Hence, benefit transfer methods are required, which add an extra source of uncertainty to the exercise. Despite these limitations, it is vital to include non-market benefits in economic feasibility studies to support investment projects with the goal of improving drinking water quality. Otherwise, these benefits are ignored, when it is well known that they make a substantial contribution to increasing human well-being.

Beyond the empirical application developed to address As contamination, the proposed economic framework is of special interest to authorities and stakeholders, since it provides a systematic tool to assess the economic feasibility of water interventions. Therefore, authorities are provided with enhanced information to support the decision-making process.

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## References

Adamowick W, Dupont D, Krupnick A, Zhanj J (2011) Valuation of cancer and microbial disease risk reductions in municipal drinking water: an analysis of risk context using multiple valuation methods. *Journal of Environmental Economics and Management* 61:213–226

Ahmad J, Goldar B, Misra S (2005) Value of arsenic-free drinking water to rural households in Bangladesh. *Journal of Environmental Management* 74:173–185

Alberini A, Kahn J (2006) *Handbook on Contingent Valuation*, Cheltenham, UK: Edward Elgar Publishing Limited

Almansa C, Martínez-Paz JM (2011) What weight should be assigned to future environmental impacts? A probabilistic cost benefit analysis using recent advances on discounting *Science of the Total Environment* 409:1305–1314

Bateman IJ, Brouwer R, Ferrini S, Schaafsma M, Barton DN, Dubgaard A, Hasler B, Hime S, Liekens I, Navrud S, De Nocker L, Sceponaviciute R, Semeniene D (2011) Making benefit transfers work: deriving and testing principles for value transfers for similar and dissimilar sites using a case study of the non-market benefits of water quality improvements across Europe. *Environmental and Resource Economics* 50:365–387

Bateman IJ, Cole MA, Georgiou S, Hadley DJ (2006) Comparing contingent valuation and contingent ranking: a case study considering the benefits of urban river water quality improvements. *Journal of Environmental Management* 79:221–231

Bonilla M, Casasús T, Medal A, Sala R (2004) An efficiency analysis with tolerance of the Spanish Port System. *International Journal of Transport Economics XXXI* 3:379–400

Bradley CJ, Yabroff KR, Dahman B, Fener EJ, Mariotto A, Brown ML (2008) Productivity costs of cancer mortality in the United

States: 2000–2020. *Journal of the National Cancer Institute* 100(24):1763–1770

Bundschuh J, Farías B, Martin R, Storniolo A, Bhattacharva P, Cortes J, Bonorino G, Albouy R (2004) Groundwater arsenic in the Chaco-Pampean Plain, Argentina: case study from Robles county, Santiago del Estero Province. *Applied Geochemistry* 19(2):231–243

Calhoun EA, Bennett CL (2003) Evaluating the total costs of cancer: the Northwestern University costs of Cancer Program. *Oncology* 17(1):109–114

Carson RT (2011) *Contingent Valuation. A Comprehensive Bibliography and History*, Cheltenham, UK: Edward Elgar Publishing Limited

Castro de Esparza ML (2006) Removal of arsenic from drinking water and soil bioremediation. *International Congress Natural Arsenic in Groundwaters of Latin America*, Mexico City, 20–24 June, 2006

Castro de Esparza ML, Noriega Pissani R, Wong Nonato M, Inamine Asato A (2005) *Estudio para el mejoramiento de la calidad del agua en pozos en zonas rurales de Puno, OPS*, Lima: CEPIS. www.bvsde.paho.org/bvsacd/cd53/puno-informe/informe.pdf (In Spanish). Accessed 29 May 2014

Chang S, Long SR, Kutikova L, Bowman L, Finley D, Crown WH, Bennett CL (2004) Estimating the cost of cancer: results on the basis of claims data analyses for cancer patients diagnosed with seven types of cancer during 1999 to 2000. *Journal of Clinical Oncology* 22(17):3524–3530

Claesson M, Fagerberg J (2003) Arsenic in groundwater of Santiago del Estero—sources, mobility patterns and remediation with natural materials. Master Thesis, Department of Land and Water Resources Engineering, KTH, Stockholm, Sweeden, TRITA-LWREX-03-05

Dupont DP, Jahan N (2012) Defensive spending on tap water substitutes: the value of reducing perceived health risks. *Journal of Water and Health* 10(1):56–68

Dutta V, Chander S, Srivastava L (2005) Public support for water supply improvements: empirical evidence from unplanned settlements of Delhi, India. *The Journal of Environment Development* 14:439–462

EPA (2013) Arsenic in drinking water web page. http://water.epa.gov/lawsregs/rulesregs/sdwa/arsenic/index.cfm. Accessed 29 May 2014

Färe R, Grosskopf S, Lovell CA, Yaisawarng S (1993) Derivation of shadow prices for undesirable outputs: a distance function approach. *Review of Economics and Statistics* 75(2):374–380

Francisca FM, Carro Pérez ME (2009) Assessment of natural arsenic in groundwater in Cordoba Province, Argentina. *Environmental Geochemistry and Health* 31(6):673–682

Gómez A (2008) Chronic arseniasis in El Zapote, Nicaragua. In: *Natural Arsenic in Groundwaters of Latin America*, Bundschuh J, Armienta MA, Birkle P, Bhattacharya P, Matschullat J, Mukherjee AB (editors), London: Taylor and Francis Group

Gomez ML, Blarasin MT, Martínez DE (2009) Arsenic and fluoride in a loess aquifer in the central area of Argentina. *Environmental Geology* 57(1):143–155

Gordon LG, Eckermann S, Hirst NG, Watson DI, Mayne GC, Fahey P, Whiteman DC (2011) Healthcare resource use and medical costs for the management of oesophageal cancer. *British Journal of Surgery* 98(11):1589–1598

Hakes JK, Viscusi WK (2007) Automobile seatbelt usage and the value of statistical life. *Southern Economic Journal* 73(3):659–676

Haller L, Hutton G, Bartram J (2007) Estimating the costs and health benefits of water and sanitation improvements at global level. *Journal of Water and Health* 5(4):467–480

Hanley N, Colombo S, Tinch D, Black A, Aftab A (2006) Estimating the benefits of water quality improvements under the water framework directive: are benefits transferable? *European Review of Agricultural Economics* 33(3):391–413

Hernández-Sancho F, Molinos-Senante M, Sala-Garrido R (2010) Economic valuation of environmental benefits from wastewater treatment processes: an empirical approach for Spain. *Science of the Total Environment* 408:953–957

HM Treasury (2003) *The Green Book Appraisal and Evaluation in Central Government*, London: TSO

Hopenhayn C (2006) Arsenic in drinking water: impact on human health. *Elements* 2:103–107

Hopenhayn-Rich C, Biggs ML, Fuchs A, Bergoglio R, Tello EE, Nicolli H, Smith AH (1996) Bladder cancer mortality associated with arsenic in drinking water in Argentina. *Epidemiology* 7(2):117–124

Hopenhayn-Rich C, Biggs ML, Fuchs A, Smith AH (1998) Lung and kidney cancer mortality associated with arsenic in drinking water in Córdoba, Argentina. *International Journal of Epidemiology* 27:561–569

Hunter RP, Pond K, Jagals P, Cameron J (2009) An assessment of the costs and benefits of interventions aimed at improving rural community water supplies in developed countries. *Science of the Total Environment* 407:3681–3685

Hutton G, Haller L, Bartram J (2007) Global cost–benefit analysis of water supply and sanitation interventions. *Journal of Water and Health* 5(4):481–501

INDEC (2012) Instituto Nacional de Estadística y Censos de Argentina. National Institute of Statistics and Census of Argentina. http://www.indec.gov.ar/ (In Spanish). Accessed 29 May 2014

Johnston RJ, Thomassin PJ (2010) Willingness to pay for water quality improvements in the United States and Canada: considering possibilities for international meta-analysis and benefits transfer. *Agricultural and Resource Economics Review* 39(1):114–131

Kaplow L, Weisbach D (2011) Discount rates, social judgments, individuals' risk preferences, and uncertainty. *Journal of Risk Uncertainty* 42(2):125–143

Kim JH, Kim SH, Joo JS, Lee KS (2012) Factors associated with medical cost among patients with terminal cancer in hospice units. *Journal of Palliative Care* 28(1):5–12

Litter MI, Alarcón-Herrera MT, Arenas MJ, Armienta M, Avilés M, Cáceres RE, Cipriani HN, Cornejo L, Dias LE, Fernández Cirelli A, Farfán EM, Garrido S, Lorenzo L, Morgada ME, Olmos-Márquez MA, Pérez-Carrera A (2012) Small-scale and household methods to remove arsenic from water for drinking purposes in Latin America. *Science of the Total Environment* 429:107–122

Litter MI, Bundschuh J (2010) Situación del arsénico en la Región Iberoaméricana. Posibles acciones articuladas e integradas para el abatimiento del arsénico en zonas aisladas. CYTED (In Spanish)

Litter MI, Morgada ME, Bundschuh J (2010) Possible treatments for arsenic removal in Latin American waters for consumption. *Environmental Pollution* 158:1105–1118

Lizama AK, Fletcher TD, Sun G (2011) Removal processes for arsenic in constructed wetlands. *Chemosphere* 84(8):1032–1043

Macioch T, Hermanowski T (2011) The indirect costs of cancer related absenteeism in the workplace in Poland. *Journal of Occupational and Environmental Medicine* 53(12):1472–1477

Maddison D, Catala-Luque R, Pearce D (2005) Valuing the arsenic contamination of groundwater in Bangladesh. *Environmental & Resource Economics* 31:459–476

Martin-Ortega J, Brouwer R, Ojea E, Berbel J (2012) Benefit transfer and spatial heterogeneity of preferences for water quality improvements. *Journal of Environmental Management* 106:22–29

Matos EL, Loira DI, Zengarini N, Fernández MM, Guevel CG, Marconi E, Spitala A, Rosso S (2003) Atlas de mortalidad por cáncer. Argentina 1997–2001. www.asarca.org.ar/archivos/AtlasMortalidadCancerArg97-01.pdf (In Spanish). Accessed 29 May 2014

McClintock TR, Chen Y, Bundschuh J, Oliver JT, Navoni J, Olmos V, Lepori EV, Ahsan H, Parvez F (2012) Arsenic exposure in Latin America: biomarkers, risk assessments and related health effects. *Science of the Total Environment* 429:76–91

Molinos-Senante M, Garrido-Baserba M, Reif R, Hernández-Sancho F, Poch M (2012) Assessment of wastewater treatment plant design for small communities: environmental and economic aspects. *Science of the Total Environment* 427–428:11–18

Molinos-Senante M, Hernández-Sancho F, Sala-Garrido R (2011a) Cost–benefit analysis of water-reuse projects for environmental purposes: a case study for Spanish wastewater treatment plants. *Journal of Environmental Management* 92:3091–3097

Molinos-Senante M, Hernández-Sancho F, Sala-Garrido R (2011b) Assessing disproportionate costs to achieve good ecological status of water bodies in a Mediterranean river basin. *Journal of Environmental Monitoring* 13:2091–2101

Montamat & Asociados (2012) Informe mensual de precios de la energía. *Carta Política* 70: 1–13 (In Spanish)

Muthke T, Holm-Mueller K (2004) National and international benefit transfer testing with a rigorous test procedure. *Environmental & Resource Economics* 29:323–336

Narrod C, Zinsstag J, Tiongco M (2012) A one health framework for estimating the economic costs of zoonotic diseases on society. *EcoHealth* 9:150–162

Pearce J, Albritton S, Grant G, Steed G, Zelenika I (2012) A new model for enabling innovation in appropriate technology for sustainable development. *Sustainability: Science, Practice, and Policy* 8(2):42–53

Pearce D, et al. (2002) Economic valuation with stated preference techniques. Summary guide. Department for Transport, Local Governments and the Regions, London. http://www.communities.gov.uk/documents/corporate/pdf/146871.pdf. Accessed 29 May 2014

Pérez A (2006) Evaluación de elementos traza en agua, suelo, forraje y leche. PhD Thesis. Centros de Estudios Transdisciplinarios del Agua. Universidad de Buenos Aires (In Spanish)

Puig I, Freire J (2011) Beneficios económicos de una mayor calidad del agua de abastecimiento en el consumo de agua envasada de los hogares (Economic benefits of the water quality supply in terms of botteled water consumption in households). *Tecnología del Agua* 31(334):60–65 (In Spanish)

Pulido-Velazquez M, Paredes-Arquiola J, Pulido-Velazquez D, Solera-Solera A, Lozano-Sandoval G, Hernández-Sancho F, Sala-Garrido R, del Saz-Salazar S, Molinos-Senante M (2008) Elaboración de una metodología y herramientas para la determinación de un programa de medidas destinadas al cumplimiento de la Directiva Marco del Agua. Estudio piloto de la cuenca del río Serpis. Working paper Jucar River Basin Authority (In Spanish)

Ready R, Navrud S (2006) International benefit transfer: methods and validity. *Ecological Economics* 60(2):429–434

Roman HA, Hammitt JK, Walsh TL, Stieb DM (2012) Expert elicitation of the value per statistical life in an air pollution context. *Risk Analysis* 32(12):2133–2151

Sarmiento M, Carignano G (2011) Rol del recurso agua en un sistema de pagos por servicios ambientales en Santiago del Estero, Argentina. Congreso Internacional Pago por Servicios Ambientales. Ixtapán de la Sal, Estado de México (In Spanish)

Steinmaus C, Yuan Y, Kalman D, Rey OA, Skibola CF, Dauphine D, et al. (2010) Individual differences in arsenic metabolism and lung cancer in a case–control study in Cordoba, Argentina. *Toxicology and Applied Pharmacology* 247:138–145

Tsai WJ, Liu JT, Hammitt JK (2011) Aggregation biases in estimates of the value per statistical life: evidence from longitudinal matched worker-firm data in Taiwan. *Environmental and Resource Economics* 49(3):425–443

UNDP (United Nations Development Programme) (2006) *Human Development Report, 2006. Beyond scarcity: Power, poverty and the global water crisis*

UNDP (United Nations Development Programme) (2012) *Progress on Drinking Water and Sanitation 2012 Update*

Um MJ, Kwak SJ, Kim TY (2002) Estimating willingness to pay for improved drinking water quality using averting behavior method with perception measure. *Environmental and Resource Economics* 21(3):287–302

Ward FA (2012) Cost–benefit and water resources policy: a survey. *Water Policy* 14(2):250–280

WHO (2001) Arsenic and arsenic compounds. Environmental Health Criteria 224. http://www.who.int/ipcs/publications/ehc/ehc_224/en/. Accessed 29 May 2014

WHO (2003) Domestic Water Quantity, Service Level and Health, Geneva

Wilking N, Jonsson B (2005) A Pan-European Comparison Regarding Patient Access to Cancer Drugs. Stockholm, Sweden: Karolinska Institutet in collaboration with Stockholm School of Economics. http://ki.se/content/1/c4/33/52/Cancer_Report.pdf. Accessed 25 Aug 2013