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ESTIMATING ECONOMIC HEALTH COSTS OF NOT CONTROLLING TOXIC WATER POLLUTION

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In the United States, the Clean Water Act gives EPA a mandate to set ambient water quality standards for all contaminants in surface water and to monitor toxic chemicals. However, EPA has "received health-related data on only 15% of the 20,000 new chemicals released in the last 30 years" (*StarTribune*). In addition, no federal laws have been enacted to directly control groundwater quality. Consequently, with the growth in new chemicals it has become increasingly important to measure the welfare costs of toxic water contamination.

Estimating the costs of toxic chemical contamination in water bodies has received relatively little attention. What makes the valuation complicated is that empirically estimated welfare costs are likely to be “situation-dependent”. That is, the estimated value and composition of welfare costs of toxic pollution may be affected by the composition of water demand and information given to the public, and as a result, the actions of private and public agents. Raucher and others found that private averting behavior depends on the extent of public notification concerning the contaminant's health risks, whether an alternative water supply is available, and whether children are in the household.

The purpose of this paper is to determine what types of information may be important in establishing the magnitude of welfare cost when a given type of contamination occurs in a specific regional setting. It suggests "scenarios" for different types of toxic pollutants that may be found in any given water body. We do this for two quite different country settings. One, a "typical" developing economy, where demand for water comes primarily from agricultural and domestic use, public and private resources are limited, and regulations on ambient and drinking water quality are not well established or enforced. The other, an industrialized economy where a large portion of water demand comes from industrial use, regulations on ambient and drinking water are well established, and monitoring and enforcement are relatively reliable.

Estimating Cost of Toxic Pollution

Prior valuation research on toxic chemicals may be classified into three broad categories: (1) avoidance-costs (AC); (2) recreational-choice (RC); and (3) cost-of-illness (COI) (work days lost plus medical expenses) or value-of-statistical-life (VSL) approaches. Since toxic pollutants may cause significant health effects, we cannot just use the AC approach.

Avoidance Costs

The avoidance costs approach is based on the idea that if one can choose a vector of averting options to optimally adjust the quality of one's "personal environment", then the welfare benefits of improving water quality can be measured by one's averting expenditures. This is true only if toxic contaminants in water can be treated to a safety level that does not affect human health over a lifetime of consumption or if there exist alternative sources of water that are both physically and economically available to the public. Provided that these conditions are met, the welfare costs of contamination may be measured in terms of avoidance expenditures. However, the approach relies crucially on underlying behavioral/informational assumptions. It assumes that consumers and regulators have the information regarding the health effects of the contaminant in question, and that a great majority of population have access to averting measures and adopts them accordingly. If people cannot use the proper measures, due to lack of information or lack of access (including lack of financial resources), then the economic value of damage to their health may be a more appropriate measure of the welfare costs. This issue is particularly important in developing countries, where health advisories and drinking water quality standards are unlikely to be well established and disseminated to public, and where people or government agencies or both have very limited resources. These issues specific to developing countries imply too large a number of observations with zero avoidance expenditures, which may preclude us from using the AC approach.

We can illustrate the problem by using a standard avoidance costs approach. Assume for simplicity that there is only one toxic pollutant of regulatory concern. We want to evaluate the welfare costs of this pollutant in drinking water sources. Let E^i be the total averting expenditures for individual *i*. Let V_u^i denote the willingness-to-pay (WTP) for clean water regardless of the contamination risk, and let C_u^i denote the WTP value for protection against this particular contaminant. In other words, V_u^i represents a portion of expenditures paid recurrently and should include a risk premium paid for avoiding unknown or unexpected water contamination, whereas C_u^i represents the WTP value to avoid the health risks associated with this contaminant. Assume that the health risks of this contaminant are well known to regulators, and thus disseminated widely to the public. Suppose further that there are *n* alternative measures for avoiding the health risks from the (potential) contamination, with costs of each measure being C_j , *j=1,...,n*. Then, the total averting expenditures for person *i* is given:

$$E^i = V_u^i + \chi^i \cdot \min\{C_1, C_2, ..., C_n, C_u^i\}$$

where χ^i =1 if *i* is aware of the contamination risk and = 0 otherwise. Note that in this formulation, it is assumed that those informed of the contamination risk would have access to averting options. We could relax this assumption by allowing the vector { C_j } to vary over individuals and to contain zero if no averting measure is available to some person. “True” welfare costs (or their ideal welfare estimates) of this toxic pollutant must have the property:

$$W^i = \begin{cases} E^i - V_u^i & if \ \ \chi^i = 1 \\ C_u^i > 0 & if \ \ \chi^i = 0 \ or\ no\ available\ options \end{cases}$$

whereas actual empirical welfare estimates using averting expenditures will result in:

$$\widehat{W}^i = \begin{cases} E^i - V_u^i & if \ \ \chi^i = 1 \\ 0 & if \ \ \chi^i = 0 \ or\ no\ available\ options \end{cases}$$

Thus, the welfare costs can be underestimated, which can be quite important if a significant part of population has either χ^i =0 or no access to avoidance options. In fact, prior avoidance cost studies have recognized that awareness of the pollution problem is one of the key determinants of averting behaviors and that those informed tend to spend more on averting expenditures. Yet, such information was not used effectively. The estimated percentage of population with χ^i =0 is no less important than the estimated averting expenditures for those whose χ^i =1, because the regulator must use the estimate of C_u^i to calculate the welfare costs for those with χ^i =0 and $E^i - V_u^i$ for those with χ^i =1.

Recreation Site Choice

Toxic water contamination may affect the welfare of people living in the vicinity of the pollution not merely from direct domestic use but also from indirect use of surface water. People enjoy clean water resources for recreational purposes such as fishing, swimming, boating, and site-seeing. Montgomery and Needelman and Phaneuf et al. estimated, in different settings, the welfare costs of toxic contamination in freshwater based on anglers' choice of fishing sites. This approach is likely to capture a significant portion of the welfare costs related to indirect use. If appropriately administered, the WTP estimate based on anglers' choice of fishing sites can be added to the welfare estimate from the avoidance costs approach.

Cost-of-Illness

We still need to estimate the welfare costs when the toxins damage human health. There are essentially three approaches to estimate these welfare effects. First, one may use the number of work-days lost and multiply it by the commensurate wage rate. Second, medical expenses paid for treating the sickness may be used. Finally, contingent-valuations are increasingly being used to estimate reduced health risks, sometimes referred to as "the value of a statistical life". The first and second methods may be used for the kinds of sickness that may be completely cured, whereas the third method is more suited for estimating the value of avoiding life-threatening illness such as cancer. A difficulty arises because many toxic water contaminants are known to have multiple health effects including carcinogenic ones. Several prior valuation studies of air-borne illness have shown that the first two approaches (cost-of-illness approaches) tend to yield lower estimates than do contingent-valuations, suggesting that the value (cost) of discomfort and suffering caused by illness may be quite large (Alberini & Krupnick). Thus, if discomfort and suffering are likely to be important, then we need to use contingent-valuations.

The list of prior valuation studies of economic losses due to toxic contaminants is surprisingly short and most of those relevant to our study are from developed countries (table 1). Yet the relative magnitude of the different valuation methods seems to be in line with our expectation, although none relate the carcinogenic effects of toxins in water to the value of statistical life. Also, no prior study has estimates for all three components, AC, RC, and COI.

Costs of Three Toxic Water Pollutants

We have selected 3 widely occurring toxic chemicals, each with different characteristic in terms of source, persistence, and toxicity. The purpose of this section is to illustrate, first, how these different chemicals can have different welfare effects under the same behavioral assumption, and second, how the same chemical can have different welfare effects under different assumptions.

Arsenic

In developing countries such as India, Bangladesh, and China, arsenic effects are widespread. In these countries, national drinking water standards are set at 0.05mg/L (or 50ug/L) (WHO). Yet private wells are an important source of drinking water and the national standards are not well enforced. As a result, "millions of people have arsenic concentrations in their drinking water above 50ug/L, with some exceeding 1000ug/L" (Smith and Smith). Even worse, the villagers have learned "from experience that it is often wise not to believe what the latest technocrats [i.e., government officials] are currently telling them" (Smith and Smith). Given such perception held by the villagers, it is very unlikely that the avoidance costs approach would capture a significant portion of the welfare costs associated with arsenic exposure, since they would not adopt appropriate measures to avoid exposure. The welfare costs of arsenic exposure in this setting would be best estimated by using the health risk approach.

In developed economies, such as the United States, only a small minority of the population is exposed to arsenic risks. In these countries, regulations on drinking water are well established and disseminated widely to the general public. Consumers are informed via consumer confidence report of the quality of water from their public water services. Thus, in these countries, the avoidance costs approach can be used to measure the people's willingness-to-pay for reduced levels of arsenic associated with drinking water. As arsenic is naturally occurring yet non-persistent as a toxic contaminant in the ecosystem, it is unlikely that recreationalists, concerned mainly with the ecological health of recreational sites, will find arsenic contamination as a key factor determining their choice of sites. As a result, the welfare costs of non-action associated with recreational uses are likely to be insignificant, and WTP estimates based on recreational demand models are, therefore, not required.

In rural areas of Asia, where surface water is highly contaminated, many villagers are afraid to shift away from their arsenic contaminated groundwater supplies. This situation establishes conditions where the costs of no action, in terms of health impacts, are likely to be quite high and affect a large number of people. For example, Bangladesh has over 4 million tube wells and studies have found that over half don't meet the World Health Organization's standard. Paul and De suggest that over sixty percent of the country's population may be afflicted with arsenic poisoning.

Mercury

In India, the national safety limits are set for drinking water at 0.001 mg/L (or equivalently, 1.0 ppb). However, a recent study conducted by the Guru Gobind Singh Indraprastha University, Delhi, revealed that the concentration of mercury in the groundwater of Delhi was significantly above the safety limit. Among 50 samples of groundwater taken randomly from along a 22-km stretch between Palla and Okhla, the mercury concentration in some samples was as high as 460 ppb (India Together). Although people in India have become increasingly aware of the health hazards of mercury ingestion, no appropriate avoidance measures have yet been disseminated to the public. Furthermore, a large percentage of the Indian population eats fish as a staple food, but "no provisions for daily or weekly mercury intake levels have been set" (India Together). In this case, the avoidance costs approach, again, will fail to capture a significant portion of welfare costs of non-action.

Many developed countries such as the United States and Japan have undergone the phase that India is now experiencing. Mercury standards and health advisories are currently well established and disseminated in these countries. As of May 2005, the EPA's maximum contaminate level (MCL) was set at 0.002 mg/L. If the levels of mercury exceed the MCL, the system must notify the public via newspapers, radio, TV and other means. Additional actions, such as providing alternative drinking water supplies, may be required to prevent serious risks to public health. Fish consumption advisories are also currently in effect for mercury in thousands of lakes and rivers. Under these conditions, the avoidance costs approach and recreational demand models can estimate the lower bound costs of mercury contamination.

Even with the limited information available, it is clear that the largest cost of not taking any action regarding mercury will be in Asia. This is because a major source of mercury in water and fish is coal-fired plants. Not only is coal an important source of energy in Asia but with the expanding demand for electricity, coal use is increasing. The United Nations Environmental Programme reports that coal-fired power plants and waste incinerators emit 1,500 tons of mercury annually of which 860 tons comes from Asia. Combine this with extensive consumption of fish and poor information, and you have the potential for high future costs. The wide spread mercury contamination in Minamata and Niigata and other fishing villages in Japan that provided the name for mercury poisoning (the Minamata disease) is a warning to other countries. Of the "2,252 patients who have been officially recognized as having Minamata disease 1,043 have died" (Harada).

Atrazine

Atrazine is an herbicide that is most commonly used on corn, which accounts for approximately 86% of U.S. usage. Atrazine offers an interesting case of hazardous water contamination because: (i) its human health impacts are not as clear as the other two toxic contaminants and is subject to controversy, (ii) its ecological impacts are known with some certainty, and as a result, (iii) its regulations on use vary among countries. For example, atrazine is still widely used in the United States, particularly in the Midwest for corn and soybeans, whereas several countries in the EU, including France, Denmark, Germany, Norway and Sweden have banned it (Organic Consumers Association). Because the scientific findings on its effects on humans are indeterminate, people's perceptions of atrazine's health risks vary significantly. Moreover, its carcinogenic classification is still under review in the U.S., and was originally classified as "possible human carcinogen". Consequently, the costs associated with atrazine contamination may be subject to two sources of uncertainty: one on whether or not people are aware of the new scientific findings and the other on how people judge the new scientific findings. Moreover, in developed countries where the occurrence of atrazine and chlorinated metabolites in water is typically below the level that causes any significant health effects, the welfare costs of atrazine contamination may be derived mainly from the ecological effects.

Policy Implications and Conclusions

The above examples show how different behavioral assumptions may change both the magnitude and composition of welfare costs estimates. Each country has varying levels of exposure to different toxic chemicals, with different policy priorities and regulatory frameworks. Thus, we need to be careful when selecting the methods to estimate the welfare cost of toxic water pollutants. The method selected should depend on the information provided to the public as well as the availability of alternatives to avoid the toxic pollutant. For example, because of imperfect information and the lack of low cost alternatives, the avoidance cost approach would underestimate the welfare costs of arsenic, mercury and atrazine in a developing country setting. In contrast, in developed countries avoidance cost would be an appropriate measure as long as there were no adverse ecological impacts and the pollutants are not persistent bioaccumulative.

For mercury, which is persistent bioaccumulative and has ecological impacts, even in developed countries, avoidance costs methods will underestimate the welfare cost. Recreational choice, cost of illness, and contingent valuation methods all may be needed to estimate the full welfare cost of mercury pollution. In contrast, in developing countries ecological damages are not likely to be high priority but cost of illness and contingent valuation models will be needed to capture the health costs of mercury pollution. The same is true for arsenic in developing countries. However, in developed countries avoidance costs models should be all that is needed to measure the full welfare cost of arsenic pollution except for low income groups or groups that are isolated and poorly informed. Finally, atrazine poses different measurement problems due to the uncertainty regarding its health effects on humans. If there are no human health effects, then the major impact is on recreational resources. In this case, atrazine would have relatively small impacts in developing countries, except where the tourist industry is important.

Another implication of our analysis is the importance of chemical characteristics, since the sources of chemicals may significantly affect the optimal choice of policies. For example, arsenic contamination typically occurs naturally and therefore, tightening emission regulations will not reduce pollution. For arsenic pollution, the best strategy would be again to disseminate reliable health-risk information along with low cost filters or other avoidance measures. Mercury, in contrast, occurs both naturally and from industrial sources with contamination problems often tied to industrial emissions.   Consequently, it is best addressed by taking stringent preventive steps (such as tightening air emission standards) and public programs disseminating information concerning contamination levels and sources.

The toxicity of chemicals is also important in determining the costs of alternative policies. For example, atrazine's health effects are still subject to scientific uncertainty, though its ecological effects are relatively well established. In this case, our policy recommendation is less clear-cut. Banning or reducing the use of atrazine compounds may be too costly if, in fact, its health risk is marginal. Again, public dissemination programs may be a good short-term strategy, as better substitutes are developed.

A final critical factor that will be important in determining the cost of not taking any action is the size of the population exposed. This will depend on the location of the polluted water as well as on the frequency and persistence of the pollutant. If action is not taken, both mercury and arsenic contamination of water are likely to impose very high future costs in Asia. Actions to avoid these costs must be given high priority in the new “water” decade.

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Table 1. Estimated Costs of Toxic Water Contamination

Study Area	Estimated Range of Costs			
	Avoidance cost per month	Contingent valuation per month	Recreational choice per season	Cost of illness per case
	-- -- US dollars -- --			
Perkasie, PA, US[1], 1987-89	17.38	--	--	--
West Virginia, US[2], 1990	26.67, 29.75, 90.83	--	--	--
Orange County, CA, US[3], 2001	--	--	--	36.58, 76.76
Seoul, South Korea[4], 1991	--	3.12-3.28	--	--
New York, US[5], 1989	--	--	63.25	--
Wisconsin Great Lakes, US[6], 1989	--	--	89.35-108.13	--

Sources: [1]Abdalla et al., 1992; [2]Collins & Steinback, 1993; [3]Dwight et al., 2005; [4]Kwak et al., 1997; [5]Montgomery & Needleman, 1997, [6]Phaneuf et al., 2000.