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What are the Economic Health Costs of Non-Action in Controlling Toxic Water Pollution?

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Abstract

This paper identifies information that may be important in determining the benefits of preventing toxic water contamination (or equivalently cost of nonaction) when a given toxification occurs. It attempts to identify information and behavior issues that need to be considered when we estimate benefits and weigh them against the costs of removing toxins. This paper also provides "scenarios" for three toxic pollutants that are found in water bodies. We make use of two alternatives--one for developing countries and the other for developed countries--to demonstrate, with specific examples of arsenic, mercury and Atrazine, how benefit estimates and control policies vary with different assumptions concerning behavior/information and type of chemical contamination. A comparison with EU evaluation experience is also carried out.

Key words: welfare costs, arsenic, mercury, Atrazine, information, water

INTRODUCTION

In the United States, the Clean Water Act, enacted initially in 1972 as the Federal Water

Pollution Control Act Amendments and amended in 1977, 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, 2005). In addition, no federal laws have been enacted to directly control groundwater quality, although other regulations such as the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA), commonly known as Superfund, and Resource Conservation and Recovery Act (RCRA), have helped to reduce groundwater contamination. Yet as the U.S. population expands and demands safe drinking water, it will become increasingly important to measure the welfare costs of water contamination by toxic substances. Concerns about the cost of toxic pollution are likely to be even more important for developing countries.

Toxins are classified based on characteristics such as expected health risks, toxicity (e.g., dose-response rate), and persistency. Currently, EPA provides nationally recommended water quality criteria for a total of 120 chemicals, and lists 85 primary contaminants along with their drinking water quality standards (EPA, 2004). In our paper we diverge slightly from these common classification practices and use toxicity, persistence, and sources as our criteria for selecting the chemicals to study.[1]

The EU followed for years a similar approach. The water framework directive (directive 60/2000) (WFD) changed dramatically the way in which water quality is regulated It requires water quality objectives to be set at the hydrographic district level (e.g., 3 to 4 hydrographic district are planned for Italy) by a local authority, through a local participated process.

However, the main toxic substances are treated differently and included in the directive under priority substances, under appendix X. The aim of the directive is to stop or gradually reduce to zero the emissions of such priority substances. For these substances, while the above is still true, the EU itself intervenes with specific regulations, including the definition of quality standards.

A distinguishing feature of the WFD, directly relevant for this paper, is that it also requires all policy measures designed to achieve the water quality objectives to be economically evaluated. This explicitly requires the estimation of costs and benefits of action/non action with respect to existing or desired water quality objectives. The WFD also introduced the concept of full cost (sum of financial cost, opportunity cost and environmental cost) as a basis for water pricing (in order to respect the polluter pays principle - PPP). In principle, in order to implement the WFD, extensive economic evaluations related to water, including the effects of toxic substances, will be needed in the future.

Yet estimating the costs of toxic chemical contamination in water bodies has received relatively little attention. A conceptual difficulty arises from the fact that toxic substances may potentially contaminate water supply systems, and thus, pose serious health risks. What makes the valuation complicated is that empirically estimated welfare costs are likely to be "situation-dependent." The estimated value and composition of welfare costs of toxic pollution are likely to be affected by the composition of water demand and information given to the public, and as a result, the assumptions concerning the actions of private and public agents. This was suggested by Raucher, 1986; Abdalla, Roach, and Epp, 1992; Collins and Steinback, 1993; and Abdalla, 1994, who 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.

One purpose of this paper is to determine what types of information may be important in establishing the magnitude of welfare benefits of removing toxic contamination (or equivalently, welfare costs of nonaction) when a given type of contamination occurs in a specific, regional setting and discuss how this information will impact the welfare benefits. The paper also suggests "scenarios" for three different types of toxic pollutants, arsenic, mercury, and atrazine, that may be found in any given water body (surface water, and groundwater). We will do this for two quite different country settings. First, we consider 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. Second, we consider 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. In the conclusion we suggest how to estimate the costs of these pollutants and how best to reduce these pollutants under the two different country settings: developed and undeveloped.

VALUATION STUDIES ON TOXINS: SITUATION/SCENARIO-DEPENDENCE

There have been a limited number of valuation studies based on the toxic characteristics of water pollutants, whereas there are an ample supply of valuations based on field parameters such as dissolved oxygen, BOD, pH, total suspended solids, and total phosphorus (See Wilson and Carpenter (1999) for a review of available valuation studies). 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. A key characteristic of toxic pollutants is that they may cause significant health effects. This distinguishes them from less hazardous water pollutants and prevents us from just using relatively straightforward valuation techniques such as 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" (Bartik, 1988), 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 alternative sources of water exist (e.g., bottled or filtered water) that are both physically and economically available to the public. Provided that the standard conditions (suggested by Bartik, 1988) are met, the welfare costs of these contaminants, related only to domestic use, may be measured in terms of expenditures to alleviate the contaminants. 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 the 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 the 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.

Another complication arises in the case of toxic contaminants. Recently, the United States and international communities expressed a growing concern over the persistent bioaccumulative and toxic (PBT) chemicals.[2] Currently, EPA lists 12 priority PBTs, many of which are still released from various industrial and nonindustrial sources in the U.S. and worldwide (EPA, 2002). One of the notable features of PBTs is that they tend to resist the natural process of degradation and accumulate in the body fat of living organisms such as fish, shellfish, and waterfowls (EPA, 2002). Long-lasting impacts of the PBTs are hard to measure and unlikely to be captured simply by averting expenditures for water consumption. Since PBTs tend to persist in the food chain, the welfare costs of nonaction must include the value of health risks, which may compound over time due to consumption of fish and other affected organisms. Yet we could use an argument similar to the one used for the avoidance costs approach, that is, if well-informed consumers adopt one or more means for avoiding consumption of contaminated water and fish, and by so doing eliminate the cumulative health risks of PBTs, then we might obtain a lower-bound estimate of WTP by adjusting the standard avoidance costs approach.[3]

Recreation Site Choice

Toxic water contamination may affect the welfare of people living in the vicinity of the pollution, not merely from direct use,[4] but 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 (1997) and Phaneuf et al. (2000) 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 because, (i) anglers may eat the fish they catch and may be concerned about the toxic levels of the fish, (ii) toxic contamination can impair propagation and survival of fish, and therefore, may significantly reduce the catch rate, (iii) anglers may enjoy the scenery while fishing, which the ecological system provides, and (iv) anglers may want to swim in the water in which they fish, but may not, because of the toxic contamination. 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 when such indirect uses are significant.

Cost-of-Illness

In addition to the two empirical approaches above, 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, economists are increasingly using contingent-valuation (WTP approach) estimates of 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 illnesses, such as cancer. A difficulty arises because many toxic contaminants listed in EPA's drinking water quality standards 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, 2000). Thus, if discomfort and suffering are likely to be important, then we need to use contingent-valuations.

We have gathered prior valuation studies on economic losses due to toxic contaminants in Table 1. Even if you add the valuation studies dealing with microbial contamination of drinking water, the list is surprisingly short (Table 2). Unfortunately, most of the studies that are relevant to our study are from developed countries. The relative magnitude of the different valuation methods seems to be in line with our expectation, though we found no valuation study relating 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. Table 3 summarizes the results of selected studies concerning the evaluation of costs from water contamination in Europe, including contamination from toxic substances.[5] Interestingly, there are no studies that estimate the cost-of-illness.

SCENARIO-BASED WELFARE COSTS OF TOXIC WATER CONTAMINATION

We have selected 3 widely occurring toxic chemicals, each with different characteristic in terms of source, persistence, and toxicity. For example, arsenic is primarily naturally occurring while atrazine is commercially introduced. The purpose of this section is to illustrate, first, how 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 or scenarios.

Example 1: Arsenic

Inorganic arsenic is a naturally occurring chemical and is found throughout the environment. "For most people, food is the largest source of arsenic exposure (about 25 to 50 micrograms per day [ug/d]), with lower amounts coming from drinking water and air" (EPA, 2005a). Inorganic arsenic is known to have both acute/immediate as well as chronic health risks. "Acute oral exposure to inorganic arsenic, at doses of approximately 600 micrograms per kilogram body weight per day (ug/kg/d) or higher in humans, has resulted in death" (EPA, 2005a). "Chronic oral exposure has resulted in gastrointestinal effects, anemia, peripheral neuropathy, skin lesions, hyperpigmentation, and liver or kidney damage in humans. Inorganic arsenic exposure in humans, by the inhalation route, has been shown to be strongly associated with lung cancer, while ingestion of inorganic arsenic in humans has been linked to a form of skin cancer and also to bladder, liver, and lung cancer" (EPA, 2005a). As of May 2005, EPA's MCL was set at 10 ug/L. High concentrations of arsenic can be found in fish and shellfish but typically as organic compounds, which are essentially nontoxic (EPA, 2005a).

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, 2004). However, for these countries 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, 2004). Yet even worse, in these countries, 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, 2004). 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, as they would not adopt appropriate measures to avoid the exposure. The welfare costs of arsenic exposure in this setting would be best estimated by using the health risk approach.

In contrast, 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 generally have good information concerning 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 nonpersistent 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 in determining their choice of sites. As a result, the welfare costs of nonaction associated with recreational uses are likely to be insignificant, and WTP estimates based on recreational demand models are, therefore, not required.

The cost will be high if no action is taken to reduce arsenic in many areas of Asia where arsenic contaminated groundwater is consumed. This includes large areas in India, Bangladesh, and China. In rural areas with surface water already highly contaminated, many villagers are afraid to shift away from their groundwater supplies that have high levels of arsenic contamination. 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 in all three countries. For example, Bangladesh has over 4 million tube wells and studies have found that over half the wells don't meet the World Health Organization's standard. Paul and De (2000) suggest that over sixty percent of the country's population may be afflicted with arsenic poisoning.

Example 2: Mercury

Mercury is a toxic persistent, bioaccumulative pollutant, listed as one of EPA's twelve priority PBTs. Certain microorganisms can convert inorganic forms of mercury to "methylmercury", a highly toxic organic form that accumulates in fish, shellfish, and animals that eat fish, including humans. Large amounts of mercury are released naturally from the earth's crust. However, industrial sources, such as combustion of fossil fuels and manufacturing activities in mining, smelting, and chemical industries, are big sources of mercury. Ingestion of mercury can cause various short-term and long-term health effects, including cancer, heart attacks, permanent damage to the brain and nervous system, and damage to stomach, kidneys, lungs and other vital organs (EPA, 2005b, 2005c, and 2000d).

In India, the national safety limits for drinking water are set 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 a 22-km stretch between Palla and Okhla, the mercury concentration in some samples was as high as 460 ppb (India Together, 2003). Though 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. Even of more concern is that 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, 2003). In this case, the avoidance costs approach, again, will fail to capture a significant portion of welfare costs of nonaction because of the significant health effects.

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 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. Furthermore, fish consumption advisories are currently in effect for mercury in thousands of U.S. lakes and rivers. Under these conditions, we can be relatively confident in using both the avoidance costs approach and recreational demand models to estimate a lower bound for the welfare costs of mercury contamination. If a significant percentage of the population did not receive adequate information or could not easily avoid eating fish, then the costs of increased health risks must also be added.

Even with the limited information available, it is clear that the largest cost of not taking any action regarding mercury will be in developing countries, particularly in Asia. This is because a major source of mercury in water and fish is mercury released into the air from 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 come from Asia. Combine this with extensive consumption of fish and poor information, and you have the potential for high future costs due to increased health problems and reduced productivity. The widespread mercury contamination in Minamata and Niigata and other fishing villages in Japan, which provided the name for mercury poisoning (the Minamata disease), should be a warning to other countries in Asia and the Pacific. Of the "2,252 patients who have been officially recognized as having Minamata disease, 1,043 have died" (Harada, 1995).

Another area of potentially high cost is in gold mining areas, such as the Brazilian Amazon, where large quantities of mercury have been released into the rivers in the effort to extract gold. Uryu et al. (2001) found that “mercury concentrations in fish appear to be high enough to cause toxification in animals at higher tropic levels in the Amazon. Wildlife in the Amazon already may be or soon will be suffering the negative effects of HG contamination” (p. 444). In Indonesia, Newmont Mining Corp and its director, face pollution charges for dumping mercury and arsenic into Buyat Bay on Sulawesi Island, which caused sickness in many local villagers (Star Tribune, Aug. 5, 2005).

The cost of mercury pollution also cannot be ignored in developed countries, as a study in the U.S. state of Minnesota highlights (Schenck, 2005). This study focused on the cost, just to Minnesota, from increased mercury pollution of water due to the 2005 U.S. EPA ruling that lowered the pollution control standard for electric utilities that have coal-fired plants. The study focused on the increased risk of heart attacks and of neurological damage to unborn children when they are exposed to high levels of mercury in their mother’s blood. The study estimates that the discounted costs imposed on the people in Minnesota due to the lower U.S. federal standard will be almost $190 million, which is 2 or 3 times more than the cost of preventing the pollution. Further evidence of the water pollution problem created by mercury in the U.S. comes from the Great Lakes. All five of the Lakewide Management plans found mercury contamination to be a problem. For example, the Lake Ontario plan, calls for projects to reduce the use of mercury as well as collection programs to dispose of products containing mercury. The Lake Superior plan also calls for collection programs as well as alternative energy strategies that will reduce the use of coal to produce electricity.

Example 3: Atrazine

Atrazine is a herbicide that blocks photosynthesis. It is registered for the control of broadleaf weeds and some grassy weeds, and is most commonly used on corn, which accounts for approximately 86% of total U.S. domestic usage in pounds. Atrazine can persist in the soil, especially under dry and cold conditions, and can move fairly easily in the soil. It metabolizes into four hydroxyatrazine compounds and three chlorinated atrazine compounds. The main break-down compounds of atrazine, hydroxyatrazine metabolites, do not move easily in the soil. However, due to their relative persistence, combined with atrazine's widespread agricultural use, it and its chlorinated metabolites are frequently found in ground and surface waters located in areas with intense agriculture (EPA, 2001a, 2001b, and 2001c).

Atrazine offers an interesting case of hazardous water contamination because: (i) its human health impacts are not as clear as other highly 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 its use (Organic Consumers Association, 2003). 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, and was originally classified as "possible human carcinogens". During the Italian debate about banning atrazine, it was suggested that banning Atrazine was just a commercial move aimed eliminating from the market a harmless product because the patent was about to expire. The above imply that the valuation of welfare costs associated with atrazine contamination may be subject to two sources of uncertainty: first, whether or not people are aware of the new scientific findings, and second, how people judge the new scientific findings. Moreover, in developed countries where the occurrence of atrazine and the chlorinated metabolites in water are typically below the level that causes any significant health effects, the welfare costs of atrazine contamination may be derived mainly from the ecological impacts.

EU EXPERIENCE

The EU experience reinforces the above findings although most of the studies conducted in the EU (table 3)c onsider only the overall water quality and do not trace back the value of a single (toxic) pollutant, except in a few cases. For the same reasons, it is often difficult to identify a single polluting sectors/industries and as a consequence, to use the evaluation data to apply the polluter pays principle.

The number of studies is small like the USA experience. This is not surprising since the European Union (EU) has not encouraged monetary evaluation of pollution due to political philosophy, legal framework and (lack of) scientific lobbying (Pearce, 2001).

The studies are very varied in terms of methodologies, methodological details (e.g. definition of users) and the object of evaluation, as well as the time of evaluation. As a consequence, it appears rather difficult (though somehow necessary) to generalize the connection between values and determinants (local income, hypotheses about water quality, etc.). In addition, the studies are usually very localized, and fail to provide a general view.

The contingent valuation approach is the dominant method used. . This may be interpreted in different ways. First, it may have been academic-driven, since many studies were implemented when the contingent approach became popular in the ‘90s. Second, it may have been an attempt to gather a number of effects altogether (ranging from avoidance to health costs) and to filter them through the perception of the pollution victims (that can help in connecting them to policy actions, including paying fees or asking for damages). Third, this could also be connected with the difficulties in gathering any reliable data on health effects, particularly for non-acute effects, that, in many cases, are likely to produce most of the damages. In some EU countries, this picture of evaluation studies could also be interpreted as the outcome of a mature legislation and pollution prevention system, where major single toxic pollution events rarely occur. Consequently, attention is focused on trade-offs concerning different levels of water quality to meet recreation demands. In a number of cases reference is made to national water quality norms, in spite of the Water Framework Directive (WFD). This may be explained by the fact that most studies where done before the implementation of the WFD, but it is also somewhat WFD-consistent since, according to the WFD, the definition of good water would still be decided locally.

In the coming years, EU countries will be obliged to implement a large evaluation exercise in order to comply with the WFD concerning full cost and the evaluation of measures. Currently, countries are largely relying on existing studies. Benefit transfer (BT) techniques are being tested in order to meet such requirements in a cost-effective manner. These attempts pay attention specifically to many of the issues discussed above about scenario dependency of evaluation of the effects of pollutants. Each region using nonlocal values for the estimation of water-relate cost and benefits will first need to choose between sources using different methodologies (or a consistent combination of them). Second, it will need to carry out the exercise to adapt values to local conditions. The size of affected population, their income, the physical features of the water body, the type of contaminant and the area affected, will be key parameters for such an adaptation. However, the ability/willingness to react to the pollution will also be a key feature of the value correction.

The current difficulty with the interpretation of existing studies emphasizes the need to improve on methodological understanding of benefit transfer, taking into account how values have been generated, what sources are used and what assumptions are made about local actions. In some cases of BT, values taken outside Europe are preferred.

A large part of these evaluations will be related to policies to prevent or reduce pollution and these policies are (too) often assumed to be effective (and sometimes efficient) when doing ex ante evaluations. Consequently, avoidance cost methods will play a major part in the evaluations. This may also be due to the fact that where treatment plants already exist it may be relatively cheap to use their cost as estimates of damages. For example, treatment costs in Italy are often used as the best estimate of urban pollution, although most people involved (and suspected by the general population although they usually are not informed) know that in most cases they do not work.

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, care must be taken 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, avoidance cost in developed countries would be an appropriate measure as long as there were no adverse impacts on recreational resources and the pollutants were not persistent bioaccumulative and difficult to measure.

For mercury, which is a persistent bioaccumulative and has recreational 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, recreational opportunities in developing countries 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 some 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. Where tourism is important, the impact of atrazine may need to be estimated using recreational choice and contingent valuation models.

Another implication of our analysis is the importance of chemical characteristics, since the source of chemicals may significantly affect the optimal choice of policies. For example, arsenic contamination typically occurs naturally; therefore, tightening regulations on ambient water quality for this chemical will not reduce pollution. For arsenic pollution, the best strategy would be to disseminate reliable health-risk information along with privately installable low-cost water filters. Mercury, on the other hand, occurs both naturally and from industrial sources. Intense mercury contamination problems are often tied to industrial emissions, particularly from coal burning plants and mining. Moreover, the welfare effects of mercury can compound over time in the food chain and can have both direct and indirect use impacts. Such chemical characteristics are best addressed by taking stringent preventative 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. Atrazine is widely used as an herbicide in the U.S. and high contamination levels are often found in the groundwater in the vicinity of its agricultural application. Banning or reducing the use of atrazine compounds may be too costly if in fact its health risk is marginal. Yet, if good substitutes are available, then bans, or use restrictions, may not have a high cost except for the producers of atrazine. Again, public dissemination programs may be a good short-term strategy along with policies to encourage the use and development of better substitutes.

A final 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 on society because of the concentration of populations in Asia who are exposed to, at least, one of these two pollutants. Actions to avoid these costs must include low cost methods for filtering arsenic out of the groundwater and the widespread adoption of technologies that effectively reduce mercury emissions from coal-fired plants and incinerators.

Finally, this paper has primarily focused on persistent, or widely occurring, toxic contaminants. There are also emergency-type contamination events, which typically pose acute, immediate health risks. These events involve toxic chemical spills into rivers or other water bodies, such as the one that occurred in Jilm in northeast China on November 13, 2005. A chemical plant explosion caused an 80 km long toxic slick, made of benzene and nitrobenzene, on the Songhua River. This type of contamination events is difficult to control, because they are usually caused by unpredictable events. Even worse, there may be “latent” contamination problems--toxic chemicals emitted from industrial facilities that sit in the ground for a long period before they enter an aquifer. Both types of contamination are likely to become more frequent as the developing world continues to industrialize.

To minimize the risks from these emergency-contamination events, we need to consider preventive as well as corrective policies. Corrective measures include (a) rapid assessment and notification of the exposed population, (b) timely clean up efforts, and (c) the provision of alternative water supplies. In many developing countries, as well as in many developed countries, these rules are not well-established. As for preventive policies, routine inspection and monitoring at chemical-intensive industrial facilities for quality of operations may be necessary to reduce the frequency of disastrous events. In addition, strict, enforceable liability rules and fines can be used to provide firms with a strong incentive to make “precautionary” investments to prevent toxic spills.

A potential further area of concern is about the accumulation in the environment of low concentrations of emissions or non-compliances small enough to not justify individual defensive action or to call for public alarm. This is the case for most of the urban wastewater systems in Italy that produce water discharges often not in compliance with existing norms. The water discharge does not normally directly contaminate drinking water sources, but enters the water cycle e.g through irrigation.

The toxic contamination in the Songhua River highlights three areas where future research and development are needed. First, we need to determine more clearly what the health effects are of different industrial chemicals, such as nitrobenzene, which have the potential of causing acute water contamination. Second, we need to estimate the risks of future acute toxic contamination events and then develop cost-effective strategies to reduce the risk. But just like flooding you cannot reduce the risk to zero. Third, the slick on the Songhua River went downstream contaminating the Amur River, and then entered Russia to contaminate Khabarovsk's water supply. For countries such as China and Russia, which share borders with others, this suggests the strong need for effective transboundary institutional arrangements to deal with pollution issues of this type.

Table 1. Estimated Costs of Toxic Water Contamination

Study Area	Avoidance cost per month	Contingent valuation per month	Recreational choice per season	Cost of illness per case
	Estimated Range of Costs			
	-- -- US dollars -- --			
Perkasie, PA, US[1], 1987-89	17.38	--	--	--
West Virginia, US[2], 1990	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.

Table 2. Estimated Costs of Drinking Water Contamination by Microbial Contaminants

Study Area	Estimated Range of Costs		
	Avoidance cost per month	Contingent valuation per month	Cost of illness per case
	-- -- US dollars -- --		
Georgia, US, 1995[1]	3.92	--	--
Milesburg, PA, US, 1989[2]	13.07-33.47	--	--
Luzerne Cty, PA, US, 1984[3]	33.90-107.70	--	858-1,255
Grande Vitoria, Brazil, 1996[4]	--	2.77-39.30	--
Kathmandu, Nepal, 2001[5]	2.94	17.36	--
West Virginia, US, 1990	26.67	--	--

Sources: [1]Abrahams et al., 2000; [2]Laughland et al., 1993; [3]Harrington et al., 1989; [4]McConnell and Rosado, 2000; [5]Pattanayak et al., 2005; Collins and Steinback, 1993.

Table 3. Estimated Costs of Toxic Water Contamination

Study Area	Estimated Range of Costs			
	Avoidance cost per month	Contingent valuation per month	Recreational choice per year	Cost of illness per case
	Us dollars			
Porretta, Bologna, IT[1], 1992		5.88		
Palmanova, Udine, IT[2], 2000		3.01-3.35		
Trentino, IT[3], 1996			5.11	
Venice,IT[4], 2002 (TC fishing)			31.50	
UK[5], 1991		1.96		
England and Wales, UK[6], (per km of coast per year)			85552-221802	
Birmingham, UK[7], 1999		0.01-0.06		
Brest, FR[8], 1993		2.29-3.08		
Alsatian region, F[9], 1993		8.85		
Denmark[10], 200?		8.94-18.58		
Netherlands, NL[11], 2002		2.44-6.01		
Austria[12], 200? (per ton COD)	59.86			
Spain[13], 1997-2002 (000 $ per ton)	109-661			
Estoril Coast, P[14], 1997 ($ per person)		44.39		
Estoril Coast, P[14], 1997 ($ per visit to beach)		7.94-10.97		
Baltic sea region, PL[15], 1994 ($ per person)		84		

Sources: [1]Stampini., 1998; [2]Marangon e Tempesta, 2004; [3]Notaro, 2001, [4]Alberini et al. (2006); [5]Hanley (1991); [6]Willis and Garrod (1996); [7]Georgiou et al. 2000; [8]Le Goffe (1995); [9] Stenger and Willinger (1998); [10] Hasler and Lundhede (2005); [11]Brouwer (2006); [12] Angst et al. (2001); [13] Sanchez-Choliz and Duarte (2005); [14] Machado and Mourato (2002); [15] Zylicz et al. (1995).

NOTES

1. Microbial contaminants typically result in immediate illness, while chemical contaminants tend to cause more long-lasting, chronic health effects. The potential health effects of a given chemical pollutant may exhibit one or more of the following generic aspects: (a) genotoxicity, (b) carcinogenicity, (c) neurotoxicity, (d) disturbance of energy transfer, (e) cause reproductive failure, and (f) effect behavior (e.g., loss of appetite).
2. The United States, the European Union, and 90 other countries signed a treaty in Stockholm, Sweden, in May 2001. Under the treaty, countries agreed to reduce or eliminate the production, use, and/or release of the 12 key persistent organic pollutants (EPA, 2002).
3. Identifying averting behaviors for contaminated fish may be difficult, since most consumers have ready access to alternative sources of fish and shellfish, without feeling any price effects.
4. Direct domestic use includes (a) drinking, (b) bathing, (c) food preparation, and (d) sanitation services.
5. Unfortunately, in most cases, the effects estimated concern toxic substances-related water pollution, but not the effects of single pollutants.

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APPENDIX

PROBLEMS WITH AVOIDANCE COST ESTIMATES

More formally, 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 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 to avoid the health risks associated with this particular 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:

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

where c^i 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 automatic 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.[1] "True" welfare costs (or their ideal welfare estimates) of this toxic pollutant must have the property:

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

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

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

Thus, the welfare costs can be underestimated, which can be quite important if a significant part of population has either c^i=0 or no access to avoidance options. In fact, it is widely recognized from the finding of prior avoidance costs studies (for example, Abdalla, Roach, and Epp, 1992; Kwak and Russell, 1994; Dickie and Gerking, 1996; Abrahams, Hubbell, and Jordan, 2000) (i) that awareness of the pollution problem is one of the key determinants of averting behaviors and (ii) that those

[1] By availability we include technological as well as financial feasibility. If a person's income is so low that he or she cannot purchase an averting measure, then the vector should contain zero.

informed of the contamination tend to spend more on averting expenditures than those who are unaware. However, in the previous studies, such information was not used effectively. From the point of views of welfare-maximizing regulators, the estimated percentage of population with $c^i=0$ is, in fact, no less important than the estimated averting expenditures for those whose $c^i=1$, because the regulator must use the estimate of C_u^i to calculate the welfare costs for those with $c^i=0$ and $E^i - V_u^i$ for those with $c^i=1$. What we want is not just the *estimates of WTP* but *the estimates of welfare costs* (or equivalently, welfare benefits of preventing or reducing toxic water contamination).