Introduction

1 Regulating chemical accumulation: an integrated approach

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The problem under consideration

In July 1980 the European Commission issued a Directive on drinking-water quality (80/778/EEC) setting a maximum admissible concentration for 71 distinct parameters. One of the most strictly regulated substances in the directive was the set of chemical pesticides. The European Commission adopted a 'practically zero' level of permissible contamination for these substances. The limit for any individual pesticide product was set at the trace level of 0.1 μg/l; a 'cocktail' standard for the allowed aggregate level of contamination by all chemical pesticides was set at 0.5 μg/l. These were levels of chemical contamination that were only just detectable under then-existing monitoring technologies. The Commission's standard was intended as a clear and unequivocal pronouncment against the accumulation of chemicals within the drinking water of the EEC.

Despite this pronouncement against chemical accumulation, pesticides have been accumulating in groundwater over the past 15 years to such an extent that several substances have breached the allowed concentration in groundwater in many of the agricultural districts across the European Union (EU) (see, generally, Bergman and Pugh, 1994). This is important because two-thirds of the EU citizenry continue to acquire their drinking-water supplies from untreated groundwater, i.e. directly from the aquifers underlying their communities. In adopting its tough stance against chemical accumulation, it had been the object of the European Commission to stimulate a comprehensive strategy of pesticide management (based on agricultural, land use and pesticide management). However, the continued accumulation of pesticides in European groundwater supplies placed the EU in the position of choosing between two poor options: either the relaxation of its earlier drinking-water quality directive or the costly treatment of groundwater prior to delivery to consumers. The latter option would generate additional costs estimated at around £10 per annum for each consumer of treated water supplies. (Söderqvist, 1994). The former option would entail a substantial loss of political and regulatory credibility. The Commission was caught between two equally unsavoury options.

The EU's approach to the resolution of this dilemma to date has been to do some of each. It has allowed the individual states to select the measures required to meet the directive's standards, in order to allow for cost-effective implementation based upon local conditions (Faure and Lefevre, Chapter 10, this volume). It has also relaxed the 'cocktail standard' for aggregate accumulation, in order to allow for the already observed additional accumulation of chemicals in groundwater sources.

The usual approach of the Member States to the problem of chemical accumulation has been to implement product-specific bans when a specific chemical has breached the EU standards. The disallowance of a market to a chemical found to accumulate in groundwater would seem to be a straightforward method for proscribing chemical accumulation. Once again it would seem to be intended to send a strong and clear signal (at national level) that accumulative chemicals are not to be allowed in use. Nevertheless agricultural chemical accumulation in groundwater supplies continues apace, even in those countries where such bans have already been implemented. The example of groundwater contamination in the maize-growing districts of norther Italy is a case in point (Sbriscia Fioretti *et al.*, Chapter 2, this volume).

The Po River Valley is an important maize-growing district with an aggressive weed problem. In the absence of an active weed control programme, it has been estimated that 31% to 38% of the average maize yield would be lost to weed encroachment. In the 1950s selective herbicide application became the primary mode of weed control, and in 1964 this strategy was extended to maize production in Italy, with the introduction of the chemical atrazine. Atrazine was a stunningly successful pesticide, providing very effective and reliable weed control for many seasons following its introduction. Of course chemical-induced selection implies the need for an evolving weed control programme, and atrazine required supplementation by other chemical products throughout the 1970s. This resulted in increasing volumes as well as increasing numbers of herbicides being applied to the Italian countryside throughout the seventies and into the eighties (see chapter 2, Table 2.3). The level of application of atrazine remained relatively constant throughout this period, even though it was being increasingly supplemented by other chemicals as well.

The Directive on drinking-water quality was finally implemented in Italy on 2 August 1985, and the monitoring of groundwater supplies was initiated on an official basis. As a consequence it was discovered that many of the communities within the agricultural district of the Po Valley were being provided with drinking water containing pesticides (including atrazine) in breach of the EU standard. In order to enforce the standard the relevant authorities (districts) initiated local, then district-level proscriptions on the application of atrazine. These product-specific bans were slow to begin (with 67 000 hectares regulated initially in 1987) but rose to include entire regions (367 000 hectares total) by 1990. Nevertheless these location-specific prohibitions were deemed inadequate and, in 1991, the product atrazine was banned from all sale or use within the state of Italy, both in those areas in which it had accumulated and in those in which it had not. A nation-wide ban of this nature will of course help to reduce the cost of enforcing the prohibition in those areas in which it is most needed. In addition, the perception was that the government was sending a signal to chemical producers and users that accumulative products were not to be tolerated, with the foreclosure of markets to those substances which demonstrably breached these standards. Bans on specific offending chemical products are often hoped to have such broad impacts on the incentives for the use of these and related chemicals (Toman and Palmer, 1997).

Despite the clarity of the policy stance against accumulation, both within the EC Directive and in the foreclosure of markets, there is little evidence that the rate of accumulation of such chemicals is slowing. Continued monitoring identifies wells newly in breach of the guidelines on account of past years of chemical applications; due to prevailing geological conditions, it is possible for maximum concentration levels to be achieved years, even decades, after application has ceased. *Even more alarmingly the newly marketed chemicals frequently exhibit characteristics equally as accumulative as those which they are replacing* (Sbriscia Fioretti *et al.*, Chapter 2 this volume; Mason, Chapter 8 this volume). After the proscription of specified chemical products on the grounds of their accumulative nature, many of the replacement chemicals used in their stead exhibit characteristics which will cause them to appear in groundwater in similar concentrations after an equivalent amount of time. The strong stand taken across and within the EU against chemical accumulation has been having little or no effect on the number or quality of accumulative substances being produced and applied within the Union. This is the primary reason that the EU was forced to relax its 'cocktail standard' on pesticide accumulation. The policy measures preventing the accumulation of specific chemical products are not having the effect of shaping the characteristics of their replacements sufficiently, and one chemical after another is accumulating in the groundwater.

How can it be the case that such strong policy measures have so little impact? It is the object of this volume to explain this conundrum. We hope to demonstrate both the reasons for the inefficacy of existing policies and the essence of an effective approach to regulating chemical accumulation. The remainder of this chapter provides an overview of our approach, and an indication of our conclusions. I recommend that the interested reader read each of the individual chapters to acquire the full story on chemical accumulation and its regulation.

An overview of the volume: empirical studies

Part I of the volume presents two chapters which attempt to dissemble the problematic pesticide into its constituent components. This allows the ensuing discussion to pursue the subject at a more fundamental level. It is not the chemical nature of the products that is problematic nor their widespread use per se, rather it is the specific characteristics of certain chemical products that gives rise to their accumulative nature. Part I of this volume identifies these characteristics, and sets forth an analysis that ascertains their relative contribution to a chemical's use and usefulness. This analysis will then be helpful later in the unravelling of the nature of the policy failures in this area, but initially it provides an excellent introduction to the nature of problematic chemical substances in general and of atrazine (and its substitute substances) in particular.

The first chapter in this section demonstrates both the need for agricultural chemicals and the need for a policy explicitly addressing the contamination resulting from their use (Sbriscia Fioretti *et al.*). Since the 1950s, chemical-based strategies have been the preferred form of weed control, and in their absence it has been estimated that up to a third of crop production would be lost. On the other hand, many agricultural chemicals have been designed in such a fashion as to ensure their accumulation in groundwater. This is because many of these chemicals (herbicides in particular) are designed to use the natural flow of precipitation to transport the chemical from the surface (where it is applied) into the soil. It is within the soil that the chemical then acts upon the germinating seeds and root matter of the weedy plant. In essence, the hydraulic cycle (from atmosphere to surface through soil and other living matter and back into the atmosphere via respiration and evaporation) is used as the transport vector through which the chemical may travel to make contact with the target organisms. For this reason, herbicides have been explicitly designed in order to react primarily with water rather than alternative media (i.e. the atmosphere or organic sphere).

The groundwater contamination problem arises because some of the natural flow of water leaks out of this cycle, and becomes relatively stagnant within various substrata. In these so-called 'sinks' the chemical substance accumulates under circumstances (out of contact with light, air or organic substances) in which it is difficult for further biodegradation to occur. The chemical's natural affinity for water has led it down a dead-end, where it will continue to accumulate so long as degradation and recharge rates are low. Groundwater aquifers are one of those dead-ends in which chemical substances are capable of being found.

For these reasons the two traits of a chemical that are most likely to determine its rate of accumulation within groundwater are: (1) its relative affinity for reacting with water relative to the other basic media (the organic sphere, the atmosphere), and (2) its absolute rate of reactivity or persistence. 'Affinity' is measured by virtue of partition coefficients which determine the rate at which the substance will react with alternative media when simultaneously exposed to them; for example, the K_{oc} coefficient states a chemical's relative affinity for organic carbon and water media. 'Persistence' is usually measured by the amount of time required for the loss of half of the original mass of the chemical substance through reactivity (the chemical's 'half-life'). The product of these two measures is combined into something termed the 'GUS index': a measure of a chemical's in-built propensity for accumulation within groundwater. Clearly, chemical substances with longer half-lives and higher relative affinities for water will have a greater proportion of their initial applications finding their way into groundwater sinks.

Of course both water affinity and persistence are in-built characteristics of useful chemicals. Water affinity provides the substance with its transport vector – to take it where it needs to be. Persistence reduces the need for multiple applications because it allows for the correct amount of the chemical to be on hand at the time that its action is needed. It does this by reducing the rate at which the chemical reacts with non-target substances; i.e. by reducing its general rate of reactivity or biodegradation. Hence, it is no accident that these chemicals accumulate in groundwater; the propensity for accumulation is a by-product of the same characteristics that render the chemical useful. This point is pursued further in an empirical analysis of the demand for the various characteristics of pesticides (Söderqvist, chapter 3, this volume). This study looks more closely at atrazine and its various substitute chemicals, and assesses the relative demand for the various characteristics which distinguish them from one another. The characteristics of useful chemicals examined there include:

(1) Persistence (half-lives).
(2) Reliability (GUS index for pre-emergents).
(3) Effectiveness (kill rate).
(4) Toxicity (lethal dose).
(5) Regulation (banned status).
(6) Age (years on market).

Unsurprisingly, this study demonstrates that the effectiveness (kill rate) of the chemical is the single most important facet of the substance; users are clearly willing to pay more for chemical substances which are more effective in removing the targeted organisms. There are other, more surprising, results from this study, but these will be addressed in the discussion later in this chapter concerning the policy studies regarding atrazine.

At this juncture, the importance of the studies in Part I is to demonstrate the nature and object of chemical design: it is a matter of in-built chemical characteristics related to very specific targets and objectives. The contest between crops and their competitors is an important and continuing one. Agricultural chemicals are not blunt instruments; they are carefully designed to perform specific functions along charted routes through the environment. This section of the volume demonstrates the complex nature of chemical design, and the range of characteristics across which chemical manufacturers must operate (persistence, affinity, toxicity, kill rate). The choices that manufacturers make regarding these various parameters are determined by what makes for a useful chemical substance in the context within which they are used. This implies that chemical accumulation is a linked outcome, not an unintended consequence, of chemical production and application. It is probably incorrect to view the societal objective as the prohibition of all accumulative substances (unless the entirety of the benefits of chemical applications are to be foregone), as opposed to the calibration of chemical design (and application) in order to balance the benefits of chemical usefulness against the cost of chemical accumulation. Part I of this volume details how the various traits of a chemical are demanded in agriculture, and how these same traits can contribute to various forms of unintended, but necessarily linked, consequences such as accumulation in the groundwater. It demonstrates the basic nature of the social problem of regulating the traits that cause chemical accumulation: the trade-off between groundwater purity and chemical effectiveness.

The valuation of resource contamination

Part II of this volume then launches into the problematic region of environmental valuation. In a previous volume (Bergman and Pugh, 1994), we discussed the importance of undertaking a cost–benefit analysis of the EU drinking-water standard for pesticide accumulation, in order to calibrate the cost of the EU policy concerning chemical accumulation against its benefits. In that volume we reported a rough estimate of the cost of the EU policy; as mentioned previously, the cost of removing pesticides from groundwater by the use of granular activated carbon filters in the Po Valley region was estimated to be around £10 per consumer per annum (Söderqvist, 1994). Now we turn to the task of estimating the benefits.

The benefits of removing these trace chemicals are more difficult to calibrate. On account of their extremely low levels of concentration in groundwater, it will require many years of continuous exposure before that exposure accumulates to levels which are toxicologically meaningful. The toxicological procedure for extrapolating an acceptable daily intake (ADI) for any given chemical based upon various indicators such as its acute toxicity is widely accepted and not under examination here, but a large amount of uncertainty must remain in a context such as this one. This is because toxicologists must operate in laboratory environments and on time scales much shorter than a normal human life-span; they are simply unable to replicate the conditions which are prevalent in the environment in assessing their likely impacts. The low level conditions of contamination prevailing in groundwater are hence not discernibly costly under standard toxicological measures; yet a large degree of uncertainty remains, precisely because these measures are not suited to the problem of long-term low dosage induced responses.

Part II describes the toxicological and the economic approaches to risk assessment under conditions of uncertainty. In addition to the uncertainty related to the definition of a toxicologically based ADI, the extremely low levels of the EU standard for drinking water (much lower than any estimated ADI) renders it technically impossible to estimate the risks of such levels of contamination based upon toxicologically relevant considerations. The toxicologists frankly admit that the EU standards are based on foundations other than the toxicological; they are 'philosophically different' from the World Health Organization (WHO) toxicology-based standards, relying upon ethical, technological as well as scientific precepts. The economists, on the other hand, advance the willingness-to-pay criterion for use in this region of profound uncertainty. If consumers are concerned about little-understood hazards such as low level groundwater contamination, then perhaps the best measure of the cost of these hazards is the willingness of consumers to pay to undertake efforts to avoid them. Willingness to pay (WTP) for avoidance of a risk is the preferred measure used by economists to calibrate the magnitude of these sorts of preference across individuals, and the economic analysis of this problem in Part II studies a range of different approaches to the estimation of this measure.

Part II includes two chapters presenting two distinct economic approaches to the assessment of the benefits of avoiding groundwater quality deterioration: (1) a survey of *indirect method* studies for quantifying individual responses to risk (Johannesson and Johansson, Chapter 5, this volume) and (2) a *contingent valuation survey* for the same purpose (Press and Söderqvist, Chapter 6, this volume). The survey on the indirect method studies reports on several different markets which contain risk assessments implicitly, e.g. workers accepting jobs with less risk exposure at lower wages. Empirical studies across such markets once again break down the marketed product or occupation into its constituent characteristics, and then ascertain the relative contribution of each characteristic to the differences in prices between the products. In this fashion the implicit price assessed by consumers to a characteristic such as potential hazardousness may be derived. Several studies have discussed this value with regard to the willingness to pay to avoid an incremental hazard resulting in the loss of one additional life, and found a range of estimates of between US$1 million and US$20 million. This approach is valuable when the relative risks of the alternatives are already known, as it is then possible to weight this risk according to the value that individuals are observed to place on risk avoidance. This is not so helpful in circumstances, such as this, where the risks are relatively low but uncertain.

Another indirect approach to valuing risks that is more applicable in this context is to use observed expenditures that individuals undertake in order to avoid the risk. For example, the risks and uncertainties of groundwater contamination may be avoided in part by, for example, drinking bottled water, installing a water filter or moving to an area with better water supplies. Obviously, some of these are better indicators of the willingness to pay for pure water, and all of them are actions replete with mixed motives. Nevertheless, avertive expenditures provide a market-based indicator of willingness to pay, and useful indicators of the potential value placed on the risk by individuals. Three studies regarding avertive behaviour towards groundwater contamination are reviewed in this volume, indicating WTP values regarding water contamination risks in the neighbourhood of US$1–10 per individual per week. (Johannesson and Johansson, Chapter 5, this volume).

The problem with these market-based indicators is that the willingness to pay measure should be geared as closely as possible to the actual environmental good that is being valued – in this context, pristine groundwater quality. Health risks (actual and perceived) are only one facet of this environmental good. For many centuries Europeans have been able to drink untreated groundwater piped directly from the aquifers, and then into their houses. The advent of intensive agricultural production and the introduction of chemical methods of weed control have now changed this for the first time. The continued application of large volumes of chemical pesticides will make it necessary to introduce drinking-water treatment, as is now the case in the most intensive agricultural districts, and it has denied Europeans something that was part of their natural heritage.

In addition, the loss of the pristine resource is something that the individual citizen might value for reasons other than health risks and uncertainty. There are also its effects on wildlife and other biota, general ecosystems, and general environmental degradation. For these reasons the market-based indirect methods of estimation are far too narrow. The true willingness to pay for pristine groundwater quality must allow for the inclusion of this wider range of characteristics and motivations that might be included in a willingness to pay for the underlying resource. This calls for different sorts of valuation technique.