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► **To cite this version:**

Francois Bonnieux, Alain Carpentier, Robert D. Weaver. Reduction in soil contamination : economic incentives and potential benefits. Seminar : Long-term perspectives for effects of rural land use changes on soil contaminants, Jun 1995, Arona, Italy. 29 p. <hal-01931604>

HAL Id: hal-01931604

<https://hal.science/hal-01931604v1>

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STATION D'ECONOMIE
ET SOCIOLOGIE RURALES DE RENNES

UNITE ENVIRONNEMENT, REVENU DES AGRICULTEURS

65, rue de St-Brieuc - 35042 Rennes cedex - France - tél. (33) 99.28.53.82 - fax. (33) 99.28.53.80

FBC29595

*Long term Perspectives for Effects of Rural Land Use Changes on Soil Contaminants
8-10 th June 1995
Arona, Italy*

INSTITUT NATIONAL DE LA RECHERCHE AGRONOMIQUE
Station d'Economie et Sociologie Rurales
DOCUMENTATION
65, Rue de St Brieuc
35042 RENNES CEDEX
Tél. : 99.28.54.08 et 09

REDUCTION IN SOIL CONTAMINATION: ECONOMIC INCENTIVES AND POTENTIAL BENEFITS

François Bonnieux, Alain Carpentier, Robert Weaver^{1*}

proposal draft June 1995

ABSTRACT

Increasing attention is focused on the problems of soil contamination in the EU. As well as having potential consequences for human health and for air and water pollution, soil pollution is an issue in its own right. Control of heavy metal emissions is a high priority and has resulted in a number of directives to reduce emissions. Pesticides pose a different problem. These chemicals are mainly applied to crops by the multitude of farmers across the agricultural sector and therefore constitute a non-point source of pollution. The limitation of pesticides use by the farm sector is primarily addressed. Various policy instruments including charges and ban are discussed.

Pesticides are used to increase expected output and also to reduce output variability. Their characteristics have implications on policy design which are addressed in this contribution. It is crucial to inform farmers on real pest exposures in order to reduce pesticide use. Information reduces uncertainty and, as a consequence, helps farmers to safely suppress unprofitable sprays. The possible

¹ Authors are listed alphabetically and are all located at I.N.R.A. Rennes. Dr. Weaver is an sabbatical leave from Pennsylvania State University.

DOCUMENTATION ÉCONOMIE RURALE RENNES



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effects of usual economic incentives such as taxes are well known in a certainty world. Their impact is questionable where risk considerations also affect producer's input choices (due to the dependence of current agricultural production on crop protection). *Ad valorem* tax would have a limited effect on pesticide use in the short run. In the long run, this tax may increase farmer's information on pest exposure valuation. Therefore, *ad valorem* tax could improve farmers crop protection and decrease pesticide use in the long run.

The benefits of decreased soil contamination are the resulting improvements in health and in a number of environmental effects (including effects on plants and animals, ecological disruption, effects on habitats and endangered species). The major objective of economic valuation is to transform estimates of changes in physical or biological effects into monetary estimates of benefits. Basic methods for valuing environmental effects (direct cost, travel cost, property value and contingent valuation) are discussed in connection with a reduction in soil contamination.

A third issue is also considered. It is concerned with the possible impacts of the common agricultural policy reform and of the accompanying measures. What are the possible impacts of the reduce in price and of the implementation of environmental schemes on agricultural output? Can we expect drastic modification in land use leading to a reduction in soil contamination?

1. Introduction

During the past three decades increasing attention has been given to the management of flows of pollution. At the opening of the decade of the 1960s, public debate of pollution was launched within the context of the post war expansions of the economies of developed countries. Industrial expansion during this period was typically accompanied by intensification in the use of limited labor. This intensification was often accomplished through expanded use of chemicals and processes that generated readily observable increases in soil, air, and water pollution. Economists quickly responded to the debate by pointing out that social welfare would not, in general, be maximized by policies which attempted to fully eliminate pollution. At the same time, the development of an extensive menu of economic policy approaches was initiated in both the theoretical and empirical literature. Somewhat later, public attention was drawn to the accumulation of pollution in the form of site specific stocks. Chemical processing, urban, nuclear, biomedical, and other forms of wastes were readily observable as polluted stocks. Importantly, the pollution of soil, water, and air by these stocks was recognized as an associated problem of significant gravity. Given the capacity of soils to chemically bond with pollutants, soil contamination emerged as a recognized form of accumulated pollution that constitutes a stock.

While the management of flow of pollution has been extensively studied from both theoretical and empirical perspectives, little attention has been given to the economics of the creation of stocks of pollution such as soil contamination. Clearly, the accumulation process involves combination of dynamic processes of diffusion of pollution through emission, transport, and disposition processes, as well as evolution of depositions of pollution through chemical processes. In contrast to this complication, the policy options for stocks of pollution are expanded by the fact that stocks are typically more readily identifiable than nonpoint

sources of pollution flows. Further, because pollution stocks are typically spatially centralized and associated with land parcels of finite dimension, private property rights exist over pollution stocks.

The objective of this paper is to present an overview of the problem, implications, and policy solutions available for management of stocks of pollution such as soil contamination. While the problem of developing policy for soil contamination is inherently more complicated than that for pollution flows, the dependence of accumulated stocks on flow processes implies flow pollution must be considered and will be discussed throughout the paper. The paper proceeds with four sections. In the next section the basic microeconomics of flow pollution is viewed and its implications for stock pollution such as soil contamination are reconsidered. In the next section, the question of soil contamination is considered within the case of pesticide use to provide a more detailed consideration of the economics of use of polluting inputs. Further, within the context of pesticides the opportunities for pollution flow policy approaches are assessed. In the fourth section of the paper, the problem of evaluation of the benefits and costs of pollution stocks such as soil contamination is reviewed. While the fundamental economics of management of pollution stocks is known at a theoretical level, in practice specification of public policy requires the estimation of social benefits and costs of reduction in stocks. The discussion in section 4 highlights the substantial complications involved in such estimation. The final section considers the economics of public policy to alter soil contamination. Here, policies focused on pollution flows are outlined as well as are approaches for prevention of stock accumulation and restoration of polluted sites. Current European policy is reviewed.

2. Microeconomics of soil contamination

This section introduces the basic concepts of pollution economics which are relevant to deal with soil contamination issues. The conflict between private and public interests is made clear in the first paragraph. This is a central theme in all analyses addressing environmental problems. The discussion of the concept of efficient contamination is based on a static analysis. This is relevant to deal with flow pollutants and most cumulative pollutants as far as stocks decay at a regularly rate. The second paragraph considers the relationship between economics and important concepts such as liability, compensation and property rights. The trade-off between efficiency and equity is also addressed. In dealing with hazardous release and major disruptions of the ecosystems, the third paragraph considers a different perspective since the static analysis is no longer relevant, uncertainty and irreversibility are then discussed in a dynamic framework.

2.1. Efficient pollution

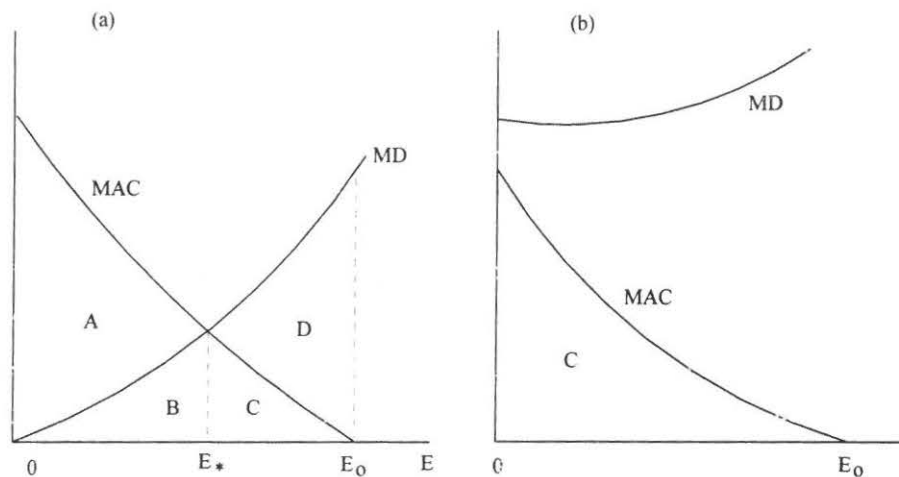
We begin by considering a very simple model that lays out the basic concepts relevant for consideration of the economics of pollution in general. It consists of a simple trade-off situation that characterizes all pollution-control activities : (i) reduction of emissions reduces the damages that people suffer from soil contamination (ii) reduction of emissions takes resources that could have been used in some other way.

To illustrate this trade-off consider a single farm (the emitter) which is contaminating soil leading to residuals into a river. These residuals have a negative impact on the activity of an aquacultural farm (the receptor) which is located downstream. This impact is expressed in monetary terms with a damage function. Upstream, the offending farmer could reduce the amount of effluent put in the river by abating pollution (or controlling pollution). Abatement includes all the many ways there are of reducing emissions : diminution in output, changes in production technology, input switching, residuals recycling, treatment, abandonment of a land. This act of abatement will require resources of some amount, the cost of which will affect the costs of production of the emitter.

Marginal abatement costs (MAC) and marginal damage function (MD) are shown in figure 1. Marginal abatement costs show the sum of all costs of achieving a one-unit decrease in emission level, or alternatively the costs saved if emissions are increased by a unit. On the horizontal axis, MAC originates at the uncontrolled emission level E_0 and slopes upward to the left. The marginal damage function shows the changes in the sum of all damages realized by society which stem from a unit change in emissions and increases with the emission level. For a fund pollutant MD originates at the origin of the axes whereas for a cumulative pollutant it originates at a point above on the vertical axis.

Because damage and its associated costs are externalities which are not borne by the emitter, it is optimal for the offending farmer not to abate emissions. Assuming marginal cost of abatement is incurred only by the polluter, Figure 1a illustrates why a profit maximizing polluter will choose the level of emissions E_0 . This private choice leads to an excessive contamination relative to that which is optimal for society. A marginal decrease in emissions from E_0 leads to an increase in abatement costs which is compensated for by a decrease in damage costs because MD is greater than MAC. As illustrated in panel (a) of Figure 1, the optimal level of abatement from society's perspective may be positive, e.g. at E^* . However, in many cases, as illustrated in panel b of figure 1, that optimal level may be zero. In both cases, the level of pollution that is optimal for society (E_*) is always less than that which is optimal for the profit maximising producer.

Figure 1. Efficient emission levels



To elaborate further let us consider panel (a) in more detail. If the emission level is E_0 , total damage costs are given by $B + C + D$ and there is not cost of abatement. If the emission level is E^* , total damage costs are given, by B and total abatement costs are given by C , thus there is a gain equal to D . The efficient level of emissions is defined as that level at which marginal damages are equal to marginal abatement costs. Higher emissions expose society, to greater costs stemming from environmental damages. Lower emissions involve society in greater costs in the form of resources devoted to abatement activities. Zero pollution is not generally required to achieve an efficient allocation for the whole society. The sum $B + C$ is a measure of the total social costs from E^* .

The difference between the private optimum E_0 and the social optimum E^* illustrates a failure of the market mechanism. This misallocation is not surprising as far as natural assets are treated as common-property resources in our legal system. The general conclusion is that soil is over exploited as waste repositories.

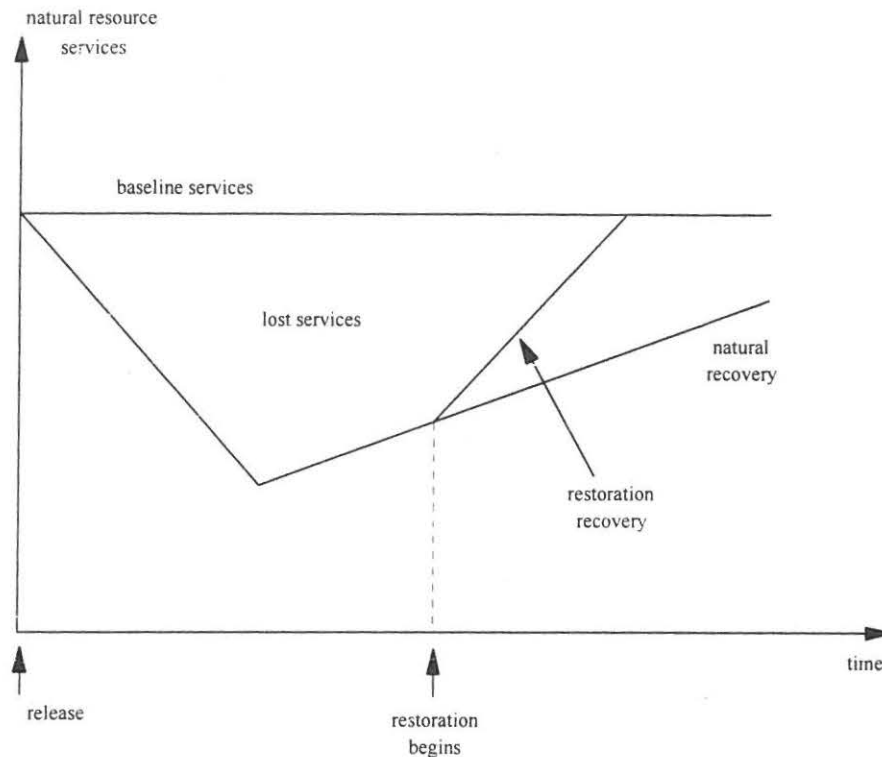
2.2. Liability, property rights and equity

One approach to environmental issues and more specifically to soil contamination is to rely on liability laws. The purpose of these laws is twofold (i) to compensate people after they have been injured (ii) to get emitters to make careful decisions. Consider again figure 1a and suppose that the actual emission level is E_0 above the efficient level E^* . Suppose further that a liability law requires the emitter to compensate the receptor in an amount equal

to the damage caused. At E_0 the amount of the compensation would be $B + C + D$. The emitter would be better off by abating emission up to E_* and then paying a smaller compensation amount equal to B . In theory then, a liability system could lead to efficient contamination. But its implementation requires a system of courts and transaction costs are a severe limitation to this approach. In general terms, transaction costs are the costs of reaching and enforcing agreements. They cover all the legal costs associated with gathering evidence, presenting a case, collecting damages ...

An alternative approach is to clearly define who has the property right over the environmental asset and then to allow bargaining among owners and prospective users. R. Coase (1960) held that, as long as negotiation costs and income effects are negligible, decentralized bargaining achieves the efficient level of emission. It must be emphasized that this outcome does not depend upon who has the right. Instead, the entitlement of property rights affects the distribution of costs and benefit among the involved parties. To make this point clear consider again figure 1a (i) if the emitter holds the right over the environment, a compensation is paid by the receptor in order to reach E_* (ii) if the receptor holds the rights the emitter has to pay to achieve the same outcome. The distribution of wealth is therefore conditioned by the allocation of property rights. Furthermore significant income effects or negotiation costs would lead to a shift in marginal abatement costs and marginal damage costs. This implies that the outcome of the bargaining process depends on the initial allocation of property rights. As we see this property rights approach may not work well when faced with the complexities of the real world.

Figure 2. Loss of natural resource services after a release and their recovery rate



2.3. Restoration issues

The economic costs of a one-time hazardous substance release are of two types (i) the costs associated with foregone or lost service flows and (ii) the costs associated with restoration (rehabilitation) or replacement of the resource. Because there is a trade-off between restoration costs and reducing lost service flows, the question arises as to the efficient level of restoration in a given case.

An important point in any quantification of damage to the environment is the resource recoverability as measured in reference to baseline level services, i.e. the conditions existing before the release. It corresponds to the amount of time needed for an injured resource to recover and reach its initial level. The ecosystem recovers at a natural recovery rate which depends on the magnitude and the toxicity of the release. This rate can be accelerated by human operations, however, the crucial issue is concerned with the choice of level of restoration (Ward and Duffield, 1992).

Figure 2 illustrates a simple time profile which describes the loss of natural resource services after a release. The baseline service could be reached after a timespan which depends on the natural recovery rate and human action. Because clean-up activities

represent the largest component of social costs of any large hazardous release, these activities raise a major question for decision - makers : how many goods and services have to be used to restore the environment ? This topic has been discussed at length in the case of oil spills (Bonnieux and Rainelli 1993a, OECD, 1982). Basically, there are three different positions taken in any debate of the optimal level of cleanup effort::

- . the laissez-faire (the position of people who are responsible for the injury),
- . the full restoration (the position of conservations and of people who suffer damages).
- . the rational clean-up effort (the position of economists).

Laissez-faire and full restoration are corner solutions to the restoration problem. Since there is a trade-off between increasing clean-up costs and reducing losses in service of natural resources, the problem is to determine the efficient level of clean-up effort, in order to achieve rational clean-up effort. The design of public policy to address the problem of hazardous substance releases has both distributive and efficiency implications. The goal of providing the injured party with an equitable compensation may conflict with the goal of efficient use of resources.

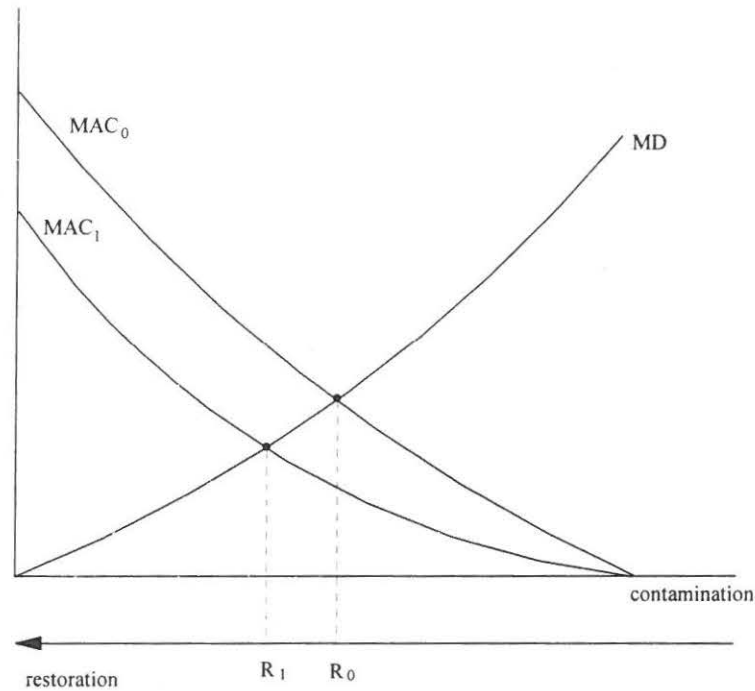
The static analysis was presented in figure 1 adressed pollution in general. The problem of cleanup is complicated by dynamic and uncertainty dimensions. (Bonnieux and Rainelli, 1993b). Basically, in a major emergency it is not possible to state with certainty the consequences of a particular measure of restoration. Will a given set of measures actually achieve their goal ? On the demand side, is it certain that the services from an injured resource will actually be desired in the future? Uncertainty on demand side and supply side poses the problem of option value. Furthermore the risk that the action taken may be too much rather than too little since public decision makers are risk averse and fear public discontent can lead to economic inefficiencies. This topic is of major concern where there are sensitive areas and where public health can be affected. Three issues must be taken unto account :

- (i) decisions are taken under uncertainty, (ii) the decision process is sequential (iii) some choices can have irreversible consequence.

The concept of quasi-option value is instrumental to rationalize this situation (Henry, 1974). It refers to the increase in expected net benefits of not undertaking a project which could have irreversible consequences. In emergency response to a major contamination it is the opposite, since the laissez-faire or a limited clean-up effort could lead to irreversible consequences. In this case the emergency response is always more costly than it would

have been with perfect information, but it induces a better protection of public health and a better saving of natural resources. This point is made clear with figure 3. Due to an increase in information there is a shift to the left of marginal abatement cost curve (or marginal cost of restoration) from MAC_0 to MAC_1 . In a two-period analysis, this leads to a decrease in the optimum level of restoration, from R_0 to R_1 towards more cleanness. The area between MAC_0 and MAC_1 can be interpreted as a monetary equivalent of the information gained between the two periods (quasi-option value).

Figure 3. Efficient restoration in a two-period analysis



3. Agricultural Pesticide Use as a Case of Study

Pesticide use has dramatically increased post World War II in the EU. For example, the monetary value of pesticide sales in France doubled during the sixties as well as the seventies and further increased by another 75% in the eighties (Brouwer et al, 1994). Exact data of the sales of pesticides for use by sectors are usually not available. This section focuses on the use pesticides by farmers. The case of the agricultural sector is interesting in the context of soil contamination for at least three reasons.

i - This sector likely uses the most part of the pesticides sold in the EU.

ii - It uses a large part of the EU members' surface.

The combined effect of these first two points is that the EU agricultural sector can be considered as one of the main contributor of soil contamination by pesticides. This provides valuable arguments to the EU public decision makers for the reduction of agricultural pesticide use.

iii - It is of peculiar interest because the agricultural sector pesticide use is essentially motivated by economic incentives.

This last point is at the core of the pesticide pollution regulation problem. While it is socially valuable to reduce pesticide use, farmers treat their crops because they find pesticide use privately profitable. In this context, public decision makers must evaluate the trade-off between the social costs and social benefits of pesticide use. The social costs benefits of pesticide use are supported by the entire society through environmental quality depletion and by the potentially polluted agricultural product consumers. The social costs benefits of pesticide use are shared by both farmers and agricultural product consumers. Farmers treat their crops because it increases their revenue. Moreover, pesticide use increases the agricultural production efficiency and, as a consequence, lowers the agricultural products market prices. This price effects are obviously favorable to consumers. Another important aspect is linked to this last point. Pesticides are used by many farmers on large areas. In this case, a restoration policy is impossible from a concrete perspective. However, the economic aspects of farmers' pesticide use provide valuable arguments in favor of the implementation of a regulation policy using economic incentives.

Within the context of agricultural use of pesticides, it is of interest to consider what kind of policy instrument should be implemented to reduce pesticide use to a given level? To begin, motivation for farmers' pesticide use must be understood. Four aspects of pesticide use microeconomics are important to note and relate to potential economic instruments.

3.1. Basic microeconomics of pesticide use and price policies

Pesticides are usually defined as damage control agents (Lichtenberg and Zilberman, 1986). They are used by farmers to protect crops against pest and disease damage. Their use increases agricultural output. However, it also can improve agricultural production quality. For example, fungicides are sometimes applied on fruits for esthetic purposes. These products must achieve visual standards to be sold on fresh markets. In this context, farmers apply pesticides if treatment cost remain below the value of the eliminated damage.

This emphasizes the key role of relative treatment cost in the farmers' pesticide use decision process.

A simple way to reduce farmers' pesticide use is to increase the cost of treatment by imposing *ad valorem* taxes on the price of this input. Taxes on input prices are easily implemented by public decision makers. Moreover, collected taxes can be redistributed to farmers in order to overcome their revenue losses, at least partially. The required tax level depends on the responsiveness of farmers' pesticide demand with respect to pesticide price changes. In the case of French crop production, a 1% increase in pesticide price is estimated to lead to a decrease in pesticide demand lying around 1.4% in the short run (Carpentier, 1995).

Relative prices of agricultural outputs are also essential determinants of farmers' pesticide use decision. An increase in output relative price tends to augment the value of pest damage and, as a consequence, pesticide productivity. In the case of the French crop sector, a 1% decrease in agricultural product relative price is estimated to lead to a decrease in pesticide demand of about 1.3% in the short run (Carpentier, 1995). In the EU context, this output price effect has played a major role, at least in the crop sector. The main instrument of the Common Agricultural Policy (CAP) implemented at the beginning of the sixties was a high price support for farmers. The combined effects of high output prices and arable land constraint led to the use of large amounts of industrial inputs such as pesticides and fertilizers. The recent CAP reform, although implemented to control the EU agricultural production increase, should lead to a decrease in pesticide use. The set-aside programs reduce planted areas, the most treated surfaces, and, as a consequence, global pesticide use. Price reductions, as shown above, should also decrease pesticide use.

The description of the above price effects assumes that the implementation of various policies does not affect market equilibrium prices. Observations of the pesticide price evolution induced by the CAP reform show that the pesticide demand decrease results in some pesticide market price decline. This last effect has substantially compensated the agricultural price reduction effect on pesticide demand. Moreover, the results presented in this paragraph assume that the economic context changes induced by price policies do not affect farmers' technology choices. The following paragraph shows that this assumption may lead to an underestimation of price effects on pesticide use.

3.2. Technological changes and pesticide policies

To illustrate the relationships existing between the economic context, technology adoption and pesticide use we develop the example of the effects on the CAP on the EU crop sector.

The output price support led to an increase of the real cost of land for which there is a physical constraint. This provided incentives to adopt techniques allowing high land productivity, namely intensive cropping technology. In case of winter wheat, this technology is characterized by large seed density, high yield seed variety, early sowing, ... (Meynard, 1991). In this context, pest and disease damage control becomes a crucial issue because the use of the techniques described above makes crop more sensitive to pest infestations. For examples, long growing seasons and large seed densities increase the likelihood of pest and fungi damage. Also, the use of intensive cropping technology creates needs for pest control.

This rapid description of the currently used cropping technology suggests two potential policy to decrease farmers' pesticide use. The first one is suggested by the dependence of the intensive cropping technology on plant protection. Pesticide use would certainly decline if farmers adopted less intensive growing techniques. In this context, the price reductions induced by the CAP reform can be seen as a first step in this direction. This policy partially eliminates the main incentive to use intensive cropping technology : high output prices. However, it should be noted that the set-aside programs strengthens the land constraint, the other main factor leading to intense use of land. The second one comes from the fact that pesticide use is not the only available mean to protect crops. The implementation of *ad valorem* taxes on pesticide price would certainly promote the use of alternative plant protection techniques by farmers. Crop rotations and biological pest management are often cited as potential substitutes for chemical pesticides applications. However, further agronomic and biological researches are needed for alternative plant protection methods to reach the actual efficiency of the chemical substances intended to prevent or combat pests (Byé *et al.*, 1991). Adoption of alternative pest control methods and/or less intensive cropping techniques would constitute a significant technological change for farmers. It can only be a long term objective.

So far our discussion has ignored the specific properties of pesticides. The effects of pesticides on agricultural production differs from those of conventional inputs. In the third and fourth paragraphs we develop this point and its implications on pesticide policy design.

3.3. Pesticide use and information use promotion

The productive effects of pesticides are measured by a reduction in the damage resulting from pests and crop disease, rather than by an increase in potential production. Due to the interactions of plant and pest growth with weather and other uncontrollable phenomena, damage induced by pest infestation occur randomly and can not be easily predicted. Therefore, the way farmers anticipate the productive effects of pesticides is

important. It defines the probability distributions used by farmers in their expectations. It depends on the amount of information they use when they choose pesticide sprays.

In developed countries agricultural scientists recognize that farmers generally apply pesticides following predetermined schedules, that is they apply fixed pesticides dosages at fixed dates without regard of the actual conditions prevailing in the fields. This shows that existing pesticides are (very) effective in controlling many serious threats to production and, with respect to this efficiency, are relatively cheap.

This simple analysis suggests that farmers would reduce their pesticide use if they were using the information generated during the production process. In fact, the information on field pest infestations (expert predictions, intermediate output levels, disease symptom observations, insect and weed scouts, ...), can be seen as an input. Its use would allow farmers to save the cost of unuseful sprays they would have applied following a rigid schedule. Therefore, it can be considered as a substitute for pesticide use in pest management processes.

This suggests the promotion of information use to be a potential policy of pesticide use reduction. This policy could be implemented through two means. Firstly, low human capital is often cited as a key obstacle to information use because it increases the information cost to farmers. Consequently, education programs intended to increase the human capital of farmers may be interesting pesticide use reducing instruments. Secondly, the United State experience shows that professional pest control consultants can concretely provide valuable information to farmers interested in improving their pest management. For example, pest control consulting (public or private) firms pest information could sell advises to farmers. In this case, the combination of two instruments seems adequate. An *ad valorem* tax on pesticide price would increase the demand of pest control information because it would increase the cost of unuseful sprays for farmers. Simultaneously, it would provide financial funds to subsidize the creation and/or the functioning of private or public pest control consulting firms.

The next paragraph deal with another aspect of lack of information use : risk considerations.

3.4. Pesticides as self-protection against production risk

Due to lack of information use, farmers almost always apply pesticides as a precaution. Because they totally or partially eliminate pest and disease damage, one of the main source of production risk, pesticides can be considered as risk-reducing inputs. Hence, farmers' attitudes toward risk need to be taken into account to understand pesticide demand.

An empirical study of a 496 French farmers growing crops sample shows that risk aversion results in a significant pesticide over-use (Carpentier, 1995). It finds that 70% of the considered farmers applies pesticides over their expected profit maximizing level following a self-protection strategy against production risk.

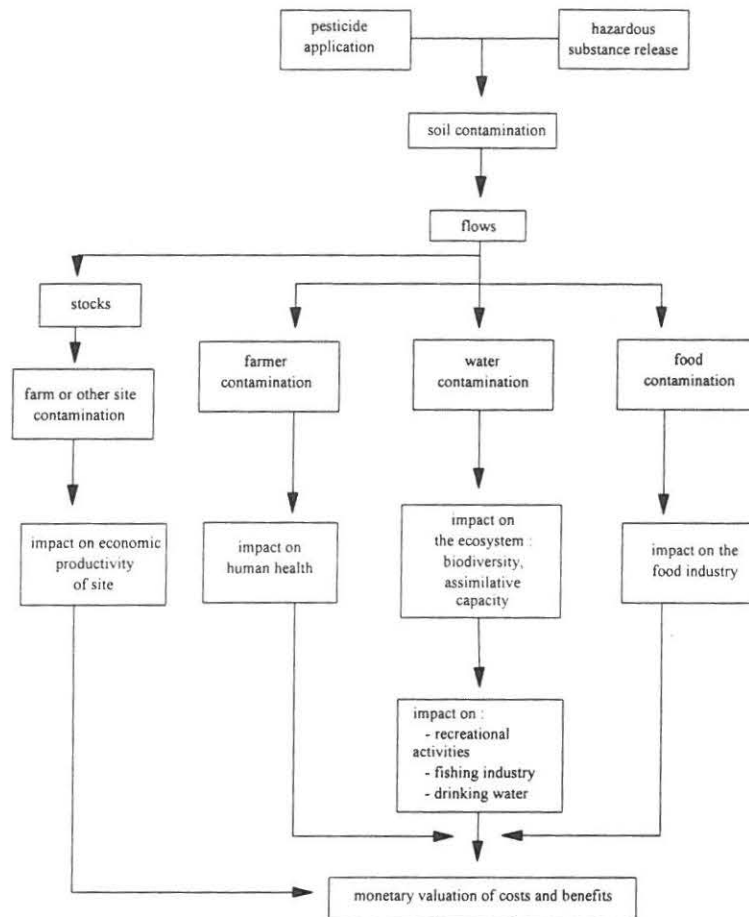
These results suggest financial crop insurance to be another potential policy to reduce pesticide use. This instrument would only be intended to replace pesticide over-use by insurance contracts for the profit variability elimination or reduction. It would have a limited impact on the less risk-averse farmers. Moreover, it is subject to moral hazard and/or adverse selection effects as shown by many studies dealing with the American Federal Crop Insurance Program (see e.g. Horowitz and Lichtenberg, 1993).

4. Estimation of social costs and benefits

4.1. A typology of costs and benefits

Figure 4 illustrates the chain of events that relate sources of soil contamination (e.g. pesticide application or hazardous substance release) to the monetary valuation of costs and benefits. This chain embodies complex physical and biological linkages well as the economic valuation process.

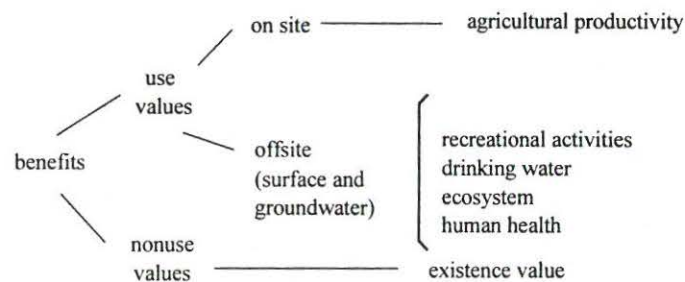
Figure 4. Economic impact of soil contamination



In general terms, the benefits of decreased pollution are the resulting improvements in health, esthetics, and reductions in damages to plants, animals and materials. Costs refer to the negative impact of increased pollution. The distinction between benefits and costs (or damages) lies in the choice of a reference point from which environmental changes are to be measured. The analysis of benefits and costs should cover the entire range of benefits and costs, from those that can be assigned a monetary value to those that can only be described qualitatively. In the recent period economists tried to extend monetary valuation to such things as human health and biodiversity which are not marketed. This economists' inclination for monetary measurement is often criticized. But real world decision making requires trade-offs and economists' effort to deal with non-marketed goods is useful. This does not imply that the public making decision process is only based on economic valuation. Other arguments from social and political approaches are also relevant.

Figure 5 emphasizes the benefits of soil restoration. This classification contains a certain element of arbitrariness but it recognizes some basic features which are embedded in most classification systems (Freeman, 1993).

Figure 5. The benefits of soil restoration



First, soil restoration yields on-site benefits which are of direct use value to humans e.g. in terms of agricultural productivity. Second, the interaction between soil quality and water resources is responsible for the functional services which provide off-site values, that is the indirect support to people economic activity and to the functioning of the ecosystem. Recreational activities such as fishing and bathing has to be mentioned, the provision of drinking water is also a crucial issue.

Benefits also stem from non-use values (existence value). These are values that are not derived from direct or indirect uses neither of the soil nor of the linked water resources. Thus, the flora and fauna that these resources support, may be valued even by humans living far away from the contaminated spot. Even if these people never actually visit the site, they may nevertheless feel a sense of benefit if restoration is carried out. Five altruistic motives for existence value have been suggested (e.g. Johansson, 1987) (i) bequest motives (future generation well-being) (ii) vicarious motives (benevolence toward relatives and friends) (iii) sympathy for people and animals (iv) environmental linkages (v) environmental responsibility.

Another perspective is provided by considering the characteristics of the goods with which benefits are commonly associated. Some benefits belong to the class of public goods (non rival goods). For this category it is generally argued that there are most efficiently provided by public agencies because such benefits will not be supplied by private markets in the absence of positive prices. Another important feature is excludability. This is related to

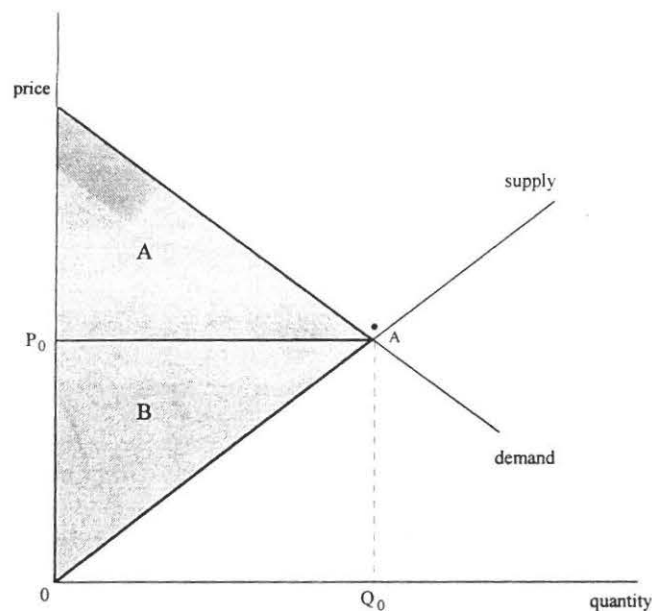
the feasibility and enforcing property rights in a given resource. Water is an important resource for which excludability is problematic.

A last perspective is related to uncertainty. The concept of option value (and option price) has received considerable attention in the literature to deal with cases where there is uncertainty on demand and supply. In a *ex ante* analysis the valuation of soil restoration could be extended to a risky world.

4.2. A conceptual approach

Due to both theoretical and empirical grounds, quantity demanded for most goods is a declining function of price, as in figure 6. Given this relationship, at a given market price level, such as P_0 , only the last unit consumed is worth this amount. The net economic benefit derived from consumption is given by the area A, above the price line and below the demand curve. This is the consumer surplus. From the supply side, producer surplus (or rent) is defined as the return to producers in excess of what is needed to get them to supply a given quantity of output. This is defined as the excess of price over firms' production costs and is shown on figure 6 as the area B, below the price line and above the supply curve.

Figure 6. Consumer and producer surpluses

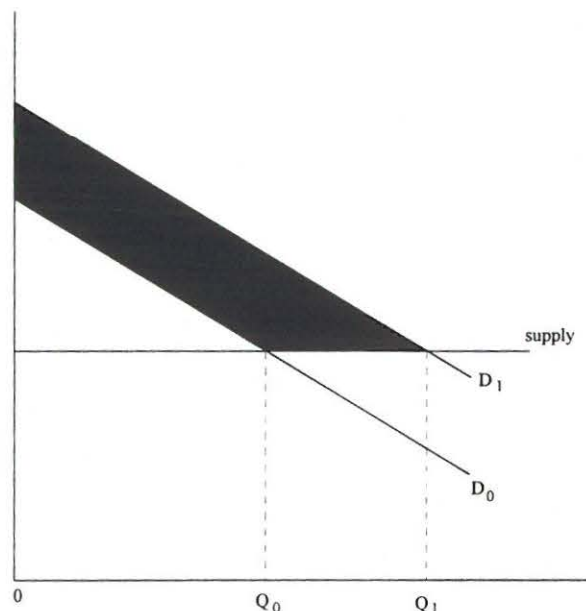


At market equilibrium given by point A in figure 6, quantity demanded equals quantity supplied. The net economic benefit derived from production and consumption is composed of two parts; consumer surplus and producer surplus (A+B). From the standpoint of society,

competitive market mechanism leads to an equilibrium for which social (or total) surplus is maximum. In this sense the mechanism is efficient.

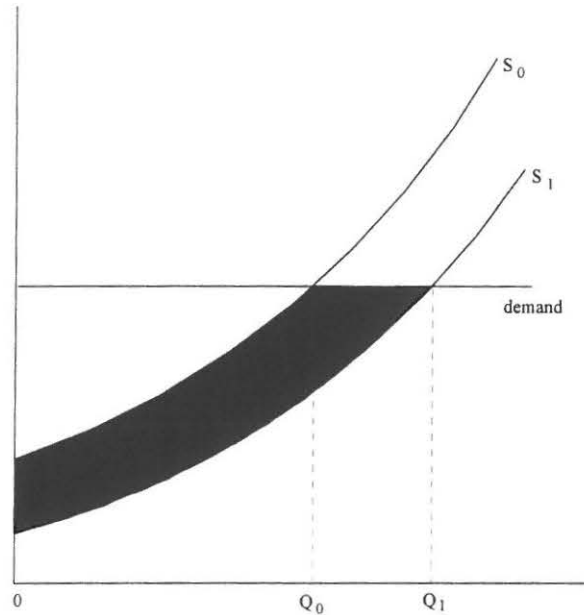
Environmental improvement and environmental degradation can effect people in many ways. For example, soils restoration can improve health or it can increase outdoor recreation opportunities. The satisfaction that people experience because of the improvement is a measure of the benefits of restoration. Assuming that individuals are aware of the effects of restoration, these benefits can be expressed in monetary terms by identifying individuals' willingness to pay restoration. This is illustrated in figure 7 where it is supposed that restoration has led to a shift in demand for a given activity e.g. angling from D_0 to D_1 . In this case it is assumed that price is constant the horizontal supply curve reflects the fact that additional anglers can fish at constant cost per individual. The gain in consumer surplus is shown by the cross-hatched area.

Figure 7. Consumer willingness to pay restoration



Producer surplus could be affected by restoration in several ways. Figure 8 shows the case where the supply curve has shifted down to a decrease in production costs. For example, because of changes in water quality, the costs associated with using this input may decrease for a given aquacultural farm. The gain in producer surplus is shown by the cross-hatched area providing by the demand curve is horizontal.

Figure 8. Producer willingness to pay restoration



The preceding approach can be extended to deal with real world situations where prices and quantities are changing but the rationale remain the same. Variations in consumer and producer surpluses provide a measure of improvements (or damages) associated with increases (or reductions) in environmental services. It is now apparent that in order to measure surplus variations one must have an estimate of the supply curve for producer surplus or of the demand curve for consumer surplus. because most problems in environmental economics are related to nonmarket resource services, specific methodologies were developed in the recent decades.

4.3. Methodological guidelines

The major division is between cost estimating methodologies and lost use valuation methodologies. The estimation of restoration costs is a conventional issue. It is based on engineering data and accounting principles to derive actual costs to restore, replace or acquire equivalent resource. Major difficulties concern benefit valuation, table 2 categorizes available methods according to how closely they are related to actual working markets.

Table 1. Methods for measuring restoration benefits

Market based	Factor income damage function
Related market	Travel cost hedonic pricing
Hypothetical market	Contingent valuation

Market based methods are of a limited application because there are to be few examples of actual markets for the involved services. Where resource is input to marketed product the factor income method can be applied to determine the variation in producer surplus. For example, this approach works for valuing the effect of soil restoration on agricultural productivity. The damage function method belongs to this category. This is a two-sept procedure. First, a dose response between exposures to a pollutant and receptors is estimated from field or laboratory data. The expected changes are then converted into monetary units. Ideally, these should be based on the observed market prices of goods and services.

The travel cost method was developed as a technique to value public recreation sites. It identifies the relationship between visits to a site and the travel costs of these visits. Because these costs vary spatially it is possible to infer a demand relationship for the site. The related market in this case is the market for travel services to the site. By identifying the effect of restoration on the demand for the site, benefits can be estimated based on the difference of the with and without restoration cases. The method can provide information only on changes that have a direct effect on the site preferences of recreationists. Specification and estimation procedures can have a significant impact on benefit estimates. Moreover there is no satisfactory estimate for the value of time which is a crucial issue.

Hedonic pricing applies in case where natural resource services are an important attribute of a marketed good such as houses and land. This method seeks to identify the indirect linkake between environmental quality e.g. soil restoration and the market price of a good e.g. land by estimating the implicit price of environmental quality. Changes in this implicit price can be interpreted as individual's willingness to pay for environmental quality. The hedonic method uses market data on property sales prices and labour wages. The assumption necessary to interpret the results as benefit measures are restrictive and in some cases implausible.

Due to its great flexibility the contingent valuation method became very popular among environmental economists during the last decade. It is based on hypothetical market, therefore it can deal with every category of value. This is the only one which is available the the non-use value case. The contingent valuation method uses direct surveys to elicit individual preferences for changes in environmental quality. In contrast to other methods which are based on actual behaviour, it reveals preferences in conditional choice situations that are depicted by the researchers. For example people would be asked their willingness to pay for a given effort of restoration. Obviously by this conditional setting can cause various problems that may bias benefit estimates. Recently in connection with the wreck of the Exxon Valdez in Alaska the debate focused on the ability of this method to estimate existence value (Hausman, 1993).

There are many situations where, for money or time reasons, it is not possible to do a site specific study. The use of valuation results from a comparable situation can be relevant but could be restricted to minor contamination cases. Application of this transfer approach requires the assumption that the resource, user and experience are in fact comparable. Because benefit transfer is a relatively low-cost procedure and can be implemented in a short time period it is receiving increasing attention (see Water Resources Research vol. 28, 1992).

Table 3. Unit day value (1992 FF) for recreational activities

picnicking	30-120
bathing	70-130
motorboating	50-230
rowing	80-350
waterfowl hunting	110-190
trout fishing	100-150
sea trout and salmon fishing	150-350
carp and perch fishing	80-130

Source : Amigues et al. 1993.

The unit day value is similar to benefit transfer, except that average values are used rather than specific estimates of a given study. Because unit values are based on average

values, their use implies that the site in question is average. In the US the Water Resources Council and the Forest Service have developed unit values that are specific for activity types. Based on extensive review of the literature, Amigues et al. (1993) have suggested a range of values for some recreational activities (table 2). These only concern use value as it is not possible to suggest unit value for non use benefits and costs. One of the biggest challenge is to determine the population affected by the environmental improvement, this is the issue of aggregation. In connection with this issue it must be emphasized that daily activities such as taking a walk or walking the dog are not highly valued but concern a large number of people. Therefore they account for large aggregate amount of money.

5. The Economics of Public Policy and Soil Contamination

The origins of soil contamination have been discussed in previous sections. In summary, two types of origins of soil contamination are important to note: point sources and nonpoint sources. Point sources result from handling, application, or storage of private inputs such as chemicals or biological products at a particular site. Typically, the site is the location at which a particular technology is operated. In agriculture, use of chemical inputs has a high potential for resulting in point source soil contamination. In handling or storage of this type of potentially polluting input, accidental loss may result. In application of these inputs, timing and conditions of application may result in accumulation of the input in soils at the site. Nonpoint sources of soil contamination have been discussed as resulting from nonpoint pollution of air, water, or soil that is then transported to a site of deposition. In contrast to point sources, sources of nonpoint soil contamination are not specifically identifiable. In considering the potential roles of public policy in managing soil contamination, the physical characteristics of soil contamination must also be noted. In particular, while soil contamination can be labeled in economic terms as a stock, the stock results from the accumulation of flows of pollution which become bound to soil. From these perspectives, we can identify a variety of bases for public policy to manage or alter the level of soil contamination. First, existent stocks of contaminated soil may be affected by policies which focus on *restoration* of contaminated soils. However, this policy role focuses on the product of contamination, rather than the process that causes contamination. The second role for public policy focuses on the contamination process and can be labeled as *prevention* policy. In this section, we briefly review alternative policy approaches for restoration and prevention of both point and nonpoint soil contamination.

5.1. Public Policy for Restoration of Contaminated Soils

The role for public policy in restoration of a contaminated site follows directly from a consideration and comparison of private vs. public interests in restoration. Because restoration focuses on a stock of soil contamination, it is useful to view the contaminated site as an asset that has negative value. From both private and public perspectives, this negative asset generates negative returns which create incentives for disinvestment in the asset through a process of restoration. In addition, the returns generated by the negative asset, e.g. a contaminated site, may be uncertain. In this case, an element of risk will be associated with the extent of the negative return associated with the polluted site. As an example, the human and animal health impacts of a particular soil pollutant may not be completely known, leaving the possibility open that a range of possible effects might occur. In this case, the negative returns associated with the site can be said to be *uncertain*. Given scientific knowledge, even an expert could not predict the level of effects that would result. Alternatively, the soil pollutant's effect on human and animal health might be related to a randomly occurring event, e.g. flood inundation. In this case, the probability of the random event might be predictable allowing the probability of an effect's occurrence to be predicted in a probabilistic sense. Here, the negative returns could be described as involving a measurable level of *risk*. In either case, private and public preferences for risk would provide a basis for assessing the optimal level of restoration of a contaminated site. Thus, two bases may exist for private and public choice to restore a contaminated site: 1) the level of soil contamination and associated effects and 2) the risk associated with the effects of the soil contamination.

From a private perspective, the optimal level of restoration of a contaminated site will depend upon a comparison of the marginal benefits and costs resulting from restoration. However, because restoration focuses on the stock of soil contamination and the flow of negative returns associated with that stock, the economic effects of restoration will alter the flow of negative returns over a time horizon that extends into the future. Thus, the decision to restore a contaminated site will consider the future stream of benefits and costs, valued in the present time period's currency. This translation of future economic values into present time period values depends upon the discount rate, or the opportunity cost of realizing an economic benefit in the future versus in the present period. Where the future stream of economic net benefits is uncertain, the decision will necessarily take into consideration this uncertainty through a consideration that involves private or public preferences for risk.

The economic rationale for a role for public sector intervention to restore contaminated sites follows directly from differences between private and public objectives and incentives. From a private perspective, only private benefits and costs will be considered and where uncertainty exists, only private preferences for risk will affect the decision to restore a site. Finally, from a private perspective, the discount rate relevant for translating future values into present values would be a determinant of private restoration decisions. From a public perspective, all benefits and costs that would accrue to society would be relevant to restoration decisions, as would social preferences for risk and the social discount rate. Thus, three bases can be distinguished which may lead private and public decisions to restore a contaminated site to differ: 1) differences in benefits and costs, 2) differences in risk preferences, and 3) differences in discount rates. Typically, the nature of these differences jointly imply that private actions for restoration will lead to less restoration than will be optimal from society's perspective.

As an example, consider the private decision to restore a contaminated site where leaking storage tanks for a pesticide resulted in contamination of ground water. From the private perspective, the benefits of restoration may be very small while for society the benefits may be very large. The costs of restoration may also differ. From a social standpoint, economies of scale and information may be such that costs of restoration may be considerably smaller than for an individual. Further, the discount rate for the individual may be very high compared to that of society, indicating that, relative to society in general, the individual would more strongly prefer to consume economic benefits in the present than in the future. Combined, these differences between private and public decisions to restore will lead the individual to find optimal restoration expenditures to be substantially smaller than the level that is optimal for society. It is this difference that provides a rationale for a role for public sector policy. Clearly, from a private perspective, the absolute extreme limit of willingness to pay for restoration of a contaminated site would be the value of site that could be recuperated through immediate sale of the contaminated site minus the gain in value of the site that could be expected from restoration.

A rich history of public policy to affect restoration of contaminated sites is available for the United States where in 1980 a law was passed to establish a fund (the Superfund) of financing to pay for restoration of sites with soils or water contaminated by pollution. This law, known as the Comprehensive Environmental Response, Compensation, and Liability Act or CERCLA) put in place an institutional mechanism for restoration of sites which offered inadequate private incentives for restoration. This law was later amended in 1986 by the

Superfund Amendments and Reauthorization Act or SARA. The objective of these laws were twofold. First, the legislation put in place a system of liability for releases of hazardous substances. Secondly, the legislation put in place a fund for financing of the cleanup or restoration of polluted sites. The liability mechanism of this legislation established a means through which private and public suits for damages could be brought against individuals or organizations responsible for pollution that caused contamination. Damages awarded would be paid to the Superfund and used to finance cleanup and restoration of sites. Further, this legislation challenged the courts to define what is meant by restoration through its requirement for a definition of what constitutes damages. Subsequent cases have resulted in definitions that view damages as including not only the cost of restoration of a site, but also the lost value of direct use, indirect use, and nonuse values associated with the site.

In conclusion, public policy approaches for restoration of contaminated soils are motivated by a divergence between private and public interests in restoration. From a longrun perspective, restoration policy corrects damage that has accumulated as a result of past private economic decisions. To the extent that restoration policy also links financing of restoration with private liability, the policy may create private incentives for private prevention of damage. Depending on the scope of liability for damages, these private incentives can be very substantial. Further, where liability is created and where damage can be detected, strong incentives may result within the context of the market for the site. In such a market, liability would create incentives for potential buyers to audit and examine the physical condition of the site to determine whether contamination exists. Where contamination is found, the bid value of the site would be altered, capitalizing the future stream of negative returns predicted as associated with the site. Again, these incentives could be substantial depending on the cost of detection and examination of the site.

5.2. Public Policy for Prevention of Soil Contamination

While restoration involves correction of damages done by soil contamination, or disinvestment of a negative asset that generates negative returns, the stocks of contamination can also be altered by changing the flow of contamination. This possibility follows directly through what was labeled above as *preventive* public policy. Here, two approaches are feasible. The first involves conventional Pigouvian policy approaches to alter the flow on pollution. These approaches involve changing incentives through the use of taxes or subsidies or changing the level of use of polluting inputs, the types of production practices used, or the level of pollution through use of regulatory standards. These policy

alternatives have been extensively discussed in the economics literature, see e.g. Baumol and Oates. In general, the efficacy of their use depends upon the ability of policy makers to estimate the marginal benefits and marginal costs associated with the contamination process. Where uncertainty exists concerning the effects of contamination, the marginal benefits of contamination cleanup would be uncertain and difficult to estimate, implying that the precision with which the level of Pigouvian instruments could be set would be compromised. While a variety of schemes have been developed to accommodate difficult to estimate marginal costs of cleanup (e.g. tradable permits), marginal benefits would remain difficult to estimate. Standards suffer from the same problems as taxes or subsidies in these regards. In both cases, the level of the instrument must be set to optimize social welfare. Where information is inadequate, incomplete, or uncertain, substantial errors can arise and the implications of such errors will be reduction in the welfare improvement achieved. Standards can often be implemented through regulation of production practices or technologies used, or through regulation of particular inputs or outputs. In all cases, such regulation must be viewed as a Pigouvian standard and the desirability of such approaches suffer from the same problems when information is limited.

The second approach that is feasible relies upon establishment of liability for the stock effect of pollution, i.e. the level of contamination. This approach involves the specification of property rights for society for an uncontaminated site. Liability approaches can operate in two ways. First, establishment of private liability can have an effect on soil contamination through the operation of private markets for the site's ownership, or for products produced at the site. This possibility is feasible when detection of contamination is relatively costless compared to the value of the site. In this case, potential buyer's of the site can inspect the site to determine the level of contamination and estimate the cost of restoration. Bids for the site and the market of the site would be directly affected by the estimated cost of restoration. However, where private detection of contamination is costly, this approach could not be expected to be feasible. In this case, a public role would have to be played for inspection of suspected contaminated sites. An example of such an institutional mechanism is presented by CERCLA and SARA in the United States. Clearly, where either private or public inspection are feasible, and where liability establishes economic liability for the full social value of damages, such a system can be expected to create substantial private sector incentives to manage soil contamination in the public interest. However, in practice, private liability will always be limited to the total value of net wealth held by the liable individuals. In this case, incentives for private prevention will be limited and will not, in general, lead to a level of prevention that is socially optimal.

It is of interest to note that liability approaches can be linked to Pigouvian instruments. For example, liability can be limited to cases where Pigouvian standards are exceeded or recommended production practices are not used. In this case, private liability can be limited transferring liability to the public sector, or society in general. Again, in the absence of full information, use of such approaches may result in disappointing improvements in social welfare.

A further issue concerning the use of liability approaches involves the nature of liability. For example, if liability is not transferable between owners of the site, then no incentives are created for buyers to inspect sites to determine the extent of their contamination. From a spatial perspective, liability can also be made joint. For example, where nonpoint pollution causes soil contamination, the collection of producers that are likely to have caused contamination might be held jointly liable. In the case of agricultural nonpoint pollution this might be a relevant strategy within a watershed.

5.3. Current EU policy

European policies stem from the Treaty of Rome and its subsequent amendments, the Single European Act and the Treaty on European Union. Current environmental policy is based on the first two treaties since the latest treaty, which was adopted on 11 th December 1991 in Maastricht, only came into force on 1st November 1993. The Treaty of Rome (1957) gives no clear legal basis for European action relating to the environment. In the early days of the EC, environmental policy was often a by-product of the collective desire to remove trade distortions. In spite of the lack of a clear legal basis, an environmental doctrine progressively emerged through Community legislation and jurisprudence.

The first directive concerned with the environment was enacted in 1967 (67/548). It set out a procedure for classifying dangerous substances according to the degree of hazard and the nature of the risks entailed as well as provisions for packaging and labelling, the purpose being to protect man, particularly in the workplace. The sixth amendment of this parent directive goes much further by adding a new classification of "dangerous for the environment" and scheme of prior notification involving tests for potential hazards before a substance is marketed. These two series of directives which regulate the use of pesticides are also based on the prevention approach. A series is relative to pesticide residues. This originated in 1976 (76/895) and is primarily intended to protect consumers by setting limits on the amounts of pesticides in food. The limits can be achieved either by limiting the application of pesticides or ensuring that the pesticides have broken down sufficiently before the food is sold. The second series generated in 1979 (79/117) and restrict or ban the use of

certain pesticides. Those that are marketed are to be appropriately classified, packaged and labelled.

In all Member States waste disposal was regarded as a local or regional problem until the early 1970s. Moreover such an approach raised problems regarding the appropriate level of responsibility for forwarding European policy. The framework directive on waste enacted in 1975 (75/442) seeks to set out a coherent set of measures applicable in all Member States, more detailed measures being provided by other regulations, such as those on toxic waste (78/319) and PCBs (76/403). The directive on sewage sludge (86/278) has several purposes including the uncontrolled spreading of sewage sludge on agricultural land. Other directives eg on the disposal of waste oils and on the reduction of pollution by asbestos have also clear implication on soil contamination.

Command and control strategies underly all the legislation passed to protect the environment. There is the same approach relative to emergency response to disasters. Directive (82/501) regarding the prevention of major accidental impacts upon the environment is a good example. It arose as a direct response to the disastrous release of dioxins from an industrial plant at Seveso in 1976. Subsequent revisions have been proposed by the Commission and Parliament following later accidents at Bhopal and the Sandoz plant in Basle.

Current European policy has arisen from an evolutionary process. There are now clear environmental provisions with the Treaty on European Union. It does seem that future policy will become increasingly influenced by fundamental ideas of environmental economics such as the polluter-pays principle. Thus a move away from emission standards and regulation towards the use of an economic incentives approach to emissions control with particular emphasis upon the use of charges and marketable permits is expected. The principle of subsidiary is also likely to influence future policy. A final principle which is liable to influence the development of policy is that of prevention rather than cure of pollution. It seems highly likely that precautionary legislation will constitute a significant element of European Policy.

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