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of the best instrument to protect human health:
a brief overview**

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From Theory to Implementation of the Best Policy Instrument to

Protect Human Health: a Brief Overview¹

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Abstract : This paper presents a survey of methods of regulations with focus on pollution abatement and of various approaches to the issue of measuring quantities such as the marginal benefit of improved health that are crucial in view of implementing the regulation. Since pollution is a public bad, in general the efficient level of pollution can only be reached by way of some sort of public intervention. The paper's focus is on so-called market-based mechanisms, which in turn are classified into price-based mechanisms (pollution taxes) and quantity-based mechanisms (tradeable permits). The basic framework for addressing the comparison between the two types of mechanisms is Weitzman (1974). In order to actually choose between regulation methods and to eventually implement chosen methods, estimates are needed of some crucial quantities, in particular of marginal costs and benefits of pollution abatement. The most problematic one is of course marginal benefit. Therefore the paper considers various approaches to the measurement of marginal benefits.

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1 Pollution Regulation to Protect Human Health

Human health problems related to environmental degradation and use of natural resources are potentially serious in many parts of the developed and developing world alike. These problems may have important economic repercussions: they may generate additional costs for patients and the social welfare system, they can lead to a loss of productivity and profits for firms and they often result in a loss of income for individuals. Then, improving human health through measures addressed at preventing or reducing environmental pollution is a very important task for policy makers and government intervention through regulation is necessary to control pollution.

[...there are too many people damaged by most emissions of pollution for them to act as a single coordinated agent. Victims of pollution damages have different tastes, incomes, education so that they cannot agree how much to control pollution. (This justifies) government intervention in absence of which the market would fail to abate emissions...] (Mendelsohn, 2002).

Government intervention to protect human health means regulations through standards or other mechanisms of pollution control. In the last decades, much of the debate on instruments for government regulation has centered on the use of market-based incentives (MBI) mechanisms as opposed to command and control approaches (CAC). In CAC regulatory mechanism, the regulator usually specifies a technology or emission standards with the aim of controlling the substances that can contribute to pollution (Ellerman, 2005). The problem with CAC mechanisms is that they do not abate pollution efficiently. The marginal cost of control is high for some firms and low for others, since as such as uniform regulations treat all firms the same way. Moreover, command and control approaches do not encourage polluters to do any better than the law demands (Gangadharan and Duke, 2001).

Among economists there is near unanimity in preferring market-based incentives since they encourage firms to abate pollution more efficiently. Among MBIs, the basic choice faced by policy makers concerns price-based versus quantity-based instruments, or in other words, pollution taxes³ and tradeable permits⁴. To establish a basis for comparison among these policy instruments, the traditional literature often relies on the following assumptions:

- the same amount of emissions from different sources have equal external costs;
- the literature ignores possible interactions with other markets;
- there is no uncertainty about the costs and the benefits of pollution control;
- a competitive structure prevails.

In this setting, it is easy to show that emission taxes and tradable permits are equivalent: the two approaches will lead to the same outcome that is the optimal level of emissions at minimum cost. In a world of perfect knowledge, marketable permits are in principle a fully equivalent alternative to unit taxes. Instead of setting the proper tax and obtaining the efficient quantity of emissions as a result, regulator could issue emissions permits. This symmetry between taxes and tradable permits, however, is critically dependent upon the assumption of perfect knowledge. In a setting of imperfect information concerning the marginal benefit and cost function, the outcomes under the

³The concept of pollution tax was developed by the British economist Arthur Pigou, in “The Economics of Welfare” (1920). The term pollution taxes otherwise known as externality taxes or Pigouvian taxes, by definition refers to a tax used to correct the misallocation of estimated damage.

⁴The term tradable permit was developed in the pioneering work of Dales (1968). Dales proposed a market of tradable permits as solution to pollution problems in which the government grants pollution rights, that should be tradable for a certain period, and in which government acts as broker for the trade monitoring the system.

two approaches can differ in important ways. When there is uncertainty either about the marginal benefits and the marginal costs the optimal level of emissions will typically not be achieved, and the goal of regulator than becomes to minimize efficiency losses (Cropper and Oates, 1992).

The choice between quantity regulation and price regulation in terms of economic efficiency under imperfect information has been shown repeatedly in the pollution control literature. The classic article in this area is Weitzman (1974)⁵. He assumed linear marginal costs and uncertainty about the level of the marginal costs and benefits (not their slopes). Under these assumptions he reaches four main conclusions. Firstly, under full information it does not matter whether taxes or individual permits are used. Both instruments secure a first best optimum. Secondly, an error in estimating the benefits function has adverse effects on welfare but does not favor one policy instrument over the other: the efficiency losses will be exactly the same for the emission tax as the tradable permits system. Thirdly, if there is uncertainty about costs, emission tax is preferred over quantity regulation if the marginal costs are steeper than the marginal benefits function. Finally, transferable permits are preferred over taxes in the case of imperfect information about costs if the marginal benefits function is steeper than the marginal costs function. These results can be summarized as follows:

$$\nabla \approx \frac{\sigma_C^2 (B'' + C''')}{2C''^2} \quad (1)$$

where ∇ denotes the relative advantage of taxes over tradable permits measured in terms of welfare. If $\nabla > 0$ taxes are preferred over quantities regulation; while $\nabla < 0$ implies that tradeable permits is preferred over taxes. B'' denotes the marginal benefits slope (with $B'' > 0$) while C''' denotes the curvature of marginal cost (with $C''' > 0$). σ_C^2 represents the uncertainty on

⁵Weitzman did not prescribe exact types of price or quantity instruments, but many authors see the issue as binary choice problem between taxes and a quantity-based regime of tradeable permits.

the cost function⁶, while the sign \approx is used to denote "a local approximation" in the traditional Taylor theorem sense.

Subsequent contributions to this topic fall into two categories:

- Modifying the assumptions in Weitzman's analysis (see Laffont, 1977; Malcomson, 1978; Stavins, 1996; Stranlund and Ben-Haim, 2006⁷)
- Comparing policy tools other than an emission taxes and tradeable permits (see Yohe, 1977; Baumol and Oates, 1988; Mckitrick, 1997; Williams, 2000; Montero, 2004).

Weitzman stresses that if the uncertainty on benefit and the uncertainty on cost function are simultaneously present and benefit and cost function are not independently distributed, the correct form of the above rule becomes:

$$\nabla \approx \frac{\sigma_C^2 (B'' + C'')}{2C''^2} - \frac{\sigma_{BC}}{C''} \quad (2)$$

where σ_{BC} represents the covariance between benefits and costs. In order to explore the full implications of the above rule, Stavins (1996) rewrite equation (2) as:

$$\nabla \approx \frac{\sigma_C^2}{C''} \left(\frac{B''}{2C''} + \frac{1}{2} - \frac{\rho_{BC} \sigma_B}{\sigma_C} \right) \quad (3)$$

where ρ_{BC} is the correlation coefficient between benefits and costs, while σ_B and σ_C are respectively the standard deviation of benefits and costs. Based on the equation (3) Stevins made the following important observations:

⁶The regulator perceives the cost function only as an estimate or approximation: $C(q, \theta)$ where q denotes the emissions reduction and θ is a disturbance term or a random variable. σ_C^2 denotes the variance of costs; as σ_C^2 shrinks to zero we move closer to the perfect certainty case where in theory the two pollution regulation mechanisms perform equally satisfactorily.

⁷Laffont (1977) examines modifications to the relative-slopes criterion when these are also uncertain. Malcomson (1978) reexamine Weitzman's rule when local linear approximations to the benefits and costs function are not appropriate. Stanlund and Ben-Haim (2006) revisit Weitzman's original work under Knightian uncertainty that is when uncertainty cannot be modelled with known moments of probability distribution.

1. when benefits and costs are not correlated, so that $\rho_{BC} = 0$, an error in estimating the benefit function has adverse effects on the welfare, but the welfare loss does not differ under taxes and tradable permits regime;
2. Given $\frac{\partial \nabla}{\partial (\sigma_B \cdot \sigma_C)} = -\frac{\rho_{BC}}{C''}$, a positive correlation between benefits and costs tends to favor tradeable permits over taxes while a positive correlation tends to favor emission taxes.
3. Given $\frac{\partial \nabla}{\partial \rho_{BC}} = -\frac{\sigma_B \sigma_C}{C''}$, the greater the benefit or the cost uncertainty and the lesser is the slope of the marginal cost function and the greater is the influence of the correlation among benefits and costs on the choice of the the best policy instrument to regulate pollution.
4. Theoretically these effects can overwhelm the usual Weitzman's relative-slopes instrument recommendation.
5. The "instrument neutrality" identified by the equality between marginal benefits from pollution reduction and marginal abatement costs disappears when benefits and costs are not independently distributed; in fact by setting $B'' = -C''$ Stavins showed that:

$$\nabla = -\frac{\sigma_{BC}}{C''} \quad (4)$$

a positive correlation favor quantities based instruments (tradeable permits) while if negative correlation between costs and benefits exists, price instruments would be optimal. Stavins presents various scenarios for statistical dependence between marginal benefits from environmental protection and marginal abatement costs. Many scenarios, however, provide examples of positive correlation, suggesting that quantity instruments would be more attractive than otherwise. For instance, he considered the weather as generator of stochastic shocks that produce

correlated impacts on marginal benefits and marginal costs of pollution control:

[... the increased ultraviolet radiation that reaches the ground level on sunny days means more ozone formation from oxides of nitrogen and volatile organic compounds. Hence the marginal cost of ambient concentration reduction (and risk reduction) would increase. Of course, on beautiful sunny days, people are more likely to be outside, exercising, and breathing the ozone-laden air; hence, the marginal benefits of ambient-reduction would also increase, yielding a positive correlation between the relevant marginal benefits and marginal costs...] (Stavins, 1996).

Williams (2000, 2002) extends Weitzman's (1974) paper by developing a model of regulation of a group of pollution sources which investigates the relative efficiency of three regulatory instruments when there is uncertainty in the regulators's knowledge of firms' costs: an emission tax, fixed quotas and tradeable permits. The general structure is similar to the model in Weitzman, but differs in two important respects: Williams's paper compares three pollution regulation instruments (tradeable permits, taxes and fixed quotas), and considers the degree of substitutability between the pollution sources. Williams results can be summarized as follows:

$$\nabla^{TQ} = \frac{B''(1-\phi) + C''}{2C''} \left(\frac{N-1}{N} \right) \sum_i \sigma_{iC}^2 \quad (5)$$

where, ∇^{TQ} denotes the relative advantage of tradeable permits over fixed quotas in terms of welfare. N denotes the number of pollution sources distinguished by location, time period or both, while ϕ represents the degree of substitutability between the pollution sources. When abatement at one location is a perfect substitute for abatement at any other location, as in the case of globally mixed pollutants, tradable permits are preferred over fixed emission quotas. However, when pollution is high localized and independent

of emissions produced by other sources, tradable permits are dominated by emission taxes or fixed quotas.

Montero (2004) considers the optimal policy instruments choice when the regulator faces several information constraints: each firms have private information about its emissions, abatement costs and production costs. Montero develops a theoretical model for an industry of heterogeneous firms that produce output and undesirable by-products. The nature of Montero's model is similar to the one in Weitzman (1974) but with important differences: he compares the performance of two quantity instruments (tradeable permits and fixed standards) and considers the effect of cost heterogeneity across firms on instrument performance. He concludes that tradeable permits is preferred over CAC regulation when cost heterogeneity across firms is large, while when heterogeneity disappears the advantage of permits reduces in favor of standards. He also examine the advantage of a hybrid policy that optimally combines permits and standards.

Weitzman and the above described subsequent contributions fixed the conditions under which each of this two instruments is to be preferred to the other in a perfectly competitive equilibrium. However, many of major polluters in the real world are large firms in non-competitive industries (oil refineries, chemical companies, and auto manufacturers) where firms are not price takers in their output markets. Buchanan (1969) called attention to this issue by showing that the imposition of a Pigouvian tax may lead to a contraction in output that under monopoly regime is below the social optimum: a tax on a polluting monopolist will reduce the generation of external damages, but it may also cause the firm to reduce further its output. Thus, there is a trade-off between the two distortions, one due to the monopolistic underproduction and the other due to negative externalities. A tax based only on negative externalities ignores the social cost of further output contraction by a monopolist whose output is already below an optimal level.

Barnett (1980) was the first to solve the problem of determining the second best optimal emission level and the corresponding second best emission tax to be imposed on a monopolist. He considers a polluter who produces a single product output q and who discharges smoke s generating external diseconomies $E(s)$. He finds that a tax rate for unit of smoke discharged which maximizes social welfare must be equal to:

$$T^* = \frac{\frac{df(p)}{dq} \frac{dq}{dT} \cdot q}{\frac{\partial s}{\partial q} \frac{dq}{dT} + \frac{\partial s}{\partial w} \frac{dw}{dT}} + \frac{dE(s)}{ds} \quad (6)$$

where $f(q)$ is the industry demand curve, while w denotes resources devoted to smoke treatment. He discusses two cases: one in which the only means to abate the external diseconomies represented by smoke is reducing output, and the second case is one in which end-of-pipe treatment is the only means of smoke abatement. In the first case, terms involving w disappear and an optimal tax is given by:

$$T^* = \frac{\frac{df(p)}{dq} \frac{dq}{dT} \cdot q}{\frac{\partial s}{\partial q} \frac{dq}{dT}} + \frac{dE(s)}{ds} \quad (7)$$

If the only means of smoke abatement is end-of-pipe treatment the polluter responds to tax by changing w , than $\frac{dq}{dT}$ is equal to zero and the optimal tax is given by:

$$T^* = \frac{dE(s)}{ds} \quad (8)$$

Only in this last case market structure is not relevant. But, market structure becomes relevant for the more general case where both w and q vary with T . Finally, Barnett reformulate equation 6 introducing the price elasticity of demand $\mu = \frac{dq}{df(p)} \frac{f(p)}{q}$ and showing explicitly its role in determining the

optimal second-best taxation:

$$T^* = \frac{-\frac{df(p)}{dq} \frac{dq}{dT}}{\frac{\partial s}{\partial q} \frac{dq}{dT} + \frac{\partial s}{\partial w} \frac{dw}{dT}} + \frac{dE(s)}{ds} \quad (9)$$

He derives two main conclusions:

- when polluters are perfectly competitive μ approaches infinity, and the value of the optimal tax rate approaches marginal external damage $\frac{dE(s)}{ds}$.
- when polluters are imperfectly competitive μ is finite, and second best optimal tax rate may be less than marginal external damage to achieve an optimal trade-off between the external diseconomies and the welfare loss associated with monopoly output contraction.

The second best tax rate is equivalent to the combination of a Pigouvian emission tax and a subsidy on production, thereby correcting both distortions. Formally, the second-best tax rate can be negative if the social damage associated to pollution is very small compared to the distortion due to the market structure. The environmental problem becomes less significant and the regulator sets a negative tax (i.e. a subsidy on pollution) to induce firms to produce more.

Chen (1990) proposes another method for regulating monopolies and their production. He assumes that the planning authority would provide to the pollutant monopolist a subsidy equal to the expected value of the total benefit from emission reduction. The pollutant monopolist chooses the abatement level which maximizes profits given the subsidy and its own private information on production costs. Chen compares, referring to Weitzman model, the abatement subsidy with two other planning mechanisms: quantity based instruments (tradable permits), price-based instruments (emissions taxes).

He derives two expressions for the comparative advantage of the subsidy relative to quantity based instruments and another for the comparative advantage of the subsidy relative to price based instruments respectively:

$$\nabla^{ST} \approx \frac{\sigma_C^2}{2(C'' - B'')^2} > 0 \quad (10)$$

and

$$\nabla^{SP} \approx \frac{\sigma_C^2 B''^2}{2C''(C'' - B'')^2} > 0 \quad (11)$$

where ∇^{ST} and ∇^{SP} denote respectively the relative advantage of subsidy over tradeable permits and the relative advantage of subsidy over price regulation mechanism in terms of welfare. Not only the abatement subsidy does dominate quantity based instruments, but also dominates price-based instruments. Chen concludes that we have the best mechanism when pollutant monopolist chooses the level of reduction in emission receiving, or internalizing the social benefits of his abatement. Subsidies however are often politically and financially infeasible and might deter the adoption of new abatement technology.

Instead, little work has been done to investigate emissions trading markets where one or more participants have market power. Much work has been done on tradeable permits as mechanisms that may themselves influence the market structure since they may be more susceptible to strategic behavior. The basic idea for the tradeable permits control is that firms will trade quotas among themselves; such trading could continue until firms have equal marginal abatement costs and there is no further incentive to trade. In equilibrium, the price in such market should be equal to marginal abatement costs of each of the firms. In practice, however, the transactions costs in the market for permits might be high and this might reduce the number of transactions and prevent marginal abatement costs from being fully equalized. It is also possible that firms could behave in an anticompetitive fashion,

for example hoarding permits in effort to drive polluting competitors out of business. (see Hahn, 1984; Misiolek and Elder, 1989; Mansur, 2006).

2 Implementation of Environmental Policy

While the theoretical literature has clarified the issue concerning the determination of the best policy instrument to regulate pollution and reduce health pollution related damages, there has been less empirical work (see Kolstad, 1986⁸; and Choi and Feinerman, 1995). Lack of appropriate data (as well as empirical research evidence) makes it difficult to quantify environmental health impacts and pollution social costs; hence, the choice of environmental instruments becomes more complicated than what would appear from the theoretical results⁹.

⁸Kolstad (1986) was the first that empirically examined the fees vs. permits issue. He evaluated policies to control sulphur emissions from power plants by creating a stochastic model of regulatory design and industrial response for taxes and permits for air pollution regulation. He found that if marginal benefits from reduced sulphur emissions were constant a price instrument would be slightly preferable, but that a slight marginal benefits slope would be enough to make permits the more desirable option.

⁹As to the environmental charges and taxes, for example, the lack of information on the damage levels and problems related to their measurement constitute serious obstacles to the practical implementation. Consider, for instance, taxes paid by road transport: in addition to the morbidity and mortality caused by vehicular emissions, road traffic leads to noise stress, loss of quality of life, water pollution etc. Often, road traffic damage is greater than the additional tax paid by road transport. Then, road transport may not pay for the social costs it generates. This may lead to a pattern of transport development which may be accompanied by excessive impacts on the environment and health.

In response to these measurement obstacles, the literature has explored some second-best approaches to policy designs which have appealing properties. Baumol and Oates developed the “environmental charges and standards approach” in “The Theory of Environmental Policy”. They suggested to first set a certain standard of pollution (emission, air and water quality, etc.) and then, through a process of trial and error, derive which level of taxes have proved to give certain outputs. Taxes would be set to achieve a certain acceptable standard rather than being based on the “unknown value of marginal damage”. They further argued that such an approach would not result in Pareto optimality but that the

[..use of unit taxes to achieve specified quality standard is the least-cost method for the achievement of these targets...] Baumol and Oates (1988).

One of the explanations for the gap in the empirical literature can be found in the difficulties that analysts encounter in estimating marginal pollution health damages or marginal benefits from reduced pollution. In fact, while measuring control costs is relatively straightforward (market exists in principle in which pollution control equipment can be bought, and such equipment will reduce pollution by measurable levels ¹⁰), health damages caused by an increase of ambient pollution or marginal benefits from a reduction in pollution concentration are much harder to measure. Many difficulties derive from the fact that individuals have different susceptibilities toward pollution: the effects on health will vary across individuals due to genetics, avoidance behavior, life-style and other several factors. Hence, quantify exactly the extent of the health damages or health benefits deriving from increasing or decreasing pollution is a very hard task: we have to calculate the associated changes in health outcomes by taking into account that pollution could easily be correlated with other factors that may be just as potent. Once we have determined health pollution damages or pollution abatement benefits, we have to put a monetary value on them. However,

[...valuing health is obviously controversial because each person may place a different value on health. The problem facing society with pollution control is that we must make decisions that are not

The regulator can also move towards the use of voluntary approach. The regulator offers the firm a contract based on a certain level of abatement effort to achieve a certain standard of pollution and the firm can accept or refuse; if firm refuses, the regulator will impose an emission charge that is a fee levied on each unit of pollutant emitted. The emission charge is not a Pigouvian tax strictly speaking; it is not based on the estimated damage, but it can be considered a legitimate interpretation of the Pigouvian concept as it is a tax implemented to combat environmental pollution circumventing the problem of the damage level measurement.

¹⁰In order to compute and to evaluate producers' marginal abatement costs, we should to calculate shadow prices of pollution from the production technology that can be derived from the estimated output distance function (Shepherd,1970; Färe et al., 1993). Shadow prices derived from estimated output distance function do not directly reflect the value of abatement to society in terms of reduced morbidity but they could be compared to independent calculations of such marginal benefits in order to guide regulatory policy.

specific to each person but rather apply to us all. It is therefore not surprising that there is such controversy about picking a single value for health...] (Mendelsohn, 2002).

Concerning the quantification of health impacts scientists have performed several epidemiological studies on the linkages between air pollution and human health and have used the air pollution dose-response function¹¹ to estimate and evaluate the effects of a change in environmental quality on health¹². Ostro (1994) presented the estimated health impact, for a given

¹¹The dose-response function relates health effects to air pollution concentrations and other factors affecting health.

¹²Many of these epidemiological studies has been conducted in developed countries and used to estimate the effects of air pollution on health in developing countries.

Ostro (1994), for instance, uses the available epidemiology literature dose-response functions from the United States, Canada and Britain, to estimate the health impacts of conventional air pollutants (particulate matter, sulfur dioxide, nitrogen dioxide, ozone) and emissions of lead in Jakarta, Indonesia. Health effects of air pollutants (such as premature mortality, hospital visits and admissions, emergency room visits, restrictions in activity, acute respiratory symptoms, acute bronchitis in children, asthma attacks, IQ loss, and blood pressure changes.) are estimated by applying these functions to ambient air pollution level.

Alberini and Krupnick (1997) use daily records from a diary-type epidemiologic study in Taiwan to fit logit equations predicting the probability of experiencing acute respiratory symptoms (and headaches) as a function of pollution and weather variables, individual characteristics, and health background and proxies for reporting effects. They find that the rate at which illnesses are reported follows the fluctuations in PM levels but remains unaffected by ozone concentrations. Their model predicts that the impact of the particulate matter effects is very small. Moreover, illness rates tend to be poorly predicted when the corresponding equation estimated for a similar study conducted in Los Angeles is used.

Alberini and Krupnick study stresses that great care must be taken in the application of these method since dose-response transfer might give very misleading results. Firstly, elderly are more sensitive to the life-shortening effects of air pollution and in particular to particulate matter. Extrapolations from the U. S. population to a population with a much younger age structure would likely lead to an overestimate of the effect of pollution on premature mortality (Cropper, et al., 1997). Secondly, measured particulate matter is a heterogeneous mixture of solids and liquid. Differences in the physical and chemical composition of particulate matter could lead to quite different relationships between measures of particulates and the health effects of concern across countries (Ostro, et al., 1996).

type of health risk, as follows:

$$dH_i = b \cdot POP_i \cdot dA \quad (12)$$

where: dH_i denotes change in population health risk, b_i denotes the slope from dose-response curve¹³, POP_i is population at risk, dA denotes the change in air pollution. To complete the benefit or damage estimation for health effects, one would calculate the economic valuation V_i of this effect as well. Therefore the total change of the social value (dT) of the health effects due to the change in environmental quality under consideration can be represented by:

$$dT = \sum_i V_i dH_i \quad (13)$$

Even assuming that we can accurately measure health effects dH_i , putting monetary values V_i on that effects is rarely easy. The ability to place a monetary value on the consequences of pollution on health remains the crucial problem of the economic approach to human health problem related to environmental degradation (Hanemann, 1994).

Environmental economists have developed methodologies to measure the value of pollution health damages; these methods can be grouped in two broad categories. The first includes methods that measure only the loss of direct income (lost wages and additional expenditures). These approaches do not include discomfort, pain, losses in leisure, and other less-tangible impacts to individual and family well-being, moreover, may seriously understate or completely ignore the health costs of people who are not members of the

[...this approach neglects differences between the United States (and other developed countries) and the target country in pollution levels, baseline health, the age distribution of the population, medical care systems, sickleave policies, and cultural factors that might affect perceptions of illness and pollution and behavioral responses...] (Alberini and Krupnick, 1997).

¹³The slope of dose-response function measures the percentage change in the health outcome for a one unit change in ambient air pollution level.

labor force. Therefore, these methods provide only the lower bound of the social costs since tend to understate the total costs to individuals. The second category is based on the willingness to pay (WTP) of some economic agents for avoiding pollution health damages. Willingness to pay reflects the individual's preferences and can be interpreted as a monetary measure of health damage.

In addition, following the conventional economic practice, we distinguish these methodologies on the basis of whether their primary focus concerns respectively nonfatal illness or rather death (or more specifically the change in the conditional probability of dying at each age, for an identified group of individuals at risk).

2.1 The Economic Value of Morbidity

Models that describe what an individual would pay to avoid illness associated to pollution are, by now, well established in the literature (Berger et al. 1987; Harrington and Portney, 1987; Cropper and Freeman, 1991). We start by sketching Berger et al.'s (1987) model in order to provide a framework for interpreting individuals' willingness to pay. Then, we critically review the methods and the research efforts that have been devoted to estimating the willingness to pay for reduced morbidity (see also Dickie and Gerking, 2002).

Berger et al. (1987), assume that a person's utility depends on the consumption of goods and services and the state of health:

$$U = U(c, q) \tag{14}$$

where U is utility, C is consumption and q is a vector of health characteristics. Individuals, however, do not know their health status with certainty. The probability of enjoying good health is influenced by choosing one's life-style, thus making better and worse health status more or less probable, and by using medical advice, pharmaceuticals, hospital treatment, etc. Although

one's current health status certainly provides some information about the likelihood of future health outcomes, the risk of getting a disease may also depend on other factors such as pollution exposure, smoking history, which are more or less independent of one's observable health state. Berger et al. (1987) assume that the probability density function for health status is:

$$h(q; X, E) \quad (15)$$

where X is preventive expenditure and E is any exogenous shift such as environmental quality change. They reasonable assume that the chances of survival can be expressed as function of health characteristics:

$$p = p(q) \quad (16)$$

In addition, they assume that health is a matter only of absence or presence of a deleterious condition and the density function $h(q; X, E)$ is discrete rather than continuous, with $q = 1$ if individuals will enjoy good health and $q = 0$ otherwise, thus:

$$\begin{aligned} h(q; X, E) &= H(X, E) \text{ if } q = 0 \\ h(q; X, E) &= (1 - H(X, E)) \text{ if } q = 1 \end{aligned} \quad (17)$$

where $H(X, E)$ denotes the probability of contracting the disease. Berger et al. (1987) assume that a person will choose preventive expenditure X in order to maximize the expected value of utility:

$$\begin{aligned} \max E(U) &= U_0 P_0 (1 - H) + U_1 P_1 H \\ \text{subject to } &M = C + X + Z \end{aligned} \quad (18)$$

where M is the income, $U_0 = U(M - X, 0)$ is the utility if free of the disease; $U_1 = U(M - X - Z, 1)$ is the utility with the disease (where Z is the cost of illness that reduces consumption without providing utility). P_0 denotes the probability of survival if the individual is free from disease while P_1 denotes

the probability of survival with disease.

From the above maximization problem Berger et al. (1987) derive a person's WTP for an exogenous reduction of the concentration of pollution. WTP can be defined as a change in income that would be required to keep the expected utility constant when there is an exogenous change. Berger et al. express WTP as sum of two terms:

$$-dM/dE = -[(U_0P_0 - U_1P_1)/\lambda](dH/dE) - (dX/dE) \quad (19)$$

the first term is the monetary value of the expected differences of the expected utilities between being healthy and ill by the change in health risk. The second term denotes the change in preventive expenditure due to an exogenous change in the environment. Finally, $\lambda = U'_0P_0(1 - H) + U'_1P_1H$ and can be interpreted as the marginal utility of income.

Referring to the model above, Berger et al. (1987) explore two techniques aimed at measuring WTP: the cost-of-illness (COI) approach and the preventive expenditures (or averting expenditure) approach, that we review below.

The cost-of-illness (COI) approach is often used to value the cost of pollution related to morbidity. COI measures any loss of earnings resulting from illness (direct costs), medical costs such as for doctors, hospital visits or days, and medication, and any other related out-of-pocket expenses (indirect costs) (see Hodgson and Meiners, 1982 for a complete description of the methodology).

A key criticism to COI approach has been that it fails to take into account individuals' preferences, disutility from illness (Harrington and Portney, 1987), averting behavior and averting expenditure (Cournat and Porter, 1981). This criticism is supported by the equation 19 that consists in two terms: a utility term which reflects the cost of illness and a second term reflecting preventive expenditure. Only if preventive expenditure does not exist or does not

change with changes in environment ($dX/dE = 0$) or, in a less plausible case, if health status does not affect directly the individual utility function, equation 19 collapses to the first term and COI can be considered a measure of individuals' WTP.

In a similar model to the one presented above, Cropper and Freeman (1991) show that willingness to pay can be expressed as the product of the slope of a dose-response relationship times the marginal value of illness. But, since the marginal value of illness includes not only the loss in productivity and out-of-pocket expenditure but also pain and suffering, defensive expenditure, and loss in leisure time, it is likely to be higher than the COI measure. Hence, COI gives only a lower bound on willingness to pay (see also Dickie and Gerking, 2002).

The *preventive expenditures* (or *averting expenditure*) approach assumes that the link between environmental quality and health damages is affected by many human choices. In Berger et al. (1987) these choices are represented by consumption of X , which can be defined in several ways such as: whether to exercise on a day with high ozone level or to install an air filter or to buy bottled water. The preventive expenditure approach infers the minimum amount people are willing to pay to reduce health risks through the amounts people living in polluted areas spend on averting measures: for instance, expenditures on air filters or bottled water can be used to infer the minimum value people are willing to pay to avoid respectively respiratory or water-borne diseases. This approach has received little attention in environmental literature since it presents many limitations (see Courant and Porter, 1981). Firstly, referring again to the equation 19, we can observe that preventive expenditure represents only a lower bound of and individual's WTP since it does not consider explicitly the cost of illness. In addition, averting measures may be difficult to define for different types of pollution.

Abdalla et al. (1992) use the averting expenditure method for valuing environmental improvements and for approximating the economic costs of

groundwater degradation to households in a southeastern Pennsylvania community. The decisions included in their study to test for averting behavior were: increased bottled water purchases among households buying it prior to the contamination, bottled water purchases by new buyers, installing home water treatment systems, hauling water from alternate sources and boiling water. The survey was conducted by mail and respondents were asked to report only those actions taken as a specific response to groundwater contamination. Their findings, obtained through a logit specification, indicate that household's knowledge of contamination, perception of risk and presence of children determine whether they undertake averting actions and that their expenditure levels are higher if young children are present. Bresnahan et al. (1997) use panel data consisting of repeated observations on 226 Los Angeles area residents during 1985-86 to explain defensive responses to air pollution using determinants predicted by an averting behavior model. Their empirical results indicate that people who experience smog-related symptoms spend significantly less time outdoors as ozone concentrations exceed the national standard. Many people also report making other behavioral changes to avoid smoggy conditions and the propensity to do so appears to increase if health symptoms are experienced. Other applications have investigated individuals' effort to reduce symptoms of air pollution exposure (Abrahams et al. 2000; Eiswerth et al. 2005). However most of these studies find it difficult to assign a cost to averting behaviors. There is no monetary price for many of these actions and no compelling reason to use wage rate for increasing time spent indoors since it may not be entirely lost.

Another method frequently used by the environmental economist for valuing reduced morbidity is the *contingent valuation method (CV)*. CV, first proposed by Ciriacy-Wantrup (1947) and first applied by Davis (1963), is a survey or questionnaire-based approach. This method can be thought of as an attempt to directly measure willingness to pay ($-dM/dE$ in the Berger et al. model presented above). In contingent valuation methods, randomly

selected samples from the general population are given information about a particular problem. They are then presented with a hypothetical occurrence such as a disaster and a policy action that ensures against a disaster; they are then asked how much they would be willing to pay — for instance, in extra utility fees, income taxes, or access fees — either to avoid a negative occurrence or bring about a positive one. The actual format may take the form of a direct question (“how much?”) or it may be a bidding procedure (a ranking of alternatives) or a referendum (yes/no) vote. Contingent valuation studies are conducted as face-to-face interviews, telephone interviews, or mail surveys. The face-to-face is the most expensive survey administration format but is generally considered to be the best, especially if visual material needs to be presented. Non-response bias is always a concern in all sampling designs. In other words, people who do not respond have, on average, different values than people who do respond.

In principle, contingent valuation methods can be used to estimate the economic value of anything, even if there is no observable behavior available to deduce values through other means. Even though the technique requires competent survey analysts to achieve defensible estimates, the nature of CV studies and the results of CV studies are not difficult to analyze and describe. However, contingent valuation methods can be very expensive because of the extensive pre-testing and survey work. Moreover, contingent valuation methods suffer from a particular lack of accuracy: the presence of many biases (these include the way in which questions are phrased, the socioeconomic profile of respondents, the amount and type of information they are given etc.).

More fundamentally, contingent valuation approach is based on the assumption that individuals have well-defined preferences over all alternative states of the world. This assumption, however, is unreasonable for children, especially for infants. One approach to valuing the health effects on children is to make the assumption of “parental sovereignty” and to value these

impacts according to the parents' willingness to pay for them (see Neidell, 2004). Freeman III (2000) however observes that

[... there is no clear reason for believing that parents' willingness to pay for changes that affect their children will be equal to the willingness to pay that the children would have for changes that affect their own well being. Some authors have noted that parents do not always seem to be the best judges of what is good for their children and sometimes engage in activities such as smoking and drinking that actually harm their children...].

Even though contingent evaluation presents these limitations, has great flexibility, allowing valuation of a wider variety of non-market goods among which individuals well-being.

Many environmental economist use this method to evaluate health benefits from reducing pollution. For instance, Alberini et al. (1997) conduct a contingent valuation survey in three cities of the Republic of China (Taiwan) to estimate willingness to pay to avoid a recurrence of the episode of illness most recently experienced by the respondent. Alberini and Krupnik (2000) conduct a contingent valuation survey to estimate WTP to avoid minor respiratory illnesses. Then they compare cost-of-illness (COI) and willingness-to-pay (WTP) estimates of the damages from minor respiratory symptoms associated with air pollution using data from a study in Taiwan in 1991-92.

Several studies have relied on a new approach in estimating the willingness to pay for improved environmental quality by relying on the *health production approach* first introduced by Grossman (1972) (see Kiiskinen, 2003). Grossman interprets a person's health as a capital stock that exogenously deteriorates at an increasing rate with age. To counteract this health deterioration, he assumes that individuals invest a portion of their assets into health production each period. By analyzing the decisions consumers make

concerning the resources allocated to health production such as medical care, time and a healthy life-style this method try to infer the value of health to the consumers and derive estimate econometrically a measure of individual willingness to pay for a reduction in pollution (Cropper, 1981; Gerking and Stanley, 1986; Dickey and Gerking, 1991; Cropper and Freeman III, 1991). Cropper (1981), for instance, extends Grossman's model of health production and health demand to incorporate pollution and estimates willingness to pay for health risks related to an index of air pollutants. Gerking and Stanley (1986) estimate willingness to pay for health risks related to ozone exposure. They compute the value of a change in health by multiplying the cost of preventive activity by an estimated ratio of marginal products of inputs in the health production function. Harrington and Portney (1987) extended the household production function model introduced by Grossman to examine explicitly the relationships among willingness to pay for a reduction in pollution. Dickie and Gerking (1991) used a set of health symptoms in estimation of functions hypothesized to be associated with air pollution. Cropper and Freeman III (1991) develop a model of health production in which the health outcome of interest is the number of hours during a time period that a person spends in sickness. They show that willingness to pay can be expressed as the product of the slope of a dose-response relationship times the marginal value of sickness time.

The health production function approach suffers from an import limitation too. The estimation of a health production function is frequently based on instrumental variables since individual's life-style and averting expenditure/behavior inputs (that in the above model is represented by X) may be endogenous. Construction of instruments, however, usually can be done in a number of ways as theory often says little about how this problem should be handled and variables used for this purpose are chose depending on what information is available together with judgement of the investigator. Different choices of instrumental variables typically can produce different estimates of

willingness to pay creating uncertainty.

2.2 The Economic Value of Mortality

Because some forms of pollution may increase mortality or shorten life expectancy, economists have to identify approaches for valuing life and the benefits of lifesaving activities. Since death is a more easily measured outcome than illness or injury (death is a one-dimensional event, whereas there are varying degrees of illness and injury) these methods are easier to apply.

In estimating the value of lifesaving, economists have followed two schools of thought. The first approach is based on measurements of the economic productivity of the individual whose life is at risk. This is referred as *human capital approach*. The second approach is based on the individuals' *WTP to reduce the risk of death* (Cropper and Freeman, 1991; Shepard and Zeckhauser, 1982; Berger et al. 1994; Johannsson, 1995).

In the standard human capital approach the value of preserving a life V_i (with reference to the equation 13) is equal to the discounted present value of lifetime earnings lost due to premature mortality. Formally the present value of lifetime earnings is given by:

$$V_i = \sum_{t=j}^T q_{j,t} (1+r)^{j-t} y_t \quad (20)$$

where $q_{j,t}$ is the probability of the individual surviving from age j to age t , y_t is the individual earnings at age t , r is discount rate and T is age at retirement from labor force. This approach has been criticized on a number of grounds: it considers individuals as units of human capital that produce goods and services for society. The values calculated are dependent on the age of death and on income, skill level, sex, race and country of residence. It omits the role of nonmarket production and because of earning differences by sex and race, it places a lower value on saving the lives of women and non

whites than on saving the lives of adult white males. Moreover, the human capital approach assigns zero value to people who are retired, handicapped or totally disabled and to children (Landefeld and Seskin, 1982). Another important objection is that the human capital approach is inconsistent with the fundamental premise of welfare economics by which individuals' preferences should constitute the cornerstone of the benefit-cost analysis (Cropper and Oates, 1992).

Individuals make decisions everyday that reflect their preferences and how they value health and mortality risks, such as driving an automobile, smoking cigarettes or living in polluted areas. Many of these choices involve market decisions. Using evidence on these market choices, which involve implicit trade-offs between risks and money, economists have developed estimates of the individuals' WTP to reduce risk of death. The *willingness-to-pay approach* is based on the assumption that changes in individuals' economic welfare can be valued according to what individuals are willing (and able) to pay to achieve that change. According to this assumption, individuals treat longevity like other consumption good and reveal their preferences through the choices that involve changes in the risk of death and other economic goods whose values can be measured in monetary terms.

One of the most used approach to measuring willingness to pay to reduce the risk of death is to infer the value from *compensating wage differentials* in the labor market (see Viscusi and Aldy, 2003 for a complete treatment). The theory behind this approach is simple:

[...The basic idea behind compensating wage differentials is that jobs can be characterized by various attributes, including risk of accidental death. Workers are described by the amount they require as compensation for different risk levels, while firms are characterized by the amounts they are willing to offer workers to accept different risk levels. The matching of wage offers

and acceptances determines the hedonic wage equation, which describes the compensation received for bearing risk in market equilibrium...] (Simon et al., 1999)

Simon et al. (1999) express the individual's willingness to substitute risk for income in the labor market as the compensation C that he would require to work at various risk levels, holding utility constant. Formally:

$$(1 - \rho)(1 - \phi)U(C + I) = k \quad (21)$$

where ϕ is the risk of death on the job, and ρ is the risk of dying from all other causes while I denotes the non-labor income. The worker's choice of risk level, ϕ , occurs where a marginal change in required compensation, $C'(\phi)$, equals a marginal change in the wage offered in market equilibrium, $w'(\phi)$, or, equivalently, where the compensation function is tangent to the hedonic wage equation, $w(\phi)$. Equilibrium in the labor market is given by the locus of tangency points between various required compensation and offer curves. This locus is the hedonic wage function, and its derivative with respect to risk of death measures the value of a small change in risk to the worker:

$$\frac{dw}{d\phi} = \frac{(1 - \rho)U(w + I)}{(1 - \rho)(1 - \phi)U'(w + I)} = V_i \quad (22)$$

Equation 22 gives the rate at which a worker is willing to substitute income for risk $\frac{dw}{d\phi}$ that is equal to his expected utility if he survives risk of death on the job, $(1 - \rho)U(w + I)$, divided by his expected marginal utility of income, $(1 - \rho)(1 - \phi)U'(w + I)$. Hence, equation 22 is a measure of the risk premium that a worker receives to compensate him for risk of death on job (we label this relationship, V_i with reference to the equation 13). Then, the compensating wage approach by providing a value V_i that is a measure of individuals WTP to avoid death risks could be used in the equation 13 to measure total change of the social value (dT) of the health effects due to the change in environmental quality. Cropper and Freeman (1991) and Cropper

and Oates (1992) stress that the compensating wage approach presents at least three problems: the first problem concerns the fact that compensating wage differentials exist only if workers are informed of job risks. A second problem related to this approach is that compensating differentials seem to exist only in unionized industries than it may provide estimates of the value of a risk reduction only for certain segments of the population. Finally, if workers have biased estimates of job risks market wage premium will yield biased estimates of the value of a risk reduction.

Moreover, compensating wage differentials approach presents other important limitations. Firstly, it tends to focus on the value adults in the prime of their life place on reducing their risk of dying, even though according to the epidemiological literature, the significant correlation between air pollutants and deaths occur among people over 65 (Ostro, 1994; Schwartz and Dockery, 1992). In addition, this method tends to focus only on immediate risk changes. However, when an environmental policy program reduces exposure to a carcinogen, while the costs of doing so are often incurred in the present, mortality risks are reduced in the future, following a latency period.

Difficulties in measuring the individual's WTP using labor market compensating wage approach and its limitations have led to the use of CV to measuring willingness to pay. As we have already seen in the evaluation of morbidity cases (section 1.2.1), CV presents great flexibility but also a number of limitations (for a complete treatment of these problems see Diamond and Hasuman, 1994).

3 Summary and Conclusions

There is a substantial amount of literature on theoretical and empirical aspects of the economic valuation of policies and instruments to improve environmental quality and human health. Here, starting with Weitzman's (1974) seminal work, we have provided a short but comprehensive overview of key

literature on the choices faced by policy makers concerning price-based versus quantity-based instruments to regulate pollution and protect human health. We have reviewed the methods employed in estimating pollution abatement costs and pollution related health damages whose comparison (with reference to Weitzman's theoretical rule) should form the basis for the choices among the price-based and quantity-based regulation instruments.

The aim of this paper is to shed light on the difficulties that analysts face in estimating the value of pollution health damages or benefits from reduced pollution. In fact, while measuring control costs seems relatively straightforward, measuring and valuing the health impacts of pollution is a very complex task: we have shown that the available methods of economic analysis are often rudimentary and the answers vary greatly depending on the method used. In recent years, however, considerable progress has been made, especially with respect to air and water pollution.

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