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Technical Report on Sustainability and Competition

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Technical Report on Sustainability and Competition

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EXECUTIVE SUMMARY

The Netherlands Authority for Consumers and Markets and the Hellenic Competition Commission – the competition authorities of the Netherlands and Greece – have jointly commissioned the following technical report on sustainability and competition. The report draws on concepts and tools mainly from environmental economics to answer the question of what forms of quantitative assessment could be applied to take account of the broader social benefits, as well as benefits for future generations, in competitive assessments. To this end, the report focuses on benefits relating to environmental sustainability; however, as the report notes, its concepts and tools are also more broadly applicable to other aspects of sustainability.

Competition law and its enforcement may address sustainability concerns in various contexts. This report takes as a leading example that of an agreement between competitors, under the presumption that the respective consideration of greater societal benefits is within the remit of a competition authority. Such an agreement may take various forms – for instance, a consensus between firms to phase out the production and sale of a less sustainable product variant. Taking recourse to established practice in environmental economics, the report shows how the benefits thus realized for society can be properly measured. In doing so, notably, one can account for the benefits to future generations.

The report is broadly organized in two parts. The first part introduces the main concepts of welfare economics in the presence of externalities. Even perfectly competitive markets fail when there is a failure to internalize such externalities, whether these externalities impact present-day members of society or future generations. The report provides readers with a presentation of the basic welfare economic analysis incorporating individuals' choice for environmental quality. The choice of a wealth maximizing criterion, in particular the Kaldor-Hicks criterion, for policy decision making is supported by arguing that when such a criterion is adopted a partial analysis may be conducted by calculating consumer welfare and its changes as well as potentially reduced environmental damages measured also in terms of welfare. Building on this, the second, main part discusses various methods to measure such changes associated with reduced environmental sustainability/environmental damages under the concept of total economic value, which encompasses the overall welfare gains attributable to improvements in environmental quality (hence referring, in particular, to the use and non-use values of any environmental asset). When this is feasible in a given context, such values can be elicited from individuals' preferences, as revealed in the market, in hypothetical choice scenarios or through stated preferences. The report discusses the underlying theoretical concepts as well as measurement techniques. When the use of an environmental good leaves a so-called behavioral trail in markets for non-environmental goods, revealed preference approaches may also be used to estimate use values, even when the considered good is itself not traded. Often, however, individuals may not be fully aware of the negative impact of certain emissions. Then, the respective impact – for instance, on health, life expectancy, or morbidity – can be determined objectively. The report also draws awareness to the need to incorporate the (changing) preferences of future members of society and introduces the reader to the various techniques of how to integrate costs and benefits arising at different points in time.

The tools discussed in this report have been widely adopted in empirical and policy works in environmental economics. Often, an analysis may, however, face time or resources constraints. The

report discusses how results from existing studies and databases – notably, determined environmental (shadow) prices – may be harnessed. It also discusses cost-effectiveness analysis as an additional approach. Thereby, the benefits of a considered agreement, for instance, may be obtained from saved abatement costs, under a given policy objective (such as an emission reduction target). In what follows, we provide a more systematic overview of the considered methods.

Overview of presented methods

Turning to a short overview, we first note that while this report is primarily concerned with methods for measuring sustainability benefits, when interventions that restrict competition lead to reductions in consumer surplus, competition assessments need to consider such effects. The general approach for estimating the welfare gains attributable to improvements in environmental sustainability is to compare the Total Economic Value (TEV) of two scenarios, one with and one without the considered restrictions to competition. For practical purposes only, we divide our measurement approaches as follows: Our first set of approaches proceeds from "primitives" – that is, from the preferences of individuals. Changes to total welfare are calculated by aggregating willingness to pay derived from individual revealed or stated preferences. Another option is to measure aggregate impacts to health or productivity (e.g. from harmful emissions). Our second set of approaches relies on a shortcut – specifically, they extrapolate from existing data, e.g. on environmental prices, or it is presumed that a policy goal, such as a cap on emissions, is an expression of societal preferences, from which shadow prices can be then derived. Table 1 (reproduced from the report's concluding remarks) provides an overview.

It should be noted that some of these methods fail to directly consider the preferences of non-consumers or the externalities that impact them. These and other specific issues associated with each method are discussed in the report. Methods that rely on the estimation and aggregation of willingness to pay elicited from individual preferences may necessitate the gathering of appropriate data, insofar as no adequate (surrogate) market exists, e.g. by conducting surveys or choice experiments. Particularly when benefits take the form of reductions in harmful substances, the welfare impact may be estimated in terms of changes to empirical measures of health benefits or medical cost savings, productivity, or, the costs individuals willingly bear to avoid harmful exposure. Such data may be available from existing databases. In Table 1 we collect separately approaches that (mainly) build on data from extant studies and databases that then need to be transferred to the specific case. This may involve less time and resources. However, when the analysis being conducted does not rely on case-specific data, but instead uses data or derived environmental prices from other studies, care must be taken to ensure that such a "benefit transfer" still adequately reflects the specifics of the examined setting, including the preferences of concerned individuals.

As a final remark, we note that different approaches may lead to different results, as we illustrate throughout the report in relation to carbon emissions. In this connection, the environmental prices derived from avoided abatement costs or carbon markets, which reflect the market evaluation of a given policy goal, typically lead to much smaller figures than alternative calculations of the global "social cost of carbon." Far from indicating contradictions between methods, this divergence highlights the need for transparency concerning measurement definitions and applied standards.

Table 1: Overview of methods presented in this report

I: Methods for environmental valuation using case-specific data		
(1)	Methods based on market choices (potentially in surrogate markets)	Examples: – Discrete choice analysis of preferences revealed from actual purchases (e.g. of products that are more or less environmentally friendly) – Hedonic prices derived from surrogate markets, e.g. real estate prices
(2)	Methods based on hypothetical choices or stated preferences	Examples: – Contingent valuation analysis based on surveys of stated preferences over hypothetical scenarios – Conjoint analysis of (pairwise) choice between different scenarios (e.g. products) – Subjective well-being valuation based on correlating stated well-being with observable (environmental) variables and monetary values
II: Valuation methods for estimating and aggregating case-specific impact		
(1)	Dose-response approaches	Example: Estimating welfare through the impact on life expectancy or morbidity
(2)	Averting and defensive behavior	Example: Estimating avoided costs of defensive expenditures
III: Valuation using data from existing studies and databases		
(1)	Benefit transfer within a calibrated model	Example: Adjusting willingness-to-pay (e.g. obtained from contingent valuation) to different socioeconomics and demographics
(2)	Environmental prices databases	Example: Using environmental prices aggregating all health-related costs from the emission of a particular substance in a specific country
IV: Valuation derived from stated policy objectives		
(1)	Using market prices for permits or taxes on emissions	Example: CO_2 prices from the EU Emissions Trading System
(2)	Use of avoided abatement costs under a cost effectiveness analysis	Example: CO_2 prices based on an analysis and ranking of the costs of alternative abatement methods

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I INTRODUCTION

The Netherlands Authority for Consumers and Markets (ACM) and the Hellenic Competition Commission (HCC) – the competition authorities of the Netherlands and Greece – have jointly commissioned the following technical report on sustainability and competition. It seeks to answer the following question:

> *Assuming a more open approach towards sustainability benefits under competition law is possible and desired, what forms of quantitative assessment could be applied in practice that take account of the broader social benefits/out-of-market efficiencies as well as benefits for future generations?*

To this end, this report draws on concepts and tools mainly from environmental economics. Its objective is to provide both a solid foundation grounded in the economics of welfare analysis, as well as an overview of applicable tools that allow one to estimate and compare the potential impairments to competition with the social benefits of increased sustainability. Given its technical nature, this report is mainly addressed to practicing economists, including in particular individuals well-versed in competition economics, but who may lack an understanding of or experience with certain concepts and methods in environmental economics. However, this report does not intend to provide a comprehensive overview of the subject. Nevertheless, given its orientation to the question posed at the outset, this survey of the material should be of strong practical value, particularly in fostering the application of concepts and tools in environmental economics that are of increasing relevance to competition authorities.

As we explore in the following, sustainability can have various meanings, ranging from the highly specific to the exceedingly broad. As the lodestone of an ethical position, it may encompass numerous domains, from ecological preservation and animal welfare to economic equality. While the concepts and methods discussed in this report lend themselves to the assessment of various dimensions of sustainability, we focus here on ecological sustainability, for three reasons: First, a market failure attributable to externalities that impair a natural resource is the most immediate problem nexus. Second, by virtue of this focus, we can immediately draw on the vast literature of environmental economics, and all discussed methods are immediately applicable. Third, there is broad public consensus on the need to promote ecological sustainability.[1] Nevertheless, our focus on ecological sustainability does not mean to imply that the sustainability concerns of competition authorities should be restricted to this domain.

[1] For instance, the draft paper of the ACM on sustainability agreements (ACM 2020) focuses on climate change and sustainability, at para. 6, but it also addresses examples of agreements on animal-friendly products or guaranteeing a fair income, at para. 30. The Staff Discussion Paper of the HCC (HCC 2020) refers to the United Nations' broader development targets of economic, financial, institutional, social, and environmental sustainability (para. 3), as well as to the broader sustainable development objectives enshrined in the EU Treaties (para. 9). Still, it is fair to say that even though much of the subsequent discussion focuses on ecological sustainability, as expressed, for instance, in the objective formulated in para. 18 (Integrating environmental concerns as broader externalities to be taken into account in competition law enforcement).

In seeking to address how sustainability benefits can be evaluated in the context of competition law, we do not concern ourselves with the specific powers invested in competition authorities.[2] Rather, we merely presume the existence of a practical and legal basis for such forms of evaluation,[3] as it would exceed the scope of this report to devote attention to the specific areas of competition law – and associated case examples – in which such methods can or could potentially be applied.

As will become evident, however, addressing such matters is not essential to the discussion of concepts and tools for measuring the benefits of ecological sustainability. Within a given welfare standard, such benefits are weighed against potential restrictions to competition, regardless of the type of such restriction, as a trade-off. In general, competition law and its enforcement may address sustainability concerns in various contexts, e.g. in the assessment of mergers and acquisitions, abuse of market dominance (Article 102 TFEU), or agreements between competitors (Article 101 TFEU). In this context, delimiting the applicable scope of the analyzed trade-off is not essential. That being said, our primary reference case in the following is of an agreement between competitors. Such an agreement can take various forms – it may, for instance, involve a consensus between firms to phase out the production and sale of a less sustainable product variant, thereby restricting consumer choice and possibly passing on a higher cost of production in the form of higher prices. Such agreements may also relate the more efficient joint utilization of a resource.[4]

This report is organized as follows. Section II introduces the main concepts of welfare economics in the presence of externalities. In particular, it addresses the need to choose a welfare criterion, as this furnishes as basis for measuring costs and benefits. Section III applies such a welfare criterion to the measurement of consumer surplus and environmental sustainability. Section III represents the core of this report, as it introduces, explains, and illustrates various methods to measure changes to environmental sustainability. Section IV concludes.

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[2] There is a considerable literature on the question of whether and in which ways competition law enforcement should consider such concerns. Advocates of a multi-goals approach see such concerns in addition to the protection of competition, which then requires to weigh up these goals in case they are conflicting (e.g. Van Dijk forthcoming). Such a calculation of societal costs and benefits is at the heart of this report. Inderst and Thomas (2020b) show that environmental benefits can also be accounted for, albeit to a lesser degree, in the context of consumer welfare analysis, so that the concepts and tools discussed in this report are indeed more broadly applicable, regardless of when the chosen goals encompass sustainability more or less widely.

[3] On this see Lianos (2018).

[4] Throughout we presume, without further discussion, that the respective restriction to competition, such as arising from an agreement between competitors, is indeed indispensable for achieving the stated benefits (see, e.g., Van Dijk (forthcoming), for a further discussion of this in the context of sustainability agreements).

II BACKGROUND: INDIVIDUAL AND SOCIETAL PREFERENCES

This section provides the background for our subsequent discussion and analysis. We first introduce and define key terminology, including in particular *sustainability* and *welfare*. Second, we delineate the scope of the present analysis. Third, we lay the groundwork for Section III, which introduces the main approaches for measuring welfare in the present context.

More concretely, Section II is organized as follows: We first define the concept of sustainability while emphasizing the inherent trade-off between economic development and environmental quality. Individual preferences regarding environmental quality are defined and aggregated to attain a social welfare function. Establishing the impossibility of constructing an objective aggregate measurement to assist us in social decision making, we introduce the criterion of Pareto efficiency (and with it also that of *potential* Pareto efficiency). In a partial equilibrium framework, individual demand, and, by extension, consumer and producer surplus are defined. Finally, we present a simple graphic depiction of the two major effects of an agreement between firms, which facilitates the presentation of the major questions that are answered in Section III.

II.1 Environmental Quality and Sustainability

Although there is evidence that the relationship between human civilization and the environment has always been fraught, prior to industrialization, the effects of human intervention on the environment remained local.[5] Over the past two hundred years, human demands on the environment have increased enormously, threatening the stability of the climate and global ecosystems. International concern gathered steam in the 1960s with the extremely influential publication *Our Common Future*, also known as the Brundtland report, named after the World Commission on Environment and Development's chair, Ms. Gro Harlem Brundtland. The report established, for the first time, the connection between economic development and environmental degradation, while highlighting the emerging challenges of climate change. The report, in describing the problem of covering the needs of the ever-growing population without degrading environmental quality, offered the most well cited definition of sustainability as the "development that meets the needs of the present without compromising the ability of future generations to meet their own needs" (WCED 1987, p. 35).

Sustainability is a very popular term, not least due to its broad applicability resulting from its very general definition. As noted in the introduction, in this report we mainly consider environmental sustainability and, thereby, the importance of externalities generated by economic activity. We acknowledge, however, that the term *sustainability* is used in a wider context; in this way, our methods and concepts need not be confined to the scope of the present analysis.

A crucial factor in defining sustainability is the assumed degree of substitutability between human-made and natural capital.[6] *Weak sustainability* requires the maintenance of a specified value of

[5] See, for instance, the discussion of the rise and fall of Easter Island in Brander and Taylor (1998).

[6] Aggregate capital consists of human-made capital (machines, buildings), human capital (knowledge and knowhow, institutions, and technology), natural resources used in the economic process, ecological capital

aggregate capital, assuming that human-made capital and human capital are substitutes for natural capital and, most importantly, natural resources.[7] Alternatively, *strong sustainability* does not allow such substitutability, requiring that certain environmental constraints are imposed on the working of the economic system.[8] Restricting attention to natural resources used as an input to the production function, historical evidence favors the weak sustainability view.[9] However, one cannot be sure that this trend will continue and, more importantly, quite the opposite happens with numerous environmental assets, such as air and water quality. The problem becomes even more complicated once it is recognized that it involves many generations with potentially different preferences and thus divergent definitions of sustainability.[10] Furthermore, in light of great uncertainty and irreversibility, it has been broadly accepted that the basic principle in guiding public policies should be to err on the side of precaution.[11]

Squaring the circle to achieve sustainability is a challenging endeavor both at the normative and the positive level. As a first step, societies have to agree on an objective – that is, what should be sustained and how economic growth/development is framed within this perspective. In economic modeling terms, these questions refer to the shape of the social welfare function, which will be briefly examined in Section II.2. From the positive point of view, societies have to understand the workings of the environmental and economic systems, including their interactions and interdependencies. Only then can they define what is to be sustained and how.[12]

At the country/market level, the optimal level of environmental quality – or, to put it differently, the optimal level of pollution associated with a particular level of market activity – has to be defined. This necessitates quantifying environmental damage – which is not a simple task, as will be discussed in Section III – and developing policies to achieve the desired environmental quality. The development of suitable policies is also a difficult endeavor, especially in markets that involve many distortions and the combination of different targets, such as sustaining a high level of competition while allowing for coordination between actors in order to achieve higher environmental quality.[13]

(including, for instance, certain species of animals or wetland) not directly used in the economic process but essential for the working of the ecosystem and social capital.

[7] This approach is based mainly on the classic contributions by Solow (1974 and 1992) and Hartwick (1977 and 1978).

[8] See, for instance, Constanza (1991), Constanza et al. (1992) and Daly (1991).

[9] For instance, in a quite famous wager, the biologist Paul Ehrlich betted, in September 1980, that prices of natural resources would increase, reflecting increased scarcity. He lost the bet to the economist Julian Simon, since prices had fallen a decade later.

[10] Social sustainability that considers issues such as equity within and between generations is even more difficult to define.

[11] Article 191 of the Treaty on the Functioning of the European Union (TFEU) details the precautionary principle. A particular application is REACH, the EU's regulatory framework for chemicals (Regulation (EC) N 1907/EC).

[12] This is currently approached using integrated assessment models, aiming to provide feasible, sustainable combinations of social, economic, and ecological states.

[13] We recognize that, as already noted in the Introduction, there are clearly jurisdictional limitations within which regulatory authorities can operate, which may have to be either amended or interpreted more broadly.

II.2 From Individual to Social Choices

In order to examine policy decisions regarding the environment, we must first consider individual preferences. Although preferences are usually taken as a straightforward given in much of the economics literature,[14] valuing the environment is not so straightforward and requires some discussion. This part of the report first examines different approaches for defining individual preferences for the environment and then examines some important issues associated with aggregating individual preferences. Section III builds on this work, providing a formal structure for the various measurement approaches that are subsequently discussed.

II.2.1 Individual Choices for the Environment

Individuals view the environment from many different perspectives. Consider, for example, the value of a wild animal. To humans it may have instrumental value[15] because it provides value as an exploitable resource (as food or labor), and/or as a source of emotional, recreational, aesthetical, or spiritual experience. In addition to the value it creates for others, a wild animal may also have value unto itself – that is, intrinsic value[16] – that needs to be recognized and respected. It has been argued that if an entity possesses intrinsic value, it "generates a prima facie direct moral duty on the part of moral agents to protect it or at least refrain from damaging it."[17] A substantial literature on environmental ethics that arose in the early 1970s[18] challenges a purely anthropocentric approach, positing new directions such as enlightened anthropocentrism, biocentrism, new animism, and deep ecology. An important point that differentiates these approaches concerns the attribution of intrinsic value – that is, whether only humans, or only animals,[19] or all natural entities including flora, mountains, and rivers, have intrinsic value. Another important point is whether comparisons between these values are permitted, that is, whether hunting or using animals in experiments should be allowed when the results of these actions provide value to humans.

As the aim of this report is to present measurement techniques that allow various preference scenarios to be compared,[20] it seems appropriate to focus on aggregate values arising under these scenarios, and not on rights. Furthermore, in order to measure the outcomes (benefits) associated with improved environmental conditions, we have to take an anthropocentric approach, defining though instrumental values broadly. That is, although we are restricting our focus to values (and

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With respect to the mandate of competition policy and authorities, see, for instance, the editorial in the Journal of European Competition Law & Practice by Kingston (2019).

[14] However, below we discuss both how changes in such preferences may be accounted for and the possibility that a change in policy or an agreement may affect also revealed preferences, e.g. as the change of behavior of others acts like a social anchor for each individual.

[15] Defined as the value of an entity as means to achieve an end.

[16] Defined as the value of an entity as an end in itself.

[17] See Brennan and Lo (2020, p. 2). On the issue of intrinsic value, see also Nash (1989) and Jamieson (2002).

[18] Building on the classic works of Rachel Carson's *Silent Spring* (1963), Paul Ehrlich's *The Population Bomb* (1968) and Dennis Meadows et al. *The Limits to Growth* (1972).

[19] Those that can experience happiness and pain, including all animal species.

[20] For instance, comparing a state of affairs in which firms are allowed to cooperate to a state of affairs in which they are not.

not considering rights) and we adopt a human perspective, we include not only instrumental but also spiritual, aesthetic, and any other values humans identify with the use or preservation of flora, fauna, and natural resources.[21]

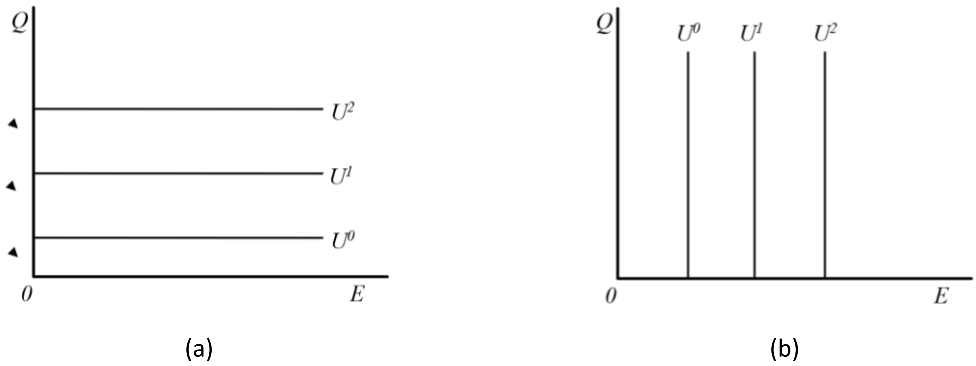
Let us now integrate the foregoing considerations in the common manner of economics to present individual preferences – that is, the utility function. Let us consider (m) commodities, that is, conventional goods and services that are traded in respective markets and (n) environmental goods and services, representing the many elements of *environmental quality* that are impacted by externalities – that is, when an individual's actions have unpriced effects on the utility of other individuals or the society at large.[22]

Assuming a society of k individuals, then individual i's utility function, where $i \in \mathrm{K}, \mathrm{K} = \{1, \dots..\mathrm{k}\}$, is defined as,

$$U_i(\boldsymbol{Q_i}, \boldsymbol{E}),$$

where, $\boldsymbol{Q_i} = (Q_{i1}, \dots, Q_{ij}, \dots, Q_{im})$ is the vector of the quantities Q_{ij} of the j commodities, $j \in \mathrm{M}, \mathrm{M} = \{1, \dots..\mathrm{m}\}$, the i^{th} individual uses (consumes) and $\boldsymbol{E}$ is the vector of n environmental goods and services available to all k individuals.[23] Individuals receive increasing value as elements in both $\boldsymbol{Q_i}$ and $\boldsymbol{E}$ increase; however, there is a trade-off between them. The different perspectives discussed above can be described by different functional forms of $U_i(\boldsymbol{Q_i}, \boldsymbol{E})$: considering $\boldsymbol{Q_i}$ and $\boldsymbol{E}$ as substitutes or complements at different levels, or even completely excluding trade-offs between their elements. Figure 1 (a) and (b) illustrate two extreme types of preferences graphically, measuring $\boldsymbol{Q}$ on the vertical and $\boldsymbol{E}$ on the horizontal axis.

Figure 1: Individual preferences over private goods and environmental goods



Note: Illustrative example.

[21] By combining competition economics (and the calculation of consumer welfare) with environmental economics (which focuses on the internalization of externalities), we can arrive at a single (monetized) metric. This would not be possible if, as noted above, we took a less anthropocentric approach, such as used in ecological economics (e.g., Common and Stagl (2005)), where such preferences are just one element of various normative criteria to be considered.

[22] This is a definition of externalities in line with Arrow (1969). For a rigorous examination of different treatments of the concept, see Papandreou (1994). It should be noticed that this definition violates one of the primary requirements of the welfare economics discussed briefly below, that of a complete set of markets.

[23] We assume that individuals care only about their own consumption of private goods and services $\boldsymbol{Q_i}$.

Part (a) of the figure presents an individual whose utility depends only on the amounts of the commodities in $\boldsymbol{Q}$ she is using. Her utility for a given vector of commodities does not change as environmental quality changes; her utility increases $U^2 > U^1 > U^0$ as she increases the use of commodities (private goods). On the other extreme, part (b) presents an individual who cares only about environmental quality, and she does not exchange environmental quality for commodities. Her utility increases $U^2 > U^1 > U^0$ only as E increases. Obviously, downward sloping convex indifference curves could better represent trade-offs between commodities and environmental quality, with different shapes indicating different individuals' preferences.

II.2.2 Aggregation of Individual Choices (Brief Overview)

Before examining individual choices, we should make some preliminary remarks on the aggregation of preferences, especially because choices about the environment (a public good) are inherently collective choices. We can define a bundle $(\boldsymbol{Q^0}, \boldsymbol{E^0})$ comprising environmental quality $\boldsymbol{E^0}$ and a specific distribution of given aggregate quantities of the m commodities to k individuals, each receiving the vector $\boldsymbol{Q_i^0} = (Q_{i1}^0, \dots, Q_{ij}^0, \dots, Q_{im}^0)$, where $\boldsymbol{Q^0} = (\boldsymbol{Q_1^0}, \dots, \boldsymbol{Q_k^0})$. In this way, we can represent society's ordering of all possible such bundles by a social welfare function,[24]

$$SW(\,U_1, \dots, U_k).$$

Such a function allows us to compare different bundles: for example, $SW^1 > SW^0$ implies that society prefers the bundle $(\boldsymbol{Q^1}, \boldsymbol{E^1})$ over $(\boldsymbol{Q^0}, \boldsymbol{E^0})$. There are different forms the social welfare function can take, reflecting different societal views and objectives, the most well-known of which are discussed in the following.

The Benthamite or utilitarian social welfare function presents society's welfare as the weighted sum of its members' utilities,

$$SW(\,U_1, \dots, U_k) = \sum_{i=1}^{k} w_i U_i,$$

where $w_i \geq 0$ are the weights representing the degree of substitutability among individuals' utilities. At one extreme, the weights could be equal. At the other extreme, the Rawlsian or max–min social welfare function presents society's welfare as the utility of its least endowed member,

$$SW(U_1, \dots, U_k) = min\,(U_1, \dots, U_k).$$

In between the unweighted utilitarian social welfare function and a complete aversion to uncertainty, as expressed by the Rawlsian social welfare function, there is a variety of different approaches.

Arrow's impossibility theorem, apart from initiating a long discussion around social welfare and giving birth to social choice theory, demonstrated that there is no objective, unambiguous way of

[24] We assume a Bergson-Samuelson social welfare function.

defining a rule to guide social choices. However, given that societies need to make choices, including those discussed in this report, some criterion is needed to provide guidance. Economists usually resort to the safe harbor of the potential Pareto optimality criterion, to which we turn next.[25]

In terms of the terminology developed above, the strict Pareto criterion states that if at least one member $l \in \mathrm{K}$ of the society prefers the bundle $(\boldsymbol{Q^1}, \boldsymbol{E^1})$ to $(\boldsymbol{Q^0}, \boldsymbol{E^0})$, that is, $U_l^1(\boldsymbol{Q^1}, \boldsymbol{E^1}) > U_l^0(\boldsymbol{Q^0}, \boldsymbol{E^0})$, while no other member objects, that is, $U_{i \neq l}^1 \geq U_{i \neq l}^0$, then $(\boldsymbol{Q^1}, \boldsymbol{E^1})$ is Pareto preferred to $(\boldsymbol{Q^0}, \boldsymbol{E^0})$ by society, that is, $SW^1 > SW^0$. The strict Pareto efficiency criterion requires that a set of policies that move the economy from $(\boldsymbol{Q^1}, \boldsymbol{E^1})$ to $(\boldsymbol{Q^0}, \boldsymbol{E^0})$ will be adopted only if the new bundle $(\boldsymbol{Q^0}, \boldsymbol{E^0})$ yields positive aggregate benefits and that a vector of transfers can be made so that at least one member of the society is better off without making any other member worse off. Despite its appeal, the strict criterion poses serious limitations: Without transfers, the applicability of the criterion is very limited since policies that create only winners, and no losers are difficult to even imagine. Requiring all transfers to be made to potential losers, requires among others that all costs and benefits to each person affected by the policies are valued and costless institutions to administer the transfers are created. It is clear that most, if not all, policies would be unable to pass the requirements of the strict Pareto criterion and potential net benefits to society will be forgone.

An alternative is the so-called Kaldor-Hicks or potential Pareto criterion, according to which, if a vector of transfers exists, such that no member of the society objects to moving to a certain bundle, then this bundle is potentially Pareto preferred, regardless of whether the transfers will actually materialize.[26] The key difference between the strict Pareto efficiency and the Kaldor-Hicks criterion is that under the latter compensation to losers is in fact not paid. That is, the Kaldor-Hicks criterion in essence decouples considerations of efficiency from those of equity. Accordingly, it is clear that although every Pareto improvement is a Kaldor-Hicks improvement, not all Kaldor-Hicks improvements necessarily map onto Pareto improvements. The potential Pareto criterion provides the underlying rationale for the Cost Benefit Analysis used in Section III.[27]

We introduce the potential Pareto criterion given its importance in economics, as it has proven invaluable for comparing two distinct outcomes – i.e. *bundles* in the preceding terminology – while focusing only on market efficiency. With a view to distributional equity, as will be explained later in this report, researchers can still measure net benefits that accrue to different groups and present possible compensatory measures to policy makers. Based on these considerations, the notion of efficiency is further developed in the following. In Section III, we then devote detailed attention to techniques for measuring environmental damage.

[25] We note, however, that also other approaches have been advocated, e.g. with a particular emphasis on concepts of fairness (see Varian (1974) or Feldman and Kirman (1974)).

[26] It is clear that transfers cannot involve environmental quality and are confined to transferable goods and services.

[27] It should again be noted that there are other alternatives that could be used in devising choice criteria, including non-consequentialist theories such as the Rawlsian, briefly presented above, or libertarianism.

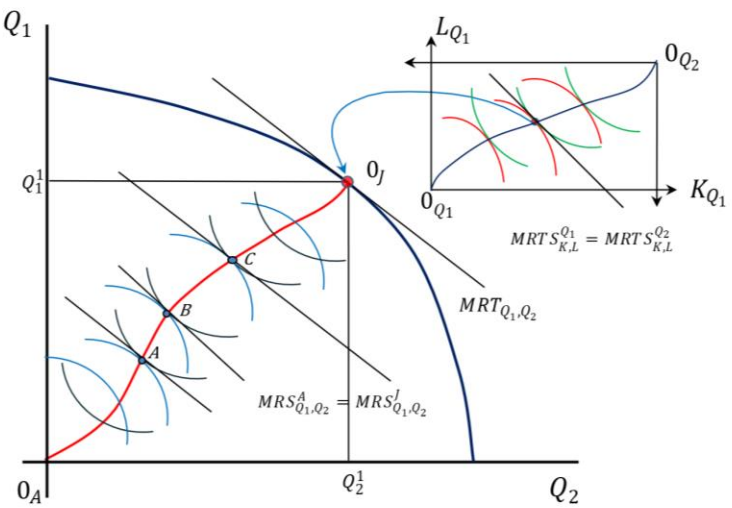
II.3 Efficiency in Markets

We define efficiency in relationship to Pareto optimality, as introduced above. We first do so in a general equilibrium framework. Subsequently, we also introduce a partial equilibrium model in order to discuss economic concepts that are closer to competition policy, such as consumer and producer surpluses.

II.3.1 Market Efficiency as Pareto Optimality

Society strives to achieve efficiency in both production and exchange. Figure 2 provides an illustration of the two basic theorems in welfare economics. In order to allow for graphical illustration, a society consisting of just two individuals, Artemis and John, is assumed, i.e. $k = 2$. Each member possesses given amounts of two productive resources, K and L, and receives value from two commodities, Q_1 and Q_2, that is, $m = 2$. We assume further that both inputs and outputs are traded in competitive markets.

Figure 2: Construction of the (efficient) production possibility frontier



Note: Illustrative example.

Given society's resources, which define the sides of the upper right small box, and technology, illustrated by the red and green isoquants,[28] technological efficiency is achieved on the points of the production contract curve showing the maximum amounts of Q_1 and Q_2 that can be produced.[29]

[28] An isoquant is a curve presenting all combinations of inputs that yield the same level of output. In Figure 2, green isoquants illustrate technology of producing Q_1 and red isoquants technology of producing Q_2.

[29] The production contract curve is illustrated in the upper right small box by the blue curve connecting the points of tangency between the isoquants of producing Q_1 and Q_2, that is, the points at which the marginal rate of technical substitution $MRTS$ in producing the two goods is equated to the ratio of the marginal products of the two inputs, $MRTS_{K,L}^{Q_1} = MRTS_{K,L}^{Q_2}$.

Mapping the points on the contract curve to the (Q_1, Q_2) space yields society's production possibility frontier.[30] Each of these efficiently produced combinations of the two goods can be allocated to Artemis and John in many different ways. For example, *(Q_1^1 , Q_2^1)* defines a box that confines all possible allocations of these efficiently produced quantities, with Artemis receiving everything at 0_J and John receiving everything at 0_A. Given their preferences over the two goods, illustrated by their indifference curves[31] the exchange contract curve defines the efficient allocations of *(Q_1^1 , Q_2^1)* between Artemis and John.[32] Starting from any other point in the box, the utility of at least one of them can be improved by moving to an allocation corresponding to a point on the contract curve. Therefore, the contract curve represents Pareto efficient allocations.

What facilitates the achievement of efficiency is competitive trading in the markets for inputs and final goods. Prices resulting from competitive trading are the signals guiding the allocation of resources along the production contract curve,[33] and the exchange contract curve.[34] Therefore, for any given initial allocation of resources between Artemis and John that is not Pareto efficient, competitive markets can lead to a reallocation that improves the utility of at least one actor. This analysis yields the two fundamental results of welfare economics: (i) competitive markets result in Pareto optimal allocations and (ii) all Pareto optimal allocations can be achieved by competitive markets.

Apart from market distortions, which competition policy aims at preventing, it is precisely the nature of environmental goods and services that poses serious challenges to achieving Pareto efficiency. This is primarily because the elements in the environmental quality vector $\boldsymbol{E}$ are definitely not privately traded, in the sense that there are no well-defined property rights and thus respective markets. Furthermore, producing and consuming some of the commodities, that is, elements in the $\boldsymbol{Q}$ vector, could create external costs to the production of other commodities and to consumers. With regards to production, an externality occurs when the production of a good results, for instance, in pollution negatively affecting the production of another good. If property rights regarding

[30] The production possibility frontier, PPF, is illustrated by the blue convex curve on the main graph. The slope of the PPF is called marginal rate of transformation MRT and is equal to the ratio of the marginal costs of producing the two goods, $MRT_{Q_1,Q_2} = \frac{MC_{Q_1}}{MC_{Q_2}}$.

[31] An indifference curve presents all combinations of the two goods that yield the same utility level, i.e. combinations between which a given individual is indifferent. In Figure 2, as Artemis's indifference curves (black curves) move further away from 0_A, this indicates increased utility levels for Artemis.

[32] The exchange contract curve is illustrated by the red curve in the main graph connecting the points at which the marginal rate of substitution MRS between the two goods is equal for Artemis and John, $MRS^A_{Q_1,Q_2} = MRS^J_{Q_1,Q_2}$.

[33] Denoting by r the price of K and by w the price of L, competitive input markets secure that $w = dL$ and $r = dK$, where dK and dL are the marginal product of K and L respectively, leading thus to, $MRTS^{Q_1}_{K,L} = MRTS^{Q_2}_{K,L} = \frac{dL}{dK}$.

[34] Denoting by P_{Q_1} the price of Q_1 and by P_{Q_2} the price of Q_2, competitive final good markets secure that $MRS^A_{Q_1,Q_2} = MRS^J_{Q_1,Q_2} = \frac{P_{Q_1}}{P_{Q_2}}$.

such pollution are not assigned, efficiency in production will not be achieved.[35] Pollution externalities affect exchange efficiency as well, since individuals cannot control the level of environmental quality $\boldsymbol{E}$, taking it as given in making their choices. Therefore, the shape of their indifference curve is affected, and exchange efficiency will not be achieved.[36]

The preceding discussion highlights two important results: first, how efficiency is achieved in perfectly functioning markets, and second, how this depends on the absence of externalities (for the *internalization* of which no such markets exist). These insights also show the limitations of competition policy alone, when such a failure of internalization occurs.

II.3.2 Defining Preferences in the Presence of Environmental Externalities

The above presentation of efficiency in a general equilibrium framework facilitated the discussion of some important challenges environmental problems pose to achieving efficiency. In order to proceed in devising a metric that will assist decision making at the market level, it is necessary to move to a partial equilibrium framework. In such a framework, one takes the allocation outside the concerned market as a given. In such a partial equilibrium framework, we now define concepts such as supply and demand.

To define market demand, one first needs to consider individual demand. It is assumed that individuals take environmental quality as given and make their choices in selected markets so as to maximize their utility, as defined by $U_i(\boldsymbol{Q_i}, \boldsymbol{E})$, subject to their budget constraints, given the prevailing prices and their income. The constrained maximization problem yields individual i's vector of demand functions, $\boldsymbol{Q_i}$**,** with elements,

$$Q_{ij} = f_{ij}\left(P_j, \boldsymbol{P_{\neq j}}, Y_i, \boldsymbol{E}\right),$$

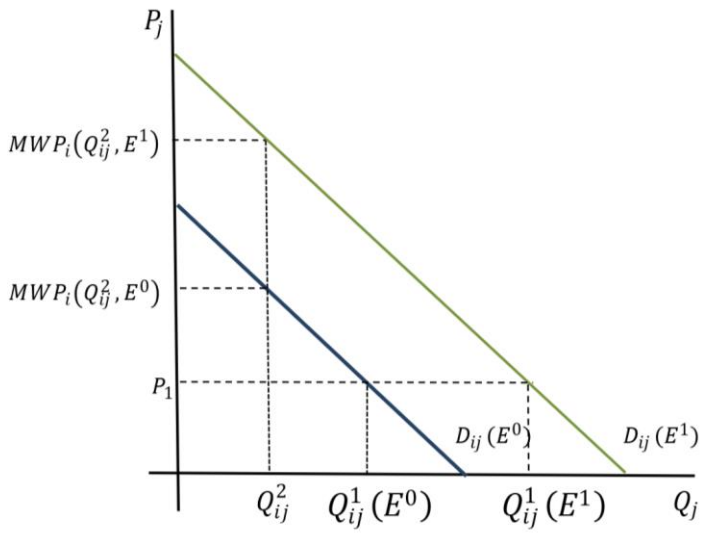
where, P_j is the price of commodity j, $j \in \mathrm{M}, \mathrm{M} = \{1, \ldots.. \mathrm{m}\}$, $\boldsymbol{P_{\neq j}}$ is the price vector of the rest of the commodities individual i uses and Y_i is her income. An individual's marginal willingness to pay for an extra unit of a particular good, say Q_j, is the maximum amount that she is willing to pay to obtain this extra unit. Her marginal willingness to pay is equal to the marginal value (benefit) she receives from using this unit of Q_j. As individuals use more and more units of the same commodity, the additional benefit they receive and thus their marginal willingness to pay, typically decreases. Individual i's demand curve for commodity j depicts the negative relationship between her marginal willingness to pay, i.e. the price of j, P_j, and the quantity she demands, in relation to given values for the rest of the parameters – that is, her income, prices of other goods, and environmental quality. At a given price prevailing in the market an individual will purchase a number of units of the commodity, the last of which will provide a marginal benefit equal to the price. Aggregating her marginal benefits – that is, her marginal willingness to pay over all units – yields her willingness to pay for the particular quantity of the good.

[35] In cases where such pollution sufficiently affects the production of other goods, the set of production possibilities could even become nonconvex and affect pricing.

[36] In the case of consumption externalities, nonconvexities in individuals' indifference curves may obtain, as analyzed in Papandreou (1994).

Increasing total production of some good j is now supposed to lead to the reduction of some elements of vector $\boldsymbol{E}$, resulting in lower environmental quality. This situation – in which a trade-off between higher production and consumption of the considered good and environmental quality exists – is arguably the interesting case in this report. Figure 3 presents a simple, linear illustration of individual i's demand for good j, in two different situations (associated, for example, with different technologies) resulting in different levels of pollution per unit of j's production, denoted as environmental quality $\boldsymbol{E}^{\mathbf{1}}$ and $\boldsymbol{E}^{\mathbf{0}}$ with $\boldsymbol{E}^{\mathbf{1}} > \boldsymbol{E}^{\mathbf{0}}$.[37,38] Her willingness to pay for any quantity Q_{ij}^2 is higher at the higher level of environmental quality – that is, she is willing to pay more to consume commodities that are associated with higher environmental quality. Alternatively, for any given price p_1, she is willing to reduce her consumption from $Q_{ij}^1(\boldsymbol{E}^{\mathbf{1}})$ to $Q_{ij}^1(\boldsymbol{E}^{\mathbf{0}})$ if environmental quality falls from $\boldsymbol{E}^{\mathbf{1}}$ to $\boldsymbol{E}^{\mathbf{0}}$.[39]

Figure 3: Individual (inverse) demand for different levels of environmental quality



Note: Illustrative example.

As the market is distorted by environmental externalities, regulatory intervention is required. Regulators should make their decisions based on aggregate demand. The latter is obtained by aggregating $Q_{ij} = f_{ij}(P_j, \boldsymbol{P}_{\neq \boldsymbol{j}}, Y_i, \boldsymbol{E})$ over the k individuals and then substituting the effect that aggregate

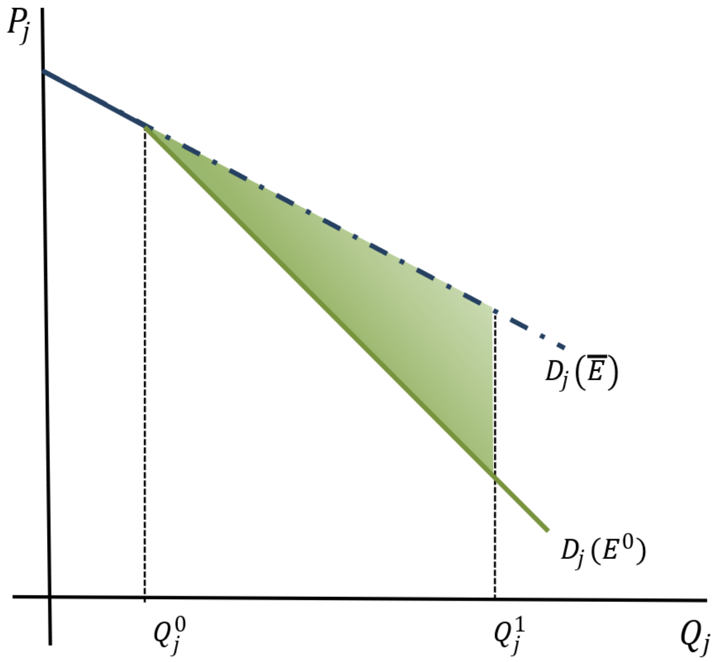
[37] We assume, first, that she has complete information about the level of environmental quality and, second, that she realizes that, although her own choices affect environmental quality, the change is insignificant when k is large. Hence, she ignores the effect of her own consumption on environmental quality. Allowing individuals to take into account the environmental damage their own consumption imposes on them, little as it might be relative to the total damage, will lead to a slight rotation, in addition to the shift, in Figure 3.

[38] The above notation implicitly assumes that production of Q_j generates pollution that negatively affects at least one element in the $\boldsymbol{E}$ vector, without improving any other.

[39] Although not unique, the above illustration is a reasonable way of presenting preferences of environmentally conscious individuals. In presenting environmental externalities, introductory economic textbooks often assume instead that individuals, realizing that their actions have little to no effect on environmental quality, they do not change their market choices at different levels of environmental quality. In Figure 3, different levels of environmental awareness result in different shifts to demand.

production $Q_j = \sum_{i=1}^{k} Q_{ij}$ has on environmental quality. Figure 4 illustrates aggregate demand for good j at two levels of environmental quality associated with the aggregate production of j.

Figure 4: Aggregate (inverse) demand for different levels of environmental quality



Note: Illustrative example.

Before addressing a shift in environmental quality, we discuss first the definition of consumer surplus derived from the consumption of the considered good. In non-technical language, if we consider a total consumption of Q_j^1 units of that good, as in Figure 4, we need to aggregate the respective willingness-to-pay. Since the derived (inverse) demand curve represents an individual's marginal willingness to pay for given quantity of the commodity, the area under the (inverse) demand curve should measure total willingness to pay. Therefore, the area now under the aggregate (inverse) demand curve up to Q_j^1 provides a measure of the aggregate willingness to pay of all market participants. If we subtract any payment they make to purchase these units we arrive at the definition of consumer surplus. We return to this in the subsequent section. Now, with the help of Figure 4, we discuss a shift in environmental quality.

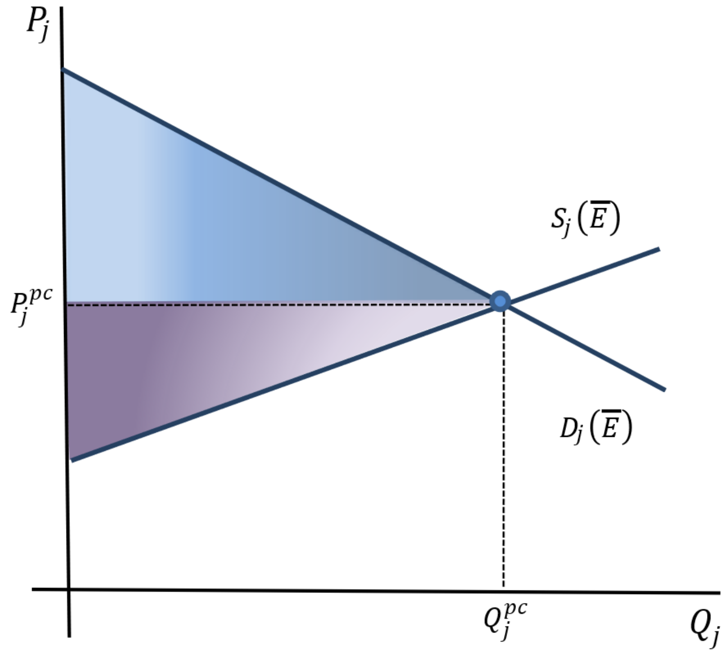
In Figure 4, environmental quality $\overline{\boldsymbol{E}}$ is associated with production of Q_j that does not harm the environment at all, while $\boldsymbol{E^0}$ is the environmental quality when production leads to some reduction in environmental quality. The level of production Q_j^0 is thus defined by the environment's capacity to fully assimilate the pollutants resulting from the production of j. Such full assimilation is no longer obtained at the higher quantity Q_j^1. If Q_j^1 units of j are produced and consumed, the integral between the two demand curves from Q_j^0 to Q_j^1, i.e. the shaded area in Figure 4, is a measure of the total willingness to pay of all those individuals $i \in \mathrm{K}$ who participate in the market for good j, for reducing the impact of the production of Q_j^1 units on environmental quality from $\boldsymbol{E^0}$ to $\overline{\boldsymbol{E}}$.

In Section III we return both to the definition of consumer surplus and to that of its difference under two scenarios – that is, with and without a possible agreement – in a more formal way, as required for the respective measurement.

II.3.3 Consumer and Producer Surplus and the Efficiency of Competitive Markets

To complete our presentation of the market, we must also define supply. The supply curve, while considering the current state of technology, defines the positive relationship between price and the quantity produced. *Producer surplus* resulting from selling a given quantity is defined as the total revenue the producer receives minus her costs. Figure 5 illustrates the demand $D_j(\overline{\boldsymbol{E}})$ and the supply $S_j(\overline{\boldsymbol{E}})$ of good j. Absent externalities, the point at which the two curves intersect defines the price P_j^{pc} and quantity Q_j^{pc} at which efficiency is achieved; there is no other combination of price and quantity which, disregarding externalities, could achieve higher value for the society, measured here as the sum of the consumer and producer surplus.

Figure 5: Consumer surplus and producer surplus



Note: Illustrative example.

Recall now our preceding discussion of efficiency in the context of general equilibrium. Presently, we consider instead only a single market in isolation, for a partial equilibrium analysis. Here, we have defined, as the intersection of aggregate supply and demand, the outcome that maximizes the sum of the consumer and producer surplus. In this partial analysis, this is also the Pareto efficient outcome. It is achieved under perfect competition, hence the index *pc*, where each market participant acts as a price taker.[40] When market power is exercised, however, this outcome is typi-

[40] Of course, this and the above graphical analysis requires a number of assumptions which in this report we need not spell out, such as homogeneity of goods or perfect information.

cally no longer realized. Competition policy's objective is therefore to achieve greater market efficiency through the restriction of market power. We now return, however, to an explicit consideration of externalities.

II.4 Consumer Surplus in the Presence of Environmental Externalities: A Graphical Illustration

Having introduced the notion of consumer and producer surplus, we re-introduce externalities to examine our case of interest: when higher environmental quality entails a trade-off, and hence a cost. In the considered case, this could entail, as we now assume for the purpose of illustration, a higher marginal cost of production for the considered good. Thus, the comparison between the two scenarios involves, first, an increase in cost, which ultimately leads to an increase in price, and a reduction in environmental damage, as caused by the production and consumption of each unit of the considered product. In Section III we return to this trade-off with the help of additional formalization.

Consider now Figure 6(a), where good j generates externalities. Pollution resulting from the production and consumption of good j notably generates damages for a wide segment of the society, including – but not limited— to consumers of good j. In order to simplify our graphic illustration, we do not consider environmentally conscious individuals, that is, aggregate demand does not rotate inwards when environmental quality is $\boldsymbol{E^0}$. Individuals participating in the market for good j completely neglect all consequences their purchases have on environmental quality and their demand thus remains at the same level it was in the absence of pollution, that is, at environmental quality $\overline{\boldsymbol{E}}$.[41] Therefore, the market demand is $D_j(\boldsymbol{E^0})$.[42] We assume an upward sloping supply curve denoted by $S_j(\boldsymbol{E^0})$. Therefore, the market yields equilibrium output $Q_j(\boldsymbol{E^0})$.

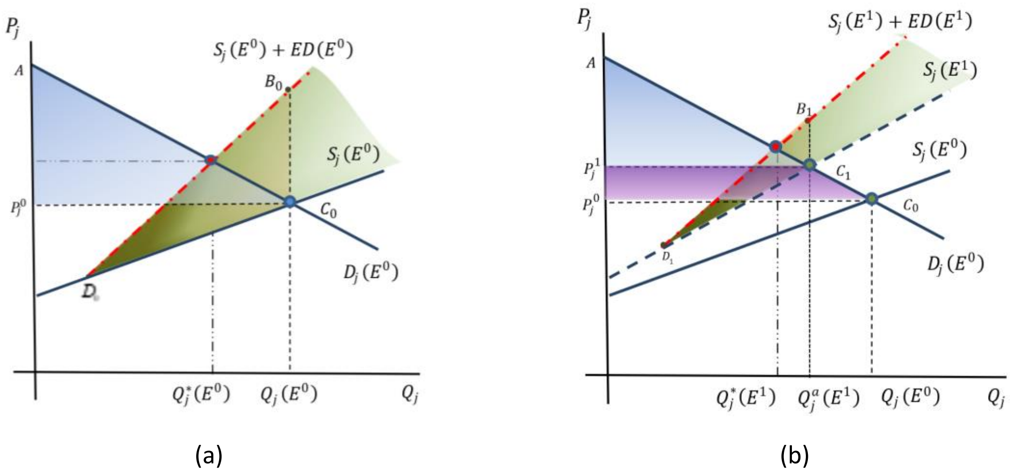
Although overlooked by market participants, pollution resulting from good j has a negative impact on society's members, including – but not limited to – market participants. A regulatory authority has to define the group of affected individuals and measure the total damage – i.e. the external costs – generated by good j's market. In order to proceed with the simple graphical illustration, let us assume that total damage has been measured, using some of the methods presented in Section III, and added to private costs. In Figure 6(a) total damage is presented by the green area, which if added to private production costs, presented by $S_j(\boldsymbol{E^0})$, shifts the supply curve to $S_j(\boldsymbol{E^0}) + ED(\boldsymbol{E^0})$ to reflect the full cost that the market for good j imposes on society. Total environmental

[41] Recall that in Section II.3.2 we considered two different levels of environmental quality, with $\overline{\boldsymbol{E}} > \boldsymbol{E^0}$.

[42] In relation to Figure 4, we assume now that the $D_j(\boldsymbol{E^0})$ collapses to $D_j(\overline{\boldsymbol{E}})$. We choose to denote the demand with the actual and not the perceived by consumers level of environmental quality.

damages, denoted by $ED\left(\boldsymbol{E^0}\right)$, include damages suffered by market and non-market participants.[43] It is evident that the social optimum requires a reduction in j's quantity to $Q_j^*\left(\boldsymbol{E^0}\right)$.[44]

Figure 6: Demand and supply under different scenarios (environmental qualities)



Note: Illustrative example.

Building on the example developed so far, consider a case in which the producers in this market consider collectively adopting a new, less polluting but more costly technology that results in an improvement of environmental quality from $\boldsymbol{E^0}$ to $\boldsymbol{E^1}$. In terms of Figure 6(b) the increased private costs are illustrated by a shift in the supply curve to $S_j(\boldsymbol{E^1})$. External costs – environmental damages – are significantly reduced,[45] leading to the social supply curve $S_j(\boldsymbol{E^1}) + ED(\boldsymbol{E^1})$ associated with environmental damages $ED(\boldsymbol{E^1})$. Assuming that firms will operate along the supply curve $S_j(\boldsymbol{E^1})$, this yields an equilibrium at $Q_j^a(\boldsymbol{E^1})$. The social optimum under the new technology, which results in environmental quality $\boldsymbol{E^1}$**,** is defined at $Q_j^*(\boldsymbol{E^1}) < Q_j^a(\boldsymbol{E^1})$.

The example ultimately leads to the question of whether the illustrated reduction in environmental damages, i.e. the improvement of environmental quality from $\boldsymbol{E^0}$ to $\boldsymbol{E^1}$, would justify the reduction in the surplus that individuals participating in this market enjoy. Specifically, we turn first to environmental benefits. Starting from the equilibrium at $Q_j\left(\boldsymbol{E^0}\right)$ in part (a) and moving to the equilibrium at $Q_j^a\left(\boldsymbol{E^1}\right)$ in part (b), the reduction of environmental damages can be measured by the difference in the areas of the triangles $D_0C_0B_0$ and $D_1C_1B_1$. Next, the reduction in consumer surplus

[43] Keeping with the linear graphic illustration, it is assumed that environmental damages increase at a constant rate as production increases. This is another simplifying assumption, since in most cases environmental damages increase at an increasing rate with production.

[44] If we considered environmentally conscious consumers, part of the total damage would be presented by a shift in the demand function, representing partial internalization of the externality created in the market. However, this would complicate the graphical illustration. Furthermore, assuming environmentally conscious consumers complicates the derivation of optimal environmental policy (see, for instance, Constantatos et al. (2021)). This simplification allows us to present the main trade-off in the simplest possible way.

[45] Figure 6 is drawn such that $ED(\boldsymbol{E^1}) < ED(\boldsymbol{E^0})$ at all levels of production.

moving from $Q_j(\boldsymbol{E^0})$ in part (a) to $Q_j^a(\boldsymbol{E^1})$ in part (b) is measured by the difference in the two areas under the demand, as shown by the purple shaded area $P_j^0 C_0 C_1 P_j^1$ in Figure 6(b).[46]

Despite its abstract nature, this simple depiction highlights important questions. First, which market equilibrium will arise with and without the considered agreement of firms (or, respectively, an alternative impediment to competition)? The calculation of the likely outcomes falls squarely into the domain of competition economics, and entails, among other things, the estimation of the respective prices in the two scenarios and the possible loss of consumer welfare (see Section III.1).

The second question of importance relates to the group of individuals who do not participate in the market but may suffer damages. Put differently, whose welfare should competition authorities take into account? Though this question is clearly outside the remit of this report, we already discussed how various welfare standards allow for the weighting of the preferences and welfare of different individuals and groups. In addition, there may be heterogeneity in marginal willingness to pay for environmental quality. As our subsequent analysis will treat consumer welfare (in the market) and environmental damages separately, if deemed appropriate such a weighting may be undertaken.[47]

Third, what do we include as costs and benefits and how can environmental damages be measured in monetary terms? The measurement of such benefits is at the heart of the second part of the subsequent section (see Section III.2 for an overview).

Fourth, how should we deal with uncertainty regarding costs and benefits, and their timing? In particular, how should costs and benefits, stretched far into the future, be accounted for and how should they be compared among themselves? We will deal with this issue separately at the end of the subsequent section (in Section III.8).

II.5 Accounting for Changing Preferences and the Time Dimension

Time is a very important dimension in many environmental problems, in part due to the time scale of environmental processes. Global warming is an example in which time plays a very important role because greenhouse gases are stock pollutants: current emissions accumulate in the atmosphere and stay active for over a century, imposing damages far into the future. In addition, the magnitude of their impact will depend in part on cumulative emissions at each point in time. Similarly, the impact of certain current choices, such as deforestation, on the loss of biodiversity, could extend even further into the future. Thus, current actions could have cumulative impacts of greater scope than considered here. More generally, environmental goods and services – which we have subsumed under *environmental quality*, or vector $\boldsymbol{E}$ — affect societal welfare far into the future.

[46] It may also be noted that the above example, as illustrated in Figure 6(b), assumes that the increase in private costs required to achieve the improvement in environmental quality is of significant size. However, a situation in which a substantial environmental improvement can be attained with relatively small cost cannot be ruled out. Such cases arguably pose less of a challenge to authorities.

[47] This relates also the question whether and to what extent consumers should be given a *fair share* in the achieved (environmental) benefits of an agreement. See also Section III.9 on an additional formal treatment of distributional issues.

Accordingly, there is a need to discount welfare levels at different points in time and derive consistent and efficient intertemporal choices.[48] In addition, when intertemporal choices are considered, various issues regarding individual and aggregate preferences arise. A couple of these preferences will be briefly examined prior to Section III.

The foregoing analysis assumed that all individuals have the same and unchanging preferences toward the environment. However, in the real world, individuals diverge in how they value environmental quality, as this divergence is mediated by various factors, including political affiliation, education level, income, gender, age, and perhaps even social status considerations.[49] While different sampling methodologies can be used to estimate aggregate willingness to pay for environmental quality – that is, total environmental damage at a given point in time – the preferences of a given society could shift over time – for example, because new information becomes available. Accordingly, one strand in the literature examines how preferences change over time, using both traditional and evolutionary approaches.[50]

However, the more salient issue is to account for the effects that current choices could have on future environmental quality, and thus on the utility of individual members of future societies. Yet even if the future effects of current actions can be predicted, the preferences of future generations regarding environmental quality – that is, their utility function – remain unknown. Although the most common way of accounting for benefits to future generations is to assume that they have the same preferences as the current generation, this is a questionable assumption, as individual preferences may change over the lifecycle – for example, due to new information regarding the effects of pollution, or the wider dissemination of such information in society. Also, individual preferences may follow broader societal shifts, given the herd behavior often visible in public opinion. Moreover, such changes could be significantly more pronounced within certain cohorts or generations.

In Figure 6(b), the agreement among producers that results in the improvement of environmental quality could yield substantial benefits to future members of the society. Crucially, a failure to take into account the benefits that the agreement could generate for future individuals might lead the proposed agreement to be abandoned, particularly if the benefits to the current generation fall short of the costs.[51]

[48] For an excellent review of these issues see Heal (2005).

[49] See, for instance, Torgler et al. (2008).

[50] See, for instance, Sartzetakis et al. (2015).

[51] Inderst and Thomas (2020a) propose including such changes in preferences in a prospective welfare analysis. Various approaches to forecast preferences and changes therein may be considered, such as the following: (i) When preference changes are most likely to due to changes in available information or in social norms – as derived, for instance, from the behavior of others – when researchers elicit preferences through contingent valuation or conjoint analysis, individuals can be primed accordingly, e.g. by providing the respective information; (ii) when a reliable causal relationship between preferences and sociodemographic variables can be established, this may be used to forecast future preferences; (iii) in addition, aggregate data on social and cultural changes can be useful.

III MEASUREMENT METHODS

In the preceding section (Section II), we introduced the basic framework of welfare analysis, particularly as it relates to environmental externalities. We also explained our focus on welfare as measured in terms of the aggregation of individual preferences. In this regard, we introduced the concept of Pareto efficiency. We then showed how, in the absence of property rights and markets, market forces alone will not lead to a welfare-maximizing outcome for public goods. In our final partial equilibrium analysis, we focused on a single market and introduced the concept of consumer surplus (both when consumers are environmentally conscious and when this is not the case) and that of producer surplus. Within our partial equilibrium framework, we also conducted an initial comparison of two scenarios, one with and one without an agreement that would lead to fewer externalities but higher production costs.

In the following, we perform a partial welfare analysis – *partial* because we only focus on two components of welfare. Specifically, when considering a producer agreement, we split the corresponding welfare impact into two components: (i) the potential welfare loss of the consumers in the affected market and (ii) the welfare gain for the society due to improved environmental stability (reduced environmental damage). Our decision to focus solely on the consumer surplus was motivated by the scope of our mandate. Accordingly, we do not take a stance on whether such a focus is generally warranted or on whether competition policy should focus mainly or exclusively on the enhancement of consumer welfare.[52]

Accordingly, in what follows we consider first in Section III.1 the measurement of (changes in) consumer surplus. This extends the preceding discussion in Section II. From Section III.2 onward we proceed to the measurement of environmental costs and benefits. The measurement of the welfare impact that results from a change in environmental sustainability is mainly associated with non-market goods. We will first discuss the respective measurement methods based on revealed or stated preference approaches. These approaches essentially follow the general welfare methodology introduced in Section II, by which individual preferences are calculated and then aggregated. Also, in light of their practical relevance, we additionally discuss measurement approaches that are oriented to a given policy objective (that presumably seeks to advance public preferences). We complete this section with a discussion of time preferences and distributional issues.

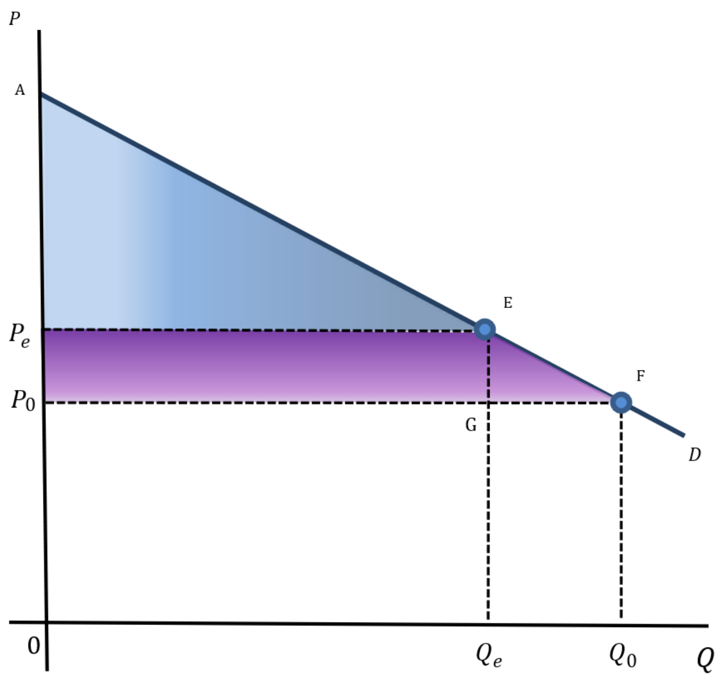
[52] Our partial welfare analysis is, with the restriction to consumer surplus, thus based on a potential Pareto efficiency criterion, formalized by the Kaldor-Hicks criterion and as discussed in the preceding section. Recall that according to this criterion, a considered alternative scenario will be socially preferable if and only if the gainers from the improved environmental conditions could hypothetically fully compensate the looser and still be better off.

III.1 Consumer Surplus

III.1.1 Measurement of Marshallian Consumer Surplus (Changes)

We previously introduced the concept of consumer surplus in Section II.3.2. We discussed how it was obtained, in a partial equilibrium analysis, from an individual's willingness to pay for additional units of a considered good. Figure 7 reproduces our graphical representation, but now we consider different price levels. In Figure 7, the consumer surplus from purchasing quantity Q_0 at the price P_0 is the area AP_0F, which is the difference between the amount that the consumer actually pays (area $0P_0FQ_0$) and the maximum amount that she/he is willing to pay (area $0AFQ_0$). An increase in the price paid by the consumer from P_0 to P_e with a corresponding reduction in purchased quantity for Q_0 to Q_e will reduce the consumer surplus by the area (P_0P_eEF). Recall also that the aggregate consumer surplus can be obtained by summing the consumer surplus of each individual.

Figure 7: Consumer surplus for different price levels



Note: Illustrative example.

In what follows, we will briefly discuss the measurement of (changes in) consumer surplus. Before doing so, we note that in all the above analysis the *Marshallian market demand curve* was used. The Marshallian demand measures both income and substitution effects.[53] We return to this below.

As the preceding graphical construction shows, the Marshallian consumer surplus can be obtained by integrating an estimated Marshallian demand function. If the demand was linear, then the graphical illustration shows how the change in consumer surplus could be easily determined (through the calculation of the respective geometric areas, more generally simply by integration).

[53] The substitution effect is the change in consumption that arises if the prices change but the agent is given enough income to maintain the same utility they had at the initial prices. The income effect is the change in consumption that arises if the consumer's income falls but if prices stay the same.

Unless it is possible to obtain elasticity estimates for sufficiently many points along the demand curve,[54] it is common to assume a specific functional form to estimate the demand curve. A widely used specification is that of loglinear demand of the general form

$$Q = A^{\gamma} P^{a} Y^{b},$$

for which one would need to estimate the parameters (a, b, γ).[55] With these variables, the change in the Marshallian consumer surplus – again, based on the considered increase in price – can be obtained as follows:

$$Change\ in\ Consumer\ Surplus = -A^{\gamma} Y^{b} \frac{(P_e^{1+a} - P_0^{1+a})}{(1+a)}.$$

III.1.2 Adjustments for Income Effects

As already noted, the discussed Marshallian demand curve does not take into account income effects. When these are non-negligible, the appropriate measures are the compensating variation (CV) and the equivalent variation (EV) defined in terms of the Hicksian demand, which is composed solely of substitution effects. These are defined as: [56]

$$CV = c(u_0, P_e) - c(u_0, P_0),$$
$$EV = c(u_e, P_e) - c(u_e, P_0).$$

Both metrics measure the monetary costs of a welfare-affecting price change from P_0 to P_e, as expressed by the respective function c, with CV using the utility level u_0 as reference (compensation returns the consumer to the original welfare level) and EV using the utility level u_e attained with price P_e.

Willig (1976) showed that the Marshallian consumer surplus (CS) is usually a good approximation of the appropriate welfare measures, CV and EV with

$$CV \geq CS \geq EV.$$

In particular, it was shown that with constant income elasticity of demand η then

$$CV \approx CS + \frac{\eta CS^2}{2Y_0},$$
$$EV \approx CS - \frac{\eta CS^2}{2Y_0},$$

where Y_0 is income. This result implies that for small income effects the approximation is indeed good and for $\eta = 0$, $CV = CS = EV$. The approximation will be good if the expenditure for the product in question represents a small portion of the consumers expenditure, so that income effects are not large.

[54] See Cohen et al. (2016) for an estimation of elasticities at several points along the demand curve using big data at an individual level.

[55] See Davis and Garcés (2009, ch. 9) for an introduction to the estimation of demand functions.

[56] The Hicksian demand is called also compensated demand function, because it takes prices and utility as arguments rather than prices and income as does the Marshallian.

Returning to the aforementioned log-linear demand model, where we previously kept income unchanged, this can now be modified. Hausman (1981) calculates the CV (or the exact consumer surplus) using the Hicksian compensated demand for a loglinear ordinary demand function as

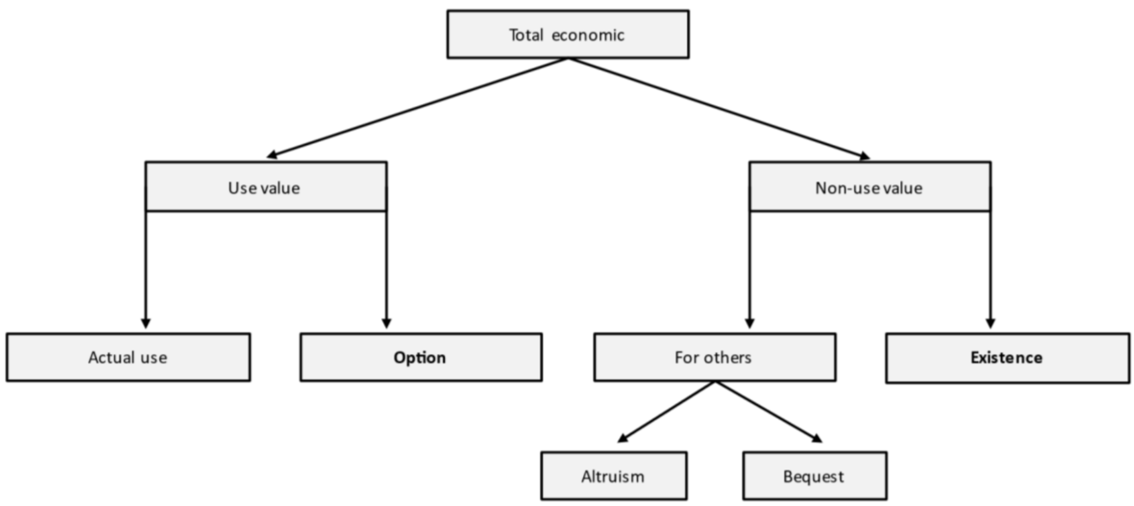
$$CV = \left\{ \left[-A^{\gamma} Y^{b} \frac{(P_e^{1+a} - P_0^{1+a})}{(1+a)} \right] (1-b) + Y^{1-b} \right\}^{\frac{1}{1-b}} - Y.$$

Clearly, when income effects do not enter into demand, with $b = 0$, we are back to the previous formula.

III.2 The Valuation of Environmental Costs and Benefits: Overview[57]

Total economic value (TEV) is a concept that seeks to capture the overall welfare gains attributable to improvements in environmental quality[58] or other factors that contribute to human well-being. TEV, which provides an all-encompassing measure of the economic value of an environmental asset, consists of both use and non-use values. Use value relates to actual, possible, or planned uses of the good in question, while non-use value refers to a valuation not based on actual, planned, or possible use by oneself (though possibly by others). Non-use values are categorized as (i) existence values, (ii) altruistic values, and (iii) bequest values. The described decomposition of TEV is shown in Figure 8.

Figure 8: Total economic value



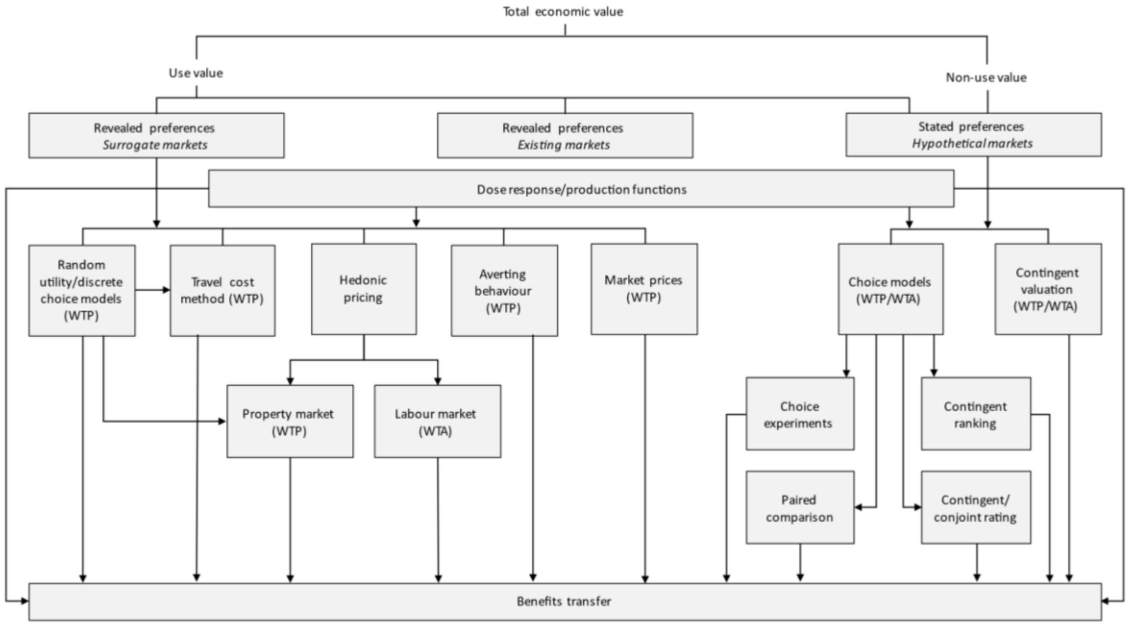
Source: Pearce et al. (2006).

For the majority of environmental goods and services, TEV is measured using valuation techniques, as shown in Figure 9. We will briefly comment on these techniques before describing them in greater detail.

[57] For detailed presentations of valuation methods for environmental goods and services see, for instance, Champ et al. (2003), Pearce et al. (2006), Hanley and Barbier (2009), Atkinson et al. (2018).

[58] Note that improvement in environmental conditions promotes environmental sustainability. Therefore, in the rest of this section the term *environmental* will be used without the term *sustainability*.

Figure 9: Total economic value measurement techniques



Source: Pearce et al. (2006).

For most measurement techniques, as shown in Figure 9, the first step is to determine revealed preferences. For example, a particular good may be available for purchase in two variants: one more environmentally sustainable, and one less so. Based on purchase data, we can extrapolate consumers' preferences for this attribute, i.e. the product's environmental friendliness.[59] The respective techniques resemble those used in the case where actual purchase data is not available and, instead, hypothetical choice decisions are analyzed. The latter are termed stated preferences. In what follows, we will mainly confine ourselves to the analysis of such hypothetical decisions. The main reason for this is that producer agreements may pertain to goods that are not yet produced.[60]

Since the use of an environmental good usually leaves a behavior trail in markets for non-environmental goods, revealed preference approaches may also be used to estimate use values even when the considered good is itself not traded (see the left-most box in Figure 9). This is done by looking at markets in which the use value of the environmental good can be revealed. However, revealed preference approaches are unlikely to elicit non-use values. Non-use values can be elicited, instead, by the aforementioned stated preference methods (see the right-most box in Figure 9).

In the following, we will discuss the measurement methods that can be used to determine revealed and stated preferences, as they appear in Figure 9. Figure 9 also mentions dose-response methods. These studies relate the dose – for example, a change in air pollution – to respective responses,

[59] We note here that obviously the use of purchase data – or, more generally, the application of surveys or choice-based methods – in relation to (potential) consumers does not allow us to extract the preferences of individuals who are not (potential) consumers.

[60] In what follows, we will, however, also comment on why the analysis of hypothetical choice data may be valuable and will provide additional information, even when the concerned products may already be traded in the market, so that real purchase data can be analyzed as well.

such as a change in medical costs or labor productivity. Again, various measurement techniques are possible in this regard, as we discuss in the following.

Our subsequent analysis goes beyond these methods, however. Specifically, we also present the newly developed method of subjective well-being valuation, which differs from the above non-market valuation methods because the estimation of value is based on the impact of changes in non-market goods on self-reported well-being measures, such as satisfaction. Subjective well-being valuation is therefore based on experience, rather than decision utility. In addition, for practical purposes we also refer to methods that make use of existing results, rather than obtaining the respective preferences from a new empirical study. Our application of the method of benefit transfer (see bottom of Figure 9), draws on data obtained in another study in order to assess environmental values in our case. We also discuss existing studies measuring various effects – for instance, environmental damages associated with particulate emissions. Finally, we complement the discussion with a reference to cases in which the respective measurement relates to a particular political objective, which may be thought of reflecting society's preferences. For instance, the imposition of a cap on carbon emissions or substances may reflect such preferences, even when the implied price may not fully reflect the respective social costs, as derived from an (alternative) impact analysis. Here, we relate the potential measurement of achieved environmental benefits to a cost-effectiveness analysis, and, in this way, compare common measures for achieving policy objectives.

III.3 Revealed Preference Methods

Revealed preference methods rely on the analysis of observed market data. As noted above, the good of interest may not be traded directly. This necessitates the examination of surrogate data from other markets. Various methods have been proposed in this regard. These methods may be specific to particular *goods*, such as the travel cost method. Others, like that of hedonic pricing, can be applied very broadly whenever market prices for another good exist (such as houses) that in turn reflect (or depend on) the considered good (such as air quality). In what follows, we introduce these methods more broadly and also to provide sufficient detail, when appropriate, to guide their application.

III.3.1 Travel Cost Method

The travel cost method (TCM) was developed to value the use of non-market goods, particularly geographical areas and locations used for recreational purposes (e.g., Parsons (2003)). The visitation of natural areas typically does not command a price in the market, and so there is a need to find alternative means of appraising their value. The TCM approach recognizes that there are various inputs to the production of a recreational experience, including travel to and from the recreational area, local accommodations, and so on. Typically, even if the recreational area itself is an unpriced good, many of the other factors employed in the generation of the recreational experience do command prices in markets, and these prices can be used to infer the value of the environmental good.

To estimate values using TCM, we need two pieces of information: (i) the number of trips that an individual or household takes to a particular recreational area over the course of a year and (ii) how much it costs that individual or household to travel to the recreational area. Such information is usually collected through surveys carried out at the recreational site. The cost of travelling to a recreational area includes two elements: (i) the monetary costs in return fares or fuel expenses, wear and tear and depreciation of the vehicle and so on and (ii) the cost of time spent travelling. Time is a scarce resource to the household, and demand for trips will be greater if it takes less time to travel to the recreational area, independent of the monetary cost of travel.[61] The TCM assumes weak complementarity between the environmental asset and consumption expenditure. This implies that when consumption expenditure is zero (e.g. when people take no trips to a forest) the marginal utility of the public good is also zero. So, if travelling to a forest becomes so expensive that no one goes any more, the marginal social cost of a decrease in the quality of that forest is also zero. This has important implications. The TCM therefore estimates use-values but cannot estimate non-use values, e.g. those associated with the existence of a forest.[62]

The valuation using the TCM is conducted with two main approaches, depending on whether a single site or multiple sites are being considered, as discussed in the next sections. This discussion also includes technical details for the method's practical application.

Single Site Models

The zonal approach entails dividing the area surrounding a recreational site to be valued into *zones of origin* from which visitors to the site are observed to travel. At the core of the empirical implementation is the estimation of a demand function, from which, as discussed above, consumer surplus (changes) can be derived.

For instance, the demand from some zone z for visits of a specific recreation side could be specified to be linear (e.g. Parsons 2003), such as[63]

$$V_z = \beta_1 TC_z + \beta_2 TC_o + \beta_3 S_z,$$

where TC_z are the travel costs from the respective zone z, TC_o is travel cost to other competing recreational sites and S_z are socio-economic variables (such as income).[64] As we already noted, once the demand function for the zonal model has been estimated, the gains in consumer surplus can be estimated by the methods described in Section III.1. To be specific, if we now take a general

[61] Empirical work suggests that time spent travelling is valued at somewhere between a third and a half of the wage rate and travel cost researchers frequently use one or the other of these values as an estimate of the price of time.

[62] Another implicit assumption made in most travel cost studies is that the representative visitor's utility function is separable in the recreation activity being modeled. That is, if the activity of interest is fishing, then the utility function is such that demand for fishing trips can be estimated independently of demand, say, for cinema trips (alternative leisure activities) or heating oil (alternative non-leisure market goods).

[63] To simplify the exposition, we only represent the deterministic part of utility.

[64] In the zonal TCM the dependent variable (V) can only take integer values which correspond to visits per year. This kind of data is known as count data and using standard Ordinary Least Squares to estimate the zonal TCM regression is incorrect. Instead, a Poisson or negative binomial regression model should be used. The Poisson model has the property that the conditional mean (the expected value) of the dependent variable is equal to the variance; if this is not true for the data, a negative binomial model should be used instead.

demand for visits $f(TC_z, TC_o, S_z)$, the aggregate consumer surplus for a given travel cost level TC_{z1} is

$$Consumer\ Surplus = \int_{TC_{z1}}^{TC_{choke}} f(TC_z, TC_o, S_z)dTC,$$

where TC_{choke} is the choke travel cost price (at which no more visits are undertaken).

Multi Site Models

The traditional approach to TCM is to determine the non-market value of recreation at a particular site under current site conditions – answering, for example, the question: "What would be the loss in welfare if a forest were to be cleared, or a national park closed to public access?" A more recent approach to TCM seeks to determine recreationalists' choice of site from a group of choice alternatives. This is the random utility choice model approach in which individuals face a choice problem that requires them to select a destination site (for example, for a mountain bike trip) from a group of close substitutes (for example, all mountain biking sites in a certain region). Here, the undertaken choices are explained by way of the respective attributes or characteristics of each site. One of these attributes is the cost of visiting the site.

In a multi-site model, the deterministic part of utility is usually assumed to be a linear function of site characteristics:

$$V_{ij} = \beta_1 + \beta_2 X_2 + \beta_3 X_3 + \cdots + \beta_n X_n + \lambda\big(Y_i - p_{ij}\big),$$

where the X_j represent attributes of the different sites that could be visited, Y_i is an individual's income, as one possible sociodemographic variable, and p_{ij} is travel costs of visiting site j for individual i. An individual's utility depends, in addition, on a stochastic part. It is assumed that the individual will visit the site that generates the highest utility. In what follows, we will frequently encounter such a framework of discrete choice analysis. In the literature, these models are made tractable by assuming that the stochastic part of individual utility exhibits a particular distribution. To be specific, using the conditional logit model, the probability that site j will be chosen over all other sites in C for individual i can then be expressed explicitly as

$$\pi_i(j) = \frac{\exp\big(V_{ij}\big)}{\sum_{k\in C}\exp(V_{ik})},$$

where the denominator represents the sum over all alternatives (belonging to the respective choice set C). Available information on individual choices, as well as on aggregate market shares of the different sites, allow us to estimate the respective parameters in the utility function. The signs of these parameters will indicate the way in which site attributes influence site choice. What is key for our purposes, however – beyond forecasting demand – is how consumer surplus changes when attributes of a given site or the overall availability of the site changes. Such a change may be brought about by an agreement between firms, or such an agreement may prevent the realization of a (negative) change. Specifically, the change in utility for a representative individual if one of the sites in the choice set is shut down is given by

$$-\frac{1}{\lambda}\left[\ln\left(\sum_{j\in C}\exp(V_{j0})\right)-\ln\left(\sum_{j\in C}\exp(V_{j1})\right)\right].$$

The respective utility changes can then be converted into a money metric using the inverse of the marginal utility of income, which is also obtained from the analysis.

III.3.2 Hedonic Pricing

The hedonic price method (HPM) estimates the value of a non-market good by observing behavior in the market for a related good. Specifically, the HPM uses a market good via which the non-market good is implicitly traded.

The starting point for the HPM is the observation that the price of a large number of market goods is a function of a bundle of characteristics. For instance, in the preceding discussion the attractiveness of a site was a function of its attributes. Likewise, the price of a car reflects various attributes, such as fuel efficiency, safety, and reliability. Similarly, the value of a particular house may depend on the number of bedrooms, whether it has a garden, and how close it is to a metro station, but also on the noise level in the neighborhood and local air quality. The HPM uses statistical techniques to isolate the implicit price of each of the characteristics related to non-market goods (e.g. local air quality). Two types of markets are of particular interest in non-market valuation: property markets and labor markets. In what follows, we look by way of example at the housing market.

The value of a unit in a given commodity class, such as a house, is a function of a vector of characteristics. Generally, these characteristics may be denoted by $\boldsymbol{Z}$, so that the price is given by $p = f(Z)$, $\boldsymbol{Z} = (z_1, \dots, z_j, \dots, z_n)$. The implicit price of a characteristic z_j is given by the respective derivative of the price function: $\frac{\partial p}{\partial z_j}$. This implicit price can be regarded as the average willingness to pay in the market in the following way: Consumers will bid an amount $B_j[z_j(.)]$ for an increase in the characteristic, depending on the value to them of that characteristic (e.g. better air quality, less noise, better views). If the market reaches equilibrium, then every consumer will be in a position where the marginal bid, $\frac{\partial B_j}{\partial z_j}$, is equal to the implicit price (that is, marginal cost) of the characteristic.

To be more specific, we specify a linear hedonic price function for housing, which can be written as follows (focusing again on the deterministic part):[65]

$$p = \beta_0 + \sum_{i=1}^{I}\beta_i H_i + \sum_{j=1}^{J}\beta_j N_j + \sum_{k=1}^{K}\beta_k L_k.$$

Recall that p is the sales price of the house. Now, we have split up the vector of characteristics as follows: $\boldsymbol{H}$ represents structural and property characteristics of the house (e.g. bedrooms, lot size), $\boldsymbol{N}$ represents neighborhood characteristics of the house, and $\boldsymbol{L}$ location characteristics such as

[65] For a detailed presentation of possible functional forms for the hedonic price function and welfare analysis, see Taylor (2003).

proximity to environmental amenities. As already noted, the implicit price for any specific characteristic z is the estimated coefficient for the variable, which for the linear specification is given by $\frac{\partial p}{\partial z_j} = \beta_j$.

We now apply this to an example. Suppose a linear hedonic price equation on houses in a given city was estimated using particulate concentrations as one of the location/environmental attributes and it was found that the coefficient on this attribute was 500 €. This shows that, on average, a one-unit increase in particulates (from the current mean level of, say, 30 micrograms per cubic meter) would reduce house prices by 500 €. A 10% reduction in particulates to 27 mg/m3 would increase house prices by 1500 € on average (500 € × 3). The average aggregate value of the reduction of particulates could be approximated by the sum of the increases in house prices in the city.

Recall now that the two types of markets that are of particular interest are property markets and labor markets. With respect to the latter, the HPM has been used to estimate the value of avoiding risk of death or injury. It does this by looking for price differentials between wages in jobs with different exposures to physical risk. Different occupations involve different risk levels; being a firefighter, for example, generally entails greater physical risks than being an office worker. Employers must therefore pay a premium to induce workers to undertake jobs entailing higher risk. This premium provides an estimate of the market value of small changes in injury or mortality risks. While our focus is on environmental sustainability, we already noted that the various methods may also be useful when agreements between firms promise to increase sustainability along other dimensions. If it was deemed appropriate to consider working conditions, including workplace safety, then the hedonic pricing approach could be used.

Given the practical relevance of the hedonic pricing approach, it seems expedient to point out some of the issues associated with its practical application. We note, however, that these issues are not specific to the hedonic pricing approach but apply equally to other measurement approaches that rely on statistical (regression) analysis. These issues include the following, as related to our example of house prices:

- Omitted variable bias: If a variable that significantly affects house prices is omitted from the HP equation, and this variable is correlated with one of the included variables, then the coefficient on this included variable will be biased.
- Multi-collinearity: Some attributes in the hedonic price function may be highly correlated with each other, such as air quality or distance from the ocean.
- Choice of functional form for the hedonic price function: Economic theory does not specify which functional form should be used for the HP equation, yet the choice of functional form will influence the value that implicit prices take.
- Market segmentation: The hedonic price function relates, in theory, to the equilibrium implicit prices for housing attributes in a single market. How big this market is in spatial terms can be difficult to assess.
- Spatial autocorrelation: This refers to the phenomenon whereby certain factors influence house prices for all properties in a neighborhood but are not observable to the researcher. The result is to make the estimates of the hedonic price equation parameters inefficient, and to bias standard errors.

III.3.3 Averting Behavior, Defensive Expenditure, and Cost of Illness

Valuation through averting behavior and defensive expenditure is used in cases in which protection from an increase in an environmental bad – or, more generally, a non-market bad – can be obtained by purchasing a market good. In this case, the price for the market good acts as a surrogate. Methods based on averting behavior are thus based on the idea that individuals and households can insulate themselves from a non-market bad by selecting more costly types of behavior that can be related to buying market goods or using other non-market goods.

Specifically, these financial outlays used to buy market goods are called defensive expenditures. The value of each of these expenditures represents an implicit price for the non-market good or bad in question. For example, households may install double-glazed windows to decrease exposure to traffic noise. Essentially, double-glazing is a market good which, in this case, acts as a substitute for a non-market good (freedom from noise disturbance). Alternatively, an individual may spend additional time indoors to avoid exposure to outdoor air pollution. In this case, the allocation of time to avoiding a non-market bad (i.e. the risk of adverse health impacts) is typically not observable and the substitute item is itself a non-market good (i.e. time that could have been used for other purposes).

The cost of illness (COI) approach is similar to the defensive expenditures method in that it focuses on expenditures on medical services and products made in response to morbidity and other health effects of non-market impacts. The health impacts[66] of air pollution, for example, can be valued by looking at expenditures for drugs to counter health effects. The difficulty with the COI approach is that changes in expenditures (e.g. for the health impacts of air pollution) are often not easy to observe directly.

III.3.4 Dose-Response Methods

In many cases the COI is determined by using an estimate of the lost output attributable to the illness. The lost output approach is related to the COI/defensive expenditure approaches, since it uses observed or estimated market prices as the measure of value. Examples include agricultural prices for changes in agricultural yields, or wage rates for changes in labor supply.

Calculating the value of lost output requires information on the link between the environmental *bad*, e.g. between air pollution levels and health impacts. These types of relationships are often estimated in the form of *dose- or exposure-response* relationships. The translation of an estimated physical response into an economic impact often requires detailed information on what the economic implications of the estimated physical response actually are. With commodities like agricultural products, the translation can be relatively straightforward, since changes in yields have direct economic impacts. With other impacts, such as those related to health effects, the translation might be less obvious.

To further illustrate this approach, let us consider the relative risk of death due to an increase in air pollution (e.g. a change in particulate matter $PM_{2.5}$).[67] A dose-response function can be written as

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[66] See, for instance, OECD (2014) or World Bank (2016).

[67] See Atkinson et al. (2018), for a detailed analysis of health valuation issues.

$$RR = e^{\beta(P_1 - P_0)} \ or \ RR = \left(\frac{P_1}{P_0}\right)^{\beta},$$

where P_1 is the current level of pollution and P_0 is some benchmark level, and the parameter β is a risk factor reflecting the severity of the health risk. The role of the *output*, as denoted by RR, now becomes transparent by the way it is used. Specifically, the proportion of fatalities in a given year due to change in pollution is given by the attributable factor $AF = \frac{RR-1}{RR}$, where the input is obtained from the respective dose-response function. If the average mortality rate in the area under study is M and the total population is POP, the total number of deaths attributable to the change in pollution will be $\Delta H = AF \times M \times POP$. If VSL is an estimate of the value of statistical life,[68] the monetary value of health damages (VHD) associated with the mortality risks due to air pollution will be: $VHD = VSL \times \Delta H$.

There exist various studies that provide an overview of estimates linking notably different emissions or the concentration of substances to the monetary value of associated health damages. We refer to these studies in more detail in Section III.6.2.

III.4 Stated Preference Methods

Stated preference approaches are survey-based and elicit people's intended behavior in constructed or hypothetical markets. In a contingent valuation analysis, such a hypothetical market is described using a questionnaire. In choice modeling, subjects are given various choices, from which preferences are derived. Both methods are described below.

Data validity and reliability are the two major concerns when using stated preference methods.[69] The primary issue is whether the hypothetical nature of the market leads to bias that may prevent meaningful intervention. In what follows, we cannot provide a sufficiently detailed discussion of these concerns. We should note, however, that the hypothetical nature of these approaches does not necessarily pose a relative disadvantage (particularly when there is no obvious alternative). For example, an agreement may envision the introduction of a new product variant that satisfies higher environmental standards. But even if this variant was already available in the market, extracting revealed preferences from current purchases may not provide a complete picture. For instance, the limited availability of this product variant and lack of information about its specific attributes relating to sustainability may allow it to command a much lower price difference than would be the case if the agreement leads to higher market shares. Also, the fact that many or most consumers will have switched to a more environmentally friendly alternative may in itself affect consumers' willingness to pay, as this provides a new anchor for their own assessment.[70]

[68] See, for instance, Viscusi and Aldy (2003) for further details.

[69] We noted that with the growth of broadband penetration and the popularity of online forums, surveys have been increasingly implemented online. Despite some shortcomings (see, e.g. the description and references in OECD (2018)) such surveys have the main advantage of being quick to implement and relatively inexpensive.

[70] This is dealt with in more detail in Inderst and Thomas (2020b).

III.4.1 Contingent Valuation

The most commonly stated preference method is the contingent valuation method (CVM). The CVM is applicable, in principle, to almost all non-market goods and is able to capture all types of benefits from a non-market good or service, including those unrelated to current or future use (i.e. non-use values). Here, the contingent market defines the good itself and its different attributes, the institutional context in which it would be provided, and the way it would be financed. A random sample of people is then directly asked to express or reveal willingness to pay (WTP) for an improvement in environmental quality, or what they are willing to accept to go without this improvement. We next describe the construction and analysis of such data in more detail.

III.4.1.1 Generating Data for Contingent Valuation

The questionnaire naturally contains the description of the good to be valued in order to reveal the most important underlying factors informing respondents' attitudes toward the good. Where applicable, there also needs to be information about the good's quality and terms of availability. Respondents are then asked questions to determine how much they would value the good given the opportunity to obtain it under the specified conditions. In addition, respondents are typically asked to reveal their socio-economic and demographic characteristics. We now turn to some of these components and stages in designing the questionnaire in more detail.[71]

Of particular relevance is the elicitation question.[72] Elicitation formats describe the type of questions that the respondent will answer about obtaining the environmental good or service. These may take various forms, such as:

Open ended: What is the maximum incremental amount that you would be prepared to pay to obtain the new product variant, as just described?

Bidding game: Would you pay an additional 5 € to obtain the new product variant, as just described?

- If "Yes": The interviewer keeps increasing the bid until the respondent answers "No". Then maximum WTP is elicited.
- If "No": The interviewer keeps decreasing the bid until respondent answers "Yes". Then maximum WTP is elicited.

Payment card: Which of the amounts listed below best describes your maximum WTP to obtain the new product variant, as just described? Answers: 0, 0.50 €, 1 €, …, > 200 €.

Single-bounded dichotomous choice: Would you pay an additional 5 € to obtain the new product variant, as just described? (The amount should be varied randomly across the sample.)

Double-bounded dichotomous choice: Would you pay an additional 5 € to obtain the new product variant, as just described? (The amount should be varied randomly across the sample.)

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[71] We thereby focus on what is of specific interest for the analysis of the considered agreements. More generally – e.g. when considering a change in policy – other aspects may be of greater relevance, such as the form of payment (through coercive payments, donations, etc.).

[72] See also Pearce et al. (2006).

- If "Yes": And would you pay 10 €?
- If "No": And would you pay 1 €?

Choosing the right response mode is important to prevent problems that may arise when analyzing the data. In this short overview we cannot cover adequately these as well as other more specific issues. For instance, the CVM may tend to overestimate true WTP when confronted with hypothetical questions (hypothetical market bias). Cheap talk can be used to moderate this hypothetical market bias. WTP estimates can also be sensitive to the amount and nature of information provided to respondents. The hypothetical nature of the posed questions and the possibility of designing the (information) environment are, however, also an advantage. In particular, they allow one to analyze how consumers' preferences depend on the respective context, so that, among other things, changes in context can be taken into account. For instance, while individuals may only have limited awareness of the implications of a particular production process or consumption behavior, this may be quickly changing. Also, individual preferences may depend on the behavior of others, e.g. due to social norms. A considered agreement may lead to considerable changes in the market outcome, and thus also change production and consumption patterns.[73]

III.4.1.2 Analyzing the Data

Analyzing the data from a contingent valuation setting consists first in applying econometric techniques[74] to the survey results to derive mean or median WTP and their most significant determinants. In a second step WTP estimates are aggregated to obtain a population total figure.

For formats like open-ended responses that elicit the WTP directly, the procedure to obtain mean or median WTP estimates is straightforward. Determinants of the WTP can be easily investigated by estimating a bid function that takes individual socioeconomic and demographic characteristics into account. If the WTP is not directly elicited, as for instance in the case of single- or double-bounded dichotomous choice formats, we can obtain estimates for WTP both parametrically and non-parametrically.

We have already introduced a random utility approach above. Again, the utility is modeled as a function of the respective attributes, including prices of the considered products, and, with respect to the individual, various sociodemographic characteristics. The underlying assumption is that the individual prefers one alternative if the respective utility, including some non-observable (stochastic) part, is higher. From this, an individual's WTP can be estimated (for more details, see our discussion of discrete choice experiments further below). Having obtained such estimates for mean or median WTP, among other alternatives, one way to obtain the population total value figure is to multiplicate them by the number of households in the population. Importantly, before implementing the CVM, both the population and time period have to be carefully chosen in accordance with the aggregate values that shall be obtained. Recall also that with respect to the time dimension – which is inherent in the considered attributes of sustainability – future WTP of the same population or a different cohort may have to be adjusted if such a change in preferences is likely.

[73] See Inderst and Thomas (2020b) for further details as well as a conceptual framework.

[74] For instance, the Turnbull estimator represents a particular non-parametric approach.

III.4.2 Choice Modeling (Conjoint Analysis)

While the CVM is the most familiar valuation technique based on stated preferences in the area of environmental economics, there is growing interest in the choice modeling (CM) and conjoint analysis (CA) approaches. In the environmental context, some of this emerging interest in CM has arisen as a response to the problems of the CVM. The CVM would typically be used to uncover the value of the total change in a multi-dimensional good. CM, however, can value changes that are multi-dimensional. Thus, if policy makers require measures of the change in each of the dimensions or attributes of the good, then some variant of CM might be considered. In addition, there may be benefits for the accuracy and reliability of estimation that are attributable directly to the subsequently described choice process, though we will also point to potential drawbacks. In what follows, we will discuss a few variants of these methods. Here, we will start with the most widely used approach: that of choice experiments or discrete choice experiments (DCE).[75] We then briefly discuss other variants.

III.4.2.1 Discrete Choice Experiments

In a DCE, respondents are presented with a series of alternatives, differing in terms of attributes and levels, and asked to choose their most preferred. A baseline alternative, corresponding to the *status quo* or *do-nothing* situation, is usually included in each choice set. The conceptual framework for DCEs assumes that consumers' or respondents' utilities for a good can be decomposed into utilities or well-being derived from the composing characteristics of the good.

DCEs are consistent with utility maximization and demand theory, at least when a status quo option is included in the choice set. If a status quo alternative is not included in the choice set, respondents are effectively forced to choose one of the alternatives presented, which they may not desire at all. If, for some respondents, the most preferred option is the current baseline situation, then any model based on a design in which the baseline is not present will yield inaccurate estimates of consumer welfare

Stages in the construction of a DCE[76]

Attributes: Given a multi-dimensional good that is to be investigated, the first step in constructing a DCE consists in selecting attributes of the specific choices, e.g. the more or less environmentally sustainable goods. Importantly, some form of monetary cost must be included into the list of attributes in order to estimate WTP or WTA. When alternatives represent hypothetical goods or services, this is immediate.

Attribute levels: Having selected a list of attributes, individual attribute levels can be chosen. These should be realistic, feasible, and ideally cover the range of respondents' preferences.

Experimental design: In the next step individual choice profiles are created by constructing combinations of levels and attributes. We hereby distinguish between *complete factorial designs,* which

[75] Generally, DCEs are a multi-attribute stated preference technique initially developed by Louviere and Hensher (1982) and Louviere and Woodworth (1983). DCEs are the only CM approach which satisfies the requirements of welfare theory.

[76] See Atkinson et al. (2018) for further details.

cover all combinations of levels and attributes, and *fractional factorial designs*, which only consist of a subset of combinations of the *complete factorial design*. The richness of the *complete factorial design* allows one to investigate the influence of the full set of attributes on individual choices. Yet given the potential multitude of different combinations, it is often too cumbersome to evaluate.

Choice sets: The final stage consists in generating so-called choice sets, where two or more alternatives (choice profiles) are compared. Choice sets are then presented to the respondents, who are asked to choose their most preferred option.

Given the data that are generated by the experiment, welfare estimates but also the influence of individual attributes can be estimated using econometric methods, such as logit or latent class models.

Formalization of the analysis

The conceptual foundation of the DCE is once again the random utility model described above. As mentioned, the indirect utility function for each respondent i can be decomposed into two parts: a deterministic element (V), on which the subsequent equation focuses, and a stochastic element representing unobservable influences on individual choice. Again, we use a linear specification, so that V is a linear function of the attributes (X) of the j different options (profiles) in the choice set (C). The (deterministic part of the) utility of profile j for respondent i is then defined as:

$$U_{ij} = V(X_{ij}) = \sum_{k=1}^{K} \beta_k X_{jk} + \beta_p p_j \, ,$$

where β_k is the preference parameter associated with attribute k, X_{jk} is attribute k in profile j, p is cost, and β_p is the parameter on the profile's cost. The probability that any particular respondent prefers option g in the choice set to any alternative option h can be expressed as the probability that the utility associated with option g exceeds that associated with all other options. Hence, the β values show the effect of attribute changes on utility, but for applied analysis, money-metric measures of WTP are required. Thus, for a marginal change in an attribute, this WTP value or *implicit price* is typically given for attribute X_k by:

$$WTP_k = \frac{\beta_k}{\beta_p} .$$

Often, in environmental economics changes in policies are considered for which an evaluation is carried out based on the extracted preferences. Of course, this can also be applied to agreements that may change the provision of products in various ways, e.g. both in terms of production and the nature of the supplied product itself. We could then consider this as, say, a choice between two alternatives, A and B, and the respective attribute vectors $(X_{1A}, X_{2A}, \dots, X_{KA})$ and $(X_{1B}, X_{2B}, \dots, X_{KB})$. The average WTP ($AWTP$) of the respective suite of changes – i.e. when A is replaced by B – is then formulated as

$$AWTP = -\frac{1}{\beta_p}(V_A - V_B) \, ,$$

where

$$V_A = \sum_{k=1}^{K} \beta_k X_{kA}, \quad V_B = \sum_{k=1}^{K} \beta_k X_{kB}.$$

Example

We next consider a purely illustrative example. In this example, consumers face a choice in which they must decide between different tuna fish products. As illustrated in Table 2, they have the possibility of choosing between tuna A, B, or C, but can also decide for no fish at all. Tuna A, B, and C differ in terms of prices as well as several other criteria that relate, among other things, to the sustainability of fish production, such as storage conditions and information about whether the fish has been caught in fishing grounds that are certified sustainable.

Table 2: Example DCE

Given the options described below, which fish would you choose?				
	Tuna A	**Tuna B**	**Tuna C**	**No fish**
Fishing technique	purse seine	longline	purse seine	
Storage conditions	frozen	frozen	fresh	
Production location	overseas	overseas	domestic	
From certified fishing grounds	no	yes	yes	
Price per 100g	3 €	5 €	6 €	
Preferred option				

Note: Illustrative example.

III.4.2.2 Other Choice Modeling Methods

Contingent Ranking

In contingent ranking experiments, respondents are required to rank a set of options that are characterized by differences with respect to numerous attributes. Taking our previous example, this could look as follows:

Table 3: Example contingent ranking

Please rank the alternative options below according to your preferences, assigning 1 to the most preferred, 2 to the second most preferred, 3 to the third most preferred, and 4 to the least preferred.				
	Tuna A	**Tuna B**	**Tuna C**	**No fish**
Fishing technique	purse seine	longline	purse seine	
Storage conditions	frozen	frozen	fresh	
Production location	overseas	overseas	domestic	
From certified fishing grounds	no	yes	yes	
Price per 100g	3 €	5 €	6 €	
Preferred option				

Note: Illustrative example.

Contingent Rating

In a contingent rating exercise, respondents are presented with a number of scenarios and asked to rate them individually on a semantic or numeric scale. In our previously introduced example, this could look as follows:

Table 4: Example contingent rating

On the scale below, please rate your preferences for this option of tuna fish.									
				Tuna A					
Fishing technique				purse seine					
Storage conditions				frozen					
Production location				overseas					
From certified fishing grounds				no					
Price per 100g				3 €					
1	2	3	4	5	6	7	8	9	10
Very low preference									Very high preference

Note: Illustrative example.

Paired Comparisons

In a paired comparison exercise, respondents are asked to choose their preferred alternative out of a set of two choices and to indicate the strength of their preference in a numeric or semantic scale. In our previous example, this could look as follows:

Table 5: Example paired comparison

Which tuna would you prefer given the two options described below?									
			Tuna A			**Tuna B**			
Fishing technique			purse seine			longline			
Storage conditions			frozen			frozen			
Production location			overseas			overseas			
From certified fishing grounds			no			yes			
Price per 100g			3 €			5 €			
1	2	3	4	5	6	7	8	9	10
Strongly prefer Tuna A									Strongly prefer Tuna B

Note: Illustrative example.

III.4.2.3 Advantages and Disadvantages of Choice Modeling

DCEs are particularly suited for situations in which changes are multi-dimensional and trade-offs between them are of particular interest. This is because DCEs can separately identify the value of individual attributes of a good or program, typically supplied in combination with one another. Thus, in this regard DCEs could be more informative than CVMs and also more appropriate for generalizations or adaptations.

CM generally avoids an explicit elicitation of respondents' WTP and instead relies on ratings, rankings, or choices among a series of alternative packages of characteristics from which WTP can be indirectly inferred. In this respect, this method may also be more akin to real choice situations, with which subjects are more familiar. One disadvantage of CM approaches lies in the cognitive difficulty associated with multiple complex choices or rankings between bundles with many attributes and levels.[77] Both experimental economists and psychologists have found ample evidence that there is a limit to how much information respondents can meaningfully handle while making a decision. One common finding is that the choice complexity or depth of a ranking task can lead to greater random errors or at least imprecision in responses. This applies in particular when the underlying choices are already highly complex, e.g. when they consist of a sequence of elements or changes that the considered agreement may bring about.[78] With respect to all of the introduced stated preference techniques, it must also be remembered that the results and welfare estimates may be highly sensitive to study design,[79] which is why an analysis of robustness – and conducting and analyzing different study designs – is preferable.

III.5 Subjective Well-Being Valuation

Subjective well-being valuation (SWB) has garnered increasing interest over the past couple of decades. It refers to self-reported measures of personal well-being obtained through surveys. The SWB valuation encompasses three key dimensions.[80]

Evaluative subjective well-being (or life satisfaction): This dimension is a self-evaluation of one's life according to some positive criterion. The degree of satisfaction can be expressed in numeric scales, e.g. from 1 to 10.

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[77] We refer to OECD (2018, ch. 5.5) for a brief account of some specific issues related to the implementation of choice experiments and how notably recent developments have made progress in overcoming some of the respective challenges (such as, for instance, respondents' selective attention or their application of different decision rules).

[78] In this context, the fruitful interaction with behavioral economics should also be noted (see, for instance, OECD (2018, ch. 4.6)). There, the focus is typically on dealing with perceived *anomalies* in stated preferences. Insights from experiments conducted in this field of economics have been used to interpret data better also from contingent valuation and choice-based experiments. As noted elsewhere in this report, notably the context-specificity of respondents' answers and choices should, however, not be interpreted as a disadvantage of these approaches. Instead, it draws attention to the fact that particularly with non-use values or when there is little a priori experience with evaluating new attributes, such context, including the availability of information, affects (stated) preferences (see Inderst and Thomas 2020b on a more detailed discussion). Still, simple errors in decision-making should be avoided, as well as the confounding influence of uncontrolled-for behavioral biases (see, e.g., the survey in the study on consumer decision-making by Chater et al. (2010)).

[79] We should note that there is some tension between the normative (welfare) context as established in the preceding section and various recent insights notably from behavioral economics (and thereby from psychology). While the standard paradigm of welfare economics posits objective preferences, which are typically also supposed to be well known to an individual, other views lean on the notion that humans act according to such *true preferences*, the existence of which does not depend on the act of choice and the underlying construction of judgment (e.g. Bernheim (2016)). While this tension does not affect the main insights of Section II, most notably the consideration of non-internalized externalities, it again emphasis the importance of carefully considering and designing the appropriate context or *decision frame* (Bernheim 2016, p. 36) when extracting individual preferences.

[80] See Atkinson et al. (2018), for further details. A detailed account of the significant increase in research in subjective wellbeing evaluation over the last decade can also be found in OECD (2018, ch. 7).

Eudaimonic subjective well-being: This dimension refers to the process of achieving a flourishing and worthwhile life where one's true potential is realized. A self-reported approach could answer to the question, "Does your life have meaning and purpose?"

Momentary subjective well-being (or affect): This dimension measures feelings, affect, or mood at a particular point in time and is affected positively or negatively by recent events.

In empirical work including non-market goods, an SWB linear function can be estimated as:

$$SWB_i = a + \beta_1 M_i + \beta_2 Q_i + \beta_3 X_i + \varepsilon_i,$$

where Q is the non-market good (e.g. air quality), M is income, and X represents other determinants of SWB. The value associated with the welfare change in the improvement of the environmental good (air quality) from state 0 to state 1 can be calculated as the Hicksian compensated surplus (CS), whereby

$$CS = \frac{\beta_Q}{\beta_M} \ , or\ CS = M^0 - \exp\left[\ln M^0 - \frac{\beta_Q}{\beta_M}\right].$$

The SWB method has been used frequently in the valuation of environmental changes such as changes in air quality, noise, climate change, and droughts (e.g. Welsch and Kühling 2009).

III.6 Additional Measurement Approaches in Practice

All of the previously introduced methods have been widely adopted in empirical research and policy studies. They are thus inherently practical. In applied policy design, the policy analyst may, however, face serious constraints with respect to time or resources. In particular, the considered issues maybe relatively broad in scope, with the considered policy – or, in our case, agreement between firms – affecting the environment in various ways. This may prevent the implementation of an original study using appropriate methods (from those described above) in order to value changes in environmental goods or services. In such a case, the analyst may use data obtained in past studies, in order to estimate values relevant for the changes brought about by the considered agreement. This may entail the use of tabulated values drawn from existing databases for various categories of environmental goods and services.

These methods are important in facilitating the valuation procedure but need to be executed with care because the transfers and the use of *ready-made values* might affect the accuracy of the results. Below we discuss such options in more detail.

III.6.1 Benefit Transfer

Benefit transfer (BT) can be defined as the transfer of existing estimates of non-market values to a new study, different from the study for which the values were originally estimated. Thus, BT uses values of a good or service estimated in one site (the *study site*) as a proxy for values of the (approximately) same good or service in another site (the *policy site*). This is the type of BT most commonly used in environmental valuation.

BT methods can be divided into four categories: unit (naïve) BT, adjusted unit BT, value function transfer, and meta-analytic function transfer.

Unit (Naïve) BT

Unit BT involves estimating the value of an environmental good or service (EGS) at a policy site by multiplying a mean unit value estimated at a study site by the quantity of that EGS at the policy site. Unit values are generally either expressed as values per household or as values per unit of area. In the former case, the aggregation of values occurs over the population that hold values for the EGS in question. In the latter case, the aggregation of values is over the relevant area.

The procedure is to *borrow* an estimate of the WTP in context S (the study site) and apply it to context P (the policy site). The estimate is usually left unadjusted: $WTP_S = WTP_P$. A variety of unit values may be transferred, the most typical being mean or median measures. Mean values allow simple transformation to aggregate benefit estimates: for instance, multiply mean (average) WTP by the relevant affected population to calculate aggregate benefits.

The main problem with naïve BT is that the conditions and site characteristics (i.e. socioeconomic and physical characteristics) that determine the WTP at the study site are unlikely to be met in a satisfactory way at the policy site.

Adjusted BT

If the study and the policy site have reasonable similarities, a widely used formula for an adjustment that may improve the accuracy of the transfer is:

$$WTP_P = WTP_S \, (Y_P/Y_S)^{\eta},$$

where Y is income per capita, WTP is willingness to pay, and η is the income elasticity of WTP. This latter term is an estimate of how the WTP for the (nonmarket) EGS in question varies with changes in income. If η is assumed to be equal to one, then the ratio of WTP at sites S and P is equivalent to the ratio of per capita incomes at the two sites (i.e. $\frac{WTP_P}{WTS_S} = \frac{Y_P}{Y_S}$).

Value Function Transfer

The value function transfer method implies that the benefit or value function estimated for S is applied to P. Thus, if it is known that WTP at the study site is a function of a range of physical features of the site, its use, and the socio-economic (and demographic) characteristics of the population at the site, then this information itself can be used as part of the transfer.

Let $WTP_S = f(A, B, C, Y)$ where A, B, C are additional and significant factors affecting WTP (in addition to Y) at site S. For instance, a linear functional form may be chosen, so that WTP_P can be estimated using the coefficients from this equation in combination with the values of A, B, C, Y at site P. Taken together:

$$WTP_S = a_0 + a_1 A + a_2 B + a_3 C + a_4 Y$$

$$WTP_P = a_0 + a_1 A_P + a_2 B_P + a_3 C_P + a_4 Y_P$$

Meta-Analysis Methods

Meta regression analysis is the statistical summarization of relationships between benefit measures and quantifiable characteristics of studies. The data for meta-analysis typically consists of summary statistics from study site reports and includes quantified characteristics of the user population, the study site's environmental resources, and the valuation methodology used, in addition to results for key coefficients.

Meta-analysis draws information from a large number of studies, and it is advisable to include a sufficient number of original studies so that statistical inferences can be made, and relationships modeled. Furthermore, the studies should be similar enough in content and context that they can be combined and statistically analyzed.

III.6.2 Transferring Results from Extant Integrative Studies and Databases

BT aims to transfer the results from a specific study to a different application. As noted, this requires sufficient comparability – such that, for instance, the adjustment could consist primarily in the scaling (of WTP) in terms of income. Often, such a comparable alternative study may not be available. In addition, unless the targeted benefit is confined to a single emission, calculations may involve different types of impact, e.g. a reduction of various harmful emissions. In such cases, researchers may draw on various integrative studies and databases in which the respective impact and social costs have already been compiled. In what follows, we briefly describe one example. This selection should not be indicative, however. The usefulness of a particular study will depend obviously on the specific application, which means that some studies may be more relevant in a specific case. Below, we thus we provide an account of additional studies and databases.

Specifically, we now consider the 2017 Environmental Prices Handbook of CE Delft (CE Delft 2018). The CE Delft Handbook provides environmental prices, which are constructed prices for the social cost of pollution, expressed in €/kg of pollutant. These prices thus indicate the loss of economic welfare that occurs when one additional kilogram of the pollutant finds its way into the environment.[81] Environmental prices provide average values in the Netherlands for emissions from an average emission source at an average emission site in the year 2015 (CE Delft 2018, p. 11). The handbook is in its 5th edition and is commissioned by the Dutch Ministry of Infrastructure and Environment. Table 6, taken from this report, provides an example for the case of environmental prices for atmospheric emissions. Note that the authors estimated an upper and a lower value for each emission so as to take into account the uncertainty that comes with these estimations.[82]

The environmental prices are average estimations for the Netherlands for an average emission area and an average emission source in 2015. The valuations are mainly based on damage cost estimations. However, abatement costs in the form of efficient prices were included for climate change related estimates (CE Delft 2018, p.14). Further, the authors differentiate between three relevant points for their valuation. There is the pollutant level, the midpoint level, and the endpoint level. The pollutant level is the type of emission that stems from an intervention and is damaging the

[81] The study thus considers various *endpoint*s, such as human health or the quality of ecosystems, for which people's willingness to pay for improvement in the form of pollution abatement are examined (based on various existing studies). These values are then calculated back to the described value for reducing the emissions themselves.

[82] CE Delft (2018) recommends the expression of estimations in ranges based on the upper and lower prices (e.g. when performing a social cost-benefit analysis). CE Delft note also, as a further example, that if a company wants to calculate specific values – for instance, for a corporate social responsibility report – they may use the central (*best estimate*) prices.

environment. The midpoint level contains the environmental categories that are influenced by the emissions such as ozone depletion, acidification, or climate change. The endpoint level captures the damages that influence human welfare, such as human health or resource availability (CE Delft 2018, p.4). For example, carbon dioxide (CO_2) is the emission on the pollutant level. It influences climate change on the midpoint level, and then impacts human health, ecosystems, and resource availability on the endpoint level. Table 6 provides an extraction for the CE Delft Handbook.

Table 6: CE Delft Handbook of environmental prices for atmospheric emissions

Substance		Lower	Central	Upper
Carbon dioxide*	CO_2	0.014	0.057	0.057
Chlorofluorocarbons*	CFC_{11}	99.6	313	336
Ultra-fine particulate matter	$PM_{2,5}$	56.8	79.5	122
Particulate matter	PM_{10}	31.8	44.6	69.1
Nitrogen oxides	NO_x	24.1	34.7	53.7
Sulphur dioxide	SO_2	17.7	24.9	38.7
Ammonia	NH_3	19.7	30.5	48.8
Volatile organic compounds	NMVOC	1.6	2.1	3.15
Carbon monoxide	CO	0.0736	0.0958	0.152
Methane*	CH_4	0.448	1.75	1.77

Values estimated for 2015, in €/kg
* The value of GHG emissions includes VAT and increases 3.5% per annum relative to the 2015 values, as detailed in Section 6.3

Source: CE Delft (2018, p. 5).

We next provide a short, simplified example for the potential use of such data. In this example, we consider emissions from heavy-duty trucks and buses, which cause a quarter of CO_2 emissions released in the EU's road transport sector and around 6 percent of total emissions per year. To address this large impact, the first EU-wide standards for heavy-duty vehicle emissions were adopted in 2019. It set targets for a reduction in average emissions by 2025 and 2030.[83] Let us consider an agreement by which the respective manufacturers would implement a 20% reduction in greenhouse gases (GHG), namely CO_2, methane, and dinitrogen monoxide. We are only concerned with the associated benefits, for which we take the respective estimated environmental prices from the CE Delft Environmental Pricing Tool,[84] as given in Table 7.

Table 7: Environmental prices for average GHG emissions in the Netherlands, used for the calculated example

Emission		Lower	Upper
Carbon dioxide	CO_2	0.0142	0.0566
Methane	CH_4	0.448	1.76
Dinitrogen monoxide	N_2O	3.75	15

Values estimated for 2015, in €/kg

[83] See Regulation (EU) 2019/1242. As an alternative to the considered *bottom-up* approach, given such targets one may also apply the subsequently discussed framework of a cost-effectiveness analysis. There, the benefits of a proposed agreement are measured in terms of saved abatement costs (such as, in the present case, an imposed restriction to traffic).

[84] The CE Delft Environmental Pricing Tool is available at https://www.ce.nl/milieuprijzen.

Source: CE Delft Environmental Pricing Tool, available at https://www.ce.nl/milieuprijzen.

For the sake of simplicity, let us now consider the respective reductions in emissions and the corresponding benefits for a single year. The respective figures for a 20% reduction in Table 8 are not meant to reflect a particular real-world scenario (though they are clearly informed by the magnitude of emissions). The obtained social benefit is estimated by multiplying the environmental prices from Table 7 with the respective reductions.

Table 8: Estimated benefits of a 20% reduction in GHG emissions for the calculated illustrative example

Reduced emission		Reduced amount (t)	Central measure	Range Lower	Range Upper
Carbon dioxide	CO_2	43 mio.	2,434 mio.	611 mio.	2,434 mio.
Methane	CH_4	33,000	58 mio.	15 mio.	58 mio.
Dinitrogen monoxide	N_2O	550,000	8,250 mio.	2,063 mio.	8,250 mio.
Values estimated for 2015, in €					

Note: Authors' calculations.

The example also allows us to draw attention to certain caveats in using such figures. Estimates in the CE Delft Handbook are based on local circumstances such as prevalent pollution levels and population density. It cannot be assumed that the average circumstances of the Netherlands are fully transferable to other EU states. Also, price estimates depend on the emission source. Finally, one should always take into account the great uncertainty involved in the respective estimates and, by extension, in the derived benefit calculations.[85]

We conclude, as discussed above, with a brief account of a selection of alternative databases and reports that provide information on environmental prices. Some of these reports are more comprehensive, such as the DE Delft Handbook, while others are targeted to particular sectors or types of emissions (such as noise). This selection primarily serves the purpose of illustration.

- The Economics of Ecosystem and Biodiversity (TEEB) database reports ecosystem service values, taking into account the cost of biodiversity loss and environmental degradation (McVittie and Hussain 2013).
- The Environmental Valuation Reference Inventory (EVRI) collects summaries of environmental and health valuation studies. The reported summaries include monetary value estimations, as well as the specific context of the respective studies (ECCC 1997).
- The Externalities of Energy (ExternE) project offers tools and data that allow one to evaluate the external cost of economic activities, focusing on energy-related activities and industrial processes (EC 2006).
- The PESETA IV study examines the effects of climate change on Europe for various climate change impact sectors and quantifies their economic impacts. It also examines policy measures for counteracting these climate change effects (Feyen et al. 2020).

[85] In fact, the authors themselves are cautious when it comes to applicability, noting that, where possible, a specific study tailored to the individual scenario should be performed in order to reflect the environmental costs and benefits (CE Delft 2018, p.12).

- The Economic Consequences of Outdoor Pollution report estimates the costs of outdoor air pollution and focuses on quantifying its impacts on human health and agriculture (OECD 2016).
- The study on Costs of Health Damage from Atmospheric Emissions of Toxic Metals quantifies the damage costs of heavy metals. It also calculates the share of damage costs attributable to waste incineration and coal-based generation in Europe (Nedellec and Rabl 2016).
- The report on Benefit Measures for Noise Abatement (Calculations for Road and Rail Traffic Noise) contains estimations for monetary abatement values for road and rail traffic noise (Andersson et al. 2013).
- The Assessment of Biodiversity Losses Project examines biodiversity losses due to energy production. It quantifies the external costs of potentially disappeared fractions due to land use changes as well as emissions of acidifying substances for 32 European countries (Ott et al. 2006).
- The report on Costs and Benefits of Nitrogen in the Environment estimates costs and benefits of reactive nitrogen emissions and thereby quantifies their damages and social costs (Brink and Grinsven 2011).

III.7 Valuation Through Policy Objectives

When a regulatory framework with market-based instruments is adopted to achieve an environmental policy target, this creates a price or shadow value (cost) for the environmental good (e.g. air quality), as a market now exists where no market existed before. If the target is set at the welfare-maximizing level, the price will fully internalize the environmental externality. In practice, however, this is unlikely; normally, the shadow price only results in partial internalization. In the following we regard the adoption of environmental targets as an expression of societal preferences, regardless of whether these preferences are a reflection of compromise between various interests or the weighing of costs and benefits for a country's own citizens with those of individuals outside a given jurisdiction.[86]

III.7.1 Climate Change: Pricing Changes in CO_2 Emissions

The aggregate cost of climate change in an economy is typically measured by a damage function as a proportion of GDP.[87] The welfare cost of emitting one ton of carbon in the atmosphere is then expressed by what is typically referred to as the social cost of carbon (SCC). The SCC is defined as the present value of the cost in social welfare caused by the emission of a marginal unit of carbon into the atmosphere. It is mainly estimated by integrated assessment models (IAM) as a global cost.

[86] Of course, if the failure to fully internalize externalities and a (shadow) price below the perceived total cost of the respective emission reflects a failure of the political process, one could not presume a full reflection of societal preferences. In this report, we do not need to take a stance on such matters. Instead, in what follows we take as a given the existence of environmental targets and discuss how these can be used for the envisaged assessment of an agreement between firms.

[87] See, for instance, Nordhaus and Sztorc (2013).

A prominent example of such a model is DICE,[88] where social welfare at each point in time is defined as the utility of per capita consumption multiplied by population.[89]

Estimates from SCC are highly sensitive to the chosen parameters, such as the time frame, the social discount rate, the equilibrium climate sensitivity, or the existence of stochastic tipping. Table 9 provides a picture of the respective sensitivity of each model.

Table 9: SCC under different assumptions

Scenarios	2015	2020	2025	2030	2050
Baseline	31.2	37.3	44.0	51.6	102.5
Optimal	30.7	36.7	43.5	51.2	103.6
2.5°C	184.4	229.1	284.1	351.0	1,006.2
Stern Review discounting	197.4	266.5	324.6	376.2	629.2
Alternative discount rates					
2.5 %	128.5	140.0	152.0	164.6	235.7
3 %	79.1	87.3	95.9	104.9	156.6
4 %	36.3	40.9	45.8	51.1	81.7
5 %	19.7	22.6	25.7	29.1	49.2

US $ per ton CO_2, 2010 international

Notes: The years at the top refer to the date at which emissions take place. Therefore, 30.0 US $ is the marginal cost of emissions in 2015 in terms of consumption in 2015. Baseline means calculations along the reference path with the current policy. Optimal means calculations are done along the optimized path of emissions. When there is a temperature ceiling damages are included along with the implicit assumption that damages are infinite if the ceiling is exceeded. Source: Adapted from Nordhaus (2017, p.1520). See also Nordhaus (2018, p.352).

Given such wide divergence in estimated values for SCC, the application of a particular estimate will have far-reaching implications for the overall assessment of a respective agreement between firms to reduce carbon emissions. In some circumstances and in some jurisdictions, there may exist a particular SCC value that is commonly used for impact assessments conducted by regulators or other bodies. This may also reflect preferences. When this is not the case, the choice of SCC needs an identifiable motivation. One important benchmark for the chosen SCC is the observed (shadow) price that may arise from an implemented target.

Setting targets in relation to climate change involves determining the maximum amount of allowable CO_2 emission within a period of time and then establishing a cap-and-trade system for emission trading. The system will result in an equilibrium price for CO_2 emissions, which can be used for

[88] The DICE (Dynamic Integrated Model of Climate and the Economy) was first developed around 1990 and has gone through several extensions and revisions. A detailed description can be found in Nordhaus and Sztorc (2013). The DICE model is a globally aggregated model that views the economics of climate change from the perspective of neoclassical economic growth theory. In this approach, economies make investments in capital and in emissions reductions, reducing consumption today, in order to lower climate damages and increase consumption in the future. The special feature of the model is the inclusion of all major elements in a highly aggregated fashion. The model contains about 25 dynamic equations and identities, including those for global output, CO2 emissions and concentrations, global mean temperature, and damages. It can be run in either an Excel version or in the preferred GAMS version.

[89] More sophisticated functions like Epstein-Zvi preferences are used in some lower dimensional analytical IAMs.

carbon pricing. As we discussed above, the fixed supply of allowances may be thought of representing societal preferences regarding climate change under certain conditions. Against this backdrop, we discuss the respective EU system.

The EU Emissions Trading System (EU ETS)[90] is based on the cap-and-trade principle. A cap is set on the total amount of certain GHGs that can be emitted by installations covered by the system in a given year. Currently, the system limits emissions from more than 11,000 heavy energy-intensive installations (power stations and industrial plants) and airlines operating between these countries. Within the cap, companies receive or buy emission allowances (EUA), which they can trade with one another as needed.[91] Currently, the EUA price ranges between 20 and 30 € per ton of emitted CO_2. However, the price was consistently below 10 € in the years leading up to 2018. A comparison of the SCC from the preceding tables shows that estimated costs typically far exceed such prices. These prices may not reflect the objectives to which European governments consented under the Paris Agreement, as they are below those proposed by the World Bank in its *Guidance note on shadow price of carbon in economic analysis* (World Bank 2017). Based on an extensive review of the estimates, the High-Level Commission on Carbon Prices, led by Joseph Stiglitz and Nicholas Stern, concluded that a range of 40–80 US $ per ton of CO_2e in 2020, rising to 50–100 US $ per ton of CO_2e by 2030, would be consistent with achieving the core objective of the Paris Agreement, namely, keeping average global warming to below 2 degrees (Stiglitz and Stern 2017).

III.7.2 Cost-Effectiveness Analysis

We would now like to extend the discussion beyond the case of CO_2 emissions. We begin with the instrument of cost-effectiveness analysis (CEA). Such an analysis compares programs or policies in order to select the one that attains a given target at the minimum cost or achieves the maximum of an objective for a given cost level. In general, CEA is suitable for comparing programs or policies for which it is difficult to determine the value of the output but where it is relatively easier to determine the cost of providing it. It is also suitable for cases in which a fixed target is set and the lowest cost way of attaining the target is required. The latter observation now ties into our preceding discussion as follows.

In the case of CO_2 emissions, we noted that next to starting from an SCC estimate, which is highly sensitive to the choice of model and parameter, we can make use of a given political target and the (shadow) price that results from it. With respect to an agreement between firms, the benefits of a CO_2 reduction may then be assessed in terms of this price. While the EU-ETS for CO_2 emissions is very specific, we note that environmental targets have been set in other areas as well, such as the improvement of air quality. Then, within the conceptual framework of the described CEA, we can apply targets for measuring the social benefits of a particular agreement.

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[90] For information about the EU ETS see https://ec.europa.eu/clima/policies/ets_en.

[91] They can also buy limited amounts of international credits from emission-saving projects around the world. With these restrictions, the program covers around 40% of the EU's GHG emissions. The cap is reduced over time so that total emissions fall. The program is currently in its third phase. The legislative framework of the EU ETS for its next trading period (2021-2040) was revised in early 2018 to enable it to achieve the EU's 2030 emission reduction targets as part of the EU's contribution to the Paris Agreement.

For instance, to stay within a given target it may be presumed that, under the most realistic scenario, restrictions to traffic are unavoidable or very likely. Consider an agreement that would credibly lead to a reduction in the respective emissions and the respective concentration of particulate matters or gases that would otherwise not materialize. The respective agreement may lead to the adoption of a common technology in some type of vehicles, which allows for such reduction even without further restrictions (albeit with higher costs and higher prices for consumers). The benefits of this agreement can then be assessed by estimating the saved abatement costs arising from an otherwise necessary restriction to traffic.[92] In the spirit of the CEA approach, we could then compare the different ways of achieving the target, including the estimated cost of price increases in terms of consumer surplus losses.[93]

We should note, however, that such an approach may not be considered in isolation when the impact of an agreement can be estimated more directly via the use of social prices for the decrease in emissions. If both approaches are possible, they can complement each other and jointly help to reduce measurement errors and assess the validity and robustness of conclusions.

Example of abatement cost calculations from the UK

The sensitivity of the SCC estimates led the UK to replace the use of SCC for policy purposes with a target-consistent approach, based on estimates of the abatement costs that will need to be incurred to meet specific emissions reduction targets (DECC 2009). This approach supports UK policy alignment with emission reduction targets. Together with other analyses from government departments, the authors of the commissioned report base their environmental price estimations on MAC (marginal abatement cost) curves, which represent the technical abatement potential of an emission reduction technology.[94] For this, the authors included a broad range of MAC estimates from different models. Using such an abatement cost method, two prices have been calculated in the UK: one for the sectors covered by the EU-ETS, which is the traded price of carbon, and one for the sectors not covered by the EU-ETS, which is the non-traded price of carbon.[95]

[92] Such restriction to traffic may require longer or less convenient travel, or it may result in loss of commerce in some areas. Of course, when such losses are compensated elsewhere, e.g. through a displacement of economic activity, an integrated approach needs to be taken.

[93] A common way of comparison is also by calculating cost-effectiveness ratios. For instance, suppose that the reduction in emissions will result in a reduction of medical incidents (e.g. respiratory problems) of A per year, while the cost in terms consumer surplus due to price increase will be $c(CS)$ and the cost in terms of traffic restrictions will be $c(TR)$. Then, the corresponding cost-effectiveness ratios will be $c(CS)/A$ and $c(TR)/A$ and the agreement will be more cost-effective than traffic restrictions if $c(CS)/A > c(TR)/A$.

[94] A part of the MAC curve can even be negative, because sometimes abatement technologies have the potential to generate net savings.

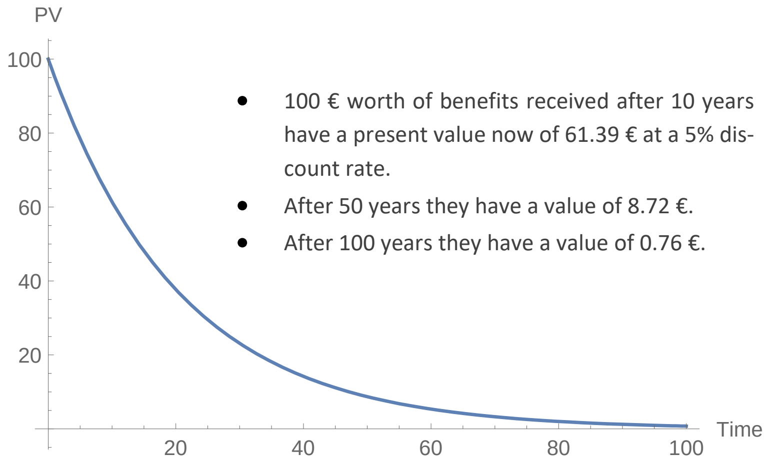
[95] In the long run, the authors of the report assume that the two prices will converge to a global price, because the carbon market is likely to become more globally integrated from 2030 onwards.

III.8 Time Preferences

Martin Weitzman's seminal paper *Gamma Discounting* (2001) argues that discounting is one of the "most critical problems in all economics." Discounting is indeed of particular relevance for the correct assessment of welfare costs and benefits for the cases examined here, since the welfare loss of consumers due to the price increase and the overall welfare gain for society due to the improved environmental conditions will have a time dimension. Moreover, gains and losses may arise at quite different points in time. For a meaningful comparison, these gains and losses need to be expressed in terms of their present value.

Discounting is the process of assigning a lower weight to a unit of consumption's future benefits and costs. The further into the future the benefit or cost occurs, the lower the weight attached to it. Let the weight that is attached to a gain or loss in any future year, t, be w_t . Discounting implies that $w_t < 1$. Moreover, discounting implies that the weight, say, 50 years from now should be lower than the weight 40 years from now. The discounting formula, assuming a constant declining rate of weights over time is $w_t = \frac{1}{(1+r)^t}$, or in continuous time, $w(t) = e^{-rt}$, where w_t (respectively, $w(t)$) is called the discount factor, while r is called the discount rate. For instance, with a five percent discount rate, $r = 0.05$, we have after 50 years a weight of only $w_t = 0.0872$. In practical terms, this would mean that a gain or loss 50 years in the future would be valued at only 8.7% of its value now. Keeping to the 5% discount rate, environmental damage 100 years from now would be valued at 0.0076 of the value that would be assigned to it if it occurred today. This arithmetic illustrates the so-called tyranny of discounting. The impact of discounting is shown graphically in Figure 10, which illustrates the value of a 100 € benefit at different points in time (years T) with a 5% discount rate.

Figure 10: Illustration of the impact of discounting



Note: Illustrative example.

Consequently, if the rate is relatively high, benefits accruing in the distant future – for example, benefits from preventing serious climate change – will have a very small present value now. This makes it difficult to use cost-benefit analysis (CBA) to justify projects that aim to prevent the impacts of climate change in the distant future. By contrast, zero discounting would imply that $r = 0$ and $w_t = 1$ for all t. This implies that the present and future have the same weight, and that intertemporal consumption values are always equal.

We will now apply the concept of discounting in our context. The main question is the choice of the specific (potentially time-varying) discount rate. In CBA, the applied social discount rate (SDR) is the rate of decrease in the social value of consumption over time. Two approaches are usually considered when determining SDRs: The first is the social time preference (STP) approach. The STP is the rate of fall in the social value of consumption. It is also known as the consumption rate of interest (CRI). The second is the social opportunity cost (SOC) approach. The SOC is usually identified with the real rate of return earned on a project in the private sector. In CBA, especially in environmental CBA, we concentrate on the CRI.

III.8.1 The Social Discount Rate: The Ramsey Formula

The SDR can be defined, by an equilibrium condition, as the discount rate at which the marginal utility of a monetary amount x € now is equal to the present value of 1 € received at future time t. Following Arrow et al. (2014), consider a social planner who is indifferent to the choice between 1 € received at time t and x € received today. Indifference implies that the marginal utility of x € today equals the marginal utility of 1 € at time t, or

$$x \cdot U'\big(c(0)\big) = e^{-\rho t} U'\big(c(t)\big) \cdot 1 \Rightarrow x = \frac{e^{-\rho t} U'\big(c(t)\big)}{U'\big(c(0)\big)} = 1 \cdot e^{-r_t t},$$

where ρ is the utility discount rate.[96] Note that the function $U()$ denotes here the respective utility function (of some representative individual). For the second step, r_t denotes the annual consumption discount rate between periods 0 and t, i.e. the SDR. Taking the natural logarithm, we obtain

$$r_t = \rho - \frac{1}{t} \ln \frac{U'\big(c(t)\big)}{U'\big(c(0)\big)} \; or \; r_t = \rho - \frac{d}{dt} \ln U'\big(c(t)\big).$$

We now illustrate this with a specific functional form. Assuming a utility function with constant elasticity of marginal utility,

$$U\big(c(t)\big) = \frac{c^{1-\eta}}{1-\eta}, \eta \geq 1,$$

where η is both the coefficient of relative risk aversion and the elasticity of marginal utility with respect to consumption, we obtain from the preceding transformations:

$$r_t = \rho + \eta g(t),$$

where $g(t) = \frac{dc(t)/dt}{c(t)}$ is the growth rate of consumption. This is also known as the Ramsey formula for the SDR. It has two key components. The first component is the utility discount rate, ρ, expressing the presumption that one values future utilities less for reasons of impatience or hazard. The second component is the wealth effect, ηg. Thereby, the weight a representative individual or the society places on the future depends on what state one will find oneself in the future (or future

[96] Following Gollier (2007) and considering a marginal investment in a zero coupon bond, which leaves the marginal utility of the representative agent unchanged, a similar argument implies: $1 \cdot U'\big(c(0)\big) = e^{-\rho t} U'\big(c(t)\big) e^{r_t t}$. Then r_t is interpreted as per period rate of return at date 0 for a zero-coupon bond maturing at date t.

generations). If society is richer in the future, $(g > 0)$, then less value will be placed on increments to consumption in the future. Thus, society values projects less that have payoffs in the future if the future is richer and there is diminishing marginal utility, as is commonly assumed. Note also that the higher ηg is, the higher the SDR is. This promotes consumption relative to investment now and vice versa.

SDR in climate policy

As in the previously discussed models to determine the SCC, various parameter specifications are used in the literature. Table 10 provides some examples, including the resulting SDR.

Table 10: Examples of SDR from the literature

Research paper	ρ	η	g	SDR
Stern (2007)	0.1%	1.0	1.3%	1.40%
Cline (1992)	0.0%	1.5	1.3%	2.05%
Nordhaus (2007)	3.0%	1.0	1.3%	4.30%

SDR under risk

If there is uncertainty, we need to take expectations into account, so that the SDR formula becomes $r_t = \rho - \frac{1}{t}\ln\frac{\mathrm{E}U\prime(c(t))}{U\prime(c(0))}$. To go beyond this formula, we need to specify the stochastic process. If the logarithm of consumption follows a stationary Brownian motion so that $d\ln c_t = \mu dt + \sigma dz_t$, which implies $\frac{dc}{c} = (\mu + 0.5\sigma^2)dt + \sigma dz$, where μ and σ are two scalars measuring the mean and standard deviation of the change in log consumption, respectively, then the Ramsey formula becomes (Gollier 2007):

$$r_t = \rho + \eta\mu - 0.5\eta^2\sigma^2.$$

The last term in the above equation is a precautionary effect: uncertainty about the rate of growth in consumption reduces the discount rate, causing more savings in the present. However, the magnitude of the precautionary effect is likely to be small, at least in the United States. Using annual data from 1889–1978 for the US, Kocherlakota (1996) estimated μ_g to be 1.8% and σ_g to equal 3.6%. This implies that the precautionary effect is 0.26%. The higher uncertainty is ($0.5\eta^2\sigma^2$), the lower the SDR. This promotes investment now relative to consumption now as a precaution against future uncertain consumption, and vice versa.

III.8.2 The Extended Ramsey Formula

We now expand on the issue of riskiness. Indeed, some countries, such as France, Norway, or the Netherlands, have adjusted the discount rate for public projects to account for project risks. There are two basic forms of project risk, one of which is important from the perspective of the SDR, and another which is not.

Unsystematic risk is the risk associated with over- or underestimating the costs and benefits of the project. In any given project, cost elements could turn out to be more or less expensive than expected, due to technical problems or other unforeseen reasons. These risks are diversifiable across a portfolio of public projects. The theory of asset pricing shows that this kind of risk ought not to affect the price of an asset, and hence the appropriate discount rate.

Systematic risk is the second type of risk. It describes a situation where risky costs and benefits are correlated to returns available in the macro-economy. Systematic risk cannot be diversified across different projects due to the macro scale of the riskiness. Asset pricing theory shows that when the project's net benefits are correlated with uncertainty regarding the wider macro economy, the discount rate should be augmented by a risk premium reflecting the project-specific risk profile of systematic risk, not the diversifiable risk.

Under the assumption that the project's net benefits and consumption growth follow a bivariate normal distribution, incorporating project risk into the appraisal of projects leads to a simple extension of the RHS of the Ramsey Formula for a risky project j:

$$SDR = \rho + \eta\mu - 0.5\eta^2\sigma^2 + \eta^2\beta_j\sigma^2.$$

The parameter β_j is the consumption *beta* which measures the correlation (covariance) between the net benefits of project *j* and systematic risk associated with consumption growth.

- If $\beta_j = 1$, then a 1% increase in consumption growth will be expected to lead to a 1% growth in the project's net benefits.
- If $\beta_j > 1$, then the project's benefits are expected to increase by more than 1% when consumption grows by 1%, introducing proportionally more systematic risk than exists in the economy.
- If $\beta_j < 0$, the project reduces risk and has insurance properties.

Project risks and discounting: The Capital Asset Pricing Model (CAPM) approach

The CAPM has been influential in social discounting. The CAPM pricing formula prices the risk associated with an asset by adding a risk premium to the risk-free rate of return in a manner similar to the consumption CAPM. The CAPM asset return formula is

$$r_j = r_f + \beta_j(r_m - r_f).$$

where r_f is the risk-free rate of return, r_m is the rate of return on the market/wealth portfolio and β_j is the project *beta* that reflects the correlation between the asset *j* and the market portfolio. The risk premium for this project is given by the market premium $(r_m - r_f)$ multiplied by the project beta, β_j.

The risk premium will be positive when β_j is positive. The logic of this pricing formula is similar to the Ramsey formula under project risk except that the covariance is with a market portfolio of assets rather than with consumption. This formula for the SDR is project specific but can be estimated by looking at suitable market returns and calculating the associated project betas.

III.8.3 Declining Discount Rates (DDRs): The Expected Net Present Value Approach to DDRs

Weitzman (1998) argued that even if every individual believes in a constant discount rate, widespread opinions on what this discount rate should be makes the effective SDR decline significantly over time.[97] Weitzman proved that computing the expected net present value of a project with an uncertain but constant discount rate is equivalent to computing the net present value with a certain but decreasing *certainty-equivalent* discount rate. Hence, if such uncertainty over the future discount rate exists, then the respective representation by declining (certainty-equivalent) discount rates effectively increases the weight that is given to future benefits and costs compared to those in the present.

III.9 Distributional Issues

Projects and policies with environmental impact inevitably have distributional consequences.[98] Environmental policies work by favoring (relative to the status quo) victims of pollution at the expense of polluters. Moreover, some policies or projects might deliver greater efficiency at the cost of distributional outcomes. Specifically, it is possible that the change in the consumer surplus (ΔCS) due to the price increase of a restriction to competition (or increased costs) and the change in the environmental benefits (ΔEB) will accrue to different income groups. In such a case it may be desirable to adjust weights given to changes in consumer surplus and changes in environmental quality.

The following ties back to the discussion of different welfare criteria (Section II.2). The most frequently used system of distributional weights in practice is a vector $\boldsymbol{a} = (a_1, \dots, a_I)$ for $i = 1, \dots, I$ income groups such that:

$$a_i = \left(\frac{\bar{Y}}{Y_i}\right)^{\eta},$$

where $\overline{\mathrm{Y}}$ is average or mean income per capita, $\mathrm{Y_i}$ is income of the i[th] individual (or group) and η is the elasticity of the marginal utility of income or society's valuation of an increment to that individual's income. In principle, η could range from $0 \leq \eta < \infty$, although conventional or unweighted approaches are equivalent to assuming $\eta = 0$, as this would result in $\mathrm{a_i} = 1$. At the other extreme, as the degree of inequality aversion grows ($\eta \rightarrow \infty$), the cost-benefit rule means always ruling out any project that adversely affects the very worse off. (Conversely, it will always rule in a project that positively affects the very worse off.) In typical applied cases, η will take values greater or equal to one[99] and less than three, as indicated in the section on discounting.

Under a weighting system like the one described above, the net welfare changes from the price change and the environmental improvement accruing to two different groups will be

[97] For a more detailed account of various rationales for assuming a DDR see also the literature discussed in OECD (2018, ch. 8).

[98] The increasing relevance of equity concerns in CBA is also expressed in OECD (2018, ch. 11), which summarizes various approaches.

[99] $\eta = 1$ implies a logarithmic utility function.

$$\Delta W = a_1 \Delta CS + a_2 \Delta EB,$$

with ΔCS and ΔEB measured by one of the methods described above as appropriate for this case. Note that ΔCS and ΔEB could be in a present value form, discounted at an appropriate rate provided that an explicit time dimension exists. Turning back to our previously introduced terminology, according to the Kaldor-Hicks criterion, for instance, an agreement among competitors will constitute a potential Pareto improvement if this measure is positive (thereby trading off $\Delta CS < 0$ with $\Delta EB > 0$.

IV CONCLUDING REMARKS

This report drew on concepts and tools from environmental economics in order to spotlight the forms of quantitative analysis that can be used in competition assessments to account for broader social benefits, including benefits for future generations. While the focus of this report was on benefits related to environmental sustainability, the presented concepts and tools are also more broadly applicable to other aspects of sustainability. As a leading example of a potential restriction to competition, we used an agreement between competitors, thus presuming that the consideration of greater societal benefits arising from such an agreement is within the remit of the competition authority. Such agreements can take various forms; for example, firms may agree to phase out the production and sale of a less sustainable product variant. As a trade-off, this may increase the prices paid by consumers, or reduce their range of choice. Drawing on established practice in environmental economics, we showed how the benefits thus realized to society can be properly measured, while also considering benefits to future generations.

In the first part of this report, we introduced key concepts in welfare economics in the presence of externalities. Even perfectly competitive markets fail when such externalities are not internalized. We discussed various welfare standards and also showed how a partial analysis, when warranted, may be conducted by comparing changes in consumer welfare to potentially reduced environmental damages. Subsequently, in the second, main part of this report, we discussed various methods for measuring changes to environmental sustainability/environmental damages using the concept of TEV.

We first introduced methods for estimating how individuals value environmental goods, as revealed by the market, in hypothetical choice scenarios, or through stated preferences. As shown, when the use of an environmental good leaves a behavioral trail in markets for non-environmental goods, revealed preference approaches may also be used to estimate use values, even when the considered good is not directly traded. As there are many variations of such assessment tools, our presentation was not exhaustive. We also discussed alternative methods that are useful, for instance, when individuals are not fully aware of the negative impact of certain production or consumption choices. In addition, we highlighted the need to consider future members of society, including potential change in environmental preferences.

Our discussion of CEA showed how the benefits of a given measure – in our example, a considered agreement – may be obtained as well from saved abatement costs, under a given policy objective (such as an emission reduction target). At this juncture and at other points in the report, we made special reference to SCC assessments. We also placed special emphasis on the extrapolation of environmental (shadow) prices from existing studies or databases, an approach which may be of particular usefulness given time or resource constraints.

In our discussion of methods, we also highlighted the information required to perform an assessment. The net welfare benefits of a considered agreement can be estimated using the methods we described given the availability of information about: (i) the specific characteristics of the market and product(s) subject to the agreement, (ii) information about how prices and demand will respond, and (iii) information about the specific environmental/sustainability benefits obtained (e.g.

reduction in air pollution). Furthermore, information is needed to support the assertion that a restriction to competition is indispensable for achieving the targeted benefits. When an agreement is then deemed to be socially desirable and when a mandate for intervention exists, these calculations can support the decision-making of antitrust authorities.

Throughout this report we presented various overviews of different methods, as taken from the academic literature. For the intended practical purposes, it seems expedient to conclude with a slightly different presentation of the discussed methods, with a classification based also on whether or to what extent case-specific data needs to be generated.

Table 11: Overview of methods presented in this report

I: Methods for environmental valuation using case-specific data	
(1) Methods based on market choices (potentially in surrogate markets)	Examples: – Discrete choice analysis of preferences revealed from actual purchases (e.g. of products that are more or less environmentally friendly) – Hedonic prices derived from surrogate markets, e.g. real estate prices
(2) Methods based on hypothetical choices or stated preferences	Examples: – Contingent valuation analysis based on surveys of stated preferences over hypothetical scenarios – Conjoint analysis of (pairwise) choice between different scenarios (e.g. products) – Subjective well-being valuation based on correlating stated well-being with observable (environmental) variables and monetary values
II: Valuation methods for estimating and aggregating case-specific impact	
(1) Dose-response approaches	Example: Estimating welfare through the impact on life expectancy or morbidity
(2) Averting and defensive behavior	Example: Estimating avoided costs of defensive expenditures
III: Valuation using data from existing studies and databases	
(1) Benefit transfer within a calibrated model	Example: Adjusting willingness-to-pay (e.g. obtained from contingent valuation) to different socioeconomics and demographics
(2) Environmental prices databases	Example: Using environmental prices aggregating all health-related costs from the emission of a particular substance in a specific country
IV: Valuation derived from stated policy objectives	
(1) Using market prices for permits or taxes on emissions	Example: CO_2 prices from the EU Emissions Trading System
(2) Use of avoided abatement costs under a cost effectiveness analysis	Example: CO_2 prices based on an analysis and ranking of the costs of alternative abatement methods

Turning to the overview provided in Table 11, we first note that while this report was primarily concerned with methods for measuring sustainability benefits, when interventions that restrict competition lead to reductions in consumer surplus, competition assessments need to consider such effects. With the considered practical applications in mind, we now divide the discussed measurement approaches as follows: Our first set of discussed approaches proceeds from "primitives" –

that is, from the preferences of concerned individuals. There, changes to total welfare are calculated by aggregating individual preferences. Another option is to measure aggregate impacts to health or productivity (e.g. from harmful emissions). Our second set of approaches discussed in this report, relies on a shortcut – specifically, they extrapolate from existing data, e.g. environmental prices. Or it is (implicitly) assumed that a policy goal, such as a cap on emissions, is an expression of societal preferences, from which benefits are then derived.

ABBREVIATIONS AND ACRONYMS

ACM	Netherlands Authority for Consumers and Markets
BT	Benefit Transfer
CA	Conjoint Analysis
CAPM	Capital Asset Pricing Model
CBA	Cost-Benefit Analysis
CEA	Cost-Effectiveness Analysis
CM	Choice Modeling
CO_2	Carbon Dioxide
COI	Cost of Illness
CRI	Consumption Rate of Interest
CVM	Contingent Valuation Method
DCE	Discrete Choice Experiments
DDR	Declining Discount Rate
DECC	Department of Energy and Climate Change
DICE	Dynamic Integrated Model of Climate and the Economy
EGS	Environmental Good or Service
EUA	European Union Allowance
EU ETS	EU Emissions Trading System
EVRI	Environmental Valuation Reference Inventory
ExternE	Externalities of Energy Project
GHG	Greenhouse Gases

HCC	Hellenic Competition Commission
HPM	Hedonic Price Method
IAM	Integrated Assessment Model
MAC	Marginal Abatement Cost
OECD	Organization for Economic Cooperation and Development
SCC	Social Cost of Carbon
SDR	Social Discount Rate
SOC	Social Opportunity Cost
STP	Social Time Preference
SWB	Subjective Well-Being Valuation
TCM	Travel Cost Method
TEEB	The Economics of Ecosystem and Biodiversity
TEV	Total Economic Value
TFEU	Treaty on the Functioning of the European Union
UK	United Kingdom
WTP	Willingness to Pay

REFERENCES

ACM (2020). *Draft Guidelines: Sustainability Agreements – Opportunities within Competition Law.* The Hague, Netherlands: Netherlands Authority for Consumers and Markets.

Andersson, H., Jonsson, L. and M. Ögren (2013). Benefit Measures for Noise Abatement: Calculations for Road and Rail Traffic Noise. *European Transport Research Review*, 5(3), 135-148.

Arrow, K. J., Cropper, M. L., Gollier, C., Groom, B., Heal, G. M., Newell, R. G., Nordhaus, W. D., Pindyck, R. S., Pizer, W. A., Portney, P. R., Sterner, T., Tol, R. S. J. and M. L. Weitzman (2014). Should Governments Use a Declining Discount Rate in Project Analysis? *Review of Environmental Economics and Policy*, 8(2), 145-163.

Arrow, K. J. (1969). The Organization of Economic Activity: Issues Pertinent to the Choice of Market versus Nonmarket Allocation. In: U.S. Joint Economic Committee of Congress, 91st Congress, 1st Session (eds.). *The Analysis and Evaluation of Public Expenditures: The PPB System*, pp. 59-73. Washington D.C., USA: U.S. Government Printing Office.

Atkinson, G., Groom, B., Hanley, N. and S. Mourato (2018) Environmental Valuation and Benefit-Cost-Analysis in UK-Policy. *Journal of Benefit-Cost Analysis*, 9(1), 97-119.

Bernheim, B. (2016). The Good, the Bad, and the Ugly: A Unified Approach to Behavioral Welfare Economics. *Journal of Benefit-Cost Analysis*, 7(1), 12-68.

Brander, J. A. and M. S. Taylor (1998). The Simple Economics of Easter Island: A Ricardo-Malthus Model of Renewable Resource Use. *American Economic Review*, 88(1), 119-138.

Brennan, A. and Y.-S. Lo (2020). Environmental Ethics. In: Zalta, E. N. (ed.). *The Stanford Encyclopedia of Philosophy.* Stanford, USA: Metaphysics Research Lab, Center for the Study of Language and Information, Stanford University.

Brink, C. and H. van Grinsven (2011). Costs and Benefits of Nitrogen in the Environment. In: Sutton, M. A., Howard, C. M., Erisman, J. W., Billen, G., Bleeker, A., Grennfelt, P., Grinsven, H. van and B. Grizzetti (eds.). *The European Nitrogen Assessment: Sources, Effects and Policy Perspectives*, pp. 513-540. Cambridge, England: Cambridge University Press.

Carson, R. (1963). *Silent Spring*. London, England: Hamish Hamilton.

CE Delft (2018). *Environmental Prices Handbook 2017: Methods and Numbers for Valuation of Environmental Impacts.* Delft, Netherlands: CE Delft.

Champ, P. A, Boyle, K. J. and T. C. Brown (2003). *A Primer on Nonmarket Valuation*. Dordrecht, Netherlands: Kluwer Academic Publishers.

Chater, N., Huck, S. and R. Inderst (2010). *Consumer Decision-Making in Retail Investment Services: A Behavioural Economics Perspective. Final Report.* Brussels, Belgium: European Commission Directorate-General for Health and Consumers.

Cline, W. R. (1992). *The Economics of Global Warming*. Washington D.C., USA: Peterson Institute for International Economics.

Cohen, P., Hahn, R., Hall, J., Levitt, S. and R. Metcalfe (2016). Using Big Data to Estimate Consumer Surplus: The Case of Uber. *Working Paper.*

Common, M. and S. Stagl (2005). *Ecological Economics: An Introduction.* Cambridge, England: Cambridge University Press.

Constantatos, C., Pargianas, C. and E. S. Sartzetakis (2021). Green Consumers and Environmental Policy. *Journal of Public Economic Theory*, 23(1), 105-140.

Constanza, R. (1991). *Ecological Economics: The Science and Management of Sustainability*. New York, USA: Columbia University Press.

Constanza, R., Norton, B. G. and B. D. Haskell (1992). *Ecosystem Health: New Goals for Environmental Management*. Washington D.C., USA: Island Press.

Daly, H. E. (1991). *Steady State Economics*. Washington D.C., USA: Island Press.

Davis, P. and E. Garcés (2009). *Quantitative Techniques for Competition and Antitrust Analysis*. Princeton, USA: Princeton University Press.

DECC (2009). *Carbon Valuation in UK Policy Appraisal: A Revised Approach.* London, UK: UK Department of Energy and Climate Change.

EC (2006). *ExternE – Externalities of Energy. A Research Project of the European Commission.* Brussels, Belgium: European Commission Directorate-General for Research.

ECCC (1997). *Environmental Valuation Reference Inventory.* Ottawa, Canada: Department of the Environment.

Ehrlich, P. R. (1968). *The Population Bomb*. New York, USA: Ballantine Books.

Feldman, A. and A. Kirman (1974). Fairness and Envy. *American Economic Review*, 64(6), 995-1005.

Feyen, L., Ciscar, J. C., Gosling, S., Ibarreta, D. and A. Soria (2020). Climate Change Impacts and Adaptation in Europe. *Working Paper*.

Gollier, C. (2007). The Consumption-Based Determinants of the Term Structure of Discount Rates. *Mathematics and Financial Economics*, 1(2), 81-101.

Hanley, N. and E. B. Barbier (2009). *Pricing Nature: Cost-Benefit Analysis and Environmental Policy*. Cheltenham, England: Edward Elgar Publishing.

Hartwick, J. M. (1977). Intergenerational Equity and the Investing of Rents from Exhaustible Resources. *American Economic Review*, 67(5), 972-974.

Hartwick, J. M. (1978). Substitution Among Exhaustible Resources and Intergenerational Equity. *Review of Economic Studies*, 45(2), 347-354.

Hausman, J. (1981). Exact Consumer's Surplus and Deadweight Loss. *American Economic Review*. 71(4), 662-676.

HCC (2020). *Competition Law & Sustainability: Draft Staff Discussion Paper on Sustainability Issues and Competition Law*. Athens, Greece: Hellenic Competition Commission.

Heal, G. (2005). Intertemporal Welfare Economics and the Environment. In: Mäler, K. G. and J. Vincent (eds.). *Handbook of Environmental Economics, vol. 3*, pp. 1105-1145. Amsterdam, Netherlands: Elsevier.

Inderst, R. and S. Thomas (2020a). Prospective Welfare Analysis – Extending Willingness-to-Pay Assessment to Embrace Sustainability. *Working Paper*.

Inderst, R. and S. Thomas (2020b). Reflective Willingness to Pay: Preferences for Sustainable Consumption in a Consumer Welfare Analysis. *Working Paper*.

Jamieson, D. (2002). *Morality's Progress: Essays on Humans, Other Animals, and the Rest of Nature*. Oxford, England: Clarendon Press.

Kingston, S. (2019). Competition Law in an Environmental Crisis. *Journal of European Competition Law & Practice*, 10(9), 517-518.

Kocherlakota, N. R. (1996). The Equity Premium: It's Still a Puzzle. *Journal of Economic Literature*, 34(1), 42-71.

Lianos, I. (2018). Polycentric Competition Law. *Current Legal Problems*, 71(1), 161–213.

Louviere, J. J. and D. A. Hensher (1982). On the Design and Analysis of Simulated Choice or Allocation Experiments in Travel Choice Modelling. *Transportation Research Record*, 890(1), 11-17.

Louviere, J. J., and G. Woodworth (1983). Design and Analysis of Simulated Consumer Choice or Allocation Experiments: An Approach Based on Aggregate Data. *Journal of Marketing Research*, 20(4), 350-367.

McVittie, A. and S. Hussain (2013). The Economics of Ecosystems and Biodiversity – Valuation Database Manual. *Working Paper*.

Meadows, D. H., Meadows, D. L., Randers, J. and W. W. Behrens (1972). *The Limits to Growth*. New York: Universe Books.

Nash, R. F. (1989). *The Rights of Nature: A History of Environmental Ethics*. Madison, USA: University of Wisconsin Press.

Nedellec, V. and A. Rabl (2016). Costs of Health Damage from Atmospheric Emissions of Toxic Metals: Part 1 – Methods and Results. *Risk Analysis*, 36(11), 2081-2095.

Nordhaus, W. D. (2007). A Review of the Stern Review on the Economics of Climate Change. *Journal of Economic Literature*, 45(3), 686-702.

Nordhaus, W. D. (2017). Revisiting the Social Cost of Carbon. *Proceedings of the National Academy of Sciences*, 114(7), 1518-1523.

Nordhaus, W. D. (2018). Projections and Uncertainties about Climate Change in an Era of Minimal Climate Policies, *American Economic Journal: Economic Policy*, 10(3), 333-360.

Nordhaus, W. D. and P. Sztorc (2013). *DICE 2013R: Introduction and User's Manual, Second Edition.* New Haven (CT), USA: Yale University.

OECD (2014). *The Cost of Air Pollution: Health Impacts of Road Transport*. Paris, France: OECD Publishing.

OECD (2016). *The Economic Consequences of Outdoor Air Pollution*. Paris, France: OECD Publishing.

OECD (2018). *Cost-Benefit Analysis and the Environment: Further Developments and Policy Use*. Paris, France: OECD Publishing.

Ott, W., Baur, M., Kaufmann, Y., Frischknecht, R. and R. Steiner (2006). *NEEDS: Assessment of Biodiversity Losses*. Zurich, Switzerland: econcept AG and ESU-Services.

Papandreou, A. A. (1994). *Externality and Institutions*. New York, USA: Oxford University Press.

Parsons, G. (2003). The Travel Cost Model. In: Champ, P. A., Boyle, K. and T. C. Brown (eds.). *A Primer on Nonmarket Valuation*, pp. 269-329. Dordrecht, Netherlands: Springer Netherlands.

Pearce, D., Atkinson, G. and S. Mourato (2006). *Cost-Benefit Analysis and the Environment: Recent Developments*. Paris, France: OECD Publishing.

Sartzetakis, E. S., Xepapadeas, A. and A. N. Yannacopoulos (2015). Regulating the Environmental Consequences of Preferences for Social Status Within an Evolutionary Framework. *Working Paper.*

Solow, R. M. (1974). Intergenerational Equity and Exhaustible Resources. *Review of Economic Studies*, 41(5), 29-46.

Solow, R. M. (1992). *An Almost Practical Step Towards Sustainability*. Washington D.C., USA: Resources for the Future.

Stern, N. (2007). *The Economics of Climate Change: The Stern Review*. Cambridge, England: Cambridge University Press.

Stiglitz, J. and N. Stern (2017). *Report of the High-Level Commission on Carbon Prices*. Washington D.C., USA: World Bank Group: Carbon Pricing Leadership Coalition.

Taylor, L. (2003). The Hedonic Method. In: Champ, P.A., Boyle, K. and T. C. Brown (eds.). *A Primer on Nonmarket Valuation*, pp. 331-393. Dordrecht, Netherlands: Springer Netherlands.

Torgler, B., García-Valiñas, M. A. and A. Macintyre (2008). Differences in Preferences Towards the Environment: The Impact of a Gender, Age and Parental Effect. *Working Paper*.

Van Dijk, T. (forthcoming). A New Approach to Assess Certain Sustainability Agreements Under Competition Law. In: *Competition Law and Environmental Stability*. Paris, France: Concurrences Books.

Varian, H.R. (1974). Equity, Envy, and Efficiency. *Journal of Economic Theory*, 9(1), 63-91.

Viscusi, W. K. and J. E. Aldy (2003). The Value of a Statistical Life: A Critical Review of Market Estimates Throughout the World. *Journal of Risk and Uncertainty*, 27(1), 5-76.

WCED (1987). *Report of the World Commission on Environment and Development: Our Common Future*. New York, USA: Oxford University Press.

Weitzman, M. L. (1998). Why the Far-Distant Future Should Be Discounted at Its Lowest Possible Rate. *Journal of Environmental Economics and Management*, 36(3), 201-208.

Weitzman, M. L. (2001). Gamma Discounting. *American Economic Review*, 91(1), 260-271.

Welsch, H. and J. Kühling (2009). Using Happiness Data for Environmental Valuation: Issues and Applications. *Journal of Economic Surveys*, 23(2), 385-406.

Willig, R. D. (1976). Consumer's Surplus without Apology. *American Economic Review*, 66(4), 589-597.

World Bank (2016). *The Cost of Air Pollution: Strengthening the Economic Case for Action.* Washington D.C., USA: World Bank Group.

World Bank (2017). *Guidance Note on Shadow Price of Carbon in Economic Analysis*. Washington D.C., USA: World Bank Group.