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Wolfgang Buchholz
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Foundations of Environmental Economics

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About this Book

According to recent analyses by the National Aeronautics and Space Administration (NASA) and the National Oceanic and Atmospheric Administration (NOAA), Earth's global surface temperatures in 2018 have been the fourth warmest since 1880. The years 2014–2018 have been, collectively, the warmest years since preindustrial times. Due to the observed global warming, serious adverse effects on economic welfare and living conditions in general are pending, and, as many fear, even an existential threat for humanity exists. Besides such global environmental challenges, also local problems like air pollution in municipalities are attracting much attention in the public debate. As a consequence, the older and thus particularly dirty diesel cars are banned from many European cities.

Environmental problems, like air pollution and global warming, are associated with significant damage cost for societies. Yet, policies to tackle these problems and to abate emissions involve costs, too. Therefore, societies observe a genuine trade-off between the costs of pollution, on the one hand, and the costs for emission reduction, on the other. To address this conflict with the help of the economist's toolbox is the basic concern of environmental economics. This includes two tasks: first of all, to evaluate the environmental harm and the costs of pollution abatement, which is a rather challenging task as the present discussion about the "correct" assessment of harmful nitrogen oxides (NO_x) pollution clearly demonstrates. When based on this assessment of costs and benefits something like the "optimal" pollution level has been determined, then as a second task the instruments of environmental policy that are appropriate to bring about this socially optimal outcome have to be devised, and their advantages and disadvantages have to be assessed and compared. The issue of instrument choice also raises many controversies as, e.g., the discussion on European climate policy and the European emissions trading system as one of its central parts shows. Against this background, our book will be organized as follows.

In ► Chap. 1, we explain the important role that environmental economics plays in dealing with environmental problems, pointing out how environmental economics serves as a valuable complement to natural sciences in finding solutions to environmental challenges. In ► Chap. 2, we then present the fundamental economic approaches for handling environmental problems giving special attention to the fundamental concepts conceived by Arthur Cecil Pigou as the founding father of environmental economics, as well as to a critical discussion of voluntary market-like approaches in the spirit of the Nobel Prize laureate Ronald Coase. For the real-world application of such mechanisms, information requirements on the costs and benefits of pollution abatement are regularly high. As market prices for most environmental goods and services do not exist, special methods for assessing the monetary value of environmental quality and its improvement were developed, which we consider in ▶ Chap. 3. There, we will mainly focus on the contingent-valuation method, which allows to capture also the so-called nonuse values (as the feeling of responsibility for environmental protection), which usually are not reflected in market transactions. After having illustrated the ways and means for getting the information required for a sound environmental policy, we take a closer look at the most important instruments of such a policy in ▶ Chap. 4. Our focus there will be on environmental taxes and emissions trading schemes, which, in Pigou's tradition, are based on pricing pollution and are strongly preferred to command and control regulations by almost all economists. In ▶ Chap. 4, it is not only explained how these instruments work and what their main advantages are but also which obstacles may impair their efficient application in the real world. Finally, in ▶ Chap. 5, we direct our attention toward transnational and global environmental problems, which present a particular challenge to environmental economics, as no central authority exists that can enforce environmental protection at the international level. Therefore, in this case, the implementation of effective environmental policy measures crucially depends on voluntary actions by sovereign and mostly selfish countries, which causes free-rider problems and severely impedes the attainment of a globally efficient environmental policy. To bring about a globally efficient protection of the environment, global collective action based on international environmental negotiations is required. Using a simple game-theoretic model, strategic issues that arise in this context will be explored.

This book aims at providing a solid basis for the education of bachelor students of economics and business administration being interested in environmental economics as well as some scientific guidance for policy-makers pursuing efficient environmental policies. Environmental problems and the economic approaches for their solution are considered both at the national and the international level. The book employs both graphical and mathematical tools that are known from basic undergraduate courses in microeconomics. It contains definitions of the most important terms and concepts used in environmental economics and a lot of control questions to check the readers' understanding. For each of the five sections of the book, a separate summary is provided.

Contents

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studied Mathematics, Economics, and Political Science at the Universities of Freiburg and Tübingen, where he finished his Ph.D. in Economics in 1982 and Habilitation in 1987. In 1988, he became an Associate Professor of Economics at the University of Regensburg. After an interplay as a Professor at the newly founded European University in Frankfurt (Oder) between 1992 and 1996, he returned to Regensburg – now as a full Professor of Public and Environmental Economics. Since 2002, he has also been a Research Professor at the Ifo Institute for Economic Research in Munich. His main fields of research are international environmental economics, the theory of public goods, intertemporal evaluation and discounting, social protection and distribution policy, and the links between economics and ethics.

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1 Introduction: Economics and Environmental Degradation

W. Buchholz, D. Rübbelke, *Foundations of Environmental Economics*, Springer Texts in Business and Economics, https://doi.org/10.1007/978-3-030-16268-9_1

The impairment of the natural environment by human activity is reflected by problems such as climatic change, local air pollution, electromagnetic radiation, and overdevelopment of the landscape. These problems cause manifold concerns and fears. People worry that the progressing environmental deterioration will not only adversely affect their health and well-being but that it will also severely threaten their descendants.

Pollution of the environment is not a novel problem. In the history, one can find several examples where human activity impaired the environment and thereby – in turn – adversely affected humans' quality of life. Already King Philip II of France (1180–1223) fainted because of effluvia from the dirt on streets (Bockendahl, 1870, quoted by Winkle, 1982). Due to the dirt on public streets of European cities in medieval times, in several cases pedestrians even had to walk on stilts in order to be able to use the roads (Winkle, 1982). Yet, humans also suffered from air pollution, which, e.g., caused a rising mortality in London of the seventeenth century (Brimblecombe, 1987). The earliest documented air pollution incident in England occurred in Nottingham, and when Queen Eleanor visited Nottingham Castle in 1257, she was forced to leave it because stench of sea-coal smoke threatened her health (Brimblecombe & Makra, 2005). Pollution remained a serious problem, e.g., in 1930, in the Belgian Meuse Valley, a ten times higher mortality was observed due to air pollution (Firket, 1936). Nemery, Hoet, and Nemmar (2001: 704) point out that this incidence in the Meuse Valley where fog on two subsequent days in December caused 60 deaths is a landmark as it "led to the first scientific proof of the potential for atmospheric pollution to cause deaths and disease, and it clearly identified the most likely causes." Nowadays, according to statistics of the World Health Organization (WHO), 4.2 million deaths every year are a result of exposure to ambient (outdoor) air pollution (information taken from the internet pages of the WHO).

Although one can observe public concern about pollution as well as regulations to combat environmental problems already in the medieval period and the early modern age,[1] public attention to the deterioration of the ecological system has increased in particular since the 1970s. Environmental protection has now become an important issue in the media and on the political agenda. There is hardly a day without some reporting on the threats to nature or our natural livelihood, e.g., in the context of natural catastrophes such as tornadoes, droughts, and flooding, and complaints about poor air quality in the cities.

Referring to processes in nature at first glance might suggest that the responsibility for addressing environmental problems can be assigned primarily to the natural sciences (e.g., biology, chemistry, meteorology, and physics). Indeed, these sciences inform us about the functioning of complex ecosystems and how the functioning of these systems is disturbed by human activities.

Yet, environmental protection has also become an important topic in economics as well as in social sciences in general. Over the last 40 years, environmental economics has

1 According to Baas (1905: 13), epidemics like those of smallpox brought about environmental regulations in the German town of Freiburg with regard to the cleaning of streets.

developed to one of the most important subdisciplines of economics. Expressed in a simplifying manner, the reason for this is that environmental protection involves monetary costs that have to be weighted against the benefits of environmental improvement.

By means of three theses, we will now discuss in more detail why environmental protection has become an important issue in economics. In doing so, we will also explain the basic concepts that are key for the understanding of environmental problems from the economist's perspective.

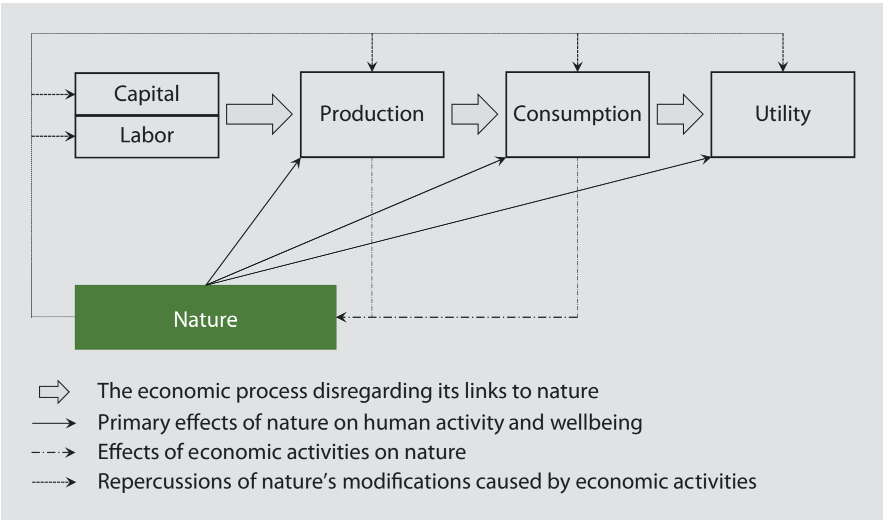
1.1 Thesis 1

The economic process (production of goods and services) requires the input of natural production factors (land, energy, etc.*) and has repercussions (release of pollutants into the air or water,* etc.*) on nature. Thus, the economy is embedded in ecological cycles.*

This thesis states on the one hand that the economic process as well as the human existence in general requires the use of natural input factors. Without breathable air, drinkable water, and sufficient supply of food, a human being can neither exist nor produce goods and services. The conditions provided by nature such as weather conditions influence economic productivity (e.g., in farming) and quality of life of human beings. At the same time, human activity modifies the natural environment in manifold ways. This – in turn – has repercussions on human beings.

The fundamental interrelations between human beings and nature are illustrated in the following ◘ Fig. 1.1.

The effects of the economic process on nature can largely be subdivided into *extraction effects* and *disposal effects*.



◘ **Fig. 1.1** Interrelations between the economic process and nature

Definition

Extraction effects are effects that arise when the current use of a natural resource impairs the options to use it in the future. Among such effects are, e.g., landscape modifications due to the establishment of water reservoirs or open brown coal pits. The destruction of tropical forests and the loss of biodiversity also belong to this category.

Both exhaustible resources like coal or other fossil fuels and regenerative resources like forests, fish stocks, and drinking water can be overused. One liter of petroleum burnt today will not be available anymore in the future. Furthermore, the available stock of this exhaustible resource will not regrow in a natural way in the foreseeable future. If regenerative resources are overused, their later availability is adversely affected, and the stocks of these resources may even be irreversibly destroyed. This case occurs when the quantities extracted exceed the quantities that grow back. In this case – albeit in a weaker form – an intertemporal trade-off between the use of these resources at different points in time also arises, and with it the sustainability of resource use is at risk.

Definition

Disposal effects result from the physical, chemical, or biological transformation of natural substances involved in consumption activities and the production process. According to the *material balance principle*, the substances that have been used in economic processes prevail after the termination of these processes albeit in a modified form.[2]

Physical transformations only cause a change in the composition of matter, while the total mass of the involved materials remains unchanged. With respect to environmental pollution, chemical transformations, e.g., caused by the burning of fossil fuels (coal, oil, natural gas), are of particular importance. The chemically transformed substances (waste products like lead, sulfur dioxide, carbon monoxide, and nitrogen oxides) are released back into the environment and alter its composition. The backflows (into environmental media like air and water) are referred to as *emissions*. Besides material emissions there are energetic emissions, too. Among these are waste heat from coal and nuclear power plants, nuclear radiation, as well as noise.

The extent to which emissions harm the nature depends on various factors, e.g., on:

- The noxiousness of the emitted substances
- Meteorological or geographical conditions (weather and landscape)
- The assimilation capacity of nature

The potential harm of a pollutant depends on the pollution load of an environmental medium, i.e. the immission (that must be clearly distinguished from the *emission*).

2 On the material balance principle, see also Perman et al. (2011: 23ff.).

Definition

Immissions are pollutants that emerge in an environmental medium and potentially adversely affect human beings, animals, plants, soil, water, and the atmosphere. The *immission level* is measured by means of objective indicators, e.g., by the concentrations of a pollutant in the respective environmental medium at a certain measuring point (e.g., 50 µg/m³ NO_2 in Dresden city center).

The immission level at some receptor point, which usually also varies over time, is – due to many factors – generally not unambiguously correlated with the emissions from a single source. To illustrate the complex relationship between emissions from different sources and the immission levels at different receptor points, let e_i denote the discharge of a pollutant at the source $i = 1, \ldots, n$. The immission q_j, i.e. the environmental load at the measurement point (discharge site) $j = 1, \ldots, m$, is measured as the concentration of the environmental pollutant per unit of the environmental medium. If the immissions are proportional to the emission levels, the immission function is $q_j = \sum_{i=1}^{n} \alpha_{ij} e_i$, where the parameter α_{ij} is the immission coefficient that indicates the impact of an emission at source i on the pollution level at the receptor j. Regarding air pollution (like sulfur dioxide) and given winds from west to east as usually observed in the northern hemisphere, the parameter α_{ij} is larger when the discharge site j is located in the east and the emitter i is located in the west.

With respect to the level of immissions exerting a harmful effect, a potentially prevailing assimilation capacity of nature has to be taken into account, which means that up to the immission level $\overline{q}_j = \sum_{i=1}^{n} \alpha_{ij} \overline{e}_i$, nature absorbs incoming emissions $\sum_{i=1}^{n} \alpha_{ij} e_i$ without any adverse effects, e.g., by the depletion of organic substances in waters. Only if immissions surpass the level $\overline{q}_j$, adverse immission effects occur. When the level $\overline{q}_j$ is exceeded, a harmful accumulation of pollutants may arise. In such a case, the pollutant is not a *flow pollutant* (like noise) but a *stock pollutant* (like carbon dioxide, which accumulates in the earth's atmosphere). Thus, $\overline{q}_j$ can be interpreted as the threshold which "is defined as the highest load that will not lead to detrimental effects on the structure and functioning of an ecosystem" (Strand, 1997: 43). For acidifying compounds damaging forests, for example, the magnitude of the critical load is expressed in milliequivalents per m^2 of the forest floor per year.

Those immissions q_j^e beyond the threshold level $\overline{q}_j$ becoming effective in harming the environment are a function of emissions as we depict by way of example in ◘ Fig. 1.2 where we assume for simplicity that all emissions have the same emission coefficient α_j and where we set $e = \sum_{i=1}^{n} e_i$.

The relationship between emissions and adverse immission effects is rather uncertain and difficult to infer, which the current controversy about the environmental harmfulness of diesel cars clearly shows. This implies significant problems for environmental economics, as the choice of a certain environmental policy instrument regularly constitutes a decision under risk implying also the danger that the true causes for an environmental problem are not identified and thus the wrong measures might be taken.

1

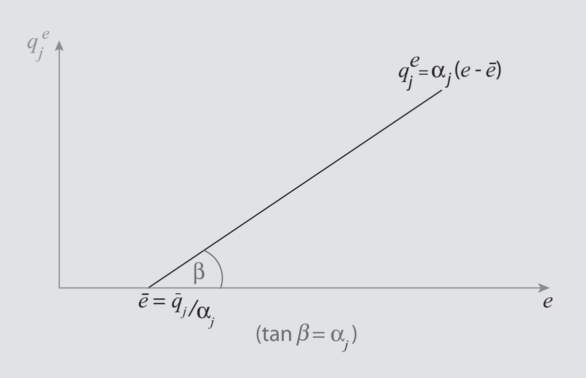


Fig. 1.2 A function of those immissions that become effective in harming the environment

Among the causes contributing to the uncertainty is that pollutants emitted into an environmental medium may interact so that synergy effects arise that are complex and not easy to grasp. An example is the reaction between nitrogen oxides (NO_x) and volatile organic compounds (VOCs) stemming from motor vehicles, causing a buildup of ozone in the presence of sunlight. One surprising observation in many urban areas is the *ozone weekend effect*. According to this phenomenon illustrating the complexity of interactions between different pollutants, ozone concentrations tend to be higher on weekends than on weekdays despite lower emission levels of NO_X and VOCs in these areas (see, e.g., Jiménez, Parra, Gassó, & Baldasano, 2005). Ozone is known of having adverse health effects, in particular the irritation of the respiratory system. Interactions between pollutants may raise their risk potential much above the level caused by an isolated pollutant.

Furthermore, many air and water pollutants are transported over a far distance and affect receptors in a hardly predictable way. Examples for such pollutants are SO_2 that causes, e.g., acid rain, and CO_2, which is the most important gas contributing to the anthropogenic greenhouse effect.

Control Question

What are emissions and immissions and what is their connection? Include an explanation of the relevance of nature's assimilation capacity!

Box 1.1: One-Source Example

Consider the example of a region where the power plant P is the only source of harmful emissions. Total emissions of P are e_P. There are different environmental media, which serve as sinks for these emissions, but we focus on forest F as a receptor. Only the part $\alpha_{PF} e_P$ of total emissions reaches this forest, while the other emissions are distributed elsewhere, e.g., because there are changing wind directions (see Fig. 1.3).

Let us assume that in the example of Fig. 1.3, the forest can absorb an input of pollutants $\bar{q}_F$ without detrimental effects on its structure and functioning as an ecosystem. Therefore, only the incoming pollutants in excess of this critical load will be harmful for this ecosystem and cause damages to the trees and the animals in the forest. The harmful immissions are $q_F^e = \alpha_{PF} e_P - \bar{q}_F = q_F - \bar{q}_F$.

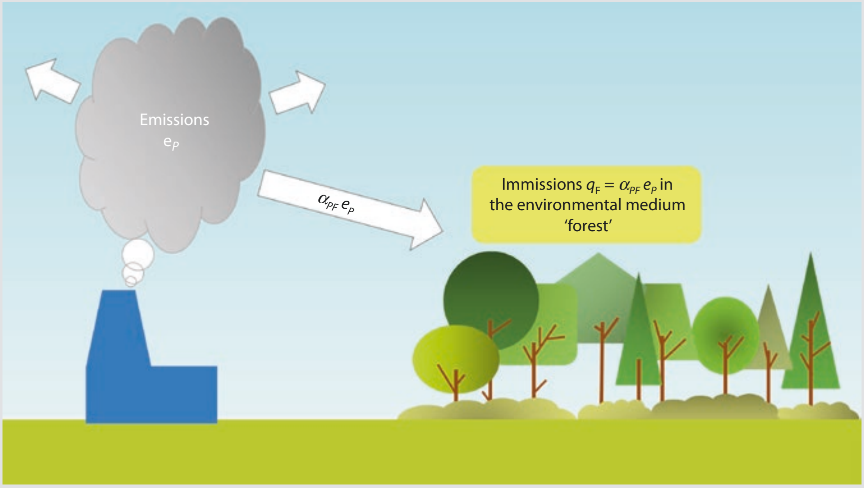


Fig. 1.3 Simplified representation of the correlation between emissions and immissions

1.2 Thesis 2

The ecological effects of economic activities provide only limited information about the need for environmental policy action. From an economist's point of view, the effects on human well-being, i.e. *on human utility, are crucial in this regard.*

These utility effects of environmental degradation can be classified as follows:

- *Direct utility effects* brought about by, e.g.:
  - Health damages (e.g., allergies or respiratory and cardiovascular diseases due to air pollution) or even premature deaths whose number amounts to 7 million per year according to a study by the World Health Organization (WHO)[3]
  - Disturbances, e.g., by malodor stemming from a chemical plant or by traffic noise
  - Impairment of recreational services provided by nature (e.g., caused by highways and wind engines)
  - Threatening of aesthetic objectives (e.g., the beauty of a landscape may be at risk due to the construction of a high-voltage power line)
  - Conflicts with ethical objectives like the desire for intergenerational fairness and for conservation of the nature (e.g., protection of species)
- *Indirect utility effects* brought about by the impairment of industrial production and the ensuing reduction of the potential to consume. These indirect utility effects are due to, e.g.:
  - Degradation of soil, water, and air, which adversely affects agricultural yields.
  - Longer absence of employees as a consequence of illnesses caused by pollution.

3 ► http://www.who.int/news-room/detail/02-05-2018-9-out-of-10-people-worldwide-breathe-polluted-air-but-more-countries-are-taking-action

- Faster depreciation of machines.
- Incurred emission abatement costs and costs of adapting to pollution. So global warming will lead to a rise of the sea levels, which necessitates the building of dams and, in the extreme, the relocation of human settlements. The production factors employed for these tasks cannot be used anymore in the production of consumption goods that would raise individuals' welfare.

From the economic perspective on environmental problems, the view reflected by Thesis 2 is highly important: *The preservation of the nature does not constitute an absolute value in itself but has to serve human interests*.

Another important insight that characterizes the economists' position toward environmental problems is that preventing environmental damages is competing with other activities that are important to improve human welfare as, e.g., individual mobility and heated rooms in winter. Like private goods, environmental protection hence must be subject to a weighing up of costs and benefits. In daily life cost-benefit comparisons are quite familiar to us with regard to private goods, e.g., when one has to decide whether to spend a given budget either on books or on cinema tickets seeking to maximize individual utility. To avoid some frequent misunderstandings, however, note that applying cost-benefit analysis to environmental problems does not imply that only tangible and egoistic objectives of individuals should be taken into consideration. Instead, individual preferences that are relevant for the assessment of environmental damages and the benefits of environmental quality improvement include ethical motivations like some feeling of responsibility for future generations. Note that paying the adequate attention to such ethical preference components is by no means at odds with the two central assumptions characterizing the *homo economicus*, i.e. rationality and self-interest (see Kirchgässner, 2008: 9 for these central assumptions).

The standard view on environmental problems prevailing among mainstream ("neoclassical") economists, however, has not remained undisputed. Rather it has been challenged by the alternative paradigm of "ecological economics" for which the preservation of ecosystems and natural cycles represents a goal in itself. From the perspective of ecological economics, nature is regarded to have an absolute intrinsic value beyond human utility considerations and preferences.

The ideal of ecological economics is that of a circular flow economy and "the very long-term health of the ecosystem, broadly defined (i.e. with human beings as part of it)" (Kolstad, 2000: 5). Ecological economics is to a large extent guided by physical objectives, e.g., by the objective of the minimization of energy use, primarily of fossil fuels. In this vein Nicholas Georgescu-Roegen (1971) as a founding father of ecological economics challenged traditional economic approaches by stressing the importance of the laws of thermodynamics, in particular the second law, the *Entropy Law*, which states: *Entropy can only be produced, it cannot be destroyed.*

Entropy is an inverse measure of the quality of energy. Every (human) activity involves the transformation of energy and with it a rise in entropy, i.e. there is a cost in terms of higher entropy or lower energy quality. As Baumgärtner (2004: 109) succinctly puts it, "with any transformation of energy or matter, an isolated system loses part of its ability to perform useful mechanical work and some of its available free energy is irreversibly transformed into heat." Such irreversible processes, however, should be avoided as much as possible in order to comply with the goal of sustainability,[4] which means the permanent maintenance of ecological-economic systems. Yet, the global system is not an isolated system, as there is, e.g., incoming solar radiation. This objective is the overarching imperative for the representatives of ecological economics for whom, consequently, very strong governmental interventions are held to be essential for coping with environmental problems. Some ecological economists even doubt whether environmental degradation can be adequately addressed in a market system at all. Therefore, it is hardly surprising that anti-globalization activists frequently sympathize with key aspects of ecological economics.

In contrast to ecological economists, neoclassical economists are more optimistic as to whether environmental problems can be adequately handled in a market system. This, however, does not mean that they expect the market mechanism alone, i.e. without external intervention, to be able to bring about solutions for environmental problems. At least in this respect, mainstream environmental economists and ecological economists agree. The shared view of both schools that the market mechanism will not provide sufficient protection for the environment is conveyed by the following thesis.

Control Question

Neoclassical environmental economics vs. ecological economics: Compare both schools of thought!

1.3 Thesis 3

The price system has a "gap" with regard to environmental pollution. In general, environmental harm is not reflected by the market mechanism. It is thus an external effect, whose neglect by the market causes welfare losses.

What is meant by a market externality can be explained by referring to the *first fundamental theorem of welfare economics*.

Definition

The first fundamental theorem of welfare economics states that – given a well-defined system of property rights – any equilibrium achieved by a competitive market with complete information of all agents involved leads to a Pareto efficient allocation of resources.

Ideally, price signals supply the individual market actors with complete and correct information about the scarcity of goods and production factors. This is in line with one of Adam Smith's key messages stating that a system of prices is coordinating the economy like an invisible hand and an individual market actor is led by this "invisible hand to promote an end, which was no part of his intention. … By pursuing his own interest he frequently promotes that of the society more effectually than when he really intends to promote it" (Smith, 1776, Book IV, Chap. 2).[5] Economists thus tend to expect a market system to be able in principle to attain a "best outcome for all," i.e. a *Pareto efficient* outcome.

4 The term "sustainability" has been coined by Hans Carl von Carlowitz (1645–1714) in a forestry context (von Carlowitz, 1713).

5 Adam Smith (1723–1790) is frequently called the "father of modern economics" (see, e.g., Sen, 1993).

Definition

An allocation is Pareto efficient if it is impossible to improve the welfare of any agent without making at least one individual worse off.

Control Question

What does the first fundamental theorem of welfare economics say? Refer to the notion "Pareto efficiency" in your explanation.

Yet, as already Adam Smith himself has remarked, the preconditions for a perfectly functioning market and thus for the applicability of the first fundamental theorem of welfare economics usually are not fulfilled in reality. Instead, the occurrence of various external effects prevents a market system from attaining Pareto efficiency.

Definition

Externalities are the effects of an individual's production or consumption activities on another party that are not compensated for through the price system.

Since environmental pollution represents the prototypical example for an externality, prices regularly fail to signal scarcity of environmental goods and services correctly. Pollution caused by polluter P and harming victim V is costless for P, so that the associated adverse effects inflicted on V are playing no role in P's decision-making. Consequently, a misallocation and welfare losses result, which are entirely at the expense of agent V. We face a market failure that will prevail as long as the adverse external effects are not internalized.

Control Question

What in general is a "market failure"? Why does environmental pollution in particular represent an example for a market failure?

Box 1.2: Natural Resource Economics

An exception to this fundamental thesis of market failure in the context of using services provided by nature is the use of privately owned natural assets. Besides land, these are natural resources like crude oil or minerals. Because these resources – unlike air in the troposphere – are not freely accessible, market prices exist for them, and the market prices tend to induce a somewhat economical use of these resources. However, it is rather uncertain whether the prices will actually reflect the long-term scarcity of the respective resources. Also in this context, market failure may arise (especially due to the lack of adequate future markets) and forecasts, on which suppliers and purchasers will base their decisions, are frequently incorrect. Another reason for misallocation on resource markets is given by market power on the suppliers' side that tends to bring about inefficiently high resource prices. Such market power or dominance has for a long time been observed with respect to crude oil and in the last few years concerning rare earth metals with China as dominant player on the world market (Massari & Ruberti, 2013). Analyzing the functioning of resource markets is the major topic in natural resource economics. Such questions will only play a minor role in this textbook on environmental economics.[6] In this book, we will instead concentrate on issues that are directly connected with environmental pollution and the external effects associated with it.

Conclusion

In this introductory chapter, we highlighted amongst others especially by means of historical examples the important role environmental problems have been playing for a long time. We discussed why, especially in the past 50 years, environmental protection has become a significant issue also for economics, what the perspective of the mainstream neoclassical environmental economics is, and what makes this dominant school distinct from ecological economics. This chapter also introduced and explained some standard economic concepts (like Pareto efficiency, market failure, and external effects), which are highly relevant in environmental economics and will thus play a central role in this textbook.

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Environmental Externalities and Their Internalization Through Voluntary Approaches

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2

Objectives of This Chapter

In this chapter students should learn:

- How the optimal pollution level can be determined in an elementary model in which the emissions of a single polluter harm a single "victim"
- How in this model of a unilateral externality bargaining between the polluter and the victim might be able to internalize the externality
- Which obstacles will prevent the success of this Coasean bargaining in reality
- Which other voluntary approaches for an internalization of environmental externalities exist
- How the situation changes in case of a multilateral externality where several polluters harm each other

In ► Sect. 2.1 of this chapter, we first of all consider how environmental pollution reduces social welfare in a laissez-faire situation, i.e. if no internalization of external effects takes place. Subsequently, in ► Sect. 2.2, we describe different approaches to internalize externalities by voluntary actions of the agents involved so that the government only plays a minor role. These decentralized and in a certain sense market-like approaches for the internalization of environmental externalities are Coasean bargaining, matching, and the establishment of social norms. Approaches with a more active government like environmental standards, taxes, or emissions trading schemes will be dealt with more thoroughly later on in ► Chap. 4.

In ► Sect. 2.2, we focus on the case of unidirectional externalities in which one polluter P is confronted with other agents, the "victims" V, who are adversely affected by P's emission. Yet, in reality, such a clear distinction between polluters and victims cannot be made throughout as polluters also frequently suffer, at least to some degree, from their own and other agents' emissions. In this case, where a harmful environmental externality hence is not unidirectional but takes effect in all directions, pollution turns into a "public bad" so that mitigation of this pollution becomes a "public good." What is exactly meant by this notion and what consequences result for voluntary internalization approaches will be briefly discussed in the Appendix to this chapter. Public goods will play a prominent role in ► Chap. 5 of this book where international environmental problems are considered.

2.1 The Socially Optimal Pollution Level in Case of a Unilateral Externality

In our basic model of a downstream environmental externality, which is produced by one agent and only causes harm to a single other agent, the level of a polluting activity is denoted x. This activity could be associated with production or consumption. The polluter P can decide about the level of this polluting activity. By choosing the level x, P achieves a net benefit (or profit) $B(x)$. The associated marginal benefit function $B'(x)$ is declining in x, i.e. formally it holds that $B''(x) < 0$. At some value of $\bar{x}$, we get $B'(\bar{x}) = 0$, i.e. the marginal benefit becomes zero, and at this $\bar{x}$, the polluter reaches his individual benefit maximum. If there is no governmental intervention that modifies P's optimization problem, he will choose the laissez-faire activity level $\overline{x}$.

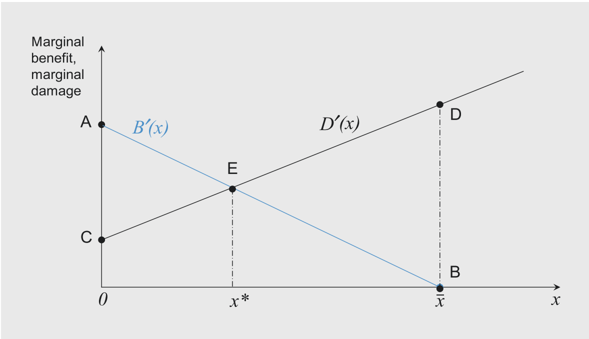
In many practical applications, the polluter P is a price-taking firm (e.g., a power plant that releases heated cooling water into a river or a pulp and paper mill whose wastewater pollutes a lake). Then, the activity level x can be interpreted as the firm's output level. In this case P's profit function is $B(x) = px - C(x)$ where p is the market price of the produced good and $C(x)$ is the firm's cost function. The decline of marginal profit in x follows from the standard assumption that $C''(x) > 0$, i.e. the assumption of increasing marginal cost of production.

The production of x is associated with an emission level of $e(x)$. To facilitate the exposition, we make the assumption that the emission level is proportional to the production or activity level; in particular we assume that $e(x) = x$.

The negative externality occurs as the emissions $e(x) = x$ cause damage to the victim V, which is described by the damage function $D(x)$. In the case of coal-fired power plants, for example, their sulfur dioxide (SO_2) emissions create acid rain, which – in turn – causes a decline in yields from crops and fisheries. Thus, the victims would be farmers and fishermen.

The marginal damage function $D'(x)$ is assumed to be increasing or at least non-declining in x, i.e. $D''(x) \geq 0$ holds. The socially optimal production level x^* is found by maximizing the social net benefit $B(x) - D(x)$, which yields the first-order condition $B'(x^*) = D'(x^*)$. The second-order condition is satisfied, too, as $B''(x) - D''(x) < 0$ holds for all $x > 0$. In ◘ Fig. 2.1, the point E, at which the marginal benefit or profit function and the marginal damage function intersect, denotes the social optimum where aggregate welfare is maximized. The maximum social net surplus, which is attained at E, is then represented by the area of the triangle CEA. Since obviously $\overline{x} > x^*$, the neglect of the negative externality in P's production decision leads – from a social welfare point of view – to a suboptimally high production level $\overline{x}$. ◘ Figure 2.1 illustrates this situation for the specific case of linear marginal damage and benefit or profit curves.

In comparison with the social optimum, the total welfare loss of P's unregulated production is measured by the area $EBD = D(\overline{x}) - D(x^*) - (B(\overline{x}) - B(x^*))$ in ◘ Fig. 2.1. The inefficiency arises because for all activity levels between x^* and $\overline{x}$, the external marginal damage of production exceeds the marginal profit or benefit. In principle, the prevention of this welfare loss would be possible in different ways.



◘ **Fig. 2.1** Marginal damage and benefit functions and the socially optimal production level x^*

Control Question

How does welfare of P and V differ between the laissez-faire outcome (at $\bar{x}$) and the socially optimal solution (at x^*)?

An internalization of external effects could clearly be brought about by merging P and V. Then, the social welfare function or aggregate profit function $B(x) - D(x)$ would become the objective function for both partners of this joint venture. Such an approach, however, is virtually impossible in most empirically relevant cases. Below we outline the basic modes of action for the most prominent internalization options.

2.2 Ways to Internalize Externalities

2.2.1 The Coase Theorem

2.2.1.1 Pigou's Call for the State as the Starting Point

When the market fails, it seems straightforward to call for action by the government. Since Arthur Cecil Pigou (1877–1959) who can be considered as the founding father of environmental economics, this has been the economists' standard approach to environmental problems.

In our simple model, this could mean that the government simply stipulates an upper limit for tolerated emissions, which clearly must be x^* to maximize aggregate welfare. If emissions exceed this threshold level, sanctions will be imposed on the polluter P. The fines for the misconducting polluter P must be so high that he prefers to refrain from the breach of the rules. More in line with the logic of the market is the application of an environmental tax, which in the true sense of the word closes the gap in the price system that is left by the externality. If the tax rate (= fee on every unit of x) is chosen as $t^* = D'(x^*)(=B'(x^*))$, i.e. as the level of marginal damage in the social optimum, P's profit function changes to

$$B(x) - t^* x \to max!$$

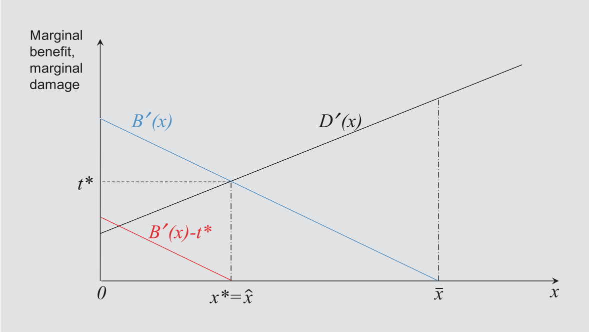
The first-order condition that characterizes P's benefit maximizing activity level $\hat{x}$ thus becomes $B'(\hat{x}) - t^* = 0$, which implies $B'(\hat{x}) = t^* = D'(x^*)$ and thus $\hat{x} = x^*$. This shows that by means of a "correctly" chosen emission tax, the socially optimal activity level $\hat{x} = x^*$ is attained (see ■ Fig. 2.2). Such a levy with a tax rate equal to the marginal damage in the optimum is called a *Pigouvian tax*.

A proportional subsidy on emission reductions below the laissez-faire level works in an analogous way. If the subsidy rate is $s^* = D'(x^*) = t^*$, the new objective function of P becomes $B(x) + s^*(\bar{x} - x)$ which leads to the same first-order condition as in the case of the Pigouvian tax. Hence, such a subsidy will also bring about the social optimum.

Control Question

Why can emission taxes and abatement subsidies have the same effect on the abatement level? How do these instruments change polluter P's welfare?

Fig. 2.2 The Pigouvian tax rate t^*



Box 2.1: The Case of Two Firms

For the sake of completeness, we will now consider also the case where not only P, i.e. the polluter, but also V, i.e. the victim, is a firm that produces a specific good. The output level of the good produced by V is y and its market price is v. V's production cost function is denoted $k(y,x)$, where $k_2 = \frac{\partial k(y,x)}{\partial x} > 0$ is supposed to hold, i.e. an increasing emission level (which is generated by P only and which is equivalent to P's rising production level) causes growing production costs for V. In this way, P exerts – by raising V's production cost – a negative external effect on V. Therefore, V has to adapt his output level to changes in the pollution level generated by P. The respective reaction function for V indicates how V will maximize his profit $vy - k(y,x)$ by choosing – for a given x – his own output level $y(x)$. The first-order condition characterizing $y(x)$ is $v = k_1(y(x),x)$, where k_1 denotes the partial derivative $\frac{\partial k}{\partial y}$, i.e. the marginal change of V's cost of production in response to an increase of its own output level.

The damage that P causes to V consists of forgone profits for V, i.e. it holds

$$D(x) = vy(0) - k\left(y(0),0\right) - \left(vy(x) - k\left(y(x),x\right)\right).$$

For the marginal damage, we obtain

$$D'(x) = -vy'(x) + k_1\left(y(x),x\right)y'(x) + k_2\left(y(x),x\right) = k_2\left(y(x),x\right) > 0$$

since $v = k_1(y(x),x)$ implies that the first two summands of the term $D'(x)$ cancel each other out.

Under plausible assumptions, it can be demonstrated that $D''(x) \geq 0$ also holds for this case. Therefore, the considerations in our basic model in which only P is a firm can easily be transferred to the slightly more complex case in which the victim V also is a price-taking (but non-polluting) firm.

2.2.1.2 The Basic Idea of the Coasean Approach

The decentralized internalization approach, which we will present now, rests upon a general idea that is fundamental to microeconomic thinking: Suppose that not all possibilities to attain utility gains are exploited in an interaction between two agents because their activities are not well coordinated. One agent is only skilled in baking bread, while the other one is talented only in brewing beer. However, both agents would like to consume some of the two produced goods, and consequently one agent lacks beer and the other one lacks bread. In a market economy, this coordination problem is solved by the exchange of goods. The exchange of goods and production factors on the market allows for exploiting benefits of specialization and division of labor and will bring about a *Pareto improvement*, which means that at least one agent is made better off without making anyone else worse off.

Against this background, the question arises as to why not also use such a market-based approach for the internalization of external effects. Without sufficient consideration of this option, the claim for governmental interventions in environmental protection seems to be at least premature. Such a differentiated view has been expressed by the 1991 Nobel laureate Ronald Coase (1960) in his famous essay *The Problem of Social Cost*, where the statement has been made that voluntary interaction between the agents involved may bring about an efficient internalization of externalities. From the perspective of this *Coase Theorem*, governmental intervention with the aim of improving environmental quality may be unnecessary as the agents can attain an efficient allocation by themselves through bargaining, which makes the exchange principle "performance and consideration" useful also for the solution of environmental problems. This has been an important reason why the Coasean approach is of much appeal and interest for environmental economists (Medema, 2014). Yet, it is not clear in advance how such bargaining processes take place and how they can be captured in the framework of an economic model. The subsequent depiction of the Coasean approach applies an elementary bargaining model, which will now be presented.

2.2.1.3 The Description of a Simple Coasean Bargaining Process

Suppose that there is no environmental policy in the beginning. This means that we face a laissez-faire system in which polluter P has the full property right regarding the environment. This implies that P is allowed to choose his activity level x without any limitations and he does so in a way that maximizes his utility, i.e. he chooses the laissez-faire level $\bar{x}$. In our simple two-stage bargaining game (see Schweizer, 1988), the victim V now wants to motivate P to alter his action by submitting the following offer to P at the first stage of our elementary bargaining game: If P is willing to reduce his activities to a level $x < \bar{x}$, he will – in exchange for this service – make the payment $Z(x)$ to P.

In the first instance, the level of this payment $Z(x)$ is undetermined. The minimum payment $Z_{min}(x)$, which is necessary to induce P to accept the offer at the second stage of the game, is equal to

$$Z_{min}(x) = B(\bar{x}) - B(x).$$

This follows, as P obviously will not accept any offer that would make him worse off. Hence, he will claim at least a compensation for the utility loss that is caused by the reduction of his activities to the level x.

The maximum payment that V is willing to pay for a reduction of P's activities to x is

$$Z_{max}(x) = D(\bar{x}) - D(x),$$

i.e. V will not pay an amount that exceeds the decrease in his damage.

In our stylized depiction of bargaining, P can either accept the offer submitted by V or he can reject it at the second stage. There are no other options for P. In this case, a utility maximizing P will accept any offer that makes him at least as well-off as he was in the laissez-faire situation. In such a take-it-or-leave-it ("tioli") offer, V can limit his offered payment in exchange for P's lowering of his activity level x to $Z_{min}(x)$.

The optimal offer of x from V's point of view can therefore be determined by maximizing his saved net costs, i.e. by maximizing damage reduction minus payments to P

$$D(\bar{x})-D(x)-Z_{min}(x)=D(\bar{x})-D(x)-\left(B(\bar{x})-B(x)\right)\rightarrow max!$$

The first-order condition that characterizes the individually efficient solution $\hat{x}$ for V is $D'(\hat{x})=B'(\hat{x})$, which yields $\hat{x}=x^*$. (The second-order condition again is satisfied because of $B''(x)-D''(x)<0$.) Consequently, the negotiating partners attain the socially optimal solution by means of the described bargaining process.

Likewise, polluter P could submit an offer to V. By doing so, P claims a payment of $\tilde{Z}(x)$ from V in exchange for lowering his activity level to some $x<\bar{x}$. The bargaining process will proceed in a completely analogous way as the one where V submitted the offer, and again the socially optimal activity level x^* can be attained.

The distribution of welfare gains from achieving the socially optimal bargaining solution depends on which agent submits the offer and thus, in a certain sense, is the leader of the bargaining game. If it is the victim V who submits the tioli offer, then he improves his well-being relative to the initial solution $\bar{x}$ by

$$B\left(x^*\right)-D\left(x^*\right)-\left(B(\bar{x})-D(\bar{x})\right).$$

In this case, V receives the total welfare gain that is generated through the bargaining. If, in contrast, it is the polluter P who submits the tioli offer, then P can usurp the whole welfare gain. Consequently, the agent who acts first by submitting a tioli offer benefits the most from the bargaining.

Control Question

Why do the gains from bargaining fall to the agent who makes the take-it-or-leave-it offer?

However, bargaining will also bring about an optimal internalization of external effects when the welfare gains are distributed more equally between P and V. Let us assume that in the laissez-faire system, V offers P the following payment in exchange for a reduction of P's activity level to x

$$Z_\alpha(x)=B(\bar{x})-B(x)+\alpha\left(D(\bar{x})-D(x)-\left(B(\bar{x})-B(x)\right)\right).$$

Here α stands for an exogenously given distribution parameter which indicates the share of the net welfare gains V leaves to P in this alternative bargaining scenario – additional to the compensation $B(\bar{x})-B(x)\left(=Z_{min}(x)\right)$ for P's benefit forgone.

In order to determine the optimal offer for V in this case, V again maximizes the net utility he gains relative to the initial situation where the activity level was $\bar{x}$. V's net utility gain is

$$D(\bar{x}) - D(x) - Z_\alpha(x).$$

This leads us to the following first-order condition

$$(1-\alpha)B'(\hat{x}) = (1-\alpha)D'(\hat{x}),$$

which again gives $\hat{x} = x^*$.

The perfect internalization of externalities by means of bargaining is also attained when the underlying legal system is not a laissez-faire type but when the polluter-pays principle is applied instead so that the victim V has the right not to be polluted at all. This implies that polluter P's activity level will initially be $x = 0$, but also in this case the socially optimal activity x^* can finally be attained through Coasean bargaining.

Control Question

How does the bargaining process under the polluter-pays principle, i.e. if P has no right to pollute, look like?

It is a key insight gained by Coase that the efficient outcome x^* arises independently of the legal positions that are assigned to the two agents, i.e. it plays no role whether we face a laissez-faire system or a system in which the polluter-pays principle applies. However, depending on the legal position, significant differences in the distributional effects occur.

Box 2.2: Bargaining in the Laissez-Faire Scenario with a Constant Compensation Rate for the Polluter

As a further variation of the bargaining process, let us consider the laissez-faire system where the victim V now offers a *constant* compensation c for each emission unit that polluter P abates. If V wants P to restrict his activity to some level $x < \bar{x}$, he thus offers to pay $c(\bar{x} - x)$ to P. On the one hand, then V's net gain as compared to the laissez-faire outcome becomes

$$D(\bar{x}) - D(x) - c(\bar{x} - x).$$

V's net gain is increasing in P's emission reduction $\bar{x} - x$ as long as $D'(x) > c$. On the other hand, P's net gain is

$$c(\bar{x} - x) - (B(\bar{x}) - B(x)).$$

P's net gain is increasing in his emission reductions as long as $B'(x) < c$.

We now assume that V chooses the compensation rate $c^* = B'(x^*) = D'(x^*)$. Then V maximizes his net gain by making the offer $\hat{x} = x^*$, which follows from taking the derivative of V's net gain and the ensuing first-order condition $-D'(\hat{x}) + c^* = -D'(\hat{x}) + D'(x^*) = 0$. The polluter will be willing to accept this offer since his net gain is increasing when in the range between x^* and $\bar{x}$, the victim V proposes a lower level of x. In this bargaining scenario, also some division of the aggregate welfare gains between the two agents P and V results (see ◘ Fig. 2.3).

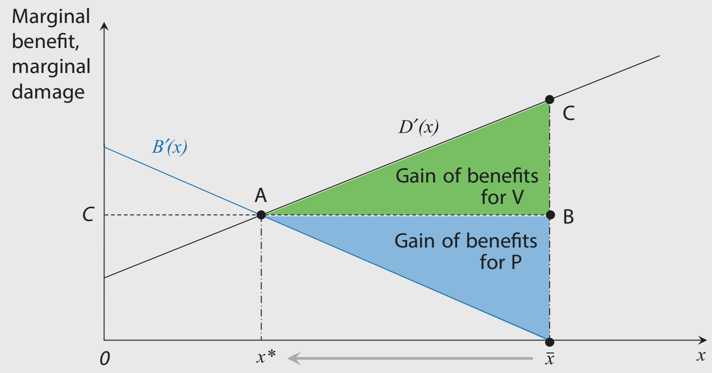


Fig. 2.3 Additional benefits generated via the bargaining scheme

The above description of the bargaining process between polluter and victim has been based on simplifying assumptions. The reasons why the Coasean bargaining approach will under realistic conditions not attain an optimal solution and hence cannot be relied upon as an instrument for solving environmental problems will be discussed in the subsequent sections.

2.2.1.4 Objections Against the Coase Theorem

Transaction Costs

In general, bargaining over the internalization of environmental externalities is different from market transactions guided by the price system – and a lot more intricate. The price system is an efficient mechanism to coordinate the exchange of goods and services, which economizes on the amount of information the market participants need to have and is associated with low transaction costs. Therefore, it is not legitimate from the outset to conclude that the advantages of the market system will prevail in the same way for Coasean bargaining – even though this bargaining is also based on the principle of exchange. Bargaining processes tend to involve considerable transaction cost (on such costs also see Endres, 2010: 43–49), especially because the information that is required for attaining an efficient allocation is not reflected by market prices, but must be acquired on a case-by-case basis. Due to the necessity of acquiring information, e.g., by means of scientific expertise to assess the level of environmental damages, additional costs arise, e.g., for fees payable to the experts, which clearly reduce the welfare gains that can be obtained through bargaining. Furthermore, due to the time requirements for information search, time delays in reaching a bargaining solution are to be expected so that the realization of bargaining benefits is postponed what causes further welfare losses.

We now provide an extreme example of fixed transaction costs that are prohibitively high and exceed the potential gains from bargaining. In Fig. 2.4, let the fixed costs of bargaining TC_P be measured by the area of the rectangle on the left-hand side and the potential gains from bargaining be given by the area of the triangle CAE. We suppose that $TC_P > CAE$ holds and that we are in the polluter-pays scenario.

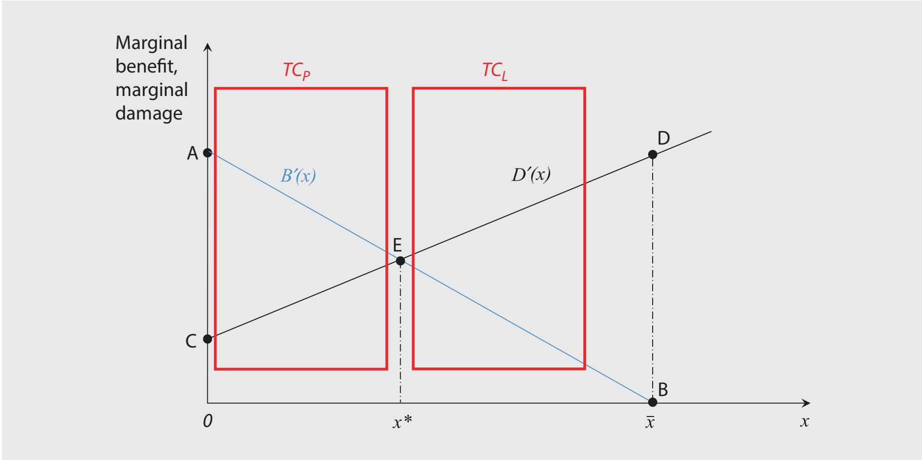


Fig. 2.4 Prohibitively high fixed transaction costs preventing Coasean bargaining

When the laissez-faire principle applies instead, transaction costs are given by TC_L, i.e. by the area of the rectangle on the right-hand side. The potential gains from bargaining are represented by the area of the triangle *EDB* in this case, and we suppose $TC_L > EDB$.

Control Question
What happens if $TC_P \leq CAE$ and $TC_L \leq EDB$?

This example just illustrates a situation in which Coasean bargaining is not able to bring about a socially efficient outcome – irrespective of whether the polluter-pays or the laissez-faire scenario applies. This negative result holds because transaction costs are exceeding the potential welfare gains that can be realized in the optimal pollution level x^*.

Against the background of the difficulties discussed above, it is conceivable that uniform environmental regulations by the government may work more efficiently to internalize externalities than the Coasean bargaining approach. Moreover, the bargaining outcome in complex situations is difficult to predict, which increases firms' internal transaction costs and investment risks and thus impedes long-term investment planning of firms. In contrast, governmental regulations are more transparent and reliable and therefore help to stabilize the economic agents' expectations, which is favorable for investment activities and thus for economic growth and employment. Increased uncertainty also motivates polluters to prefer flexible technological solutions especially for abatement measures allowing for adjustments to unexpected developments. These flexible technologies, however, may ex post not be the most efficient ones. Finally, uniform standards set by the government will ensure a level playing field for companies and avoid strategic abuse of Coasean bargaining, so that environmental regulation by the government might be in the interest of polluting firms.

Asymmetric Information Between Polluter and Victim

The attainment of efficient bargaining outcomes is generally impeded by incomplete information of agents about the opponent's costs and benefits (see Farrell, 1987;

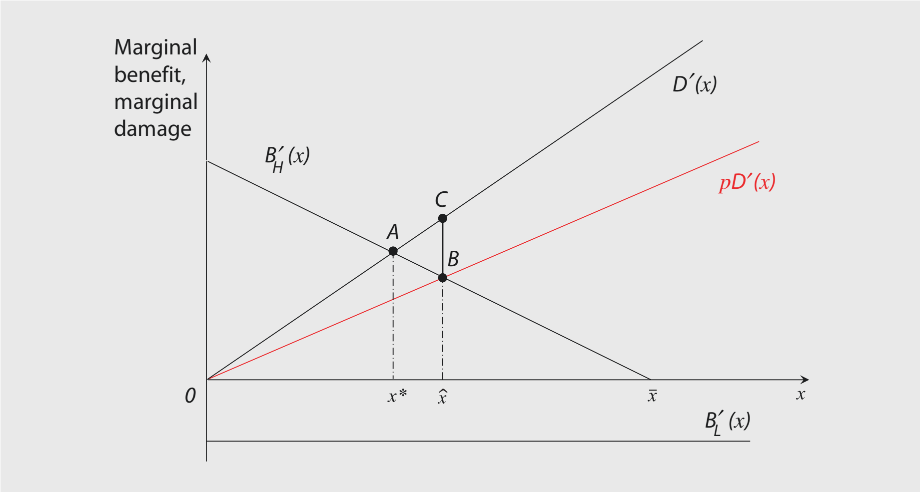


Fig. 2.5 The outcome of Coasean bargaining under asymmetric information

Buchholz & Haslbeck, 1991/92; Illing, 1992; Demougin & Illing, 1993). In case of asymmetric information, it becomes possible to cheat and thus to usurp benefits at the other agent's expense. How this affects the outcome of Coasean bargaining over environmental externalities is discussed next.

Let us consider a laissez-faire system in which the victim is not fully informed about polluter P's marginal benefit (profit) function $B'(x)$. V does only know that with probability of p, polluter P's marginal benefit curve is the declining curve as assumed in the preceding sections (see Fig. 2.5.). In this case, P is said to be of the H-type, where H stands for a high marginal benefit for P. If there is no governmental environmental policy, such a polluter P will clearly choose the activity level $\bar{x}$, at which the marginal benefit curve is cutting the x-axis. However, with the probability $1 - p$, the marginal benefits of P are everywhere zero or even negative. In this case, P has no incentive to choose any positive activity level. Then, P is said to be of the L-type, where L stands for a low marginal benefit.

Let us first consider the benchmark case in which V is, in the laissez-faire scenario, informed about whether P is of the H- or L-type, so that he is able to condition his offers to the type of P. Then he clearly would not offer to pay anything to an L-type, as such a type would not harm him anyway. P's activity level thus will be zero. Given an H-type, however, V's offer would lead to a reduction of P's activity level to x^* as described in the treatment of Coasean bargaining above. This differentiation of offers between L- and H-types leads to the first-best outcome.

Yet, in the scenario with asymmetric information in which V does not know whether his counterpart P is of the H- or the L-type, such a differentiation of offers is no longer possible. Instead V will have to offer the amount $B(\bar{x}) - B(x)$ in any case if he wants to make sure that he will induce the H-type polluter to limit his activity to some level $x < \bar{x}$. The reason is that a strategically acting L-type polluter will not be willing to reveal his true type, i.e. that any positive activity level would not pay off for him at all, and rather will pretend to be an H-type. Otherwise, he would give away the payment $B(\bar{x}) - B(x)$ that he could have easily obtained by cheating about his true L-type characteristic. The payment $B(\bar{x}) - B(x)$ is an *information rent* for the L-type polluter which is received for his false pretense.

Asymmetric information then distorts V's decision about submitting an offer to P in the following way: The payment $B(\bar{x}) - B(x)$ to P only brings about a decline in V's damage costs if the polluter is of the H-type. In the case of an L-type, the payment has no effect as an L-type always chooses the activity level zero causing no harm to V. When we assume for simplicity that V is risk neutral, his objective function then becomes

$$p\left(D(\bar{x}) - D(x)\right) + (1-p)0 - \left(B(\bar{x}) - B(x)\right).$$

Thus, in order to maximize his bargaining gain, V would choose his activity level $\hat{x}$ that satisfies the following first-order condition

$$B'(\hat{x}) = pD'(\hat{x}).$$

Since $pD'(x) < D'(x)$, the activity level $\hat{x}$ exceeds the optimal level x^* that would have arisen from bargaining with an H-type polluter under complete information. ◘ Figure 2.5 illustrates this case.

The activity level $\hat{x}$ resulting from such bargaining is the higher, the lower the probability p that the polluter is of the H-type. The less likely the occurrence of an H-type, the less worthwhile V considers his payments to be, as they only reduce V's environmental damages when he indeed faces an H-type polluter.

In comparison with the first-best solution, the degree of internalization is suboptimally low in case of asymmetric information, i.e. pollution will be suboptimally high. The loss of surplus in comparison with the first-best solution when P is of an H-type is described by the area of the triangle *ABC* in ◘ Fig. 2.5. The expected value of the welfare loss due to asymmetric information therefore is $p \cdot ABC$.

? Control Question

How does the bargaining outcome change if the conjectural probability p that P is of an H-type increases?

Strategic Manipulation of Property Rights: An Extortion Game

Even when a state of complete and symmetric information prevails, incentives to act strategically in bargaining may exist. These incentives can be attributed to the fact that – depending on his legal position – at least one of the negotiating partners has the chance to manipulate the starting point of the bargaining in his own favor and thus to "extort" the other partner. (This controversial issue was early raised by Kneese, 1964 and Wellisz, 1964. Also see, e.g., Schlicht, 1996, Buchholz & Haslbeck, 1997, and Medema, 2015). In order to describe how this extortion can be brought about, we suppose that the polluter P has the choice between two different technologies T_1 and T_2, which represent two production technologies in case the polluter is a company, for pursuing his activity. The respective marginal benefit curves B_1' and B_2' have the shape as depicted in ◘ Fig. 2.6.

Again, the laissez-faire principle is assumed to apply, but now it is P who makes the offers. We next ask how P will act in this situation if bargaining with V either is excluded or is possible.

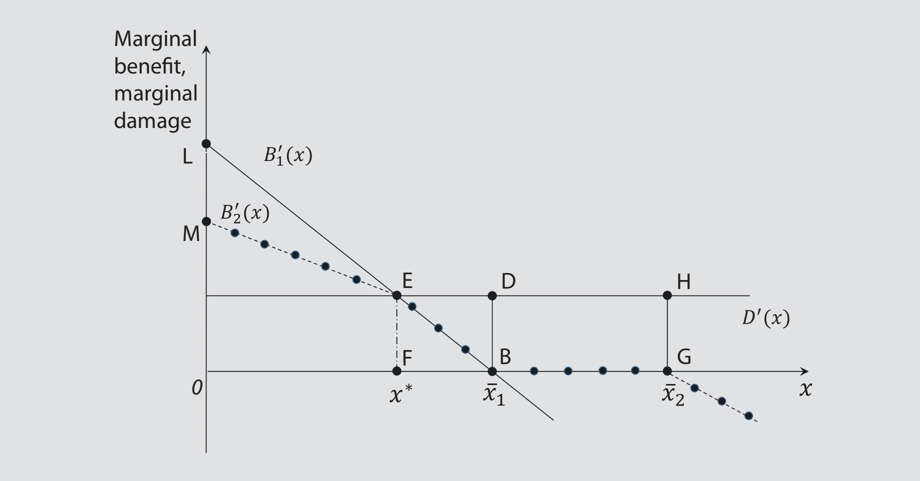


Fig. 2.6 An extortion technology

Without the option of bargaining, P would choose technology T_1 and then $\bar{x}_1$ as his activity level since if he chose T_2 instead, he would lose the surplus as measured by the area of the triangle *LME*.

In contrast, if bargaining is an option and P anticipates the outcome of bargaining, P will possibly choose the less efficient technology T_2. The choice of this technology then is due to the strategic motive that the application of the economically disadvantageous technology T_2 allows for an extension of his activity level from $\bar{x}_1$ to $\bar{x}_2$ without a loss in surplus. With technology T_1, however, such a threat would not be credible because P's benefit function $B_1'(x)$ becomes negative beyond the activity level $\bar{x}_1$ so that any increase of activity would make P worse off. Yet, if P has chosen technology T_2, the victim V becomes willing to pay an additional amount as measured by the rectangular *FGHE* for P's activity reduction from $\bar{x}_2$ to the level x^*. In comparison with the case where T_1 is chosen, P's surplus will thus change by *BGHD* – *LME* through the choice of T_2, which is the difference between the gain in P's surplus *BGHD* resulting from his improved bargaining position after the choice of T_2 and the decline in P's surplus *LME* due to the technological disadvantage resulting from the choice of T_2 instead of T_1.

If *BGHD* > *LME*, it thus follows that P will choose technology T_2 for purely strategic motives in order to extort an additional surplus from V. This has consequences for the aggregate social welfare and its distribution among P and V:

The choice of T_2 is associated with a decline in social welfare relative to the first-best solution since $B_2'(x)$ lies below $B_1'(x)$ up to the socially optimal activity level x^* so that T_2 is economically unfavorable as compared to T_1. This welfare loss is measured by the area of the triangle *LME* in Fig. 2.6.

The creation of bargaining options and the ensuing choice of technology T_2 brings about a decline in V's surplus by *BGHD* relative to the status quo without bargaining with the activity level $\bar{x}_1$ and a redistribution of wealth from V to P. Consequently, bargaining does not achieve a Pareto improvement, which would require that no agent is made worse off. If in addition to $BGHD > LME$ also $BDE < LME$ holds, bargaining will even result in a decline in aggregate social welfare relative to the status quo in which no attempts at internalizing the externality are made and in which technology T_1 is chosen and the activity level $\bar{x}_1$ is realized. The reason for this unexpected outcome is that the loss of welfare due to the lacking internalization of externalities is smaller than the welfare loss due to the application of the unfavorable technology T_2. This result contradicts the economic intuition according to which one would expect that bargaining always brings about a welfare improvement (or at least no decline).

? Control Question

How does the outcome of the extortion game change when the point G (i.e. $\bar{x}_2$) in ◘ Fig. 2.6 is shifted to the right?

How can the seemingly paradoxical result be explained that the enabling of bargaining entails a welfare loss? It is well-known that exchange processes require the assignment of secure property rights to the traded goods. In the case of private goods, the initial endowments are clearly limited in a natural way by the available amount of goods that are either in possession of the trading partners or that are producible with given scarce resources. In the case of Coasean bargaining considered above, things are quite different. The "good" that P offers to V are emission reductions. In a pure laissez-faire system, polluter P can extend his initial endowment arbitrarily by reducing V's use of the good "clean environment." In the real-life context, one would simply call such a procedure a "theft." The strategic manipulation of property rights, which drives the results in this chapter, is questionable from an ethical viewpoint but may also be economically inefficient causing a welfare decline as shown by our theoretical argument.

To put it differently: The laissez-faire system in Coasean bargaining over environmental pollution does not rest upon a solid system of property rights, but actually is granting the polluter the right for an even unlimited appropriation of the scarce good "clean environment." This clearly shows why the Coasean bargaining model with laissez-faire system can by no means be considered as equivalent to the conventional exchange model with fixed property rights.

The polluter's right to expand his production indefinitely would in addition allow P to exploit V to the extent that V might finally be fully impoverished. If V anticipates such an extortion, he will have no incentive anymore to generate property when this property is in danger of being taken away. This confirms that also in the context of Coasean bargaining, solid property rights are a crucial precondition for welfare-enhancing capital accumulation and economic growth.

? Control Question

What does securing property rights mean for Coasean bargaining?

These arguments clearly show that it is wrong to believe that efficient environmental protection can be attained solely by bargaining and without governmental regulation, e.g., by setting some upper limit for the emissions that are permitted for P. Without such restrictions, bargaining can even bring about a welfare loss.

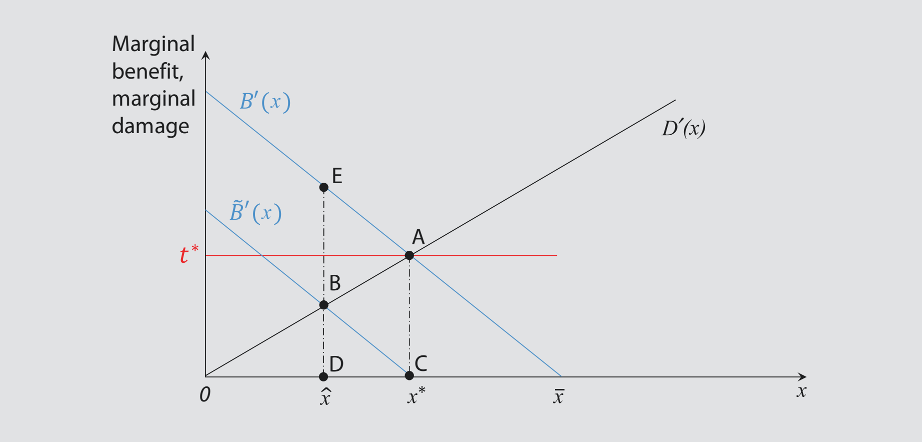


Fig. 2.7 Marginal benefit functions with and without the Pigouvian tax

Incompatibility of Pigouvian Taxes and Bargaining

If the Pigouvian tax is levied with the Pigouvian tax rate t^*, polluter P's marginal benefit function will shift in parallel downward, i.e. it becomes $\tilde{B}'(x) = B'(x) - t^*$. Again we suppose that a laissez-faire system applies in which V submits an offer to P. Because of the tax-induced reduction of P's marginal net benefits, the payments required to compensate P for his loss in benefits also decline. A tioli offer submitted by V will therefore bring about an activity level $\hat{x}$ that is determined by $\tilde{B}'(\hat{x}) = D'(\hat{x})$ and that is thus lower than the optimal level x^*. As Fig. 2.7 illustrates, V will pay an amount equal to the area of the triangle *BDC* to polluter P. The social welfare loss resulting from the incompatibility of both environmental policy instruments is equal to the area *AEB*. Hence, from the point of view of economic welfare, it does not hold that "two are better than one."

Control Question

What bargaining outcome will result when the emission tax rate is smaller than the Pigouvian tax rate?

Free Riding

Until now we assumed that there is only a single victim, which however is not the case for most empirically relevant problems of environmental pollution. Yet, if there is more than one victim, the conditions for Coasean bargaining change considerably (see Ellingsen & Paltseva, 2016, as a recent contribution). For an explanation we assume that the polluter P is a steelworks whose profit function exhibits a constant marginal benefit b up to the production level $\bar{x}$, while marginal benefits become negative beyond $\bar{x}$. Let the original victim V be the owner of an orchard whose fruit crop yields are impaired by the pollution of the steelworks. The marginal damage for V is also assumed to be constant and has the value $d > 0$. Furthermore, it holds that $d > b$, but $\frac{d}{2} < b$. In the laissez-faire system, P's initial production level is $\bar{x}$ as depicted in Fig. 2.8. Through

2

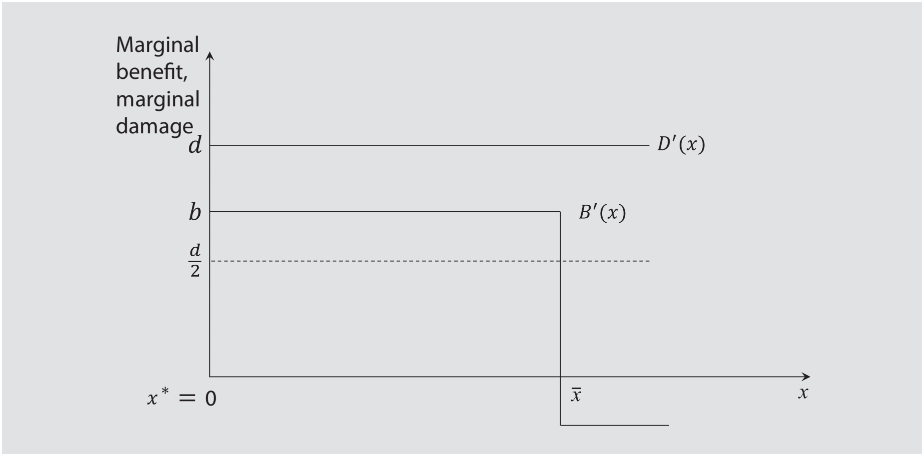


Fig. 2.8 Graphical illustration of a case with free-rider incentives

Coasean bargaining with tioli offers by V to P, we clearly get a bargaining outcome with the activity level $x^* = 0$, i.e. P's steel production entirely ceases, P is fully compensated for his profit loss $b\overline{x}$, and V's net advantage is $(d-b)\overline{x}$.

Let us now suppose that the owner of the orchard passes away and that his two daughters V_1 and V_2 inherit the orchard in equal parts. The problem, however, is that both sisters do not get along well and therefore are not able to act cooperatively when making Coasean offers to the steelworks P. This implies that each of them will submit her tioli offers to P individually. In line with Nash behavior, each sister V_1 and V_2 will adapt her own offer in an optimal way to the offer of the other sister. Under the simple assumptions we have made, the determination of these optimal responses is quite easy. So, V_2 will always have to pay $Z_2 = b\Delta_2(x)$ if she strives for an additional reduction of P's production by $\Delta_2(x)$ irrespective of which offer Z_1 has been made by V_1. In order to find her optimal response, she will compare her benefits of the steelworks' additional production reduction $\frac{d}{2}\Delta_2(x)$ with the costs $b\Delta_2(x)$ of her additional payment. As $\frac{d}{2} < b$, a positive offer hence is not worthwhile for V_2.

Hence, not to pay anything to polluter P is the dominant strategy for the two sisters V_1 and V_2 when they act in an uncoordinated way. The consequence is that no internalization of externalities via bargaining will take place. The production remains on level $\overline{x}$, and each of both sisters V_1 and V_2 will be faced with a damage of $\frac{d}{2}\overline{x}$.

This inefficiency is due to a "social dilemma," which is caused by the split of the orchard. The transfers that the victims are willing to pay individually are too low for inducing the polluter P to change his behavior. The reason for this is that each sister appropriates only half of the total benefit that is generated via her payment to P (and the induced reduction in damage), while the other sister enjoys the other half (free of charge). Each victim therefore would prefer to take a free ride at the expense of the other victim, i.e. she would like to benefit from the other victim's payment. Expressed differently, after the parcelling out of the orchard, mitigation of the environmental harm represents a public good for V_1 and V_2 that is provided at a suboptimally low level (which even is zero in our case) when they act non-cooperatively. The problem associated with free-riding incentives becomes the more severe, the higher the number of victims. Hence, especially for larger groups of polluted agents, for which coordinated action is hard to achieve, we cannot expect that Coasean bargaining will bring about an efficient outcome.

Control Question
How does the bargaining model change when there is a higher number of victims?

2.2.1.5 Overall Assessment of the Coase Theorem

In the scientific literature, there is no full consensus about what the Coase Theorem precisely states. Coase himself has not presented a theorem in its true sense (i.e. a statement that is derived from specific assumptions) but has only demonstrated by means of examples that – without governmental intervention – bargaining as a market-like exchange process can bring about an internalization of externalities. The basic (and uncontestable) message was that one should at least take a closer look at the diagnosis of a market failure through externalities than Pigou did when he rapidly called for governmental intervention. Coase himself was well aware that in reality bargaining will be impeded by a multitude of hurdles, which he all classified as transaction costs and considered their absence as an important precondition for the perfect functioning of the bargaining process. The absence of transaction costs, however, is quite unlikely in real-world cases, and therefore the Coasean approach should be treated with caution when it comes to its application as an internalization device in reality (see, e.g., Allen, 2015, for a defense of the Coase Theorem and Medema, 2018, for an extensive review of the literature and a profound assessment of the Coasean approach.)

Since the level of transaction costs depends on the initial distribution of property rights, the government might in some cases try to improve the prospects for bargaining over externalities by allocating the property rights in a way that minimizes transaction costs. Yet, in most empirically relevant cases, there are many beneficiaries of pollution abatement. Due to free-riding incentives and the difficulties of self-organization of large groups, effective environmental protection might be provided by means of governmental intervention at the lowest transaction costs. Just from the perspective of transaction costs, the government therefore might turn out as the superior actor in many cases when the internalization of environmental externalities is at stake.

There are still other problems with the Coasean approach. So strategic behavior may pose a threat to the success of bargaining. Negotiating partners have a strong interest to manipulate the distribution of costs and benefits of environmental protection by strategic maneuvers to their own advantage – and they have the possibility for doing so in particular when monitoring is difficult and costly. Extortion strategies can also be easily pursued, which do not only lead to an unfair distribution but also may cause welfare losses: The theoretical result we obtained in this context has shown that agents anticipating environmental negotiations may have reduced incentives to develop and to apply better technologies if the use of worse technologies provides them with better extortion options. Additionally, negotiating partners must agree on the distribution of the costs and benefits of environmental protection, which may be difficult and time-consuming when notions of fairness differ.

Despite its limitations an important merit of the Coasean approach nevertheless is that it has raised the awareness that environmental regulations based on the almost self-evident polluter-pays principle are not at all mandatory. Even though they may seem ethically questionable, solutions where the victim pays a compensation to the polluter may come about faster than those where the polluter has to carry the financial burden of environmental protection. Especially in the international context, environmental improvements frequently become feasible only when the victim pays. In this respect, Coase has contributed to a higher acceptance of more effective methods of tackling environmental problems. Principles of justice that otherwise look well-founded may actually hinder the implementation of welfare-enhancing solutions, which represents one further variation of the deep conflict between efficiency and justice.

2.2.2 Other Voluntary Approaches

2.2.2.1 Matching

Decentral matching (see, e.g., Guttman, 1978; Rübbelke, 2006) is an internalization approach that resembles the instrument of subsidies. Like subsidies, the matching mechanism changes the effective price of environmental protection, but in contrast to the case of subsidies, it is not the government that makes a payment to the polluter P but the victim V. Given a matching rate $s > 0$, the polluter P's total benefit becomes $B(x)+s(\bar{x}-x)$ when his production level is x so that at x, the "price" (= marginal foregone benefit) that P has to pay for a reduction of his emissions is reduced from $B'(x)$ to $B'(x) - s$. Then V takes a part of the losses P has to incur when he reduces his production. Under the influence of matching, P thus maximizes his total benefit if he chooses the production and emission level $\hat{x}(s)$, which is characterized by the marginal condition $B'\left(\hat{x}(s)\right)=s$. Clearly, $\hat{x}(s)$ becomes smaller when the matching rate s is increased. The total cost of the victim V, which consists of environmental damages and matching payments to P, then is $D\left(\hat{x}(s)\right)+s\left(\bar{x}-\hat{x}(s)\right)$.

To minimize these total costs, the victim V will apply a matching rate s^* for which the first-order condition is $D'\left(\hat{x}\left(s^*\right)\right)\hat{x}'\left(s^*\right)-s^*\hat{x}'\left(s^*\right)+\bar{x}-\hat{x}\left(s^*\right)=0$. Inserting $B'\left(\hat{x}\left(s^*\right)\right)=s^*$ gives

$$D'\left(\hat{x}\left(s^*\right)\right)-B'\left(\hat{x}\left(s^*\right)\right)=\frac{\hat{x}\left(s^*\right)-\bar{x}}{\hat{x}'\left(s^*\right)}>0,$$

where the inequality holds since $\hat{x}\left(s^*\right)<\bar{x}$ and $\hat{x}'\left(s^*\right)<0$. Consequently, the resulting production level $\hat{x}\left(s^*\right)$ lies right to the optimal production level x^*, i.e. $\hat{x}\left(s^*\right)>x^*$. This means that the victim V would not strive for a complete internalization of the environmental externality if he has to bear the costs of matching.

2.2.2.2 Establishment of Social Norms

Behavior of individuals and partly firms is not only influenced by material incentives but also by nonmonetary motivations as those generated by ethical values and social norms. Many people have strong feelings of responsibility for the protection of nature and for future generations, which affect their behavior toward the environment. In the context of norm-driven behavior, a distinction has to be made between descriptive and injunctive norms, which motivate individuals through different channels.

Descriptive norms are people's perceptions of how human beings typically behave and lead – often without any deeper reflection – to adaptive action. As Cialdini et al. (1990: 1015) put it: "If everyone is doing it, it must be a sensible thing to do", which, e.g., means that if I am observing other people carefully separating their waste, I will consider such a behavior as normal and then imitate it. Thus, descriptive norms involve perceptions of which behavior is typically performed.

In contrast, *injunctive norms* are people's perceptions of what behavior is deemed right or wrong and thus have an essentially normative content. Many people would feel guilty if they violated the norm and threw away garbage on streets or in parks or if they still were using plastic bags. They also disapprove such a behavior by others, which causes a deterrent effect for people whose moral preferences are not so strong.

Norms can be established and reinforced, e.g., by education already at schools and information campaigns, not only by the government but also by private organizations as environmental associations and NGOs like Oxfam or World Wildlife Fund (Buchholz, Falkinger, & Rübbelke, 2014). In reality, it has been confirmed by various studies that social norms may indeed promote the choice of sustainable consumption patterns (see, e.g., Demarque, Charalambides, Hilton, & Waroquier, 2015) and the willingness to recycle (Cialdini, 2003) and to save energy in households and hotels (Allcott, 2011; Goldstein, Cialdini, & Griskevicius, 2008).

As important the encouragement of environmentally friendly behavior through norms is in some fields of environmental protection, voluntary action guided by norms cannot replace environmental laws and regulations by parliaments and governments. Injunctive norms, however, exert a probably much stronger influence on the shaping of environmental policy through another channel, i.e. through the formation of political will in democracies. Yet, also note that attempts of the government to change normative attitudes and approaches to "nudge" (see Thaler & Sunstein, 2008) people to act in a socially desirable way are, as an instrument of "libertarian paternalism", often criticized as being not in accordance with the image of the autonomous and mature citizen.

Not only individuals but also firms voluntarily engage in environmentally friendly activities to demonstrate a sense of "corporate social responsibility" and to avoid the bad reputation of being greedy mudslingers. Yet, such moves are often strategically motivated to pre-empt stricter governmental regulation or simply represent some kind of "greenwashing" aiming at hiding other not environmentally sound activities.

? Control Question

What environmental commitments by firms do you know? Give some examples.

Conclusion

You have learned in this chapter:

- That in the absence of internalization measures, unidirectional environmental externalities entail too high emission levels and thus lead to welfare losses.
- How in a laissez-faire scenario internalization can be attained by bargaining where the victim makes take-it-or-leave-it offers to the polluter.
- That for various reasons (transaction costs, asymmetric information, free-rider incentives in the case of several polluters and victims, simultaneous application of other environmental policy instruments), Coasean bargaining cannot be expected to generate an efficient outcome.
- That Coasean bargaining under the laissez-faire rule might be abused for extortion, which may bring about a loss of social welfare even in comparison with the situation without bargaining.

Appendix: Multilateral Externalities, Public Goods, and Mixed Goods

Until now we have only dealt with the case of a downstream, i.e. a unidirectional, externality where there is a clear distribution of roles between the polluter and the victim. Yet, in many cases of empirically relevant environmental problems, agents are affected both by the emissions of all other agents and by their own pollution, so that essentially everyone at the same time is a polluter and a victim. A prominent example for such a "multilateral externality" is climate change where the emission of greenhouse gases by car drivers or consumers of fossil-fuel-based electricity contributes to global warming and thus harms the emitters themselves like anyone else. Around the globe, there is non-rivalry and non-excludability (see ► Box 2.3 below) in receiving the adverse effects of global warming, and hence this warming due to the emission of greenhouse gases is a global public bad.

To describe this situation in a model, we assume that there are two agents A(dele) and B(ridget). Each of both uses fossil fuels and thus reduces environmental quality for herself as well as for her counterpart. Analogously as before in the case of a unidirectional externality, the material benefit from their own environmentally damaging activity x_i is denoted by $B_i(x_i)$ for agents A and B, respectively. However, now the environmental damage that A and B suffer depends both on her own pollution and the pollution caused by the other agent. Hence, agent A' s environmental damage function is $D_A(\alpha_{AA}x_A + \alpha_{BA}x_B)$, and that of agent B is $D_B(\alpha_{BB}x_B + \alpha_{AB}x_A)$ where $\alpha_{ij}(i,j = A,B)$ indicate the impact of both agents' emissions on the environmental quality enjoyed by them. If $\alpha_{AA} = \alpha_{BB} = \alpha_{BA} = 0$, but $\alpha_{AB} > 0$, the case of a unidirectional externality is obtained as a special case of this general model, in which agent A is the polluter and agent B the victim. In the following, we focus on the case $\alpha_{AA} = \alpha_{BB} = \alpha_{AB} = \alpha_{BA} = 1$ where the environmental impact of both agents' emissions is completely symmetric so that pollution becomes a true "public bad." Henceforth, abatement is a "public good," which simply means that abatement activities

(= reductions of the polluting activities x_i) have the same effect on environmental quality for each agent irrespective of where abatement efforts take place. If both agents act independently in this situation with reciprocal externalities, they will – according to the Nash hypothesis – adapt their own emission level to that of the other agent, i.e. agent A maximizes her total benefit $B_A(x_A) - D_A(x_A + x_B)$ for any given x_B by choosing x_A, while agent B analogously maximizes $B_B(x_B) - D_B(x_A + x_B)$ for any given x_A. In this case the non-cooperative Nash equilibrium, i.e. the laissez-faire solution with activity levels $\bar{x}_A$ and $\bar{x}_B$, is attained when the reactions of both agents coincide, i.e. the two marginal conditions $B'_A(\bar{x}_A) = D'_A(\bar{x}_A + \bar{x}_B)$ and $B'_B(\bar{x}_B) = D'_B(\bar{x}_A + \bar{x}_B)$ simultaneously hold. The optimal activity levels x_A^* and x_B^* that maximize aggregate welfare $(B_A(x_A) - D_A(x_A + x_B)) + (B_B(x_B) - D_B(x_A + x_B))$ of the two agents instead are characterized by the marginal conditions $B'_A(x_A^*) = D'_A(x_A^* + x_B^*) + D'_B(x_A^* + x_B^*)$ and $B'_B(x_B^*) = D'_A(x_A^* + x_B^*) + D'_B(x_A^* + x_B^*)$. For aggregate polluting activities of both agents, we then have $x_A^* + x_B^* < \bar{x}_A + \bar{x}_B$, i.e. that – not quite surprisingly – aggregate emissions in the optimal solution are lower than in the laissez-faire outcome. This follows by an indirect proof: Let us assume that $x_A^* + x_B^* > \bar{x}_A + \bar{x}_B$. Then it follows from $D''_i(x_i) > 0$ that $B'_i(x_i^*) = D'_A(x_A^* + x_B^*) + D'_B(x_A^* + x_B^*) > D'_i(\bar{x}_A + \bar{x}_B) = B'_i(\bar{x}_i)$ would hold for both agents $i = A, B$. Since $B''_i(x_i) < 0$ this, however, would give $x_i^* < \bar{x}_i$ for $i = A, B$ and thus a contradiction.

Box 2.3: Public Goods vs. Private Goods

The consumption of a private good like ice cream is characterized by excludability, i.e. one can be excluded from consuming this good, e.g., the iceman can refuse one's consumption of ice cream as long as the price for it is not paid by the potential customer. Moreover, there is rivalry in consumption, which means in the ice-cream context that if the (paying) customer finally consumes his dish of ice cream, no one else can consume the very same dish of ice cream. If others would take away a share of this dish, this will reduce the customer's benefit from eating his ice cream.

In contrast, potential consumers of pure public goods cannot be excluded from the consumption of this good (see Samuelson, 1954, Musgrave, 1959, and Cornes & Sandler, 1996, for the historic development of this concept). Take climate protection as an example. If Australia reduces its greenhouse gas emissions and thus is slowing global warming, not only Australia benefits, but automatically also the rest of the world. Non-excludability in consumption prevails because Australia cannot prevent others (e.g., the European Union or the USA) from enjoying the positive effects of mitigated climate change. At the same time, there is non-rivalry in consumption, i.e. Australia's own benefit resulting from its abatement measures is not adversely affected when other countries also enjoy the positive effects of Australia's mitigation efforts.

For the sake of completeness, it has to be noticed that there exist also mixed goods, i.e. goods which lie somewhere between the extremes of the pure private good and the pure public good. On the one hand, e.g., the use of an uncrowded bridge is not subject to rivalry until its capacity limit is reached, but there well exists the possibility to exclude agents from crossing the bridge and thus consuming this "good." Such goods are called club goods, and a toll may be charged for their use (Musgrave, 1959). On the other hand, and of more relevance for environmental economics, there are

	Non-rivalry	Rivalry
Excludability	Club good	Private good
Non-excludability	Pure public good	Common-pool resource

Fig. 2.9 Types of goods

common-pool resources like international fishing grounds (see Hardin, 1962) for which there is rivalry, but exclusion of users is difficult if not impossible. In the case of such commons, the risk prevails that they will be overexploited and not "conserved for the benefit of all" (Ostrom, 2008: 3573). A classification of these different types of goods is provided by Fig. 2.9.

Another category of goods having both private and public characteristics is that of impure public goods or joint-production goods, where an agent gets some additional private benefit from his public good contribution (Cornes & Sandler, 1984, 1996; Kotchen, 2005). So climate policy measures – by inducing improvements in energy efficiency – reduce the burning of fossil fuel and thus the emissions of local or regional pollutants like particulate matter and sulfur dioxide. Consequently, such abatement activities do not only contribute to the global public good "climate protection" but also generate beneficial effects like health improvements at the regional level (Pittel & Rübbelke, 2017).

In principle, internalization of reciprocal externalities is conceivable by the same voluntary approaches as in the case of unilateral externalities, i.e. through bargaining and matching. Both theoretical modelling and practical implementation of these approaches are getting much more complicated in the case of reciprocal externalities. Yet, in the context of global public goods, as in particular climate change mitigation, overcoming suboptimal provision is not possible without bargaining between states as there is no international coercive authority that could enforce globally efficient climate protection regulations. Hence, the voluntary approaches are of much importance in this field. That especially norms may play a dominant role for successful collective action on voluntary public good provision is stressed by Nobel laureate Elinor Ostrom. "Increasing the authority of individuals to devise their own rules may well result in processes that allow social norms to evolve and thereby increase the probability of individuals better solving collective action problems" (Ostrom, 2000: 154). We will later come back to these issues in ► Chap. 5.

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Monetary Valuation of the Environment

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Objectives of This Chapter

In this chapter students should learn:

- The microeconomic foundations of the monetary valuation of environmental benefits and damages
- The basic features of the contingent valuation method (CVM) through which agents' preferences for environmental quality are elicited by survey questions
- The potential advantages of this widely applied stated preference technique over other revealed preference approaches for environmental quality assessment
- The issues that have to be addressed and the decisions that have to be made when a CVM study is conducted
- The conceptual and practical problems that are connected with the CVM and the limitations of this technique
- How the observance of some practical guidelines may help improve the quality and validity of CVM studies

Internalizing environmental externalities through Coasean bargaining would have the merit that a central compilation and handling of information by the government would not be necessary. However, when Coasean bargaining does not take place, which is very likely in the case of real-world environmental problems, and therefore governmental interventions are required, the environmental authorities must have information about costs and benefits of environmental protection measures. While their costs can frequently be estimated through the observation of market transactions and prices, the benefits of environmental protection measures are more difficult to assess. The reason is that these benefits often depend on individuals' subjective valuations of changes in environmental quality, which are not adequately reflected by market prices. The monetary valuation of these benefits requires sophisticated procedures, the most important one being the *contingent valuation method (CVM)* which is based on interviews with people being affected by environmental damages. As a consequence of the Exxon Valdez oil spill that occurred in Prince William Sound (Alaska) in 1989, this method for environmental damage assessment has become mandatory in the USA. Before we will describe the contingent valuation method in detail (and some other procedures rather briefly), we first of all explain some fundamental theoretical concepts that are related to the monetary evaluation of the environment. In spite of valuation methods' relatively broad range of applications, the attempts at monetary valuation of environmental quality have not remained undisputed. In this chapter, we will therefore also have a closer look at the validity and the practical usefulness of valuation studies.

3.1 Theoretical Background

3.1.1 Price Changes

To get started let us consider a simple partial model that is well known from any basic microeconomics course. In this model there is only one single good z with an inverse demand function $p_d(z)$. The increase in utility that consumers enjoy from the rise in the provision of

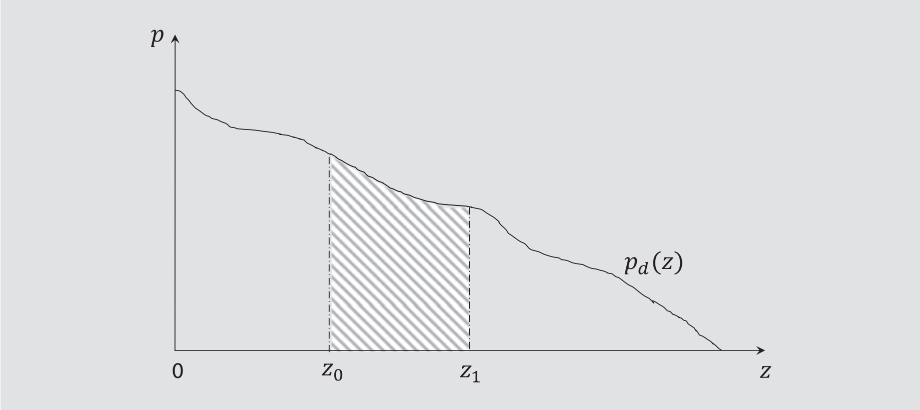

Fig. 3.1 Inverse demand function for good z

the good to be assessed ("environmental quality") from its initial level z_0 to a new level $z_1 > z_0$ is depicted in Fig. 3.1 by the area below the inverse demand function between z_0 and z_1.

In order to explain this measurement of the utility derived from consumption of a good, let $B(z)$ stand for the gross utility that individuals enjoy from the consumption of the amount z of the good under consideration. Given the price p for the good and supposing that the consumers act as price takers, they will demand that amount $z_d(p)$ at which their net utility $B(z) - pz$ is maximized and at which thus the first-order condition $B'(z_d(p)) = p$ holds. If one takes the amount z of the good instead of price p as the independent variable, this condition translates into $B'(z) = p_d(z)$ for marginal utility. Through integration, we get $B(z) = \int_0^k p_d(\tilde{z})\, d\tilde{z}$ for the increase in consumer surplus, which results when the consumed amount of the good rises from z_0 to z_1. In the next subsection, we will show how monetary assessment of environmental quality changes can be conducted in the standard microeconomic household model with several goods.

Control Question

Why can in an elementary partial model the value of the inverse demand function for a good be identified with marginal utility?

3.1.1.1 Basic Concepts Explained in a Two-Goods Model

Employing a purely partial model in the context of utility assessment is practical but not very precise. The appropriate starting point for monetary utility valuation are – like in the microeconomic household theory – the individual preferences with respect to the consumption of several goods. For simplicity, we confine our analysis to the case of only two goods, where x is the amount of a private good consumed by the considered representative individual and z is the provision level of another good, which can either be private or public. The utility function of the individual is $u(x,z)$.

From microeconomic theory, it is well known that the measurement of utility via the utility function $u(x,z)$ allows only ordinal comparisons. Consequently, it can only be ascertained whether an individual prefers a bundle of goods to another bundle, but it cannot be determined by how much this bundle of goods is higher valued by the

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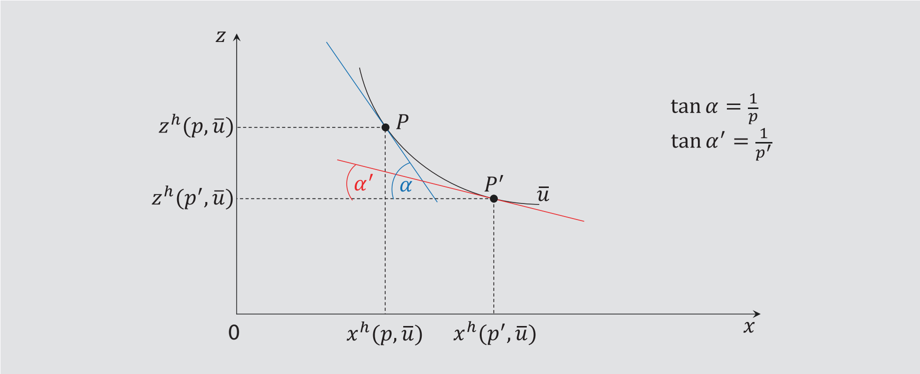


Fig. 3.2 The Hicksian demand for goods z and x

individual. Something like this, however, is needed when one wants to provide a monetary evaluation of changes of environmental quality. In order to cope with this problem, specific tools from microeconomic theory (such as the "Hicksian demand function" or the "expenditure function") have to be employed to get a better understanding of the problems that arise in the context of the monetary evaluation of environmental quality (see, e.g., Cornes, 1992 or Jehle & Reny, 2011).

The standard (uncompensated) *Marshallian demand function* $z^m(p,y)$ describes how the demanded quantity of good z changes with a varying price p of this good, given that the income level y of the individual is kept constant. Our introductory remarks on the partial model suggest that this demand function could be used as the base for monetary valuation. Yet, as Freeman III et al. (2014: 54) stress, "[a]lthough the Marshallian consumer surplus has some intuitive appeal as a welfare indicator […] it is not a measure of gain or loss that can be employed in a potential compensation test." Instead, *Hicksian demand functions* have to be used to define correct measures for the monetary value of environmental quality and its changes.

A Hicksian demand function indicates the demand effects of price changes when a certain given level $\bar{u}$ of the representative individual's utility is fixed (instead of the income level as in the case of Marshallian demand). Therefore, the Hicksian demand for z indicates the amount $z^h(p,\bar{u})$ that will be demanded if the price for z is p and the individual is kept at the utility level $\bar{u}$. Graphically (see Fig. 3.2), $z^h(p,\bar{u})$ results as the ordinate value of the point in which a straight line with the slope $-\frac{1}{p}$ (viewed from the x-axis toward the z-axis) is tangent to the indifference curve for the utility level $\bar{u}$.

The individual is in a household optimum for a given p if his marginal rate of substitution between z and x is equal to p (or, equivalently, the marginal rate of substitution between x and z is equal to $\frac{1}{p}$).[1] If the price of z increases, then the tangent turns

1 The marginal rate of substitution is equal to the slope of the indifference curve and therefore it is negative. However, for simplicity reasons, the minus sign is frequently omitted. Mathematically correct would be to state above that the absolute value of the marginal rate of substitution must be equal to p.

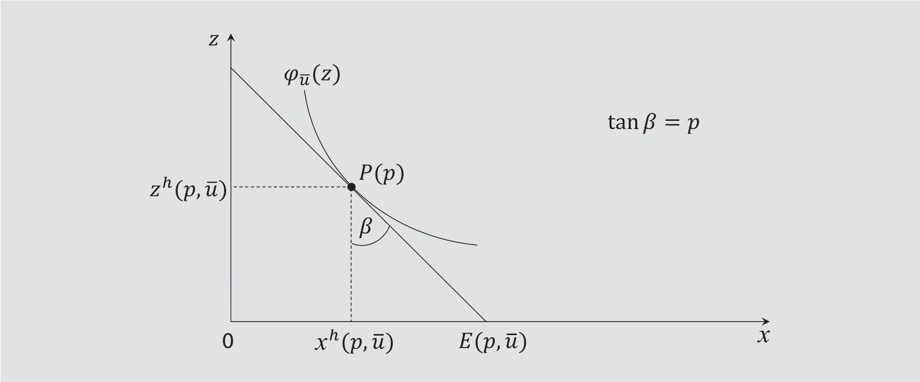


Fig. 3.3 The expenditure function

anticlockwise along the indifference curve, i.e. from P to P' when the price of z increases from p to p' (see Fig. 3.2). Then, given a convex indifference curve, the Hicksian demand for z clearly declines. Analogously, $x^h(p,\bar{u})$ describes the Hicksian demand for the other good x, whose price is normalized to unity so that x becomes the *numéraire good*.

The Hicksian demand point $P(p) = \left(x^h(p,\bar{u}), z^h(p,\bar{u})\right)$ can obviously be represented as the individual's Marshallian demand if the individual's income is adapted in an adequate way, which means that when a certain fixed income level is given, an income compensation would be required. This explains why Hicksian demand functions are often labelled as *compensated demand functions*.

Then, the level of the hypothetically adjusted income is determined by the *expenditure function* $E(p, \bar{u})$. Graphically, its value is determined by the point of intersection between the tangent through the Hicksian demand point $P(p)$ and the x-axis (see Fig. 3.3). Hence, the formal expression for the expenditure function reads

$$E(p,\bar{u}) = x^h(p,\bar{u}) + pz^h(p,\bar{u}) = \varphi\left(z^h(p,\bar{u})\right) + pz^h(p,\bar{u}),$$

where $(x =)\varphi(z) = \varphi_{\bar{u}}(z)$ describes the indifference curve for the utility level $\bar{u}$ as a function of z.

By taking the derivative with respect to the price p, we determine the change in the expenditure function resulting from a marginal increase in the price p of good z:

$$\frac{\partial E(p,\bar{u})}{\partial p} = \frac{\partial \varphi}{\partial z}\frac{\partial z^h}{\partial p} + p\frac{\partial z^h}{\partial p} + z^h(p,\bar{u}) = z^h(p,\bar{u}).$$

The first two terms cancel each other out since the absolute value $-\frac{\partial \varphi}{\partial z}$ of the slope of the indifference curve is equal to price ratio p in a household equilibrium. The identity derived in this way constitutes a central result in *duality theory* and is denoted *Shephard's Lemma*. Expressed verbally it states that the derivative of the expenditure function with respect to the price of a good is equal to the Hicksian demand for this good.

3

Box 3.1: The Formal Optimization Approaches

The formal optimization approaches to determine the Marshallian and Hicksian demands are as follows:

Marshall

Utility is maximized subject to a budget constraint: in Fig. 3.4, Marshallian demand is given by the tangent point between the given budget line with income $\bar{y}$ and slope $-1/p$ and the highest attainable indifference curve u_2.

Hicks

The income that is needed for attaining a given utility level $\bar{u}$ is minimized: in Fig. 3.4, the Hicksian demand attainable with minimum income y_2 is given by the tangent point between the indifference curve for $\bar{u}$ and a budget line with slope $-1/p$.

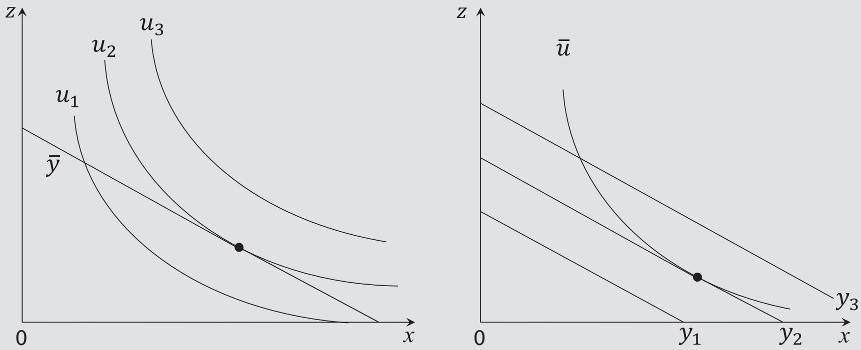


Fig. 3.4 Marshallian demand (left) and Hicksian demand (right)

Control Question

What is the difference between Marshallian and Hicksian demand functions? What is the connection between them?

3.1.1.2 Two Concepts to Measure Changes in Utility

In a next step, we will apply the basic concepts explained in the preceding subsection to provide a monetary valuation for the utility change that results when the price of a private good z is affected by environmental quality. This specifically means that an improvement in environmental quality from G_0 to G_1 will cause a decline in the price of z from $p_0 = p(G_0)$ to $p_1 = p(G_1)$. As a simple example, imagine that an improvement in air quality leads to an increase of fruit supply so that the market price of fruits falls. Due to the lower price of fruits, a representative agent will attain a higher utility level $\bar{u}_1$ after the improvement of environmental quality than before, where his utility has been $\bar{u}_0$. Keeping the income level y of the representative agent constant the points P_0 and P_1 in Fig. 3.5 then indicate the household optima resulting for these two prices p_0 and p_1, respectively. We now can define two measures for the monetary valuation of the utility change that is associated with the move from P_0 to P_1:

The *compensating variation CV* indicates the maximum amount of income the representative individual is willing to pay for the improvement of environmental quality

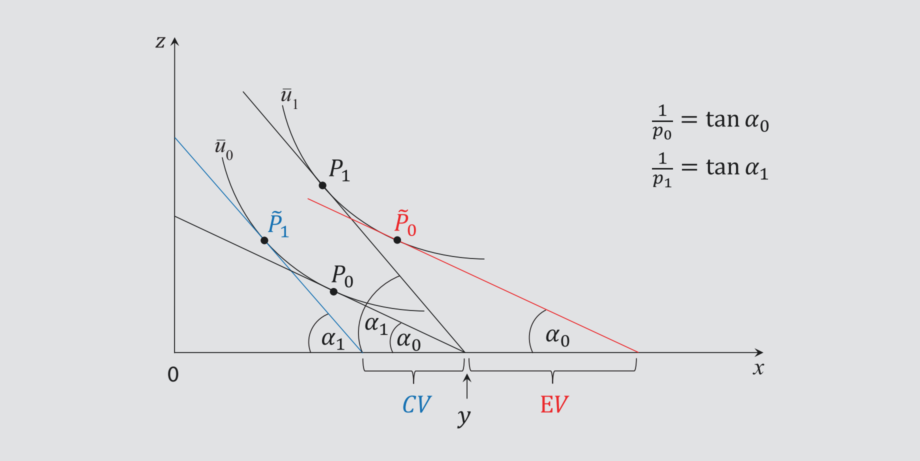


Fig. 3.5 Graphical representation of compensating variation (CV) and equivalent variation (EV)

bringing him from G_0 to G_1. This means that the representative agent makes the payment CV, which enables him to benefit from the lower price p_1 while staying at his original utility level $\bar{u}_0$. Using the expenditure function, the compensating variation thus becomes

$$CV = E\left(p_0, \bar{u}_0\right) - E\left(p_1, \bar{u}_0\right).$$

An alternative to the CV is the *equivalent variation* EV, which is formally defined as

$$EV = E\left(p_0, \bar{u}_1\right) - E\left(p_1, \bar{u}_1\right).$$

The equivalent variation stands for the amount of income that the representative individual will at least claim as a compensation when the environmental quality improvement does not take place and consequently the price of good z does not fall. To put it differently, EV measures the amount of money that would have to be given to the individual in the initial (more polluted) state to make him as well-off as in the situation where environmental quality has improved and where he consequently enjoys a lower price of good z. The reference level for the agent's utility in case of EV is $\bar{u}_1$.

While CV is a measure of the willingness to pay for the improvement of environmental quality leading to a decline in the price for a private good, EV measures the willingness to accept the nonappearance of this improvement. In Fig. 3.5, CV and EV are depicted in an x-z diagram.

Control Question

What makes the difference between compensating and equivalent variation? Why can the compensating variation be interpreted as a willingness-to-pay measure and equivalent variation as a willingness-to-accept measure in the context of an improvement of environmental quality?

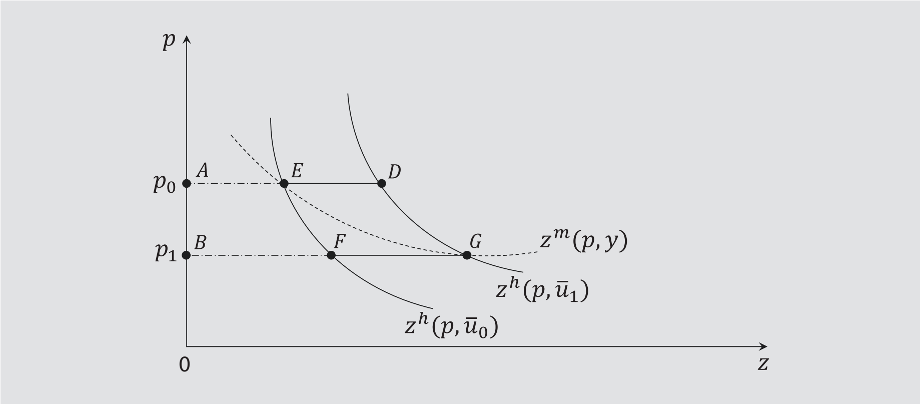


Fig. 3.6 Graphical representation of *CV* and *EV* with Hicksian demand curves

The measures *CV* and *EV* can also be displayed in a price-quantity diagram by means of Hicksian demand functions (see Fig. 3.6). So, by applying Shephard's Lemma $\left(\frac{\partial E}{\partial p} = z^h\right)$, we obtain

$$CV = E(p_0,\bar{u}_0) - E(p_1,\bar{u}_0) = \int_{p_1}^{p_0} z^h(p,\bar{u}_0)\,dp = ABFE,$$

$$EV = E(p_0,\bar{u}_1) - E(p_1,\bar{u}_1) = \int_{p_1}^{p_0} z^h(p,\bar{u}_1)\,dp = ABGD.$$

Note that in Fig. 3.6 we have taken into account that the Hicksian demand function $z^h(p,\bar{u}_1)$ is located to the right of $z^h(p,\bar{u}_0)$ provided that z is a normal good. Normality of a good – by definition – means that for a fixed price, its Marshallian demand is increasing in the income level. This directly implies that for a certain price p, Hicksian demand for good z is increasing in the level of utility.

In this situation the relationship between Hicksian and Marshallian demands is given by the identities $z^m(p_0,y) = z^h(p_0,\bar{u}_0)$ and $z^m(p_1,y) = z^h(p_1,\bar{u}_1)$, which imply that the Marshallian demand function $z^m(p,y)$ passes through both points P_0 and P_1. Consider the *gross consumer surplus*, which is defined as the area under the inverse Marshallian demand function, i.e.

$$CS = \int_{p_1}^{p_0} z^m(p,y)\,dp = ABGE.$$

Figure 3.6 thus directly shows that $CV < CS < EV$. Therefore, the consumer surplus *CS* represents some kind of an in-between compromise between *CV* and *EV*. At the conceptual level, this gives a justification for using *CS* as an approximate welfare measure in the case of price changes. At the practical level, *CS* has the merit of being determinable in a straightforward way, since in contrast to Hicksian demand, Marshallian demand can be observed directly.

Control Question

Why can the level of the equivalent variation be expected to exceed that of the compensating variation?

Box 3.2: The Case of a Deterioration of Environmental Quality

We have until now considered the case in which environmental quality is improving. Figure 3.7 instead describes *CV* and *EV* in the opposite case where environmental pollution causes an increase in the price of good *z*. Now, the representative agent is worse off due to the price increase. In this case *CV* represents the minimum compensation that he would require to accept the deterioration of environmental quality and the ensuing higher price of good *z*. The reference point is again the agent's initial welfare level $\bar{u}_0$. In contrast, *EV* is the individual's maximum willingness to pay to avoid the reduction of environmental quality. Here, the reference point is the utility level $\bar{u}_1$ in the post-change situation.

Table 3.1 compares the cases of the improvement and the deterioration of environmental quality.

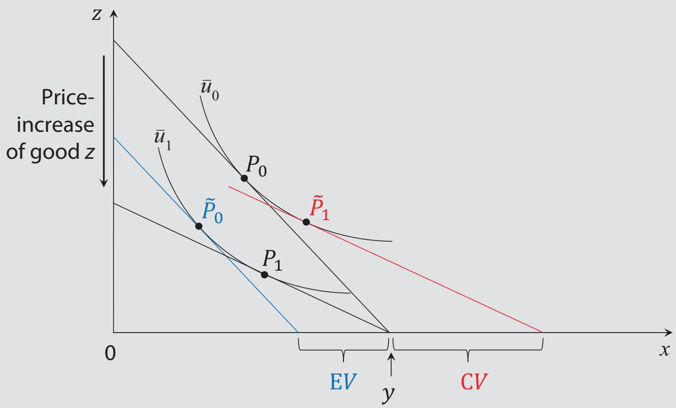


Fig. 3.7 Compensating variation and equivalent variation when the price of good *z* increases

Table 3.1 Comparing *CV* and *EV* in different cases

CV	*EV*
1. Improvement of environmental quality → WTP (willingness to pay)	3. Absence of an improvement of environmental quality → WTA (claim for compensation)
2. Deterioration of environmental quality → WTA (claim for compensation)	4. Avoidance of a deterioration of environmental quality → WTP (willingness to pay)

3

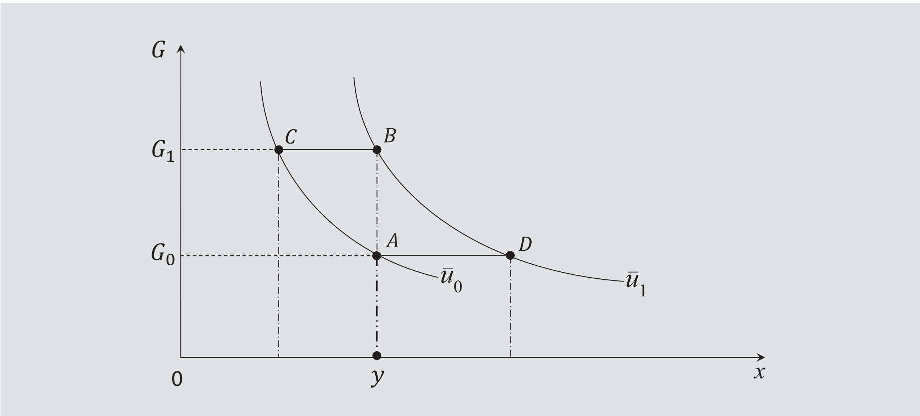
3.1.2 Quantity Changes

When changes of environmental quality have a direct effect on an agent's utility (i.e. not – as before – an indirect one through changes of the price of a private good), monetary evaluation of these utility changes can be conducted in a similar way. We now suppose that z represents the level of environmental quality, i.e. the public environmental good whose supply level again is denoted as G. To keep the analysis simple, we assume that private good consumption x is equal to the exogenously given income y so that the costs that an individual possibly has to bear for the improvement of environmental quality are not taken into account.

Let, as before, the level of the environmental good G increase from its initial level G_0 to some higher level G_1. In ◘ Fig. 3.8 the distance AD gives a monetary measure of the corresponding change of utility, which is analogous to the monetary utility measure EV in the case of a price change: an individual will claim at least the amount AD as a compensation when he waives the environmental quality improvement from G_0 to G_1. Obviously, the agent would also demand at least AD to become willing to accept the environmental degradation from G_1 to G_0. Consequently, AD is a willingness-to-accept (WTA) measure either for not having an environmental quality improvement or for tolerating an environmental quality deterioration.

In ◘ Fig. 3.8, the distance CB corresponds to the monetary utility measure CV for an environmental quality improvement as described above in a different setting: in order to enjoy better environmental quality G_1 (instead of G_0), the individual is maximally willing to pay an amount CB. Moreover, CB represents the willingness to pay (WTP) for avoiding a decrease of environmental quality from G_1 to G_0.

In the case of quantity changes, WTA and WTP measures can again be depicted by means of Hicksian demand functions. How this works exactly will be shown – among many other things – in the subsequent subsection, where we discuss several problems that have to be taken into consideration in the context of monetary valuation of environmental quality in general and the contingent valuation method in particular.



◘ **Fig. 3.8** *CV* and *EV* in case of quantity changes

Control Question

Why does the WTA measure indicate the minimum amount of compensation for an environmental damage that has to be paid to the affected agents?

Conclusion

You have learnt in this section:

- What Hicksian demand functions describe and how they can, with help of the expenditure functions, be related to the standard Marshallian demand.
- How the compensating variation CV provides a willingness-to-pay measure for an improvement of environmental quality that entails a price reduction for a private good.
- How the equivalent variation EV analogously provides a willingness-to-accept measure for an environmental damage.
- How CV and EV can be described by means of Hicksian demand functions.
- That under standard assumptions on preferences WTA will exceed WTP.
- How willingness-to-pay and willingness-to-accept measures can be defined when changes of environmental quality directly affect individual utility (and not via changes of the price of a private good).

3.2 The Contingent Valuation Method: Measuring Utility by Means of Surveys

3.2.1 The Idea

The monetary valuation of changes in environmental quality by means of direct questioning of the affected agents has a long tradition. The basic idea underlying the *contingent valuation method* has already been developed by German-American economist Siegfried von Ciriacy-Wantrup (1947). Since the 1970s, this valuation method has found broader application in the context of various environmental problems like water and air pollution. Particularly important fields of application of the CVM are the measurement of welfare losses caused by the impairment of landscapes, loss of biodiversity, and the destruction of natural habitats (see, e.g., Carson, Flores, & Meade, 2001, Carson & Hanemann, 2005, Mäler & Vincent, 2005, and Alberini & Kahn, 2009 for broad overviews). As already mentioned above, the CVM gained particular importance in the USA where since the early 1990s the government can – quasi as a trustee of its citizens and the nature – take companies to court and claim compensations for environmental damages. In doing so, methodically well-grounded procedures for the valuation of damages are mandatory, which raised the empirical relevance of the CVM significantly.

Since the 1950s, also indirect valuation methods have been frequently applied for a monetary assessment of environmental quality. These approaches like the *travel cost approach* or the labelled *hedonic pricing method* are based on the observation of actual market transactions (see, e.g., Phaneuf & Smith, 2005, and Palmquist, 2005, for detailed descriptions of these indirect approaches).

Box 3.3: Alternative Evaluation Methods

Among the other methods applicable to valuate environmental assets or damage are in particular hedonic pricing, the travel cost method, choice experiments, and asking for expert opinion. Furthermore, there exist variations of the CVM.

- *Hedonic pricing*: In many cases, the market value of an asset is affected by pollution. The hedonic pricing method infers the monetary equivalent of the damage that an increasing pollution level has caused from the change of the asset's market value. Imagine the following case: people own houses and property nearby an area where an airport is constructed. The airport will bring about some noise disturbance, which will cause a reduction in the demand for these houses and property on the market. The consequence is a dropping market price for these assets. From the respective price decline, a monetary measure for damage of the airport's noise pollution is inferred.
- *Travel cost method*: This method especially is applied to measure the value of a scenic beauty or a recreation site: imagine that after environmental pollution has eliminated a recreation site, an evaluator assesses the costs and time that people had to bear for a trip to this area before it was destroyed. These costs can then provide a lower bound of the use value of that recreational site and hence of the damage caused by pollution.
- *Choice experiments*: In choice experiments, the evaluator presents a number of discrete alternatives. To be more precise, different bundles of nonmarket goods are described in terms of their attributes and levels of these attributes. Respondents are then asked to state which of the alternatives they prefer. As Hanley, Adamowicz, & Wright (2005: 228) explain: "By incorporating price as one of these attributes, marginal utility estimates from probabilistic choice models can be converted into willingness-to-pay (WTP) estimates for changes in attribute levels." The methodology of the choice experiment approach combines Kelvin J. Lancaster's (1966) characteristics approach with random utility theory (see Thurstone, 1927; McFadden, 1974). According to Lancaster's theory, people derive utility not from a good itself but from the attributes (characteristics) of this good. According to random utility theory, people make their welfare-maximizing decision in a rational and well-informed way. Yet, an analyst does not know utilities with certainty and therefore treats them as random variables.
- *Asking for expert opinion*: Hausman (2012: 44) criticizes the CVM and argues that "public policy will do better if expert opinion is used to evaluate specific projects, including nonuse value, and to set appropriate financial incentives to reduce the risk of accidents such as the Exxon Valdez and BP disasters."
- *Variations of the CVM*: Instead of assessing the willingness to pay for an improvement of environmental quality (or for a prevention of its deterioration), one may explore the maximum willingness to work WTW for the same purpose (see, e.g., Ahlheim et al., 2010). The applicability of this method is of particular importance in countries with very low private household incomes, where asking for monetary contributions does not provide reliable results. A problem, however, is that working time cannot be easily converted into monetary values so that interpretation of WTW studies is not straightforward.

  One may also use – instead of money or labor – staple food (like rice) as measure for the valuation (see, e.g., Shyamsundar & Kramer, 1996).

In contrast to these indirect methods, the CVM has a decisive advantage since not only *use values* but also *non-use values* can be measured by direct surveys. Such non-use values are of high importance in the context of environmental problems. Let us now explain these different forms of values in some more detail.

Use values stand for the direct benefits that agents obtain from "consuming" an environmental good as, e.g., enjoying low levels of noise or air pollution. Non-use values, as already emphasized by Krutilla (1967), can be classified in the following categories, which, however, cannot be clearly separated from each other:

- *Option values* refer to agents' positive willingness to pay for their potential future uses of environmental goods, which means: "Even if I will not visit a natural park today, I would like to preserve the possibility to do so at a later point in time." In a similar way, the desire for preserving biodiversity in order to allow future use of the protected genetic pool, e.g., for the development of drugs against serious illnesses, can be interpreted as such an option value.
- *Bequest values* stand for an individual's positive willingness to pay for the use of the environment by future generations. The use value of environmental goods and services enjoyed by posterity thus becomes a component of the utility perception of individuals living today. The bequest value is derived from "altruistic" motives so that caring for the well-being of other individuals (i.e. in this special case of those living in the future) affects one's own valuation of the environment.
- *Existence values* reflect the positive willingness to pay for the preservation of natural assets without establishing an immediate connection to any concrete benefit flowing to human beings. In particular, neither the potential own use (as in the case of the option value) nor the use by future generations (as in the case of the bequest value) matters. Rather, the preservation of nature and the maintenance of the integrity of creation perhaps inspired by religious beliefs constitute values in itself.

Preferences of individuals regarding the natural environment are only fully captured if the "soft" utility components being related to these non-use values are adequately taken into account. However, non-use values are rarely reflected in market transactions, and consequently indirect valuation methods are failing from the outset. The CVM technique instead opens a way out of this dilemma because individuals can directly reveal their non-use values through their responses to the questions posed in a CVM survey. Furthermore, the CVM may make it easier to control for the influence of factors which determine individual actions but which are not interrelated with the environmental good under consideration as, e.g., the availability of a good restaurant near a scenic beauty. Therefore, key problems of indirect methods that rely on agents' observed behavior can a priori be avoided by applying the direct CVM approach.

Control Question
What are use and non-use values of environmental goods and services? Provide some examples! What kinds of non-use values do you know?

Control Question
Can you imagine situations in which non-use values are finding expression in market transactions?

3.2.2 The Procedure

In most cases, it is not clear from the outset how a survey for eliciting the value of a certain environmental asset has to be organized to obtain meaningful and reliable answers. On different layers, various design options for a CVM study exist, which in particular means that answers to the following questions have to be found by the investigators and decisions have to be made:

- *How should the situation be arranged through which the investigator elicits the answers of the respondents?*
  On the one hand, it has to be decided how to describe the environmental good (its original state and its potential change) to be valued; on the other hand, a precise *payment vehicle* has to be specified. In doing so, one has to keep in mind that providing the respondent with detailed and vivid information about the object of the survey (perhaps via photos of a polluted lake or an endangered animal species) causes the danger that emotions are stirred up too much. The test person then might feel provoked to declare a too high valuation of the environmental good. Moreover, many environmental problems are so complex that individuals without some background information about intricate ecological interdependencies may not be able to give useful answers at all. In addition, one has to take care that the suggested payment vehicle does not provoke negative sentiments and useless protest answers. Due to such psychological effects, many individuals might be motivated not to reveal their true willingness to pay, but they will, e.g., state too low values as a protest against payment schemes that are considered unfair because others are held to be responsible for the environmental problem at stake. A similar phenomenon can arise if the payment scheme resembles conventional taxes so that a general resistance to taxation is triggered, which also distorts the answers. As part of the elicitation process, it may also be helpful to inform the interviewees about per capita expenses of the government for different other purposes (education, defense, etc.) in order to give them some point of reference for a realistic assessment of their own willingness to pay.
- *What should the elicitation method look like*? *In which format should the questions be presented to the test person?*
  Direct *open-ended questions* without any additional specifications seem to be an obvious approach. However, they tend to overstrain the intellectual capacities of the respondents in fields where they cannot rely on previous experience and thus have no anchor for their answers. In contrast, *bidding games* begin with some initial value for the willingness to pay, which then is revised (lowered or increased) until it is finally accepted by the test person (also see Boyle, Bishop, & Welsh, 1985). When the *payment card method* is applied, the interviewees are confronted with a list of possible values for their willingness to pay from which they have to select a particular one.

  Finally, the *referendum format* confronts different subgroups of individuals with the question whether they would accept a certain predetermined value to represent their willingness to pay. So the question may read: Would you be willing to pay 50 € for the preservation of the Danube wetlands? The big advantage of this dichotomous choice method is that individuals are familiar with such decisions as they resemble decision-making at political elections and referenda. The disadvantage, however, is that the interviewer could confront each subgroup of respondents with only one single potential willingness-to-pay value, which then has to be accepted or rejected. The high total number of tests that are needed with this approach would entail high costs of the CVM study.

- *Which medium should be used to transmit the questions and test persons' answers*?
  Postal or e-mail interviews are easy to handle and relatively cheap to conduct, but it cannot be guaranteed that the respondents actually understand the questions posed. In order to tackle this problem, the interviewer may confine himself to rather simple questions. This, however, might interfere with the complexity of the environmental problem under consideration so that one would not obtain meaningful results. Furthermore, the response rate for postal and e-mail interviews is generally rather low, and there is a high risk of biased results. Obviously, it can be expected that primarily those individuals will respond who have an especially high interest in the respective environmental good. Then, the aggregate willingness to pay is overestimated. *Telephone surveys* are also relatively cheap. However, the accessibility by phone is not equal among different population groups, and, in the age of mobile phones, phone numbers are no longer readily accessible. Hence, it cannot be ensured that the group of respondents is representative. Moreover, telephone surveys do not allow for visual illustrations of the environmental problem under investigation. Due to the development and use of new technologies (like Internet telephony and video conferencing), this problem, however, can now be mitigated or even be overcome. In *personal interviews,* the possibility for interviewees to ask questions helps to solve problems of comprehension, which improves the reliability of the results. However, apart from the relatively high costs of interviews, there is a risk that the information provided by the interviewer causes some distortion of the responses.
- *How should the sample of respondents be composed*?
  Especially with regard to non-use values where no immediate own involvement of the respondents prevails, serious problems of defining the appropriate set of interviewees arise. In this case, the circle of persons that is potentially affected by the evaluated environmental damages may become very large. People in industrialized countries are also concerned about deforestation and biodiversity loss in developing countries, for example, and therefore should also be included in a CVM study addressing these problems.

? Control Question

What are the most important design features of CVM studies? What do you think: Is it possible to decide on these features in an objective way?

3.2.3 Problems with Contingent Valuation Studies

Although the survey method looks very attractive at first glance, its application involves several fundamental problems. That is why time and again doubts are expressed whether these surveys can indeed produce meaningful results (see as an early example of a fundamental critique Diamond & Hausman, 1994). Therefore, we will now present and discuss the most important objections that have been raised against the CVM.

3.2.3.1 Strategic Bias

The results of surveys could be biased as interviewees may try to attain a higher utility level by intentionally making false statements concerning their preferences. Due to the public good characteristic of the good "clean environment," each person would like to take a free ride on others' contributions to environmental protection (see, e.g., Cornes & Sandler, 1996: 30). Thus, if an individual expects his expressed willingness to pay to be positively correlated with real financial burdens for environmental protection falling on him, he has an incentive to understate his preference for the good "clean environment." If, however, an individual instead supposes that he can pass on the costs for environmental protection to someone else (perhaps to the polluters provided the polluter-pays principle is applied), an overstatement of willingness to pay will result.

Economic theory, fortunately, shows that at least the problem of strategically motivated false reporting of preferences does not constitute an insurmountable obstacle for CVM studies. Rather, by a smartly designed incentive scheme, response behavior can be influenced in such a way that an interest in giving wrong answers no longer exists. Nobel laureate William Vickrey (1961) was the first to design such an incentive mechanism for ensuring truthful reporting of preferences that later on lead to the Vickrey-Clarke-Groves VCG mechanism (see Cornes & Sandler, 1996: 221–229 and Jehle & Reny, 2011: 461–465 with references also to the original papers by Clarke & Groves).

We now describe a simplified version of this preference-revelation mechanism in which we consider an economy with only two individuals $i = 1, 2$. Generalization to the case of an arbitrary number n of individuals, however, is completely straightforward. In this economy, a welfare-maximizing government has to make a dichotomous decision whether it should provide a specific environmental public good or not. The costs of producing this public good are denoted by c. The true benefit b_i that individual $i = 1, 2$ receives from the public good is assumed to be only known to the respective individual, but not to the other individual and the government, so that a situation of asymmetric information prevails. Yet, the government, seeking to maximize social welfare, has to have correct information about the individual benefits b_1 and b_2. How the VCG mechanism is able to prevent incentives for false reporting and thus to provide the required information about the true individual benefits will be demonstrated next.

To this end we assume that agent 1 (agent 2) reports an individual public good benefit $\tilde{b}_1$ ($\tilde{b}_2$), where the tilde indicates that $\tilde{b}_i$ are the benefits as *declared* by agent *i*. These stated preferences may well deviate from his *true* benefits b_i, which is the source of the problem.

Depending on the stated preferences $\tilde{b}_1$ and $\tilde{b}_2$, the VCG mechanism now specifies the government's actions in two respects: on the one hand, whether the environmental good should be provided or not and on the other hand which tax payment should be imposed on agent 1 and agent 2, respectively.

- If $\tilde{b}_1 + \tilde{b}_2 < c$, the environmental public good is not provided, and none of the agents has to pay a tax.
- If $\tilde{b}_1 + \tilde{b}_2 \geq c$, the environmental public good is provided, and the agents have to pay a "Clarke tax" according to the following rules:
  - Individual 1 pays nothing if $\tilde{b}_2 \geq c$. Otherwise, i.e. if $\tilde{b}_2 < c$, he has to pay $T_1 = c - \tilde{b}_2$ as a tax to the government.
  - Individual 2 pays nothing if $\tilde{b}_1 \geq c$. Otherwise, i.e. if $\tilde{b}_1 < c$, he has to pay $T_2 = c - \tilde{b}_1$ as a tax.

Hence, the level of the Clarke tax that an individual has to pay equals the difference between the costs c of producing the public good and the individual benefit that the other agent declares. An agent has to pay a Clarke tax only if the decision whether to provide the public good or not changes due to his preference statement. Note that the tax payments that are the crucial element of this preference-revelation scheme are just incentive taxes which are not raised with the intention to finance the public good. Even though the Clarke tax revenues can also be employed for this purpose, there is no direct connection between them and the costs of public good provision.

The task now is to check which public good benefits rational agents will report when they correctly anticipate the effects of the VCG mechanism. We consider the behavior of individual 1 and determine (as is usually done in the non-cooperative game theory) his optimal reactions to some stated preferences $\tilde{b}_2$ of individual 2. In particular, we explore how – given the rules of the mechanism – individual 1 maximizes his utility via his response behavior. For some given $\tilde{b}_2$, different possible cases have to be distinguished.

Case 1 If $\tilde{b}_2 \geq c,$ then the environmental public good will be produced, regardless of the preferences that individual 1 declares. Individual 1 pays no incentive tax and attains the net benefit b_1. A false reporting of preferences could not raise individual 1's benefit level beyond b_1. Consequently, he has no incentive to lie.

Case 2 If $\tilde{b}_2 < c,$ then individual 1 decides through his preference statement whether the public good will be provided or not. We distinguish two sub-cases:

- *Sub-case 2a:* $b_1 + \tilde{b}_2 < c$
  Given the true valuation of individual 1, the provision of the environmental public good is not worthwhile.
  Let us now examine whether the truthful reporting of $\tilde{b}_1 = b_1$ is the optimal choice for individual 1: if the true valuation is reported by individual 1, the public good will not be provided. Then, individual 1 does not have to pay anything, and his net benefit is zero. If he states untrue preference $\tilde{b}_1$ instead and $\tilde{b}_1$ remains in the range $\tilde{b}_1 < c - \tilde{b}_2,$ then he will attain the same result as if he had declared the truth. The false declaration would not give him any advantage.
  Finally, if individual 1 reports an untrue preference $\tilde{b}_1$ that is located in the range $\tilde{b}_1 \geq c - \tilde{b}_2$ and thus overstates his appreciation of the public good, then this would induce the provision of the public good by the government. Yet, in this case, agent 1 will have to pay the incentive tax of $T_1 = c - \tilde{b}_2$ so that individual 1's net benefit becomes $b_1 - T_1 = b_1 - c + \tilde{b}_2,$ which is smaller than 0 as $b_1 + \tilde{b}_2 < c$ holds in this sub-case. Consequently, the net benefit, which is attained through lying in this case, is lower than the net benefit (of zero) that can be received by truthful reporting. Hence, lying never pays in this sub-case.
- *Sub-case 2b:* $b_1 + \tilde{b}_2 \geq c$
  If individual 1 correctly reveals his preferences for the environmental public good, i.e. if $\tilde{b}_1 = b_1$, then the government produces the public good, and individual 1 has to pay the Clarke tax $T_1 = c - \tilde{b}_2$. His real after-tax net benefit thus becomes $b_1 - T_1 = b_1 - \left(c - \tilde{b}_2\right) > 0,$ since $b_1 + \tilde{b}_2 \geq c$ has been assumed for this sub-case. If

individual 1 instead declares a high false benefit $\tilde{b}_1$ in the range $\tilde{b}_1 \geq c - \tilde{b}_2$, then he would attain the same outcome as with a truthful answer. Consequently, lying will not provide any advantage in this range.
If individual 1, however, reports a low false benefit $\tilde{b}_1$ in the range $\tilde{b}_1 < c - \tilde{b}_2$, there will be neither a provision of the public good nor a tax payment by individual 1. Then his net benefit becomes zero, while truthful reporting gives individual 1 the positive net benefit $b_1 - \left(c - \tilde{b}_2\right)$. Hence, false reporting would harm individual 1 in this sub-case.

In a completely analogous way, we could also show for individual 2 that lying about his true preferences for the public good never pays. In game-theoretic terminology, truthful revelation of preferences thus constitutes a (weakly) dominant strategy for both agents. The unique Nash equilibrium of this game, i.e. the outcome where each agent maximizing his personal welfare gives his best response to the best responses of the other agents is where both individuals will make correct statements. Therefore, the described VCG mechanism operates as desired. Consequently, from the theoretical perspective, no fundamental reasons exist why asymmetric information should distort the outcome of CVM studies throughout.

? Control Question

Why is truth-telling on all sides the Nash equilibrium in the non-cooperative preference elicitation game under the VCG mechanism?

In large groups incentives for strategic behavior may be of minor extent, even when there is no such complicated incentive scheme like the VCG mechanism applied. Referring to the political process in a democracy, the Norwegian economist Leif Johansen (1977) argued that an agent's strategy of understating his preferences for the public good in order to lower his share in financing the good is unlikely "to succeed in an open political decision-making process involving elected representatives" (Johansen, 1977: 147). The main reason for that simply is that – in a naturally intransparent process – people may weight the danger of non-provision of the public good higher than the danger of having to carry high costs. Hence, Johansen's consideration can also be of some relevance for CVM studies.

3.2.3.2 The Absence of an Unambiguous Valuation Standard

When the environmental quality or the provision of an environmental public good changes, WTA and WTP represent two equally plausible measurement concepts. From the perspective of microeconomic household theory, it might seem to be irrelevant whether WTA or WTP is applied since the value of a good as measured by its market price reflects both WTA and WTP. However, empirical studies (see, e.g., Horowitz & McConnell, 2002) regularly show that the WTA measure is much larger than the WTP measure.

From the perspective of microeconomics, it might seem obvious to consider these empirical findings as preference-theoretic anomalies, i.e. as a deviation from economically rational behavior of a homo economicus. In this vein, behavioral economics has traced the difference between WTA and WTP to an *endowment effect* (as coined by Thaler, 1980,

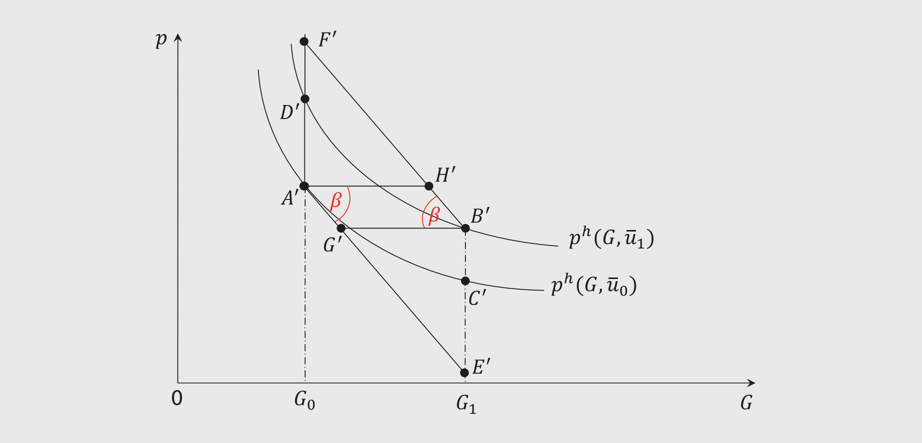


Fig. 3.9 The difference between WTA and WTP

and empirically demonstrated by Kahneman, Knetsch, & Thaler, 1990) and to *loss aversion* as theoretically described by the *prospect theory* of Kahneman & Tversky's (1979). *Endowment effect* means that people value goods they possess more than goods they have to acquire. *Loss aversion* implies that losses against a reference point are weighted more than gains (Kahneman et al., 1991).

This reference to behavioral approaches, although being interesting in itself, may to a certain degree not be required. Rather, if the level of the environmental good increases by more than a marginal amount, it can be expected even from the point of view of standard neoclassical household theory that the WTA can become much larger than the WTP. To demonstrate this also for the case of a direct quantity change of the environmental good G, we now again apply Hicksian demand curves (see Fig. 3.9).

Going back to Fig. 3.8, the two indifference curves belonging to utility levels $\bar{u}_0$ and $\bar{u}_1$, respectively, are described by the two functional forms $\varphi_0(G)$ and $\varphi_1(G)$ depending on G. Then, as $WTA = \varphi_1(G_0) - \varphi_0(G_0)$ and $WTP = \varphi_1(G_1) - \varphi_0(G_1)$, the difference between WTA and WTP can be expressed as

$$WTA - WTP = \left(\varphi_1(G_0) - \varphi_0(G_0)\right) - \left(\varphi_1(G_1) - \varphi_0(G_1)\right)$$

$$= \left(\varphi_1(G_0) - \varphi_1(G_1)\right) - \left(\varphi_0(G_0) - \varphi_0(G_1)\right)$$

$$= \int_{G_0}^{G_1} \left(-\varphi_1'(G)\right) dG - \int_{G_0}^{G_1} \left(-\varphi_0'(G)\right) dG$$

$$= \int_{G_0}^{G_1} \left(p^h(G,\bar{u}_1) - p^h(G,\bar{u}_0)\right) dG,$$

where $p^h\left(G,\bar{u}_i\right)$ again is the inverse Hicksian demand function corresponding to the utility level $\bar{u}_i$.[2] In ◘ Fig. 3.9, the difference between WTA and WTP hence is – analogous as in the case of price changes – represented by the area between both inverse Hicksian demand functions within the limits G_0 and G_1, i.e.

$$WTA - WTP = A'C'B'D',$$

if the inverse Hicksian demand function $p^h\left(G,\bar{u}_1\right)$ lies to the right of the inverse Hicksian demand function $p^h\left(G,\bar{u}_0\right)$. Yet – as we have already noted above – normality of the environmental good implies $G^h\left(p,\bar{u}_1\right) > G^h\left(p,\bar{u}_0\right)$ for all p, which means that $p^h\left(G,\bar{u}_1\right) > p^h\left(G,\bar{u}_0\right)$ (for all G). In this case, $A'C'B'D'$ has a positive value such that in the case of a normal good, indeed the WTA will be higher than the WTP.

We will now go one step further and ask on which factors the extent of the difference between WTA and WTP depends (see also Hanemann, 1991; Ebert, 1993). For this purpose we take another look at ◘ Fig. 3.9. By μ we denote the maximum absolute value of the slopes of both inverse Hicksian demand functions $p^h\left(G,\bar{u}_0\right)$ and $p^h\left(G,\bar{u}_1\right)$ on the interval $[G_0, G_1]$. When the angle β is given by $\mu = \tan\beta$, then the area $A'C'B'D'$, which describes the difference between WTA and WTP, is enclosed by the parallelogram $A'E'B'F'$. Going back to ◘ Fig. 3.8 shows that, as a consequence of normality, the indifference curve $\bar{u}_1$ is flatter in point B than the indifference curve $\bar{u}_0$ in point A so that $p^h\left(G_1,\bar{u}_1\right) < p^h\left(G_0,\bar{u}_0\right)$ holds. Hence, in ◘ Fig. 3.9 the point B' lies below A', which gives $A'H' < G_1 - G_0$. The area of the parallelogram $A'E'B'F'$, which is

$$\left(G_1 - G_0\right)\cdot A'F' = \left(G_1 - G_0\right)\cdot \mu \cdot A'H',$$

thus is smaller than $\mu \cdot (G_1 - G_0)^2$. All in all we obtain an upper bound for the difference between WTA and WTP, i.e.

$$WTA - WTP < \mu \cdot \left(G_1 - G_0\right)^2.$$

Thus, the difference between WTA and WTP can only become large if $\mu = \tan\beta$ is large. A high value of $\tan\beta$, which results from steep inverse Hicksian demand functions, indicates that the Hicksian demand for the environmental good reacts little to a change in price and therefore is highly price-inelastic. In particular, this is the case if the good G cannot easily be substituted by other goods so that a low degree of substitutability proves to be a necessary condition for a large difference between WTA and WTP.

Note in this context that steepness of the Hicksian demand curves is not a sufficient condition for getting a large difference between WTA and WTP: for a quasi-linear utility function of the type $u(x, G) = x + \varphi(G)$, the Hicksian demand curves are the same for all utility levels such that $p^h\left(G,\bar{u}_0\right) = p^h\left(G,\bar{u}_1\right)$ holds for all $G > 0$. Then, area $A'C'B'D'$ in ◘ Fig. 3.9 vanishes. Nevertheless, our analysis highlights the importance the price elasticity of environmental goods has for the size of the difference between WTA and WTP.

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2 Recall that in a household optimum, the absolute slope of the indifference curve has to equal the price ratio between the two goods and thus the value of the inverse Hicksian demand function.

Control Question
Why is the slope of the Hicksian demand functions responsible for the size of the difference between WTA and WTP? Why is this result important especially with respect to the assessment of environmental quality changes?

Especially in the case of environmental goods, a low degree of substitutability does not seem to be unusual so that a large difference between WTA and WTP does not come as a surprise in these cases. The destruction of landscapes and the extinction of animal species are irreversible processes that destroy unique natural assets for which no direct substitutes exist. Already Krutilla (1967) has emphasized the low degree of substitutability of environmental goods and, in a quite informal way, even gave some hints that this may cause a difference between willingness to pay and willingness to accept. So he notes that the:

> Maximum willingness to pay … may be significantly less than the minimum which would be required to compensate such individuals were they to be deprived in perpetuity of the opportunity to continue enjoying the natural phenomenon in question. (Krutilla, 1967: 779–780)

In contrast, the WTA and WTP measured in the case of goods like candy bars are relatively close to each other if the sweets that were given to the test persons in a WTA scenario can be easily purchased again in a shop around the corner so that substitutability is high. In an experiment, which compares WTA and WTP, Shogren et al. (1994) find that the discrepancy between WTP and WTA disappears with repeated exposure to the market and experience with market transactions (see also List, 2003). The difference between WTA and WTP in general tends to be small for market transactions. In this context, Randall & Stoll (1980) bring forward the argument that goods sold in competitive markets with zero transactions costs resemble money and thus represent perfect substitutes.

However, for CVM studies in environmental economics such observations are only of indirect relevance. Rather it has to be taken serious that just for environmental goods, wide gaps between WTA and WTP are by no means an exception and should cause no irritation and should not necessarily be attributed to preference anomalies (Kahneman et al., 1991). Yet, in face of this expectable outcome, it has to be carefully observed in a CVM study whether in the specific case one is interested in the individuals' WTP or in their WTA. The choice of the valuation standard that matters less in valuation of ordinary market goods may make a big difference in case of environmental goods.

3.2.3.3 The Special Role of Ethical Benefit Components

Ethical values represent special non-use values whose adequate recording is an important objective for CVM studies. However, a strong influence of ethical motives may cause significant problems with respect to the validity of CVM studies.

For many individuals, ethical behavior is increasing their personal welfare, i.e. individuals draw a satisfaction from applying their ethical principles and from the demonstration of their ethical motives. Ethical behavior (toward charities and the environment) then can be interpreted as *warm-glow giving* (Andreoni, 1990) where *warm glow* stands for an individual's positive feeling when he does something good. Striving for such a warm glow and thus for moral satisfaction, "the public good is a means to an end – the consumption is the sense of moral satisfaction associated with the contribution" (Kahneman & Knetsch, 1992: 64).

In contrast to donations to a public good that imply real payments by the donor, CVM surveys even allow the respondents to get some warm glow free of charge. Consequently, it has to be expected that the respondents overstate their preferences for a specific environmental good and use CVM studies for some kind of *cheap talk*, i.e. to support a good cause by a statement of intent that is non-binding and hence does not trigger payments. Empirical studies confirm the relevance of this phenomenon, which distorts the results of CVM surveys: in several CVM studies (see, e.g., Seip & Strand, 1992), test persons were asked in a first round, which amounts of money they were willing to donate for an environmental protection organization. In a second round, the interviewees received membership forms for the same environmental protection organizations. It turned out that the actual payments (membership fees and donations) of many respondents were far below the willingness to pay they had stated in the first round of the study. Therefore, a discrepancy between stated and revealed preferences clearly emerged.

Due to the influence of ethical benefit components, also double counting may occur: if two environmental goods are valued independently of each other and with a time lag, the respondents may include a general (and diffuse) ethical utility component ("the protection of the environment is something important and valuable, which I intend to support") twice in their assessment. If one would evaluate the two items simultaneously instead, the individuals would declare this general ethical utility component only once.

The risk of double counting clearly does not imply that ethical value components should not be taken into account at all. Ethical values are important components of individual preferences in particular in the context of environmental problems, which from an economist's point of view are in principle as significant as the values resulting from the concrete usage of an environmental good. However, their precise assessment and evaluation involve significant demarcation problems, which are not only of technical nature.

Control Question

What does *warm-glow of giving* mean? Why does it imply a risk of double counting in CVM studies?

3.2.3.4 Delimitation Problems

This problem area consists of two parts, the first one of which relates to the set of interviewees and the second one to the number of projects to be evaluated.

Particularly in the case of environmental assets, the beneficiaries (or in the case of environmental harm the victims) as a whole are frequently hard to determine. The high importance of ethical (non-use) utility components implies that also long-range effects on utility have to be taken into account for a correct assessment of environmental benefits and damages. So, inhabitants of industrialized countries are also concerned about the biodiversity loss in tropical forests, for example. Possibly, a CVM study on this topic actually ought to mainly focus on this group because the individuals in industrialized countries may even have a stronger interest in the respective environmental problem than the people living in or nearby the forest – who may even benefit from deforestation. However, the problem arises that the number of potentially affected people that should be interviewed may become extremely large and thus unmanageable because of its size. With respect to non-use values, frequently an unambiguous criterion does not exist that helps to relate a clearly defined circle of affected people to a particular environmental problem.

Control Question

Why is the set of people that is affected by a change of environmental quality often not well-defined? Give some examples.

Moreover, it is not clear in which way the environmental benefits and damages should be elicited that fall on people not yet born, who clearly cannot be interviewed today. Empirically, many environmental problems (as climate change or ozone layer depletion) affect future generations significantly so that a complete monetary evaluation of environmental quality changes is not possible without taking future costs and benefits into account. To this end, two conceptually different approaches are available.

Approach 1 Transferring utility values articulated by individuals living today to the members of future generations.

Extrapolating stated preferences from the present into the possibly far-distant future raises the question how to deal with uncertainty, which considerably increases with growing temporal distance. Future generations will perhaps have quite different preferences for environmental goods than people living today. It is also conceivable that biotechnological progress can enable the substitution of disappearing environmental assets, which might render it possible that extinct animal species are to some extent reproduced by genetically modified organisms. Furthermore, it is hard to predict the demographic development and thus the exact number of individuals which will consume an environmental good affected in the future. Yet, this number crucially determines aggregate utility of future generations and thus the results when the extrapolation method is applied.

Another important question that cannot be discussed here in detail is to which extent utility of future generations should be discounted. As Thomas C. Schelling (1995: 395) explains, "[e]conomists who deal with very long-term policy issues, like greenhouse gas emissions over the next century or two, are nearly unanimous that future benefits … need to be discounted to be commensurable with the consumption earlier forgone to produce those benefits." Ethical reasons as the claim for an equal treatment of all generations would instead require that future generations essentially get the same weight in a valuation as the present generation (see, e.g., Roemer, 2011; Stern, 2015: 151–184). From this perspective, only as far as there is some risk that the world and with it future generations will cease to exist, a positive discount rate is justified.

If one – for whatever reason – denies the postulate of an equal treatment of all generations and considers discounting as basically justified, it remains unclear how high the level of the social discount rate should be chosen. Should the social discount rate be derived from market interest rates (see, e.g., Nordhaus, 2013: 182–194) or should it at least partly be based on ethical reasoning? The problem is further complicated as environmental quality may have an impact on the discount rate, e.g., individuals living in a clean environment tend to enjoy the future more than individuals living in a polluted one. "Assuming that the rate of time preference is negatively correlated with the state of nature, economies that are endowed with a very low stock of natural resources would face high rates of time preference" (Pittel, 2002: 83).

Approach 2 Utility of future generations is only taken into account via the altruistic preferences of the current generation.

The basic justification for this approach is that only individuals living today can articulate their preferences in a CVM study – and only they are capable of making decisions on the management of environmental assets. This pragmatic concentration on the present generation clearly does not mean that the welfare of future human beings is completely neglected. Rather, the interests of posterity are indirectly taken into account because currently living people also benefit – due to their intergenerational altruism and their, e.g., religiously conditioned feeling of responsibility, as important ethical non-use values – from rising welfare of future generations and from avoiding environmental risks for them. Even though this approach might help to avoid some of the problems discussed above, it causes serious ethical problems especially as it does not grant future generations an independent value. Their interests only count through the perception of these by the present generation.

Which of these two approaches should be preferred is a philosophical question that cannot be answered without invoking value judgments. However, it is obvious that the assessed utility values will differ significantly between both approaches.

Control Question

What approach for taking the interest of future generations into account in CVM studies would you personally prefer? Give a reasoned reply.

Yet, the result of a CVM study does not only depend on the set of interviewees but also on the sequence and composition of the items addressed in a survey. The ensuing *sample bias* results from a general phenomenon that is well known from microeconomic theory: if an individual, who has a fixed budget, is offered two goods gathered in a basket, he is willing to pay less for these goods than he would be if these goods were offered to him separately. Hence, it is not clearly determined what can be regarded as the individual's valuation of a single good. The higher the number of goods being offered at once, the bigger is this discrepancy and thus the related bias.

The phenomenon that the willingness to pay decreases with a growing number of offered goods occurs both in connection with public and private goods. However, in the case of private goods, this implies less trouble because the market generates a preselection of goods. In marketing studies that are conducted before the market launch of a consumer good, the good to be valued is already specified. Furthermore, the aim of the valuation procedure is not to determine an objective value of this good but more modestly to obtain information about its market opportunities. In contrast, in CVM studies valuing environmental assets, the interviewers in principle can more freely decide on the sample, i.e. the nature and scope of the projects being included in the survey and how a single project is "embedded" in the entire sample. Hence, "the designer of a contingent valuation survey may be able to determine the estimated value by the choice of a level of embedding. This potential for manipulation severely undermines the contingent valuation method" (Kahneman & Knetsch, 1992: 64). These problems caused by the *embedding effect* do not simply arise from imperfections in specific applications of the CVM approach but rather are fundamental ones that cannot be overcome by further refinements of the evaluation technique.

Related to the embedding effect is the *scope effect*, which refers to differently sized samples of the same environmental good. A stark example is a survey study that asked for the willingness to pay for the protection of rare birds. It turned out that the reported willingness to pay for the protection of 100.000 birds was only insignificantly higher than the willingness to pay for the protection of 1.000 birds. In this context, the warm-glow effect also may have strongly influenced the outcome (see Desvousges et al., 1993). In a similar vein, Diamond & Hausman (1994) give an example where the willingness to pay to clean one lake is approximately equal to the stated willingness to pay to clean up this lake and additional four ones.

Control Question
Why is the valuation of a single good dependent on the basket in which it is presented?

3.2.3.5 Badly Informed Interviewees and the Hypothetical Bias

In general, individuals tend to be less well informed about public goods, in particular about their quality attributes and costs, than about the private goods they consume. This disparity of information is a consequence of rational behavior of individuals: if an individual acquires information about a private good that he intends to purchase, this helps him to make a better choice. The costs of information procurement, e.g., through consulting Consumer Reports in the USA or the Stiftung Warentest in Germany, thus are offset by the utility gains resulting from a well-founded decision.

With respect to a public good, the individual's situation is entirely different: irrespective of his information about a specific environmental good, an individual knows that he will, as a single member of a large group, have only a minor influence on the decision whether the considered public good is provided or not (see, e.g., Johnston, 2006). From a rational agent's point of view who compares costs and benefits of information procurement, the acquisition of precise information about the considered environmental good therefore is not worthwhile. Consequently, a rational agent will use his scarce resources like time and money for other purposes instead. This problem of a generic information deficit is aggravated by additional factors:

- Because of the uniqueness of the decision about environmental assets, individuals lack the possibility of a valuation based on experience.
- In many cases, the procurement of information about private goods is much cheaper than the acquisition of information about public goods: for individuals the correct valuation of environmental benefits and damages frequently requires special knowledge of natural or engineering sciences, e.g., on the assimilation capacity of a spoilt river.
- The assessment of preferences in CVM studies is mostly of a hypothetical nature, i.e. its connection with environmental policy measures that are actually taken is not very close. Hence, the influence on real decisions that is perceived by the

respondents is even smaller than in the usual political process as, e.g., in referanda, which further reduces the incentives for acquiring detailed information on the subject of investigation. The hypothetical nature of CVM surveys in general makes it likely that answers are not well-thought expressions of stable preferences but are rather based on "gut instinct," which in particular may lead to an overstatement of preferences (see, e.g., Hausman, 2012). Results that follow from "thinking fast" (in the sense of Kahneman, 2011) do not seem to provide a sound foundation for welfare judgments.

Due to the initially prevailing lack of information, there is the necessity to instruct the interviewees sufficiently about the environmental project that is to be evaluated. Only then the interviewee is able to give really useful answers. Yet, there is a risk that the survey results are not independent of the provided information, so that there is a danger of some *information bias*. Only because the interviewer has emphasized the importance of the environmental problem to be assessed, a respondent may get the impression that he is expected to express a high willingness to pay. In general, the danger cannot be excluded that the interviewer, even unconsciously, exerts some moral pressure on the test persons to give "politically correct" answers that do not reflect their real attitudes. An anonymous survey would in this respect be less prone to such biases.

Control Question

What makes the "choice" you are making in a CVM study different from your choice as a consumer on a private goods market?

3.2.4 Conclusions for the Design and Validity of CVM Studies

In the USA, the Oil Pollution Act of 1990 requires monetary valuation of environmental harm caused by accidents of oil tankers. Therefore, as already mentioned above, also non-use values have to be taken into account. In order to improve the CVM for the assessment of environmental damages, the US authorities had established an expert commission in the early 1990s, the so-called NOAA panel with the Nobel laureates in economics Kenneth Arrow and Robert Solow as two of its members (see Arrow et al., 1993). The report of this commission has – after a critical discussion of the various problems associated with the CVM technique – formulated guidelines for the implementation of CVM studies. The compliance with the rules as suggested by the NOAA panel should bring about results as reliable as possible and should support the further development of the CVM technique.

Some of these rules are common requirements for respectable empirical research in social sciences like sufficient pretesting of the questionnaire, precise description of the evaluated project, and checks of understanding. According to the NOAA panel, also general questions, e.g., with respect to the general attitude toward the environment and big business, should become part of the survey as they "help to interpret the responses to the primary valuation question" (Arrow et al., 1993: 4609). Furthermore, the panel has recommended the use of the referendum format, i.e. of yes-no questions on the acceptance of certain proposed levels of valuation and to add a no-answer option to them, i.e. the possibility to refuse an answer. The respondents also should be made aware not only of their budget constraints but also of possibilities to "substitute" the spoilt environmental asset by other undamaged ones – as well as of the "future (potentially restored, W.B. and D.R.) state of the same natural resource" (Arrow et al., 1993: 4608–4609). Interestingly, to avoid extreme answers, the NOAA panel strictly prefers elicitation by willingness-to-pay questions even though damage assessment would – at least at first glance – rather suggest the willingness-to-accept format.

As plausible as the recommendations of the NOAA panel may appear, they have also been criticized, mainly because many of them are not based on theoretical arguments but on purely pragmatic consideration reasons or just on common sense. Moreover, a thorough discussion of the deep conceptual difficulties that are related to the embedding effect is missing. In this context, a clear statement would have been helpful that a definite monetary value of a certain natural asset does not exist – because of unavoidable double counting of feelings of warm-glow giving, the inevitable information deficits of the interviewees, the impossibility of clearly demarcating the circle of affected people, the generally unsolvable embedding problems, etc. In face of these problems, one has to concede that the outcome of CVM studies is time and context dependent and that the CVM technique sensibly can only be applied to environmental problems of limited scale and scope.

In the case of legal enforcement of claims for compensation for lost non-use values, which gave the very reason for establishing the NOAA panel, many of these problems fortunately are of less importance as many features of the survey are clearly determined already from the outset. Therefore, in the planning phase of a CVM study, fewer decision problems, whose solution unavoidably would require some arbitrariness, arise than in CVM studies on general environmental problems. Consequently, some sources of error that tend to invalidate the results of CVM studies are eliminated:

- The environmental damage to be valued has actually occurred.
- The circle of affected people is legally specified as the group of US citizens.
- The respondents know already a large part of the relevant information from the media.

Even though observation of the NOAA guidelines in CVM studies clearly will not deliver absolute truths, following these rules can provide useful information that improves decision-making in concrete situations. At least one gets some rough estimate of the order of magnitude the monetary value of an environmental asset may have.

With the discussed reservations in mind, in the context of CVM, it indeed holds that "some number is better than no number." Nevertheless, one should be fully aware of the limitations of this valuation method. Almost 20 years after the controversy between Hanemann (1994) on the one hand and Diamond & Hausman (1994) on the other, scientific opinion on the usefulness of stated preference elicitation by means of the CVM method still has remained strongly divided. So, in a relatively optimistic mood, Kling, Phaneuf, & Zhao (2012: 23) state that their "sense is that the last 20 years of research have shown that some carefully constructed number based on stated preference analysis is now likely to be more useful than no number in most instances for both cost-benefit analysis and damage assessment."

But this favorable assessment is confronted with a harsh verdict on the CVM by Hausman (2012: 54) who concludes that " 'no number' is still better than a contingent valuation estimate" and "that unless or until contingent valuation studies resolve their long-standing problems, they should have zero weight in public decision-making."

Control Question

What do you think: Is in the context of the monetary evaluation of environmental quality changes "some number better than no number"? Try to give a personal judgment using the knowledge you have acquired in this chapter.

Meanwhile there exists a vast variety of CVM studies dealing with different environmental problems, protection of landscapes, and habitats or species conservation (see, e.g., Willis & Guy, 1993; Kotchen & Reiling, 2000; Jakobsson & Dragun, 2001; Hammitt & Zhou, 2006; Ahlheim, Börger, & Frör, 2015; Jones et al., 2017).

A significant number of contingent valuation studies address questions that merely originate from fundamental scientific interest, e.g., the role of the embedding effect is investigated. In some studies, both applied political and scientific theoretical aspects are analyzed at once. A comprehensive introduction to the large and growing literature on contingent valuation is provided by Carson (2012). Oerlemans, Chan, & Volschenk (2016) review the literature on the use of contingent valuation for measuring willingness to pay for green electricity.

Conclusion

You have learnt in this section that:

- The CVM technique seems to be suitable for capturing non-use values that are of particular importance for the evaluation of environmental goods and services.
- A CVM study is specified by a lot of design elements on which decisions have to be made by the investigators.
- External factors in general affect and possibly distort the results of a CVM study.
- The positive difference between willingness to pay and willingness to accept can be expected to be especially large for environmental goods since they are difficult to substitute. The choice between the WTP and WTA approach therefore becomes vitally important.
- It is often not possible to delineate the circle of affected people and to assess the quite relevant ethical values for environmental protection in a clear and objective way.
- Pragmatic guidelines as those from the NOAA panel may improve the quality of CVM studies considerably but do not overcome their deep conceptual problems.

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A Comparison of Environmental Policy Instruments

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Objectives of This Chapter
In this chapter students should learn:
- What the main instruments of environmental policy are
- What advantages price-based instruments, i.e. emission taxes and cap-and-trade systems, have over command and control approaches
- Which problems may occur with emission pricing in specific situations and which may thus put their benefits into perspective
- Which practical experiences have been made with the application of these instruments
- Why – despite their basic advantages – price-based instruments of environmental policy have to be complemented by other measures

4.1 Background

In order to reduce emissions and to improve environmental quality, the government can apply different environmental policy instruments. Empirically still most important are *command and control (CAC) instruments* by which the government directly regulates an emitter's behavior and inflicts sanctions on the emitter in case of misconduct. Such CAC instruments can take different forms. In many cases, an emitter is only allowed to release emissions into the environment up to a certain maximum level, or he is forced to employ a specific abatement or production technology. As we will see below, from an economist's point of view, CAC instruments are associated with grave disadvantages, which are mainly due to their lack of flexibility, regularly causing inefficiency of environmental protection.

Among economists, there is instead a deep-rooted preference for market-based environmental instruments, which use prices to signal the scarcity of the good "clean environment" and which allow emitters to flexibly adapt when choosing their abatement measures. The standard price-based instruments are emission levies or emission taxes whose theoretical ideal is the Pigouvian tax, which we have already presented in ► Chap. 2 of this book. Yet, even under less perfect conditions than required for the application of a Pigouvian tax, emission levies still possess significant merits. We will consider these advantages as well as disadvantages in this chapter.

However, in order to use price signals for internalizing environmental externalities, it is not necessary that prices are exogenously set by the government since a market for emission permits or emission allowances can be established. In such cap-and-trade systems, where the "cap" is the permissible level of aggregate emissions, the emissions price is determined endogenously through trading these permits. How such emissions trading systems work and which problems are associated with them will be another major topic of this chapter.

For the evaluation of different environmental policy instruments, one can employ various criteria. An environmental policy instrument should in particular exhibit:
- *Environmental effectiveness*. "Policies that achieve specific environmental quality goals better than alternative policies can be said to have a higher degree of environmental effectiveness" (IPCC, 2007: 751).

- *Low transaction costs*. This especially means that neither the government nor the polluters should have to bear high costs for acquiring the information that is necessary for the application of and the adaptation to an environmental policy instrument. Furthermore, effective control of polluters should be possible at reasonable costs.
- *Static efficiency* or, synonymously, *cost-effectiveness*, which means that environmental goals should be attained at minimum total costs.
- *Dynamic efficiency*, which means that technological progress bringing about a reduction of the cost for environmental protection should be promoted.
- *Equity* requires benefits and costs that are associated with environmental policy to be distributed fairly among the agents involved, i.e. between polluter and victim, different types of emitters, rich and poor people, different regions, etc.

Conclusion
You have learnt in this part of the chapter what criteria for the assessment of environmental policy instruments exist.

Control Question
What do you think about political feasibility as an additional evaluation criterion?

4.2 Command and Control Instruments

4.2.1 Types of Command and Control Instruments

Two main types of command and control (CAC) policies being addressed to polluters are of particular importance, i.e. technology mandates and performance standards.

Technology mandates or *technology standards* are specific requirements regarding the production process and equipment (Goulder & Parry, 2008: 157; Jaffe & Stavins, 1995: S45), e.g., that all coal-fired electric utilities have to install scrubber types to desulfurize flue gas and that in cars catalytic converters for exhaust gas cleaning have to be applied.

Performance standards require emission levels to be kept within certain limits, which gives the polluters some flexibility in achieving the target. Such standards do not only refer to emissions in absolute terms (e.g., tons of carbon dioxide per year) but also to emissions per unit of economic activity (e.g., limits for carbon dioxide and nitrogen oxide emissions of cars per km). Emissions then can be reduced by means of fuel switching (e.g., from "dirty" coal to "cleaner" natural gas), by improving the technological efficiency of engines, or by a change of technology, i.e. through a transition to electric mobility.

4.2.2 Advantages and Disadvantages of Command and Control Instruments

CAC instruments appear to be appropriate tools for the government to combat environmental problems as they explicitly prevent undesired behavior, i.e. activities harming the environment: actions that impair or threaten other agents are forbidden, as one knows it from many other fields of life. Simply think of regulations on road traffic or food safety.

CAC instruments obviously comply with the polluter-pays principle since they make the polluters responsible for the environmental harm caused by them by obliging them to meet the stipulated emission limits. Thereby, an intuitively comprehensible distribution norm is observed according to which the costs of environmental protection are imposed on the polluter and not on the victim.

Most importantly, CAC instruments promise to guarantee a high level of environmental effectiveness, as they restrict environmental pollution to levels in a direct way. This obvious advantage, however, is thwarted since the connection between the emission level of a single plant and the emissions' harmful effects is not clear-cut but also depends on:

- Various natural factors (as explained already in ► Chap. 1 of this book). Wind force and other ambient conditions (e.g., streets in the countryside vs. deep street canyons), for example, have a strong influence on the concentration of the waste gases in the local atmosphere. These factors may vary strongly over time.
- The interplay between different pollutants, e.g., in the presence of solar radiation, carbon monoxide (CO), methane (CH_4), non-methane volatile organic compounds (NMVOCs), and nitrogen oxide (NO_4) react with other chemical compounds to form ozone.
- The already prevailing environmental load, e.g., the number of cars polluting the concerned location.

The relationship between the actual environmental damages and the emission limits for a single car therefore is quite loose. In the same vein, a cleaner car clearly will have a higher environmental impact if it is in operation over a longer period of time. For all these reasons, the idea that CAC instruments naturally lead to a high level of environmental effectiveness needs to be put into perspective.

With respect to the costs of controlling the polluters' compliance, large differences between various types of CAC instruments exist. Monitoring the observance of emission limits may become very expensive, especially if it is carried out continuously and not on a random basis. In many cases of technology standards (e.g., the obligation to install filters or catalysts in a machine or vehicle), a one-time and thus much cheaper certification before operation starts is sufficient. A certification could be granted not only for particular polluting units, but for general types, which would further reduce control costs but might open the door for abuse and even outright fraud, as the recent diesel scandal shows.

However, from an economist's point of view, the main disadvantage of CAC instruments lies in their great uniformity and the ensuing lack of flexibility. As, what is mostly the case, the particular conditions of an individual polluting unit are not adequately taken into account by a CAC instrument, environmental policy becomes unduly expensive, and welfare losses result. We will take a closer look at this major problem of cost-inefficiency being associated with CAC instruments in a simple model.

Let us assume that there are two firms, which are characterized by their abatement cost functions $R_i(v_i)$ $(i = 1, 2)$. Here, v_i stands for the abatement level of firm $i = 1, 2$. Firm i's marginal abatement costs are $MAC_i = R_i'(v_i)$, and its initial emissions level is e_i^0.

The aggregate abatement level, i.e. the sum of emissions that both firms reduce altogether, that is aimed at by the government is denoted by $\bar{V}$. Given $\bar{V}$, the permissible aggregate emission level is $\bar{E} = e_1^0 + e_2^0 - \bar{V}$.

The objective of environmental policy, i.e. attaining the level $\bar{V}$ or equivalently $\bar{E}$, may be derived – as discussed in ▶ Chap. 2 – from maximizing social welfare by equating marginal benefits of environmental protection with marginal abatement costs. Due to the huge information problems mentioned before such an approach, which strives for reaching a first-best optimum, usually is much too ambitious. For pragmatic reasons, it therefore is frequently assumed that the total abatement target $\bar{V}$ is simply determined by a political process, without solving an intricate optimization problem.

The government's task then is reduced to the minimization of total abatement costs when $\bar{V}$ is to be attained. Formally, this means

$$\min_{v_1, v_2} R_1(v_1) + R_2(v_2)$$

subject to $v_1 + v_2 = \bar{V}$. The corresponding Lagrangian function reads

$$L = R_1(v_1) + R_2(v_2) - \lambda(v_1 + v_2 - \bar{V}).$$

The first-order conditions that have to be met in a cost-minimizing allocation (v_1^*, v_2^*) of abatement efforts of the two firms then are

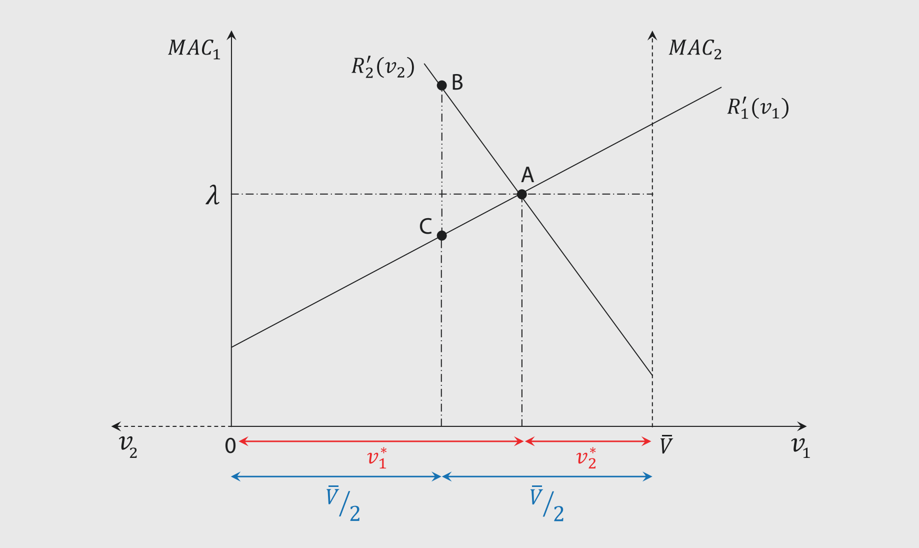
$$\frac{\partial L}{\partial v_1} = R_1'(v_1^*) - \lambda = 0 \text{ and } \frac{\partial L}{\partial v_2} = R_2'(v_2^*) - \lambda = 0.$$

These conditions directly imply that in the optimal outcome in which both firms' aggregate abatement costs are minimized and thus cost-efficiency results, the marginal abatement costs of both firms must be identical, i.e.

$$R_1'(v_1^*) = R_2'(v_2^*) = \lambda.$$

Here, the Lagrangian multiplier λ is the shadow price of emission reductions, which indicates the additional abatement costs that would occur if the environmental target $\bar{V}$ was tightened by a marginal unit. In ◘ Fig. 4.1, the point of intersection between the marginal abatement cost curves of both firms depicts the minimum of aggregate abatement costs.

We can observe from ◘ Fig. 4.1 why a uniform CAC solution, which requires the same emission reduction for both firms, usually does not bring about a cost-effective outcome for the whole society: If each firm is obliged to provide half of total abatement, i.e. $\bar{V}/2$, then – as in ◘ Fig. 4.1 the marginal abatement cost curve of firm 2 is steeper than that of firm 1 – $R_1'(\bar{V}/2) < R_2'(\bar{V}/2)$ results, i.e. the marginal abatement costs of firm 2 exceed that of firm 1. Therefore, a reallocation of abatement efforts from firm 2 to firm 1 would reduce total abatement costs. The potential for cost saving through such an arbitrage is fully exploited when marginal abatement costs are equalized, i.e. firm 1



■ **Fig. 4.1** The distribution of abatement efforts among two polluters

abates v_1^* and firm 2 abates v_2^*. The welfare loss (= avoidable costs of emission reduction), which arises in the undifferentiated CAC solution $(\bar{V}/2, \bar{V}/2)$ compared to the cost-minimal solution (v_1^*, v_2^*), is described by the area of triangle ABC.

Control Question

Why do uniform emission standards normally not lead to a minimization of aggregate abatement costs? Give an intuitive explanation.

Box 4.1: The Social Marginal Abatement Cost Curve

Minimization of aggregate abatement costs can also be graphically described in an alternative way, which can – in contrast to ■ Fig. 4.1 – easily be extended to an arbitrary number n of polluters. To keep it simple, we, however, confine the demonstration again to the case $n = 2$ for which we determine the whole economy's aggregate ("social") abatement cost function $C(V)$ given that the aggregate abatement level V is attained at minimum cost. Depending on V, firm 1's abatement level in this cost-effective solution is denoted by $v_1(V)$, and that of firm 2 by $v_2(V)$. Following the same cost-minimization calculus as above (and letting $V = \bar{V}$), $v_1(V)$ and $v_2(V)$ satisfy the first-order condition

$$R_1'(v_1(V)) = R_2'(v_2(V)) = \lambda(V),$$

where now also the Lagrangian multiplier is made dependent on V. The aggregate abatement cost hence is

$$C(V) = R_1(v_1(V)) + R_2(v_2(V)),$$

4

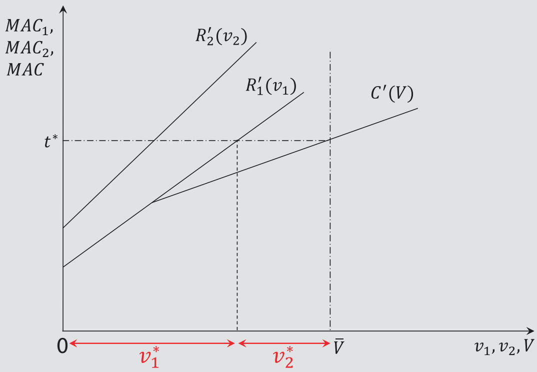
which, by applying the chain rule,[1] gives

$$C'(V) = R_1'(v_1(V))v_1'(V) + R_2'(v_2(V))v_2'(V) = \lambda(V)$$

for the marginal social abatement cost curve. In the calculation, it has been taken into account that $v_1'(V) + v_2'(V) = 1$, which follows from differentiating $v_1(V) + v_2(V) = V$, and that $R_1'(v_1(V)) = R_2'(v_2(V)) = \lambda(V)$ holds.

The marginal social abatement cost function $C'(V)$ above is graphically described by horizontal aggregation of the two individual firms' marginal abatement cost functions $R_1'(v_1)$ and $R_2'(v_2)$ (see Fig. 4.2): if, as required in the cost-efficient solution, these two marginal cost functions attain the same value $\lambda(V)$, then we know from the reasoning above that also $C'(V)$ must take on this value if $V = v_1 + v_2$.

Fig. 4.2 Aggregation of marginal abatement cost curves



Control Question

How is the marginal social abatement cost function obtained when there is an arbitrary number of polluters? Give an intuitive explanation.

From the diagrams presented above, we directly conclude that different slopes of marginal abatement cost curves of different polluters require the abatement levels of polluters also to differ in order to attain cost-effectiveness. Consequently, a uniform CAC instrument leads to a welfare loss in such a case. It is this static inefficiency of CAC instruments which primarily motivates the economists' fundamental skepticism toward these instruments of environmental policy. On these grounds, economists prefer environmental policy instruments that employ an artificially introduced price mechanism and grant the polluters – in comparison to CAC instruments – a greater flexibility when adapting to environmental policy measures. A particular type of such price-based environmental instruments are environmental taxes. We will next discuss in detail the potential advantages of environmental taxes.

1 According to the chain rule it holds for the derivative of a function $f(x) = g(h(x))$ that $f'(x) = g'(h(x)) \cdot h'(x)$.

Conclusion

You have learnt in this section:

- Which basic types of environmental policy instruments exist.
- Why uniform command and control instruments usually imply too high aggregate abatement costs and thus welfare losses.

4.3 Advantages of Environmental Taxes

4.3.1 Cost-Effectiveness in Abating Emissions ("Static Efficiency")

In the model described by ◘ Figs. 4.1 and 4.2, we have demonstrated the lack of cost-effectiveness implied by a uniform CAC instrument. We now show in the framework of this model how the cost-effective outcome can be attained through taxation of emissions. To that end, we assume that an emission tax with the tax rate $t^* = R_1'\left(v_1^*\right) = R_2'\left(v_2^*\right)$ is applied so that total private costs for polluter $i = 1, 2$ are

$$R_i\left(v_i\right) + t^*\left(e_i^0 - v_i\right).$$

In order to minimize his costs, polluter i will choose that abatement level $\hat{v}_i$ for which the derivative of this expression with respect to v_i becomes zero, i.e. for which the first-order condition $R_i'(\hat{v}_i) = t^*$ is met. Since $t^* = R_i'\left(v_i^*\right)$, we thus have $R_i'(\hat{v}_i) = R_i'\left(v_i^*\right)$ and consequently $v_i^* = \hat{v}_i$. A "correctly" chosen emission tax rate (namely, one that is equal to the shadow price λ of emissions in the optimum) gives every polluter the incentive to choose the abatement level v_i^* that is required to attain aggregate abatement $\bar{V}$ at minimum total costs. Further governmental measures are not needed in this ideal scenario to ensure cost-effectiveness.

Note that the targeted total abatement level $\bar{V}$ is not required to coincide with the socially optimal level where – according to the Pigouvian approach – the social marginal abatement cost curve and social marginal abatement benefit curve intersect. Hence, emission taxes also make it possible to attain efficiency gains if the marginal benefits of abatement activities are not exactly known and thus to have "(static) efficiency without optimality" in pollution control.

This pragmatic *price-standard approach* has been devised by William J. Baumol and Wallace E. Oates (1971) who recommended a *pricing* of emissions in order to achieve a predetermined set of *standards*, i.e. a reduction of emissions by some predetermined level (see also Baumol & Oates, 1988, Chap. 11).

Definition

The *price-standard approach* assumes some level $\bar{V}$ of total emission reductions as the target of environmental policy, which is to be attained with minimum total abatement costs.

The information requirements for this approach are much lower than those for the more ambitious Pigouvian approach, which in practice "rarely proved feasible because of our inability to measure marginal social damage" (Baumol & Oates, 1971: 42). That is why the price-standard approach is of high empirical relevance.

Control Question

What is the difference between the Pigouvian approach and the price-standard approach? Why does an emission tax lead to minimum aggregate abatement costs?

So far, the only option for some firm i to adapt to environmental taxes has been the implementation of technical abatement measures. In addition, a firm usually has the option to reduce emissions by lowering its production level. How firms will combine these two options for emission reduction can be shown by an extension of our basic model which leads to the firm's aggregate abatement cost curve $S(v)$.

Box 4.2: Combining Technical Abatement with a Reduction of the Production Level

We consider a polluting firm, which acts as a price-taker on the market for the good that it produces. The price of this good is fixed and equal to p. The firm's costs of producing the output x are described by the cost function $C(x)$ with $C'(x) > 0$ and $C''(x) > 0$; i.e. the total and the marginal production costs rise with increasing production level. When the firm chooses the production level x, then its profit is $G(x) = px - C(x)$. The proportional emission function $e(x)$ that indicates the emissions resulting from producing the output x is given by $e(x) = x$. As before, the firm's technological option (= option a) for emission reduction (e.g., through installations of filters) is described by the abatement cost function $R(v^a)$, where v^a indicates the emission reduction being attained by technological means.

The firm's costs resulting from the second abatement option (= option b) are denoted by $Q(v^b)$, where v^b indicates the emission reduction that results from lowering the production level. $Q(v^b)$ is the decline in the firm's profits as a result of the production cut, i.e.

$$Q\left(v^b\right) = G(\bar{x}) - G\left(\bar{x} - v^b\right),$$

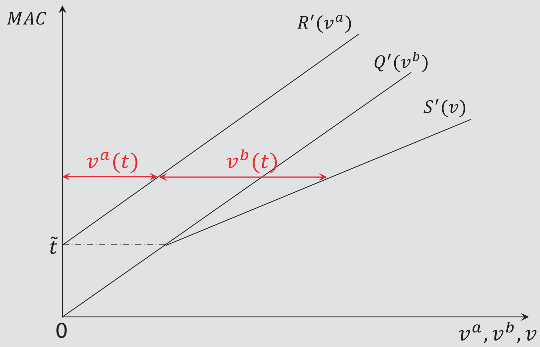
where $\bar{x}$ stands for the production level that would maximize the firm's profits if no cuts of production were made, which is characterized by condition $p = C'(\bar{x})$ (price = marginal production costs). The first and second derivatives of $Q(v^b)$ are $Q'\left(v^b\right) = p - C'\left(\bar{x} - v^b\right) > 0$ and $Q''\left(v^b\right) = C''\left(\bar{x} - v^b\right)$, which means that the marginal abatement cost curve $Q'(v^b)$ is increasing in v^b given the assumptions made for cost function $C(x)$. At $v^b = 0$, we have $Q'(0) = p - C'(\bar{x}) = 0$. The firm has to decide how to distribute the total abatement level v between the two abatement options, i.e. to choose $v^a(v)$ and $v^b(v)$ with $v^a(v) + v^b(v) = v$ to minimize its total costs for achieving the emission reduction v. In the cost-minimum solution, the marginal costs of both abatement options obviously must coincide, i.e.

$$R'\left(v^a(v)\right) = Q'\left(v^b(v)\right)$$

must hold. This implies that the firm's aggregate marginal abatement cost function $S'(v)$ is graphically obtained by the horizontal aggregation of the individual marginal abatement cost functions $R'(v^a)$ and $Q'(v^b)$. In this way, the observations we made before in the case of two different polluters are transferred to the case of two different abatement options for one and the same polluting firm.

Note that usually it cannot be expected that $R'(0) = 0$, since already the very first technically abated unit of emissions incurs cost. Therefore – in contrast to $Q'(v^b)$ – the function $R'(v^a)$ generally will not start in the origin of coordinates so that for low levels of emission reduction and thus for a rather unambitious environmental policy, abatement activities will be pursued through a cut in the production level. Then this kind of adaptation to environmental policy represents the cheaper option from a firm's point of view.

Fig. 4.3 Combining a polluter's abatement options



Control Question
What does curve $S'(v)$ in Fig. 4.3 describe? Explain!

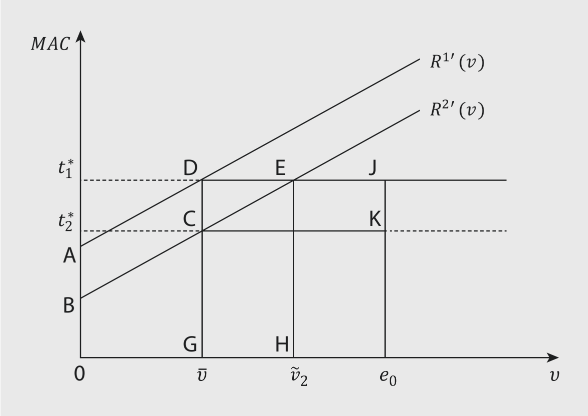
If now an environmental tax with the tax rate t is levied, the firm will choose the abatement level $v(t)$ for which $S'(v(t)) = t$ holds and its abatement efforts are distributed between the two abatement options according to the condition $R'(v^a(t)) = Q'(v^b(t)) = t$ (see Fig. 4.3). If the tax rate is below the threshold value $\tilde{t}$, then the abatement of emissions will exclusively be accomplished via a cut in the production level. At the macroeconomic level, jobs may be lost in this case: on the one hand, employment is reduced by the decline in production activity, while, on the other hand, there is no new job creation due to the development and production of abatement technologies (as they are not used by the firm). From this perspective – somewhat surprisingly – a more ambitious environmental policy also might have some economic advantages.

Against this background, we can summarize the effects of environmental taxes as follows: they bring about an equalization of marginal abatement cost *among firms*, and, at the same time, they induce *within each firm* a cost-efficient distribution of abatement efforts between different abatement options.

4.3.2 Stronger Incentives for Innovation ("Dynamic Efficiency")

Emission taxes not only induce the efficient application of a given abatement technology but also have a positive effect on the polluter's choice between several available abatement technologies, i.e. he may switch to yet unapplied "better" technologies (for other effects of environmental policy instruments on technological change, see, e.g., the surveys by Löschel (2002) and Requate (2005)). To make this precise, we assume that the

Fig. 4.4 Incentives for changing the abatement technology: a comparison of instruments



polluter initially employs the abatement technology 1 that is characterized by marginal abatement costs $R^{1\prime}(v)$. Alternatively (and not additionally!) the polluter can implement another abatement technology 2, whose marginal abatement costs $R^{2\prime}(v)$ are below $R^{1\prime}(v)$, i.e. $R^{2\prime}(v) < R^{1\prime}(v)$ holds for all $v > 0$ (see, e.g., Downing & White, 1986). The transition from technology 1 to technology 2 incurs fixed costs $F > 0$, e.g., because the installation of an improved desulfurization device involves investment costs. Consequently, there are two opposing effects on abatement costs when the new technology 2 is applied: on the one hand, there is a reduction in running costs, while, on the other hand, there is an increase in costs due to the investment costs.

Let us now consider whether the polluter has an incentive to replace the old abatement technology 1 with new technology 2. In doing so, we compare different types of environmental instruments.

First of all, we deal with the case in which the government uses a CAC instrument, i.e. an abatement standard $\bar{v}$, that has to be met by the polluter within a specified period of time. To determine how the polluter can fulfil this abatement requirement with the lowest possible costs, let us regard Fig. 4.4.

Through the transition from the old abatement technology 1 to new abatement technology 2, the running costs for abating $\bar{v}$ fall by the area between the old marginal abatement cost curve $R_1'(v)$ and the new one $R_2'(v)$, i.e. by the area $ABCD$ in Fig. 4.4. The polluter will compare this cost reduction with the fixed costs F incurred by the new technology, which implies that the polluter will change the abatement technology as long as $ABCD > F$. If $ABCD = F$, he will be indifferent between both technologies; if $ABCD < F$, he will stay with the old technology.

In a next step, let us consider how an environmental tax will influence the technology choice in the same situation if the tax rate is fixed at $t_1^* := R^{1\prime}(\bar{v})$, i.e. at that level which would induce the abatement level $\bar{v}$ given the initial abatement technology 1. If the polluter shifts to the new technology 2, then he will choose the higher abatement level $\tilde{v}_2$, which is characterized by condition $R^{2\prime}(\tilde{v}_2) = t_1^*$. He will thus have additional current abatement costs which are described by the area $GHEC$ in Fig. 4.4. At the same time, due to the lower marginal abatement costs of the new technology 2, he saves current abatement costs as given by the area $ABCD$, and, in addition, tax payments reflected by the area $GHED = t_1^*(\tilde{v}_2 - \bar{v})$. In total the technology switch reduces the polluter's running costs by the area *ABED*. Consequently, the environmental tax with the tax rate t_1^* brings about a transition from old technology 1 to new technology 2 if and only if $ABED > F$ holds.

The area *ABED* (cost savings in the case of an emission tax) exceeds the area *ABCD* (cost savings in the CAC case) by the area of the triangle *CED*. If $ABCD < F < ABED$, a change in technology takes place in the tax case but not in the CAC case. Hence, as compared to a fixed emission standard, the development and application of – in terms of running abatement costs – "cheaper" abatement technologies get more likely when an emission tax is applied and, moreover, the abatement level will be higher. This simple consideration leads to the conclusion that environmental taxes can be expected to promote environmentally friendly technological progress more effectively than CAC policies and induce more abatement activities. These advantages are of significant ecological and economic importance in particular in the longer term.

Control Question

Which are the two partial cost-saving effects, which a shift to the second abatement technology entails in the case of a fixed emission tax rate?

However, in face of our previous analysis, we have to notice that the abatement target $\overline{v}$ will no longer be attained with the new technology. Hence, some conflict arises between the strong dynamic incentive effect of the environmental tax on the one hand and the objective of the price-standard approach on the other hand. This problem can be addressed in the following way:

- One can interpret the abatement target of the price-standard approach in an asymmetric manner: it is not allowed to undershoot the emission abatement target, but an overshooting is appreciated for reasons of the precautionary principle. As O'Riordan & Jordan (1995: 193) explain, the precautionary principle emerged during the 1970s and "[a]t the core of early conceptions of precaution […] was the belief that the state should seek to avoid environmental damage by careful forward planning." It is now found in many international documents such as the Rio Declaration on Environment and Development from 1992. Principle 15 of this Declaration states: "In order to protect the environment, the precautionary approach shall be widely applied […]. Where there are threats of serious or irreversible damage, lack of full scientific certainty shall not be used as a reason for postponing cost-effective measures to prevent environmental degradation."
- One applies the consideration above to the case in which only a small group of polluters has the possibility to switch to a new abatement technology with lower marginal abatement costs. Since the increase of the abatement activities of the members of this group that results from the introduction of an environmental tax only has a marginal effect on total abatement, in this case a government following the price-standard approach has no incentive to adapt the emission tax rate.

In the other extreme, a polluter as described by ◘ Fig. 4.4 causes the lion's share of emissions. If the government strictly pursues the price-standard approach in this case, it will react to a change in the abatement technology and subsequently modify the tax rate. If there is only one polluter at all, the government would respond to the technology switch (from old abatement technology 1 to new abatement technology 2) with a reduction of the tax rate from t_1^* to $t_2^* := R^{2\prime}(\bar{v}_2)$. If the polluter anticipates this reaction, he can even expect a saving in his running costs amounting to the area *ABCKJD* from the change of his abatement technology. The area *ABCKJD* clearly is larger than the area *ABED*, which describes the saving in running costs due to the technology switch in the case of an unmodified tax rate t_1^*. The reason for this increase in cost savings is that in the case of a "big" polluter, the switch in technologies also brings about a decline in the environmental tax rate. Therefore, the likelihood of a shift to a cost-saving technology is further enhanced when the polluter – acting as a leader of the game – can expect a cut of the emission tax rate. This game structure is analogous to that of the Stackelberg game between two firms, which is well-known from basic courses in microeconomics.

? Control Question

Why is the incentive for a change of the abatement technology greater when the big polluter expects an adjustment of the emission tax rate?

If there are several polluters, an individual polluter has at most a limited influence on the governmental adaptation of the environmental tax rate. Then, an individual polluter may want to act as a free rider by sticking with the old technology 1 and hoping that the other polluters change their abatement technologies, thereby inducing a tax rate decline. Later, in connection with markets for emission permit trading, we will discuss this scenario in more detail.

Conclusion

You have learnt in this section:

- How polluters adapt to emission taxes.
- That under ideal conditions, emission taxes lead to a minimization of abatement costs and spur environmentally friendly technological progress more than emission standards do.

4.4 Problems with Environmental Taxation

According to the arguments presented above, emission taxes are considered to be superior instruments of environmental policy. Yet, the reality is more complex so that in less ideal scenarios, the advantages of emission taxes have to be put into perspective. This sets some limits to the applicability of environmental taxes in reality. At least it is necessary to inspect carefully the specific characteristics of the environmental problems that are addressed by environmental taxation. Some problems that are associated with environmental taxes will now be discussed with help of various simple models.

4.4.1 Differences of Environmental Impact

In many cases, the reduction of emission levels as such does not represent the adequate objective of environmental policy. From an ecological perspective, it rather is the environmental load that is caused by emissions which is of main importance, i.e. the mitigation of those immissions that become effective in harming the environment (as discussed in ▶ Chap. 1 of this book) should be the ultimate goal of environmental policy. In general, the relationships between emission levels and harmful immission levels are very complex and vary strongly from one pollutant to another.

Let us, for simplicity, consider a pollutant for which the nature's assimilative capacity is zero and for which a linear relationship between emission and immission levels prevails. Then – as already described in ▶ Chap. 1 – the adverse immission level q_j^e that is caused at receptor j by emissions from source i is given by $\alpha_{ij}e_i$, where α_{ij} is the immission coefficient.

Let us now investigate how divergent α_{ij}'s affect the design of environmental tax schemes aiming at cost-effective abatement. We do this for the simplest conceivable case where there are two polluters with abatement cost functions $R_1(v_1)$ and $R_2(v_2)$, respectively, and only one single receptor of pollution. The immission coefficients are $\alpha_1 := \alpha_{11}$ and $\alpha_2 := \alpha_{21}$. Let ΔI stand for the targeted reduction of environmental load at the receptor. To find the socially optimal solution, we again have to minimize the total abatement costs $R_1(v_1) + R_2(v_2)$, but this time, the constraint becomes $\alpha_1 v_1 + \alpha_2 v_2 = \Delta I$. The Lagrangian function thus reads

$$L = R_1(v_1) + R_2(v_2) - \lambda(\alpha_1 v_1 + \alpha_2 v_2 - \Delta I),$$

so that we obtain the following first-order conditions for the cost-minimizing abatement levels v_i^* (with $i = 1, 2$) of polluter 1 and 2, respectively:

$$\frac{\partial L}{\partial v_1} = R_1'(v_1^*) - \lambda\alpha_1 = 0 \text{ and } \frac{\partial L}{\partial v_2} = R_2'(v_2^*) - \lambda\alpha_2 = 0.$$

From this we get

$$\frac{R_1'(v_1^*)}{\alpha_1} = \lambda = \frac{R_2'(v_2^*)}{\alpha_2},$$

where λ denotes the shadow price for the use of the environment. Hence

$$\frac{R_1'(v_1^*)}{R_2'(v_2^*)} = \frac{\alpha_1}{\alpha_2}.$$

If we face a situation where $\alpha_1 > \alpha_2$ and the marginal abatement costs are increasing, we get – in a hypothetical comparison with the case of identical coefficients (i.e. $\alpha_1 = \alpha_2$) – a higher abatement level for polluter 1 and a lower one for polluter 2. Furthermore, the polluter i (with $i = 1, 2$) will choose that abatement level for which his marginal abatement costs become equal to the environmental tax rate t_i that he faces.

If one strives for the social cost minimum with the associated abatement levels v_1^* and v_2^*, then different tax rates $t_1^* = R_1'(v_1^*)$ and $t_2^* = R_2'(v_2^*)$ have to be applied to the two polluters. From the first-order condition for cost-minimization derived above, we observe how the tax rates have to differ. It must hold that

$$\frac{t_1^*}{t_2^*} = \frac{\alpha_1}{\alpha_2}.$$

Consequently, the individual environmental tax rates have to be proportional to the immission coefficients so that the polluter generating the higher effective environmental load effect has to be charged the higher tax rate. Only if the immission coefficients are identical for all polluters, a uniform environmental tax rate for all emitters would bring about a minimization of total abatement costs.

Control Question

Why do different immission coefficients require different emission tax rates? Give an intuitive explanation!

Such a situation of identical immission coefficients prevails in the case of greenhouse gas (GHG) emissions like those of carbon dioxide (CO_2). Before CO_2 emissions from different sources contribute to global warming, these emissions are mixed up in the atmosphere. CO_2 but also the GHGs methane and nitrous oxide are "well distributed in the atmosphere across the globe, simplifying a global monitoring strategy" (Karl & Trenberth, 2003). Thus, the particular location of CO_2 emissions is irrelevant for their damaging effects so that different emission taxes are not needed.

However, if immission coefficients are different, the government may fail in designing an optimal environmental tax system as it may not be able to collect the required information about the effects of specific emissions on environmental quality. Consequently, in such a situation, it is unlikely that an environmental tax scheme will bring about the allocation through which the desired environmental goal is attained at least cost.

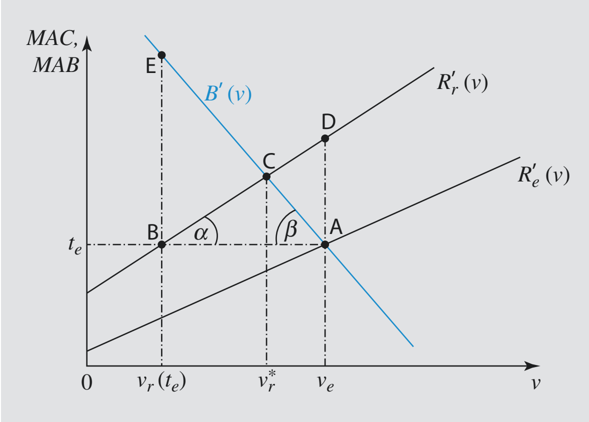
4.4.2 Unknown Marginal Abatement Costs

Independently of whether one pursues the Pigouvian or the price-standard approach, the government requires information about the polluters' marginal abatement cost curves in order to determine the appropriate environmental tax rates. In many cases, the available information will not be very precise, as only the polluters have full knowledge about their potential abatement technologies and have no interest in reporting this information correctly to the environmental authority.

In such a situation of asymmetric information between the government and the polluters, in the framework of the Pigouvian approach the possibility arises that CAC instruments are superior to environmental taxes. How this can happen has been described by Martin L. Weitzman (1974) in a seminal paper (see, e.g., Hepburn, 2006, for some further discussion).

In this model it is supposed that the government decides about environmental policy parameters on the basis of a, for the sake of simplicity, well-known marginal abatement benefit curve $B'(v)$ and an *expected* marginal abatement cost curve $R_e'(v)$. (The marginal abatement benefit curve $B'(v)$, which is dependent on the abatement level v, is

Fig. 4.5 Weitzman's (1974) model with an unknown marginal abatement cost curve



obtained from the marginal damage function $D'(e)$, which is dependent on the emission level e, by letting $B'(v) = -D'(e_0 - v)$, where e_0 stands for the initial emission level of the regarded polluting firm.)

The government – following the Pigouvian approach – then wants to attain the presumed socially optimal outcome A that in Fig. 4.5 is located in the point of intersection between the $B'(v)$ curve and the $R_e'(v)$ curve. To this end, the government can either use a CAC instrument, i.e. a quantity-based instrument that directly stipulates the abatement level v_e, or alternatively an environmental tax, i.e. price-based instrument with the tax rate $t_e := R_e'(v_e)$.

Now assume that the real abatement cost curve $R_r'(v)$ deviates from $R_e'(v)$ so that the expectations of the environmental authority have been wrong. Then the actual social optimum with abatement level v_r^* is located in point C, where the marginal benefit function $B'(v)$ and $R_r'(v)$ intersect, and the price-based and quantity-based environmental policies are no longer equivalent in their effect on the abatement level. While with the CAC policy the emission level v_e is still attained, this is no longer true when an environmental tax with rate t_e is applied. Instead, the polluting firm will adapt to t_e by choosing the abatement level $v_r(t_e)$ for which $R_r'(v_r(t_e)) = t_e$ holds, which leads to point B in Fig. 4.5. In both cases, i.e. with the emission standard v_e and with the emission tax rate t_e, welfare losses in comparison to the actual optimum C with the optimal abatement level v_r^* result.

In order to assess which of these instruments is to be preferred in this situation, we have to compare these two welfare losses: the welfare loss in the CAC case with the emission standard v_e is represented by the area of the triangle CAD in Fig. 4.5, while the welfare loss associated with the environmental tax rate t_e is measured by the triangle CEB. Obviously,

$$CAD = ADB - ACB$$
$$CEB = AEB - ACB.$$

Because the triangle ACB is part both of CAD and CEB, the welfare loss under the emission standard v_e thus is smaller (larger) than the welfare loss under the emission tax rate t_e if $ADB < AEB$ ($ADB > AEB$).

In ◘ Fig. 4.5 the marginal abatement benefit curve $B'(v)$ as well as the marginal abatement cost curves $R_e'(v)$ and $R_r'(v)$ are depicted as straight lines (in the case of small changes of v one can justify this as a linear approximation of more complex curves). If α stands for the gradient angle of $R_r'(v)$ and β for the gradient angle of $B'(v)$, then one gets

$$ADB = \frac{1}{2}\overline{BA}\cdot\overline{AD} = \frac{1}{2}\overline{BA}\cdot\tan\alpha\cdot\overline{BA} = \frac{1}{2}\overline{BA}^2\cdot\tan\alpha$$

and

$$AEB = \frac{1}{2}\overline{BA}\cdot\overline{BE} = \frac{1}{2}\overline{BA}\cdot\tan\beta\cdot\overline{BA} = \frac{1}{2}\overline{BA}^2\cdot\tan\beta.$$

Therefore, the welfare loss associated with the emission tax rate t_e is larger (smaller) than the welfare loss associated with the emission standard v_e, if $\alpha < \beta$ ($\alpha > \beta$), i.e. if the actual marginal abatement cost curve $R_r'(v)$ is flatter (steeper) than the marginal abatement benefit curve $B'(v)$. Hence, the CAC instrument will be more advantageous than the tax instrument when the marginal abatement cost curve is rather flat and the marginal abatement benefit (or marginal damage) curve is relatively steep. We can explain this as follows:

A flat actual marginal abatement cost curve, on the one hand, implies that small changes of the environmental tax rate will have a strong impact on the abatement level, which then is very price elastic. A steep marginal benefit curve, on the other hand, indicates that variations in the abatement level will bring about large changes in the damage caused by pollution. If in this situation the government wants to take precautions against the risk of errors, it therefore should pursue the CAC approach and stipulate the emission standard v_e. This reasoning makes it also clear why environmental taxation is never an appropriate environmental policy instrument for dealing with highly toxic substances with a significant hazard potential for human health.

? Control Question

Why can an emission standard be superior to an emission tax when the abatement cost curve is unknown? Give an intuitive explanation. What about the case when the marginal environmental benefit curve is unknown?

If the price-standard approach is adopted, ignorance about actual marginal abatement costs gives rise to another problem: a government that does not know the marginal abatement costs is not capable to determine the appropriate tax rate for bringing about the targeted abatement level. Baumol & Oates (1971: 45) also recognized this challenge and explained that environmental authorities could receive the required information by means of iterative adjustments in the tax rates: "if the initial taxes did not reduce the pollution […] sufficiently to satisfy the preset acceptability standards, one would simply raise the tax rates. Experience would soon permit the authorities to estimate the tax levels appropriate for the achievement of a target reduction in pollution."

Whether such a method of iterative adjustments really is practicable and economically meaningful is questionable. Firstly, the variation of environmental tax rates is time-consuming: before an environmental authority is able to make the appropriate adjustment of the environmental tax rate, reliable information about the polluters' reactions to the initially chosen tax rate has to be collected. Secondly, abatement measures are frequently associated with long-term investments in abatement technologies, which clearly lowers the flexibility of polluters in future periods of time. Polluters will not be willing to change their abatement activities by modifying their environmental protection technologies unless their initial technologies are not entirely depreciated. Consequently, an adjustment of the environmental tax rates may exert only negligible influence on the polluters' abatement activities in the short run anyway.

Furthermore, even in the long-term, the iterative adjustment process might not bring about the desired abatement cost minimization because the initial tax rate was chosen completely wrong (e.g., much too high) by the government. The misdirected polluters may have initially invested in a "wrong" (e.g., too expensive) environmental protection technology and are now locked-in into this technology.

Yet, another scenario is also conceivable: if the polluters anticipate adjustments of environmental tax rates from the outset, then they might act more carefully with regard to their investments in abatement technologies in order to reduce the risk of bad investments. They therefore may choose more flexible abatement options, but such a strategy does not ensure the application of least-cost abatement technologies in the long-run.

Box 4.3: Some Explanations on the Tangent

Let us consider a topographic survey. You are standing in a certain distance from a mountain that is 1.5 km high. As Fig. 4.6 illustrates, you see the top of the mountain from your position with an elevation angle of 19°. How can we determine the distance d in this case?

For the ratio between opposite leg and adjacent leg, it holds, by definition, that

$$\tan\alpha = \frac{\text{length of the opposite leg}}{\text{length of the adjacent leg}}.$$

Since

$$\tan\left(19^\circ\right) = \frac{1.5\,\text{km}}{d},$$

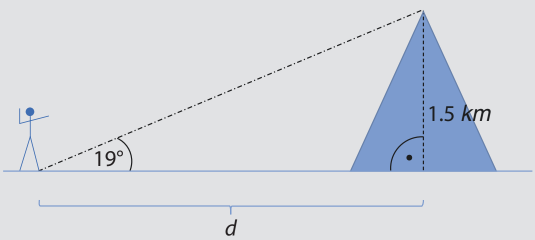
we have

$$d = \frac{1.5\,\text{km}}{\tan\left(19^\circ\right)} \approx 4.35\,\text{km}.$$

Therefore, the distance d looked for is about 4.35 km.

Knowing these geometrical relationships, in our formal analysis above, we have substituted $\tan\alpha \cdot \overline{BA}$ for $\overline{AD}$ where $\overline{AD}$ is the length of the opposite leg in Fig. 4.5. The same line of reasoning applies where we have substituted $\tan\beta \cdot \overline{BA}$ for $\overline{BE}$ above.

Fig. 4.6 Practical example of working with trigonometry



4.4.3 Effects of Environmental Taxation in Case of a Supplier's Monopoly

The welfare effects of environmental taxation heavily depend on the competitiveness on markets. Therefore, we will next examine in a small model how an emission tax works that is – in a specific situation – levied on a monopolistic firm's emissions (see Buchanan, 1969, Barnett, 1980, as seminal papers on environmental taxation in case of imperfect competition, and Ebert, 1991/1992 for some extension).

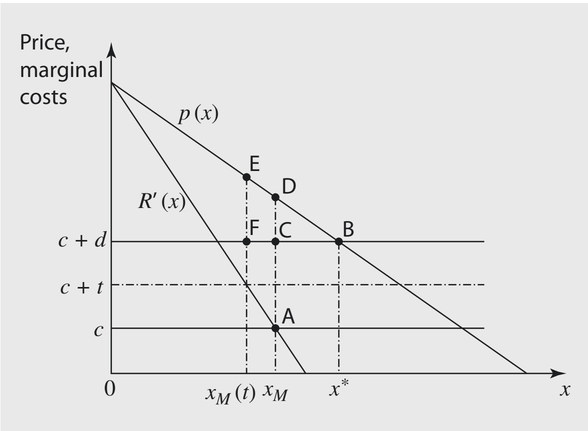
In this model, the amount of the good that is produced by the monopolist is denoted by x. The marginal production costs are assumed to be constant and equal to c. The inverse demand function, which describes consumers' behavior, is $p(x)$. Production of x is associated with emissions $e(x) = x$, and the marginal damage caused by these emissions is also constant and equal to d. The major simplifying assumption in this model is that the polluting monopolist can abate emissions only via reducing his production level, i.e. there are no end-of-pipe technologies for pollution abatement available.

If there is no environmental policy, the monopolist produces the amount x_M of the good for which the marginal revenue is equal to his marginal cost of production, i.e.

$$R'(x_M) = p'(x_M)x_M + p(x_M) = c,$$

which is described by point A in ◘ Fig. 4.7.

Now assume that an emission tax with some tax rate t is introduced. As depicted in ◘ Fig. 4.7, output then will decline to $x_M(t)$, where $R'(x_M(t)) = c + t$ holds and welfare will be lowered by the area $CDEF$. This shows that in the case of a monopoly, environmental taxation may not increase but reduce welfare. The simple cause for this, perhaps somewhat surprising, outcome is that in the situation as described by ◘ Fig. 4.7, the marginal benefit $p(x)$ of consuming x in x_M, and to the left of it, exceeds the aggregate marginal costs $c + d$ of production.



◘ **Fig. 4.7** Welfare losses through environmental taxation in the case of a monopoly

This constellation results if the output level x_M chosen by the unregulated monopolist is below the socially optimal output level x^*, which is characterized by

$$p\left(x^*\right) = c + d.$$

At x^* aggregate welfare, i.e. gross consumer surplus $\int_0^x p(\tilde{x})$ minus total cost $(c + d)x$ of production (private costs cx plus social costs dx), is maximized. In ◘ Fig. 4.7, the optimal solution is represented by point B. If $x_M < x^*$, the profit-maximizing monopolist has already restricted his production level by so much that a more than sufficient improvement in environmental protection has been brought about.

If, however, $x_M > x^*$, i.e. if the unregulated monopolist produces more than the socially optimal level, environmental taxation is well able to induce a welfare improvement. Yet, in this case the environmental tax rate t_M^* implementing the socially optimal outcome still deviates from the Pigouvian tax rate t_p, which would equal the marginal environmental damage d. In fact, we have $t_M^* < d = t_p$, i.e. the optimal environmental tax rate is lower than the marginal environmental damage. This is illustrated in ◘ Fig. 4.8.

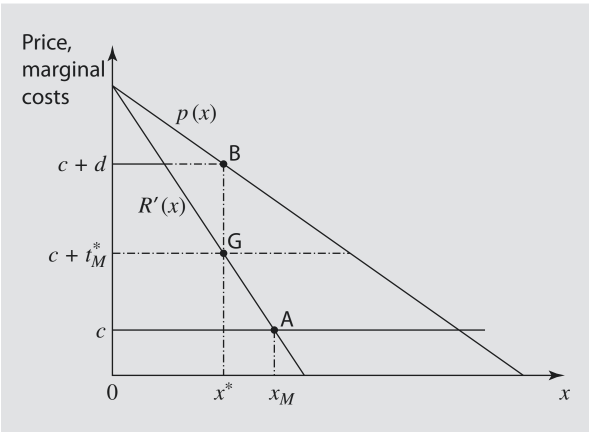
Control Question

Use ◘ Fig. 4.7 to explain why in the case of perfect competition the optimal solution is attained through an emission tax levied at the Pigouvian tax rate.

The socially optimal environmental tax rate that actually induces the monopolist to produce the socially optimal output level x^* is then characterized by the following first-order condition:

$$R'\left(x^*\right) = c + t_M^* \qquad \text{or}$$

$$t_M^* = p'\left(x^*\right)x^* + p\left(x^*\right) - c.$$



◘ **Fig. 4.8** The optimal emission tax in the case of a monopoly

In Fig. 4.8, this condition is fulfilled in point G. Because of

$$p\left(x^*\right) = c + d,$$

we obtain

$$t_M^* = d + p'\left(x^*\right)x^* \text{ or}$$

$$t_M^* = d + \frac{p^*}{\varepsilon\left(p^*\right)}$$

where $p^* = p(x^*)$ and $\varepsilon(p^*)$ stands for the price elasticity of demand at the price p^*.

For understanding this derivation of t_M^*, note that the elasticities of a function and its inverse are simply reciprocals to each other. Therefore, the demand elasticity of the price $\frac{p'(x)x}{p(x)}$ is the reciprocal of the price elasticity of demand, i.e.

$$\varepsilon\left(p^*\right) = \frac{1}{\frac{p'\left(x^*\right)x^*}{p\left(x^*\right)}} = \frac{p\left(x^*\right)}{p'\left(x^*\right)x^*}.$$

The expression for t_M^* allows us to make a direct comparison between t_M^* and the Pigouvian tax rate $t_p = d$. Thus, we see that the smaller the price elasticity of demand, the higher is the deviation of the socially optimal tax rate from the Pigouvian tax rate. This is quite plausible: if the demand for the good is only little price elastic, the monopolist has considerable market power, which he exploits by choosing a relatively low supply level and thus, voluntarily, reduces emissions and environmental harm. In principle, even the outcome $x_M = x^*$ may result. Then, by chance, the monopolist would choose the socially optimal production level on his own initiative.

Control Question

"The higher the monopoly power of a polluter, the lower is the optimal environmental tax rate." Explain this statement! Under which conditions will the optimal tax rate become negative so that the tax turns into a subsidy?

These considerations can easily be generalized to the oligopoly case in which there is a limited number $n \geq 2$ of polluting firms. In doing so, it becomes apparent that the socially optimal environmental tax rate approaches the Pigouvian tax rate if n goes to infinity and the market structure converges to the one with perfect competition.

Control Question

Why will the level of the optimal environmental tax rate decrease when the number of polluting oligopolistic firms increases?

4.4.4 The Possibility of Rising Emissions Through Environmental Taxation

In the previous models, environmental taxes at least brought about a reduction of emissions, which however did not always entail a welfare improvement as we could observe in the monopoly case. Yet, if one takes adjustment processes into account that take effect only in the long-term, then it even becomes possible that taxation of emissions gives rise to growing emission levels. Next, we describe this paradoxical effect in a standard model of a long-term competitive equilibrium.

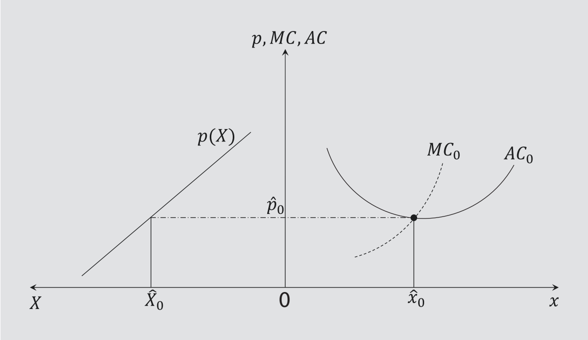
We assume that all firms are identical and $C(x)$ stands for the cost function of one single firm. The marginal cost function $MC_0 = C'(x)$ is growing in the output level x, and the average cost function $AC_0 = \frac{C(x)}{x}$ is supposed to be U-shaped. The output level of an individual firm for which average total costs are minimized is denoted by $\hat{x}_0$. In $\hat{x}_0$, the marginal cost function MC_0 and average total cost function AC_0 intersect, i.e. $C'(\hat{x}_0) = \frac{C(\hat{x}_0)}{\hat{x}_0}$. (This basic result on cost functions follows since by minimizing $\frac{C(x)}{x}$ we get $\left(\frac{C(x)}{x}\right)'\Big|_{x=\hat{x}_0} = \frac{C'(\hat{x}_0)\hat{x}_0 - C(\hat{x}_0)}{\hat{x}_0^2} = 0$, which clearly is equivalent to $C'(\hat{x}_0) = C(\hat{x}_0)/\hat{x}_0$.)

The inverse demand function for the respective good is denoted by $p(X)$, where X stands for the total amount of the good produced and traded on the market. Based on these assumptions we will now make use of the concept of the long-term competitive market equilibrium, which is defined as follows:

> **Definition**
>
> The *long-term competitive market equilibrium* is characterized by the following properties:
> (i) Each individual supplier at the market produces the average total cost-minimizing amount $\hat{x}_0$.
> (ii) The market price of the traded good in the equilibrium is $\hat{p}_0 = C'(\hat{x}_0) = C(\hat{x}_0)/\hat{x}_0$.
> (iii) The total output is $\hat{x}_0$ determined by the condition $p(\hat{X}_0) = \hat{p}_0$.
> (iv) The number $\hat{N}_0$ of firms supplying their output on the market is endogenously determined by the condition $\hat{N}_0 \cdot \hat{x}_0 = \hat{X}_0$. Thus, we get $\hat{N}_0 = \hat{X}_0 / \hat{x}_0$.

These properties can be motivated as follows: given perfect competition, an individual firm acts as price-taker, i.e. it maximizes its profits $px - C(x)$ for any given price, which leads to $p = C'(\hat{x}_0)$ as first-order condition. Furthermore, in the long-term equilibrium on the competitive market, the zero-profit condition $p\hat{x}_0 - C(\hat{x}_0) = 0$ or, equivalently, $p = \frac{C(\hat{x}_0)}{\hat{x}_0}$ must be satisfied. Otherwise, in the case of a positive profit, other firms

Fig. 4.9 The long-term competitive market equilibrium



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would be attracted to enter the market, and in the case of a loss, i.e. a negative profit, firms would give up and leave the market. In both cases, the outcome cannot persist so that there could be no equilibrium in the long-run. Taken together it follows that the price $\hat{p}_0$ in the long-term competitive market equilibrium has to satisfy $\hat{p}_0 = MC_0 = AC_0$. Since condition $MC_0 = AC_0$ only holds at the minimum of average costs, this implies that in the long-term competitive market equilibrium, each firm will produce $\hat{x}_0$, which then directly leads to the properties (ii), (iii), and (iv) as described above. The features of the long-term competitive market equilibrium are illustrated in Fig. 4.9.

Control Question

Give a brief explanation of the distinction between a short-run and a long-run competitive market equilibrium.

Let us now examine how an environmental tax with a rate t changes this long-term competitive market equilibrium when the production level x of an individual firm generates emissions $e(x)$. As in the previous subsection, we suppose that the firms' only option to reduce emissions is a cut in production, i.e. the use of end-of-pipe abatement technologies is not possible. Regarding the type of the emission function $e(x)$, two cases are distinguished.

Case 1 Emissions are proportional to the output level x, i.e. $e(x) = ex$.

Under the influence of the environmental tax, an individual firm's cost function then becomes $C_t(x) = C(x) + tex$ so that the new average total cost function is $AC_t = C(x)/x + te$. Graphically this means that the average cost curve (as well as the marginal cost curve) is shifted upward in parallel by te. Therefore, the output level entailing the least average costs and thus the individual firm's output in the long-term competitive market equilibrium is not modified through the environmental tax, i.e. $\hat{x}_t = \hat{x}_0$ holds (see Fig. 4.10).

Consequently, also the equilibrium price rises from $\hat{p}_0$ to $\hat{p}_t = \hat{p}_0 + te$. While the output and emission levels of each individual firm remain unchanged, for the new aggregate output level $\hat{X}_t$ we have $p\left(\hat{X}_t\right) = \hat{p}_t > \hat{p}_0$ and thus $\hat{X}_t < \hat{X}_0$. Therefore, the aggregate output level falls, and the number of firms that are active on the market shrinks to $\hat{N}_t = \hat{X}_t / \hat{x}_t = \hat{X}_t / \hat{x}_0 < \hat{X}_0 / \hat{x}_0 = \hat{N}_0$.

Case 2 Average emissions $e(x)/x$ are declining with an increasing output level x.

Such a situation prevails, e.g., when the emission function is linear and takes the form $e(x) = e_0 + ex$. Here, e_0 stands for a fixed level of emissions that already arise before

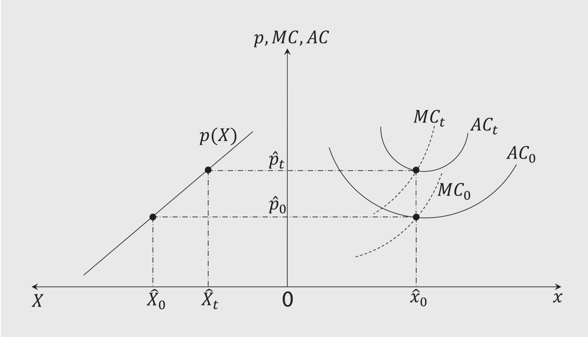

Fig. 4.10 Changes to a long-term competitive market equilibrium in case of a proportional emission function

the first unit of output can be produced. The assumption of such fixed emissions reflects many empirically important situations, e.g., as production plants and machines frequently have to reach a certain minimum operating temperature. The required energy use (e.g., through burning coal or oil) regularly leads to environmentally harmful emissions like CO_2 or SO_2. Alternatively, one can imagine that e_0 stands for those emissions that are incorporated in the capital goods being needed for the production of the good considered in our model.

If the output x is produced by a firm, its tax payments then are $te(x)$, which changes its average cost curve to

$$AC_t(x) = \frac{C(x)}{x} + t\frac{e(x)}{x}.$$

In order to avoid formal complications, we assume that the curve AC_t is also U-shaped, which is ensured if the tax rate t is small. Because of the increase in costs for all output levels x, the curve AC_t is located above the original average cost curve AC_0. In order to determine the minimum of the average cost curve $AC_t(x)$, we calculate its derivative at $\hat{x}_0$, i.e. the output level for which the original average costs $AC_0(x)$ are at their minimum:

$$\left.\frac{\partial AC_t}{\partial x}\right|_{x=\hat{x}_0} = AC_t'(\hat{x}_0) = \left.\left(\frac{C(x)}{x}\right)'\right|_{x=\hat{x}_0} + t\left.\left(\frac{e(x)}{x}\right)'\right|_{x=\hat{x}_0}.$$

This derivative is negative because the first summand on the right-hand side is equal to zero in $\hat{x}_0$, while the second summand is – in accordance with the assumption on the emission function – negative. This means that the AC_t curve is still declining when it passes through $\hat{x}_0$. The U-shaped AC_t curve therefore reaches its minimum to the right of $\hat{x}_0$, i.e. $\hat{x}_t > \hat{x}_0$. Hence, we indeed observe the paradoxical effect that the introduction of an emission tax leads to an increasing output level of an individual firm and consequently to increasing emissions.

Since $AC_t(x) > AC_0(x)$, the equilibrium price increases, i.e. $\hat{p}_t > \hat{p}_0$, which implies that the aggregate output level is reduced to $\hat{X}_t < \hat{X}_0$. Figure 4.11 illustrates these effects. The number of firms then must decline to

$$\hat{N}_t = \frac{\hat{X}_t}{\hat{x}_t} < \frac{\hat{X}_0}{\hat{x}_0} = \hat{N}_0.$$

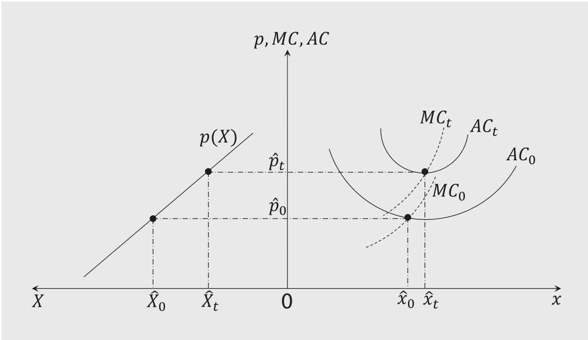

Fig. 4.11 Changes to a long-term competitive market equilibrium in case of decreasing average emissions

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While the emissions of a single firm increase, the aggregate emissions decrease nevertheless from $\hat{E}_0$ to $\hat{E}_t$ since

$$\hat{E}_t = \hat{N}_t \cdot e(\hat{x}_t) = \hat{X}_t \cdot \frac{e(\hat{x}_t)}{\hat{x}_t} < \hat{X}_0 \cdot \frac{e(\hat{x}_0)}{\hat{x}_0} = \hat{N}_0 \cdot e(\hat{x}_0) = \hat{E}_0.$$

This inequality holds as $\hat{X}_t < \hat{X}_0$ and $\frac{e(\hat{x}_0)}{\hat{x}_t} < \frac{e(\hat{x}_0)}{\hat{x}_0}$, which follows because average emissions are decreasing in x and $\hat{x}_t > \hat{x}_0$. Summarizing, in the case of declining average emissions, the introduction of an emission tax has the following effects:
- Fewer firms are active on the market.
- The individual pollution levels of these firms increase.
- The total emissions in the regarded economy decline.

Therefore, the environmental tax only has the desired effect on emissions at the aggregate level while it is counterproductive at the local level. Yet, note that these considerations do not reveal how the environmental tax actually affects economic welfare in the economy. We will next consider how the welfare effects depend on the kind of pollutant that is emitted.

Environmental taxation definitely increases welfare in case of pollutants whose damaging effect depends on the aggregate of emissions as in the case of greenhouse gases. A decline in aggregate emissions then reduces environmental damages even though emission levels may rise in different regions.

In other cases, the total damage may increase due to rising emissions of individual firms although aggregate emissions in the whole economy decrease. In order to describe the conditions for this undesired outcome, we assume a local damage function $d(e(x))$ that describes the environmental damage caused by the production activity of an individual firm in its surroundings. Given an environmental tax rate t, total damage is

$$\hat{N}_t \cdot d\big(e(\hat{x}_t)\big) = \hat{X}_t \cdot \frac{d\big(e(\hat{x}_t)\big)}{\hat{x}_t}$$

where again $\hat{N}_t$ denotes the number of firms in the long-term competitive market equilibrium after taxation. Differentiating this expression with respect to t (and omitting for the sake of simplicity arguments and indices) yields

$$X\frac{d'e'x-d}{x^2}\frac{\partial x}{\partial t}+\frac{d}{x}\frac{\partial X}{\partial t}.$$

Because $\frac{\partial x}{\partial t}>0$ and $\frac{\partial X}{\partial t}<0$, the total damage declines after an increase of the tax rate if $d'e'x - d < 0$. This condition is fulfilled in the important case where local damages are proportional to emissions.

If instead $d'e' > d/x$, i.e. if the marginal environmental damage caused by an individual firm's additional output exceeds the average environmental damage generated by this firm, total damages could rise with an increasing environmental tax rate.

Control Question

What effects are caused by environmental taxation in the long-run competitive market model when average emissions are increasing in the production level of a firm?

4.4.5 An Excessive Dynamic Incentive Effect

At first glance, it seems to be a clear advantage of environmental taxes that they generate stronger incentives to promote environmentally friendly technological progress than CAC instruments. Yet, in specific situations, these stronger incentives may bring about welfare losses since some overshooting of emission reductions may occur. How this may happen will now be described.

As before, $R_1'(v)$ stands for the initial marginal abatement cost curve of a polluter, while $R_2'(v)$ is his new marginal abatement cost curve after the transition to a new abatement technology. Again we assume that the new marginal abatement costs are throughout lower than the initial ones and that the switch to the new abatement technology triggers fixed investment costs F. Unlike the treatment of dynamic efficiency before (► Sect. 4.3.2), we now include, as a variation of the Pigouvian approach, a declining marginal abatement benefit curve $B'(v)$ into the model.

In this situation, the transition toward the new abatement technology may increase welfare only if in ◘ Fig. 4.12 the area $ABCD$ is larger than the investment costs F. If instead $ABCD < F$, the new technology definitely is not profitable for the economy. The optimal solution then is given by the point D where the initial abatement technology is applied and the abatement level is v_1^*, i.e. that level at which the marginal abatement cost curve $R_1'(v)$ and the marginal environmental benefit curve $B'(v)$ intersect. This optimal solution would be implemented by the Pigouvian tax rate t_1^* if only the initial abatement technology is available or if $ABED < F$ holds. Yet, if the polluter could use the alternative abatement technology and $ABED > F$, then the tax rate t_1^* would induce the polluter to switch to the new abatement technology although this change of technology is undesirable. Given t_1^* the polluter would not only use the "wrong" (new) technology,

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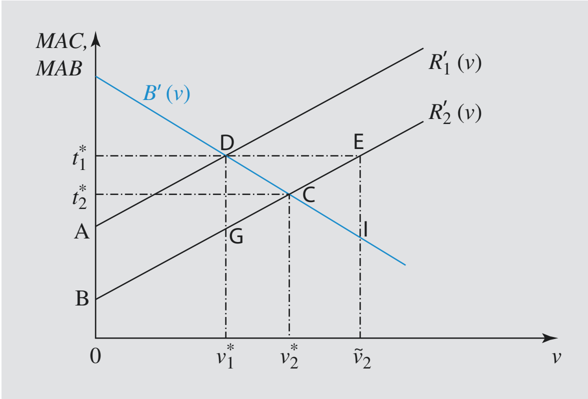


Fig. 4.12 The possibility of an excessive promotion of environmentally friendly technological progress

but he would even choose a too high abatement level given the new technology, i.e. he would be in the point E where the abatement level is $\tilde{v}_2 > v_2^*$. The total welfare loss as compared to the optimal solution then becomes $F - ABCD + CIE$. Even if the tax rate could be adjusted to the correct level t_2^* after the technology switch, a net welfare loss of $F - ABCD$ remains. Of course, the shift to the "wrong" abatement technology could be prevented by choosing a lower environmental tax rate as a second-best solution. This, however, would imply a "wrong", socially suboptimal abatement level. The message is that in the scenario considered here, environmental taxation is not capable of attaining the first-best solution, i.e. abating v_1^* with the old technology.

Control Question

How can the second-best optimal environmental tax rate be determined? Which welfare loss results with this tax rate as compared to the first-best solution?

By means of a CAC instrument stipulating v_1^* as the required abatement level, the socially optimal solution would be implemented under the same conditions. Since $ABGD < ABCD < F$, the polluters prefer to stay with the "right" old abatement technology and clearly abate the "right" level v_1^* of emissions with this technology.

As Fig. 4.12 shows, the misallocation through environmental taxation as described above is most likely when the marginal abatement benefit curve is sharply decreasing in the relevant range. Then, an extension of the abatement level only produces a relatively small additional damage reduction, so that the investment in the new abatement technology is not profitable from a social welfare perspective. In contrast, if marginal abatement benefits (or marginal damages of pollution emissions) are constant, the possibility of a too strong dynamic incentive effect of the environmental tax is precluded.

4.4.6 Welfare Losses Due to the Double Burden of Environmental Taxes

Again in the framework of the Pigouvian approach, we assume that $B'(x)$ denotes the marginal profit function of a polluting firm whose fixed costs are $F > 0$. (To avoid misunderstandings, these fixed costs differ from those considered in the previous section

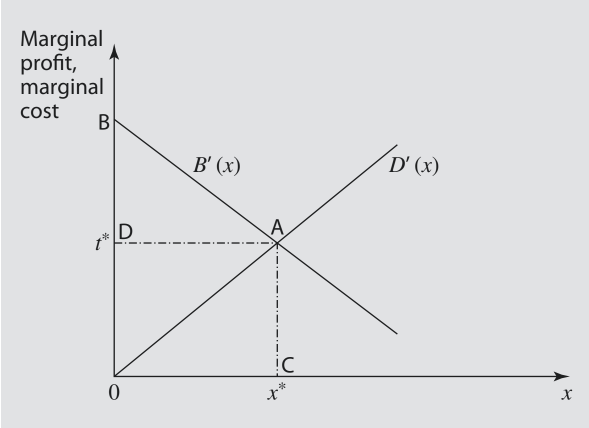


Fig. 4.13 The possibility of misallocation due to "double burden"

although the same symbol is employed!) $D'(x)$ stands for the marginal environmental damage resulting from the firm's production level x.

Starting production is socially profitable if $0AB > F$, i.e. if the difference between current profit of the polluting firm and environmental damage in the optimal solution with the output level x^* exceeds the fixed costs of production (see Fig. 4.13). To attain x^* by means of environmental taxation clearly requires the tax rate $t^* = D'(x^*) = B'(x^*)$. The firm's tax payments then are $t^*x^* = 0CAD$, which reduce its current profits to DAB. As DAB is smaller than $0AB$, it becomes possible that

$$DAB < F < 0AB$$

holds. Then production will not be started, and hence the socially optimal production level x^* will actually not be attained. The reduction of the firm's potential profits through the emission tax exceeds the damage caused by the firm's production, i.e. $0CAD > 0CA$, and consequently net profits would not be high enough to cover the fixed costs. The double burden then is $0CAD - 0AD$.

Definition

The *double burden* that is imposed on polluters through environmental taxation is caused by the tax payments, which add to the abatement costs as the polluters' first burden.

In the situation considered here, these problems could be avoided so that the socially optimal output level x^* would be attained if a CAC instrument stipulating a maximum production level x^* were implemented instead.

The extent of the double burden depends on different factors that vary from case to case. As can be inferred from Fig. 4.13, the flatter the marginal damage curve, the lower the double burden. In the case of constant marginal damage, there is no double burden at all. (Just imagine that the marginal damage curve turns clockwise around point A.) Relatively flat marginal damage curves occur rather frequently in reality because many environmental problems (especially in the context of air pollution) are caused by a large number of polluters, and, moreover, the emissions strongly intermix in the environmental media.

The extent of the double burden is also mitigated if the polluting firms can pass on the burden to other economic agents like customers or subcontractors. The extent to which such burden shifting is possible depends to a considerable degree on the conditions on the output market. Especially firms that compete internationally and have to take the global market price of their output as given will be much affected. If an environmental tax is levied in a single small country, shifting the tax burden is virtually infeasible so that firms have to bear the full burden themselves. In times of globalization, the double-burden argument therefore has to be taken more seriously than in former times where firms had been exposed to international competition to a much lesser extent. As a consequence, heavily burdened firms will demand tax exemptions and other favorable conditions from the government, and they mostly will be successful with their claims as they can threaten to cut jobs.

Without special tax conditions, the risk prevails that firms have to exit the market because of the double burden, which entails a devaluation of physical and human capital. The ensuing welfare loss will be particularly large when investments are sunk, i.e. when the capital cannot be used for other purposes.

Things may become even worse. If the domestic industry loses its competitiveness, the production may move to countries where environmental standards are lower. Globally, environmental damage therefore may even increase due to unilateral taxation of pollution emission by one country as it triggers "leakage effects". Then it may even happen that a country raising its environmental tax rates will be harmed in a twofold way: economically by a loss in domestic production and jobs and ecologically by a deteriorated environmental quality.

The double burden caused by environmental taxation might generate further adverse effects. So the payment of taxes for residual emissions reduces a firm's possibilities for self-financing its investments. If profitable expansion and rationalization investments fail to appear for this reason, economic growth will slow down causing adverse employment effects. There is even a danger that firms lack the money to finance the abatement investments. This reduces demand for abatement technologies and thus employment also in these industries and may as well retard research and development in environmental technology, so that it becomes more difficult to increase environmental standards in future periods.

One could counter this argument by stressing that external sources of funding are available and can serve as a substitute for self-financing of investments. However, a closer look reveals that the use of this alternative or complementary form of funding is also very limited: due to the double burden, profitability declines. Combined with their weaker equity position, this increases the affected firms' bankruptcy risk, which makes borrowing money harder and more costly. In a comprehensive model of capital allocation, it can be demonstrated that an additional financial load for an industry will cause the capital stock of this industry to decline in the equilibrium.

Especially in small owner-managed firms, another effect has to be considered as well: the withdrawal of financial means due to the double burden causes the owners of firms to become more risk averse as empirical findings suggest. Guiso & Paiella (2008), for example, showed in an empirical analysis using household survey data that individuals who are more likely to become liquidity constrained are characterized by a higher degree of absolute risk aversion. In the risk-theoretical literature, such a behavioral pattern is coined *decreasing absolute risk aversion*, which says that the absolute amount that is invested in a risky project will fall when the investor's wealth is reduced (see, e.g., Gollier, 2001). If firms' willingness to invest in innovative and thus naturally risky projects declines, this is neither advantageous for technological progress, nor for economic growth, nor for the creation of new jobs.

Control Question
Which factors make it more likely that welfare losses are caused by the double-burden effect of environmental taxation?

At the end of this subsection, we briefly restate the core idea of the double-burden argument: the improvement of environmental quality causes costs, which are either due to the implementation of abatement technologies or to a cut in production. An environmental policy that is guided by the polluter-pays principle so that the polluters have to bear these costs will in general increase welfare. If such environmental policy reduces the profitability of polluting firms and capital is withdrawn from the polluting industries, this is inevitable from a welfare perspective. An efficient internalization of external effects necessitates to a certain extent an economic structural change toward environmentally sound production patterns and outputs. In view of the double-burden argument, it is, however, much less justifiable that the polluting firms also have to bear costs which go beyond the expenses required to achieve the objectives of environmental policy. This is even potentially welfare reducing: the profits in the affected industries decline more than necessary, and the induced reallocation of capital will be too strong. The costs of environmental protection then become inefficiently high, which in turn will make people less willing to accept and to vote for a more ambitious environmental policy. These risks associated with environmental taxation have to be weighted against the clear advantages such a policy instrument has.

Conclusion
You have learnt in this section that – despite their basic advantages – the application of uniform emission taxes will involve problems. In particular emission taxes:

- Have to be differentiated when different sources of pollution have a different impact on environmental quality.
- May be inferior to emission standards if marginal abatement costs are unknown.
- Must be lower than Pigouvian taxes (and even may turn into a subsidy) if there is imperfect competition between polluting firms and especially if a monopolistic market structure prevails.
- May increase local pollution levels in a long-run competitive market equilibrium and thus have a counterproductive effect.
- May promote environmentally friendly technological progress too much. Then a situation might occur in which the first-best solution cannot be implemented through environmental taxation (while it is well possible with an emission standard).
- Cause an additional financial burden that, through various channels, may lead to welfare losses and lower growth and employment levels.

4

4.5 The Double Dividend Hypothesis: The Idea of "Greening" the Tax System

In the 1990s, an additional argument in favor of environmental taxation has gained much attention in environmental economics and partly also in politics: taxation of emissions is not only expected to entail static cost-effectiveness (i.e. to minimize total abatement costs) and to contribute to dynamic efficiency (i.e. to promote environmentally friendly technological progress), but it also may constitute the basis for creating an overall more efficient – and therefore from an economic point of view "better" – tax system. From this perspective environmental taxation promises to have an advantage in addition to the improvement of environmental quality, i.e. a second or a "double dividend." The environmental tax reform, which was initiated in Germany at the turn of the century, was strongly influenced by such ideas. Before the pros and cons of the double dividend hypothesis are discussed in more depth, its theoretical foundations have to be outlined.

4.5.1 The Idea of the Double Dividend

Environmental taxes do not only lead to the internalization of external effects but, besides this primary goal, also generate some tax revenue, which can be used by the government to finance its expenditure. If total government spending is kept at a constant level, i.e. the condition of *revenue neutrality* is satisfied, environmental taxation renders it possible to reduce traditional taxes like income and consumption taxes, thus reducing their *excess burdens*. The expectation of lower excess burdens and, henceforth, an increased efficiency of tax collection is the normative basis of the double dividend hypothesis, which sometimes is even used to justify a comprehensive tax reform that assigns eco-taxes the central role in the tax system.

■ *What Is the Excess Burden of Taxation?*

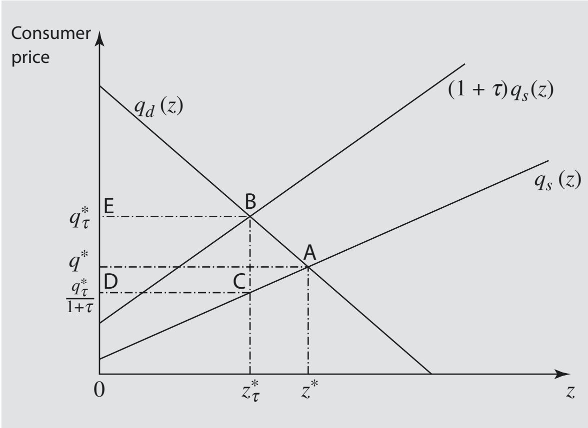
In order to understand what is meant by excess burden, let us consider a market on which a good z is traded, whose consumer price is q. By $q_s(z)$ we denote the increasing inverse supply function for this good, while $q_d(z)$ is the decreasing inverse demand function that indicates the price at which consumers are just willing to buy a certain quantity of the good. The competitive market equilibrium (with the quantity of good z^* and the price q^*) where supply and demand coincide is given by the intersection of $q_s(z)$ and $q_d(z)$, i.e. by the point A in ◘ Fig. 4.14.

Let us now consider the case in which a value-added tax is levied on good z, which leads to a proportional surcharge on the *producer price*. If the tax rate is τ, the consumer then has to pay the price $(1 + \tau)q_s$, so that the producers realize the net price q_s per unit of good z. In ◘ Fig. 4.14, as a result the inverse supply curve turns upward. The market equilibrium after taxation, i.e. point B in ◘ Fig. 4.14, is given by the intersection between the new inverse supply curve $(1 + \tau)q_s(z)$ and the unchanged inverse demand curve $q_d(z)$.

Control Question

Why is the excess burden a quadratic function of the tax rate in the linear model described in this section?

Fig. 4.14 Graphic depiction of excess burdens



The quantity of good z in equilibrium is reduced to z_τ^*; the consumer price increases to q_τ^*, while the producer price decreases to $q_\tau^*/(1+\tau)$. The tax payment $\left(\tau q_\tau^*/(1+\tau)\right)\cdot z_\tau^*$ is described by the rectangle *DCBE* in Fig. 4.14. However, the tax not only charges consumers and producers by this amount but in addition causes a real loss of welfare, the *deadweight loss* of taxation, that in Fig. 4.14 is given by the Harberger triangle *CAB*. These excess burdens result as the consumption tax drives a wedge between the producers' and the consumers' prices thus preventing the market mechanism to reach the optimal allocation in point *A*.

Definition

The *excess burdens of taxation* are the real welfare losses that are implied through the distortion of relative prices caused by taxation.

What Is the Double Dividend?

Looking at Fig. 4.14, the excess burdens for consumers and producers can be reduced by lowering the tax rate τ since then the Harberger triangle *CAB* clearly shrinks. However, if governmental expenditure is not reduced and, at the same time, public debt is not increased, additional revenue from other taxes is required to compensate for this loss of government's funds. Taxation of environmentally damaging activities seems to be an ideal candidate for filling this gap, as they bring about improvements in environmental quality and social welfare and thus, in this respect, are fundamentally different from the standard taxes. In a certain sense, they bring about a positive and not a negative distortion. Therefore, it is understandable that a tax reform, which strengthens the importance of environmental taxes, is considered a kind of panacea that not only is useful for coping with ecological problems but also helps to mitigate the negative welfare effects resulting from conventional taxation.

Definition

The *double dividend* of an environmental tax is given by the reduction of the excess burdens that results when revenue of the environmental tax is used to lower standard taxes with distorting effects.

Besides the reduction of the excess burden of taxation, the quantity effect, i.e. the increase of z when τ is reduced, can also be a goal for policy makers. This is especially true if ◘ Fig. 4.14 depicts the labor market and z symbolizes labor demand and supply. A reduction of the wage tax, made possible by introducing or raising environmental taxes, will entail an increased level of employment and thus be helpful in fighting unemployment. This effect can also be interpreted as a form of the double dividend phenomenon.

4.5.2 The Level of the Optimal Environmental Tax When the Double Dividend Is Taken into Account

If environmental taxation is advantageous in a double sense, it might be plausible to assume that the environmental tax rate, which turns out to be optimal when also the reduced excess burdens of other taxes are taken into account, lies above the Pigouvian tax rate, which depends solely on the ecological welfare component, i.e. the marginal environmental damage. On closer inspection, this hypothesis proves to be incorrect as the following theoretical consideration shows:

Let x denote the quantity of a "dirty" good, whose production or consumption is associated with environmental damage $D(x)$. The costs of the production of x are $C(x)$, and $p(x)$ again represents the inverse demand function of good x. Now an environmental tax with tax rate t is levied on the emissions caused by the production of x, which leads to a new competitive market equilibrium with the quantity of goods $x(t)$. If emissions are proportional to the quantity of produced goods and the emission coefficient is normalized to one, the revenue of the environmental tax is $T(t) = tx(t)$. For a given environmental tax revenue $T(t)$, $L(T(t))$ is assumed to indicate the welfare gain which is achieved through a reduction of other conventional taxes and their excess burdens (given that total tax revenue is kept constant). In ◘ Fig. 4.14 this means that the tax rate τ is reduced to a lower level $\tilde{\tau}$ for which the difference between the old tax revenue rectangle and the new one just equals environmental tax revenue $T(t)$. The Harberger triangle, which is implied by the new tax rate $\tilde{\tau}$, then completely lies within the old Harberger triangle. The difference between the two triangles then describes the decline in excess burdens of taxation on the "clean" good z, i.e. $L(T(t))$. The larger $T(t)$, the lower the new tax rate $\tilde{\tau}$ and thus the larger the difference between the two Harberger triangles will be. In general, this means that $\partial L/\partial T > 0$, i.e. that with rising environmental tax revenues (of good x) the avoided excess burdens (from taxation of good z) are increasing.

Including this welfare component, social welfare as a function of the environmental tax rate t is

$$W(t) = \int_0^{x(t)} p(\tilde{x})\, d\tilde{x} - C(x(t)) - D(x(t)) + L(T(t)),$$

where again $\int_0^{x(t)} p(\tilde{x})\, d\tilde{x}$ measures the gross consumer rent, i.e. the gross benefits from consuming the quantity x. The marginal change in social welfare that results from a marginal increase of the environmental tax rate then is

$$W'(t) := \frac{\partial W}{\partial t} = p\left(x(t)\right)\frac{\partial x}{\partial t} - \frac{\partial C}{\partial x}\frac{\partial x}{\partial t} - \frac{\partial D}{\partial x}\frac{\partial x}{\partial t} + \frac{\partial L}{\partial T}\frac{\partial T}{\partial t}.$$

Hence, the socially optimal environmental tax rate t^* is characterized by the first-order condition $W'(t^*) = 0$.

We now examine under which conditions this tax rate t^* is larger or smaller than the Pigouvian tax rate t_p, which in turn corresponds to the marginal damage in the efficient outcome. To this end, we determine the sign of the derivative $\partial W/\partial t$ at $t = t_p$, which gives

$$W'(t_p) = \left[p\left(x(t_p)\right) - C'\left(x(t_p)\right) - D'\left(x(t_p)\right)\right]\frac{\partial x}{\partial t}\bigg|_{t=t_p} + \frac{\partial L}{\partial T}\cdot\frac{\partial T}{\partial t}\bigg|_{t=t_p}$$

where $C'(x) = \frac{\partial C}{\partial x}$ and $D'(x) = \frac{\partial D}{\partial x}$. We now observe that $t_p = D'(x(t_p))$ and that $p(x(t_p)) = C'(x(t_p)) + t_p$, which characterizes the competitive market equilibrium as modified by the environmental tax. There, the market price must equal the producers' private marginal costs, i.e. marginal production costs plus environmental tax rate. The first term in the above expression for $W'(t_p)$ therefore becomes zero, so that it reduces to

$$W'(t_p) = \frac{\partial L}{\partial T}\cdot\frac{\partial T}{\partial t}\bigg|_{t=t_p}.$$

Because $\partial L/\partial T$ has a positive sign, $W'(t_p)$ has the same sign as $\frac{\partial T}{\partial t}\Big|_{t=t_p}$. It therefore follows that a marginal increase in the environmental tax rate t beyond the Pigouvian tax rate t_p leads to a welfare increase if the environmental tax revenue increases. In this case $t^* > t_p$ is ensured – and the intuitive expectations are confirmed.

A priori, however, it cannot be ruled out that the tax revenue does not grow but falls when the environmental tax rate is increased. The Pigouvian tax rate then is so high that it is already in the falling range of the *Laffer curve*, which indicates the relationship between the level of the tax rate and the revenue flowing from this tax.

Definition

The Laffer curve of a tax indicates how the tax revenue varies with the tax rate.

It is generally assumed that the Laffer curve is bell-shaped with its maximum at the tax rate t_{max} (see ◘ Fig. 4.15). The decline in tax revenue when tax rates are high can be traced back to behavioral changes of the taxed agents. They try to avoid the additional burdens resulting from tax increases by restricting the extent of their taxed activity and substituting the dirty good by other goods. The higher the tax rate in the initial state, the more likely it is that the negative tax base effect will overcompensate the positive tax rate effect when the tax rate is increased. Formally, this *Laffer effect* occurs if

$$\frac{\partial T}{\partial t} = t\cdot\frac{\partial x}{\partial t} + x(t) < 0 \text{ holds, if } -\frac{\frac{\partial x}{\partial t}\cdot t}{x(t)} > 1,$$

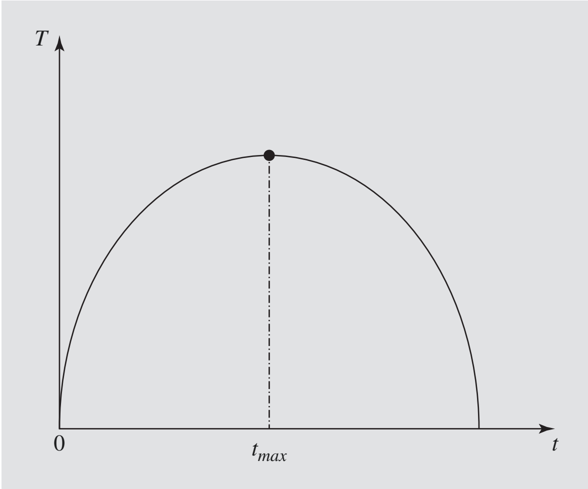


Fig. 4.15 Laffer curve depicting the relationship between tax rate and tax revenue

i.e. if the tax elasticity of demand for the dirty good x is sufficiently high, which means that there is a sharp fall in demand and thus a strong tax base erosion effect as a result of a price or tax increase. Then, a welfare increase is achieved by lowering the environmental tax rate against the Pigouvian tax rate so that $t^* < t_p$ holds.

In the case of emission taxes, high environmental tax rates are correlated with high levels of abatement measures. In view of the double dividend argument, emission reduction and thus protection of the environment therefore might come into conflict with the goal of attaining a high revenue from environmental taxes and, henceforth, the possibility of cutting other distorting taxes.

Remarkably, this shows that an environmental tax rate t^* will lie below the Pigouvian tax rate t_p when the environmental tax has to have a strong steering function and should lead to a considerable improvement in environmental quality. In this situation, the inclusion of the double dividend welfare component would lead to a deterioration in the quality of the environment, which may seem paradoxical at first sight.

Control Question

Why does a high marginal environmental damage make it more likely that the optimal environmental tax rate is smaller than the Pigouvian tax rate? Why does this seem paradoxical?

4.5.3 Challenging the Existence of Double Dividends

Although the double dividend argument appears very plausible, the existence of this additional advantage of environmental taxation has not remained undisputed. Part of this controversy is simply based on a nontrivial terminological ambiguity, i.e. the term "double dividend" is used in different ways which, not surprisingly, is leading to some confusion (see Bovenberg, 1999; Goulder, 1994, 2013 for detailed reviews of this discussion).

4.5.3.1 The Semantic Confusion About the Concept of Double Dividends

The semantic confusion in the debate on the double dividend originates from the fact that environmental taxes usually also lead to excess burdens in the conventional sense. This becomes particularly clear when environmental taxes are levied as taxes on specific "dirty" goods (such as mineral oil) rather than as taxes on emissions. The Harberger triangle that arises from taxation of the "dirty" good x is illustrated in Fig. 4.16, where, for the sake of simplicity, we are assuming constant marginal costs of production c and constant marginal damage of environmental pollution d (and, again, $p(x)$ denotes the inverse demand function). Point A (with the production level x_0) is the competitive market equilibrium without any governmental intervention, while point B (with the production quantity x^* being characterized by $p(x^*) = c + d$) is the optimal solution, which is implemented with the Pigouvian tax $t_p = d$.

Let us now assume that an environmental tax with an arbitrary tax rate t is levied, which leads to $x(t)$ as the production level in the competitive market equilibrium and, in the usual way, to excess burdens as measured by the Harberger triangle ACD. A welfare increase nevertheless results because the environmental damage that is avoided by the environmental tax exceeds these excess burdens, i.e. $AEFD > ACD$ holds (see Fig. 4.16). (This observation applies to not very high environmental tax rates t, and in particular to all t subject to $t \leq t_p = d$.) That an environmental tax is economically advantageous means not at all that it goes without excess burdens, which however are smaller than the gains in environmental quality.

Environmental tax revenue then is $DCGH(=t \cdot x(t))$. If, with this environmental tax revenue, excess burdens of $L(DCGH)$ can be saved by reducing other taxes, total net welfare amounts to

$$AEFD - ACD + L(DCGH).$$

The difference $AEFD - ACD$ is the standard Pigouvian welfare effect, and $L(DCGH)$ is the potential double dividend component, which would be zero if tax revenues were returned to individuals in a lump-sum way as an "eco-bonus" (of a fixed amount inde-

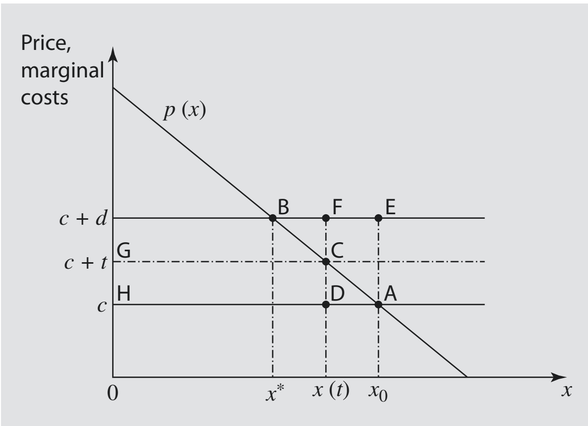


Fig. 4.16 Environmental taxes as taxes on particular goods causing an excess burden

pendent of economic entities' behavior). The welfare component $L(DCGH)$ is positive when tax rates of distorting taxes are reduced as described before, which trivially implies

$$AEFD - ACD + L\left(DCGH\right) > AEFD - ACD,$$

i.e. a positive welfare effect that arises additional to the Pigouvian effect indeed results. The double dividend hypothesis in its original form was related precisely to this fact – and this interpretation has been followed in the previous subsection. Yet in the same context, one could also define another form of a double dividend, which in contrast to the conventional *weak double dividend* has been coined as *strong double dividend*. Such a terminological distinction is completely harmless if it were simply made to indicate that even more benefits of environmental taxation might exist beyond the weak double dividend.

Definition

A revenue-neutral shift of the tax burden from standard taxes to environmental taxes yields a *strong double dividend* if the total excess burden is reduced by this tax reform.

The strong double dividend would arise if $L(DCGH) > ACD$ holds. In this case, the reduced excess burdens of other taxes would not only be positive, but even greater than the excess burdens of the environmental tax. Then the total excess burdens of the entire tax system would be reduced by a transition to environmental taxes, so that some kind of a "third dividend" of environmental taxation would arise. In this situation, a welfare gain hence would materialize independent of the environmental improvement, which would make it possible to completely refrain from referring to the positive ecological effects when one advocates for an environmental tax (eco-tax) reform. The prefix "eco" would then perhaps only be an instrument of political marketing, in order to be able to better enforce a tax reform that promotes efficiency of taxation anyway. Emphasizing the concept of a strong double dividend at least implicitly would entail that ecology takes a back seat, while considerations about optimal taxation are moving into the center.

Such a shift of attention toward the partial welfare effects of an eco-tax reform that can be attributed to changes in excess burdens clearly is legitimate. A problem, however, arises if the strong double dividend is identified with the double dividend as such, and the absence of a possible third advantage is used to question the meaningfulness of the double dividend hypothesis at all. The argument against environmental tax reforms concerning the possible or even probable absence of a strong double dividend alone is hardly a particularly serious one.

From a theoretical point of view, the observation that a double dividend in the strong sense is not always to be expected is not surprising either. Whether $L(DCGH) > ACD$ actually applies depends to a large extent on the taxes which are replaced by the environmental tax and the magnitude of their excess burdens. A rough criterion for predicting whether there is a third dividend of environmental taxation can already be inferred from our simple market models: on the one hand, a high level of

$L(DCGH)$ results if the demand function of the good for which taxes are abated is flat, i.e. if the demand for this good is characterized by a high price elasticity. A low value for ACD, on the other hand, is obtained when the price elasticity of demand for the dirty good, which is subject to the environmental tax, is small. Taken together, both partial effects lead to a decrease in total excess burdens so that the strong double dividend occurs. Should the reverse be the case, an increase of aggregate excess burdens would result instead.

Control Question

What is the difference between the weak and the strong version of the double dividend? Why could one be interested in the existence of a strong double dividend?

Despite all the criticism, the discussion about the double dividend of environmental taxation has considerably improved the understanding of the factors on which the welfare effects of environmental tax reforms depend. These factors will now be explored in some more detail.

4.5.3.2 Determining Factors for the Occurrence of a Strong Double Dividend

For a long time, the *theory of optimal taxation* (see, e.g., Atkinson & Stiglitz, 1980; Koskela & Schöb, 2002; Wendner & Goulder, 2008) has examined what determines the extent of excess burdens in specific cases.

Definition

The *theory of optimal taxation* examines which properties of taxes are suitable for minimizing the excess burden of taxation.

One of the basic findings is that price elasticities of demand play a central role in this context. The importance of price elasticities of demand for the optimal design of a tax system is most clearly reflected in Ramsey's *inverse elasticity rule*, which states that optimal (ad valorem) tax rates for two different goods are to be chosen inversely proportional to the goods' respective price elasticities of demand. As a result, optimality of taxation demands that goods with low price elasticities are subject to higher tax rates.

Definition

The *inverse elasticity rule* says that excess burdens of taxation become smaller when goods or activities with a low elasticity of demand are taxed (instead of those with a high elasticity of demand).

The example of energy products such as oil or coal, which in reality are an important field of application for environmental taxes, shows that the price elasticity of demand depends heavily on the period of time under consideration. At least in the case of moderate and thus politically acceptable fuel taxes, only slight changes in demand can be expected in the short-term since people are not easily ready to change their habits and have to drive to work anyway. In the long run, the price elasticity of demand is much higher, since higher fuel prices motivate people to pay more attention to fuel consumption when buying a new car. The automobile industry clearly will take these changed customer preferences into account and produce more fuel-efficient cars. In the longer run, it is therefore less likely that a strong double dividend can be realized.

Another general argument from the theory of optimal taxation is also not favorable for the emergence of a strong double dividend. Total excess burdens are usually the lower the more the tax burden is spread evenly on various goods and activities. (In the simple competitive market diagrams that we have used in our theoretical treatment of the double dividend, this result simply follows because the area of the Harberger triangle for a single good or activity is a quadratic function of the tax rate.) Yet, an environmental tax reform might lead to a concentration of the tax burden on specific environmentally harmful goods so that total excess burden will increase as compared to a uniform tax on all consumption goods. This reservation against the occurrence of a strong double dividend also holds true if an increase of an eco-tax is accompanied by a revenue-neutral reduction of a labor income tax, which can be considered as being equivalent to a general consumption tax.

Another prominent result in the theory of optimal taxation is the *leisure complementarity rule*, which reads as follows.

Definition

The *leisure complementarity rule* says that excess burdens of taxation are low when the taxed goods or activities are *complements* to leisure but are high if they are *substitutes*.

This rule can be understood as an implication of the postulate that tax burdens should be widely spread in order to avoid too high excess burdens. From this point of view, it would in particular be desirable to extend taxation also to leisure as some kind of consumer good.

Control Question

Explain in some more detail why a broad spread of taxation over many goods and activities leads to lower aggregate excess burden. What is the relation to the leisure complementarity rule?

Since direct taxation of individual leisure does not appear to be feasible in reality, the *indirect* taxation of leisure by taxing goods whose use is positively associated with leisure and which therefore are complements to leisure can be seen as an approach to improve efficiency of tax collection. The taxation of goods which are substitutes to leisure (and, thus, complements to labor) instead would lead away from such an implicit broadening of the overall tax base and therefore would contribute to an increase in total excess burden.

"Dirty" energy products as a major field of application of environmental taxation and labor are, to varying degrees, complementary inputs in the production process. Energy thus mainly constitutes a substitute to leisure, although many leisure activities as vacation trips are associated with the use of energy. Therefore, an energy tax is more likely to reduce the indirect taxation of leisure, which limits the chances to benefit from the double dividend in its strong form.

For the extent of excess burden resulting from the introduction or increase of environmental taxes, the initial state of the entire tax system also is of much importance. Since the conventional taxes reduce overall economic activity, they also indirectly dampen environmentally damaging activities thus pre-empting part of the required internalization. This implies that in view of pre-existing taxes, the correct environmental tax rate does not correspond to marginal damages of pollution, i.e. does not equal the Pigouvian tax rate, but rather is lower. We could already observe a similar phenomenon in our discussion of optimal environmental taxation in the case of a monopoly. (Note that this argument for an optimal eco-tax rate below the Pigouvian tax rate is based on a tax interaction effect (see, e.g., Bovenberg & Mooij, 1996) and thus is conceptually different from that we presented in ► Sect. 4.5.2.)

Employment could also – contrary to original optimistic expectations – be falling instead of rising in the course of an environmental tax reform in which a tax swap between eco-taxes and the labor income tax is made. How this may happen already becomes clear from our theoretical analysis in ► Sect. 4.5.2: assume that in the initial state a proportional labor income tax is levied, which as an indirect uniform consumption tax also affects the dirty good. Now assume that an extra eco-tax on the dirty good is introduced. If the substitution effect for this good is so strong that the effective tax revenue from the dirty good decreases, i.e. a Laffer effect results, revenue neutrality requires a higher labor tax rate, which in any case leads to an increase in aggregate excess burden and hence to an absence of the strong double dividend. In the standard case in which the labor supply curve is upward sloping, the lower net wage then entails less employment. Only in the exceptional case of a decreasing labor supply curve a positive employment effect would be brought about by the eco-tax reform. In a very simple model, we thus realize that the slope of the labor supply curve is of much relevance for specific effects in the context of the double dividend.

Control Question

"An environmental tax reform will reduce unemployment." How can this assertion be qualified? What can be said against it?

In extreme cases, it is even possible that taxation of a "dirty" good, accompanied by an income-neutral fiscal relief for labor and capital, can lead to an increase in emissions. In this case, there is not even a first dividend, i.e. a welfare increase resulting from an improvement in environmental quality. However, due to their complexity, we cannot discuss such – quite surprising – results in more depth. For a more extensive discussion of this issue, see, e.g., Goulder (2013).

4.5.4 Conflicts of Environmental Taxes with Other Economic Objectives

If environmental taxation entails positive welfare effects and even a strong double dividend leads to an overall more efficient tax system, a "green" tax reform nevertheless might come into conflict with distributional objectives. Especially these problems are the main obstacles to the implementation of environmental taxes in the political process.

A major reason for adverse distributional effects of environmental taxes is that energy consumption of households only rises disproportionately with household income. This applies to fuel consumption of vehicles ("a four times as expensive car usually requires less than four times as much gasoline"), to electricity consumption of electrical appliances ("the differences between individual refrigerators are not that large"), as well as to energy consumption of heating systems ("the living space does not grow proportionally with household income"). The distributional effects of energy taxes are therefore *regressive*, i.e. the tax burden relative to household income falls with increasing income.

Definition

The distributional effects of a tax are called *regressive (progressive)* if a taxpayer's tax burden increases less (more) than proportionally with his income.

In contrast to the past, this regressive effect of eco-taxes is no longer mitigated by the fact that people with low incomes do not possess energy-intensive devices like cars, refrigerators, central heating, etc. In addition, air travel with high kerosene consumption is nowadays not a privilege of the rich anymore.

The regressive effects of environmental taxes are intensified in many cases (at least in the short-term) by high investment costs of energy saving measures, which, therefore, can only be afforded by earners of higher incomes. This applies both to the purchase of vehicles with lower or even zero fossil fuel consumption ("electric cars are still more expensive than cars with petrol engines") as well as to thermal insulation of buildings and alternative heating systems like heat pumps. If budget constraints (combined with limited access to credit financing) prevent the poor from making energy-saving investments, they might in addition be hit harder than the rich by environmental taxes. That low-income households are overly burdened by environmental taxes is further supported by the following observations:

- Higher-income earners are statistically more likely to own residential property and therefore are more interested in making investments for saving energy. Tenants, on the other hand, are dependent on the investment decisions by their landlords who also benefit from the increased property value.
- With increasing income, the demand for personal services (e.g., visiting restaurants), cultural activities (e.g., theaters), and high-quality handmade products (e.g., furniture), whose production is associated with relatively low energy consumption, increases disproportionately. Hence, the difference between consumption patterns of poor and rich households amplifies the regressive distributional effect of environmental taxes.

There are a lot of approaches how to mitigate these regressive distributional effects of eco-taxes. One of them, as applied in Switzerland, is the use of an eco-bonus system which essentially means an equal per-capita rebate of the tax revenue. However, environmental taxation not only distributes tax burdens unequally among different income groups, it also exhibits interregional distributional effects, e.g., fuel taxes are particularly affecting residents of rural areas who often have to commute to their workplaces over long distances because sufficient public transport is nonexistent.

A group of people that, in this respect, is particularly disadvantaged by environmental taxes are families with children who have moved to the countryside because there is no affordable housing for them in large cities. In the same context, also a “hold-up” problem arises: households that have committed themselves by buying a house in the countryside can easily be exploited (and then feel deceived) by the government by later increases of fuel taxes. They additionally suffer since the value of their residential property in a rural area will fall due to a higher fuel tax, while prices of real estate near the city center or at least close to public transport will increase.

Equity of taxation not only refers to the distributional effects of different taxes but also to the rules according to which an individual’s tax burden is assessed. In this context, especially observance of the *ability-to-pay principle* is required, which means that everyone should be taxed according to his or her personal capability to do so. By using taxes on emissions or on the production or consumption of dirty goods, it is, however, not possible to take the individual situation of a taxpayer (as, e.g., his family or health status and the implications for his actually disposable income) into account so that one has to refrain from the ability-to-pay principle and the corresponding idea of fair taxation. Yet, this problem is not specific for environmental taxation, but already occurs in the framework of conventional taxation when consumption taxes are applied whose tax rates are not differentiated enough between essential and luxury goods.

Concerning the problems related to environmental taxation, one has to note as well that for the government there is a potential conflict between the primary *incentive effect* of environmental taxes, i.e. the reduction of emissions and the improvement of environmental quality, on the one hand, and the government’s desire to collect a certain tax revenue on the other. Hence, there is a danger that a government, being mainly interested in high tax payments, will lose interest in additional emission reduction and the advancement of environmentally friendly technical progress. Opportunities for a continual improvement of environmental quality may be lost in this way.

Even though a complete erosion of the environmental tax base is not to be expected (since neither a Pigouvian tax nor a tax according to the price-standard approach are aiming at zero emissions)[2], revenue of eco-taxes will not develop along with GDP, while it can be expected that the demand for additional public services will increase with a growing GDP. Moreover, since the incentive effects of environmental taxes are generally not well predictable, tax revenues become more uncertain in an extended eco-tax system, which makes government’s budget planning more difficult.

In the decoupling of tax revenues from economic growth, however, one could also see an advantage of environmental taxation. If one assumes that the share of the public sector is too high anyway, environmental taxes could be regarded as a kind of automatic brake for further expansion of state activity. However, there is also some risk that free-spending politicians will react to a decline of the environmental tax base with an increase in the tax rate. By anticipating this possibility, planning security on the part of taxpayers is undermined.

The expectations of individuals about the government’s future behavior play a significant role for the acceptance of environmental taxation. In reality, it is hard for the government to make a credible promise that an ecological tax reform will be revenue neutral (“when environmental taxes are levied other taxes are reduced”). So citizens cannot

2 Except for the taxation of greenhouse gases.

escape the fear that environmental taxes are (ab)used as a vehicle for increasing the overall tax burden (see Fairbrother, 2019). Therefore, it could be helpful for the implementation of an ecological tax reform if all relevant political parties had acquired the reputation of operating economically and restraining government spending. From this perspective, thrifty parties that have acquired such a reputation of obeying budgetary discipline could ultimately be more successful in the introduction of ecologically motivated taxes.

Control Question

What are the pros and cons of an environmental tax reform? Try to give a summary!

Conclusion

You have learnt in this section that:

- Emission taxes have a "double dividend" as their revenue can be used to reduce the usual taxes and the welfare losses ("excess burdens") caused by them.
- Somewhat paradoxically the optimal eco-tax may become lower when this double dividend is taken into account.
- Beyond this *weak* double dividend, even a *strong* double dividend may arise, which means that total excess burden of the entire tax system decreases.
- The occurrence of the strong double dividend depends on specific factors that are known from the theory of optimal taxation.
- Eco-taxes may come into conflict with distributional objectives.

4.6 Emission Trading: An Alternative Incentive-Based Instrument of Environmental Policy

In the case of environmental taxes, the price of emissions is exogenously determined by the government. It is only modified from time to time, e.g., in order to adjust it to new technological conditions or to altered preferences of individuals. In order to cope with environmental problems, one could alternatively delegate the determination of the emission price to markets. Already in the 1960s, the American economist John Dales (1968) sketched how such markets, on which allowances for emissions are traded, could be established.

Meanwhile, emission trading has developed from a theoretical idea to important practical applications. One of the prototypical examples was the market for sulfur dioxide emission allowances that was introduced in the USA in 1995 to cope with the acid rain problem. Since then other emissions permit markets especially for greenhouse gas emissions have become increasingly important whose most prominent example is the European emissions trading system for carbon dioxide emission permits.

In such an emission allowance scheme, the government fixes the aggregate level of emissions (the "cap") that is tolerated in a certain period of time and divides this total amount into individual units or certificates, which grant the right to emit a certain level of the respective pollutant. As these certificates are tradable among the polluters, a *cap-and-trade system* is created. The market price of permits that is established on this market for emission allowances provides – just as a tax on emissions – a scarcity signal to polluters, which encourages them to abate emissions. Essentially, an emissions trading scheme thus exerts the same allocative effects as an environmental tax scheme. In this section, we will use a simple model to explain the general functioning of permit markets and their prospective advantages. Thereafter, we will discuss the problems that cap-and-trade schemes entail and which adversely affect their attractiveness.

4.6.1 The Functioning and the Prospective Advantages of Cap-and-Trade Systems

4.6.1.1 Selling Permits to Polluters

We assume that there are n polluters $i = 1, \ldots, n$, with initial emission levels e_i^0 and individual abatement cost curves $R_i(v_i)$, where v_i is the abatement level of the ith emitter. His residual emissions are given by $e_i = e_i^0 - v_i$. Based on this, we can describe the *demand function* $E^d(z)$ for emission permits, which indicates the total amount E of emission permits the n polluters wish to purchase when the permit price is z. In a first scenario we make the following two assumptions:

- The government *auctions* a certain amount of emission permits.
- The emitters can only obtain permits by purchasing them in the auctioning process.

Thus, in this scenario the emitters do not have an initial endowment with permits but have to buy it from the government. If an emitter has purchased a certain amount e_i of permits, he is allowed (for a specified period of time) to release the respective quantity of emissions. Let us now determine polluter i's individual demand function for permits when the permit market is assumed to be competitive so that each polluter acts as a price-taker.

Given a permit price z, polluter i faces total costs of

$$R_i\left(e_i^0 - e_i\right) + ze_i,$$

where e_i indicates his residual emissions, i.e. $v_i = e_i^0 - e_i$ is polluter i's abatement level. Then $ze_i = z\left(e_i^0 - v_i\right)$ are his expenses for the purchase of the permits (= residual emissions) that he needs. The minimization of an emitter's total costs yields the following first-order condition:

$$R_i'\left(e_i^0 - e_i\left(z\right)\right) = z.$$

In order to minimize his total costs, polluter i adapts to the given permit price z by buying that amount of allowances $e_i(z)$ for which his marginal abatement costs (MAC) become equal to z. Thus, graphically, the individual demand function $e_i(z)$ for emission permits can be depicted by drawing the marginal abatement cost curve $R_i'\left(v_i\right)$ starting in e_i^0 from the right to the left (see Fig. 4.17).

As usual we assume that marginal abatement costs $R_i'\left(v_i\right)$ are increasing in the abatement level v_i. The polluter then abates emissions up to the level where marginal

Fig. 4.17 An individual demand curve for emission permits

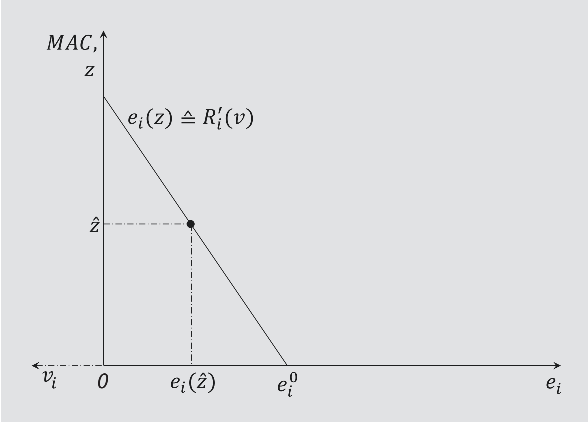


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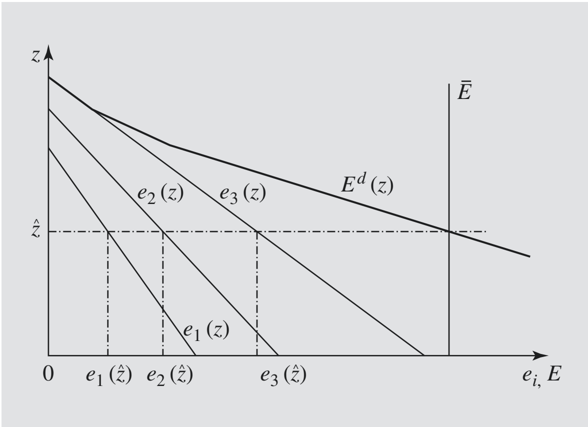
Fig. 4.18 The overall demand function for emission permits



abatement costs equal the permit price z, i.e. where the abatement of this unit and the purchase of permits for one additional unit of emissions become equally costly. To buy more permits would be more expensive than abating emissions. It is obvious from Fig. 4.17 that, as a consequence of increasing marginal abatement costs, the demand function $e_i(z)$ is decreasing in the permit price z.

The aggregate demand function $E^d(z)$ that describes total permit demand of all polluters is obtained by horizontal aggregation of the individual demand functions (see Fig. 4.18), i.e. $E^d(z) = \sum_{i=1}^{n} e_i(z)$.

Control Question

How is polluters' demand for emission allowances determined when permits are sold by the government?

If the government fixes a cap $\bar{E}$ of tolerated aggregate emissions and issues the corresponding volume of emission allowances, this leads to a supply curve for permits that is a parallel to the z-axis and thus is completely price inelastic. The competitive permit market equilibrium with the permit price $\hat{z}$ then is located at the point of intersection between the demand function $E^d(z)$ and the supply curve, which is a vertical straight line at $\bar{E}$.

In the market equilibrium, the marginal abatement costs of all polluters are equal to the equilibrium price $\hat{z}$. This, in particular, implies that all polluters face the same marginal abatement costs in the market equilibrium. Hence, a *cost-effective outcome* is obtained in which the specified total abatement level $\bar{V} = E^0 - \bar{E}$ (with $E^0 = \sum_{i=1}^{n} e_i^0$) is attained with minimum total abatement cost. This shows that cost-effectiveness of abatement cannot be realized exclusively through environmental taxation, but also through the creation of an emissions trading system.

In contrast to environmental taxation, it is ensured by the application of a permit scheme that the emission target $\bar{E}$ is achieved – since it is directly stipulated by the government. There is no need for the government anymore to set a price "fitting" exactly its emission target $\bar{E}$. Therefore, the government is freed from performing a task that it could have hardly fulfilled because of limited information. Consequently, the permit scheme avoids a serious problem associated with environmental taxes from the outset, which is considered to be a key advantage of emission trading over environmental taxation. Furthermore, those welfare risks are excluded that may result (as described in the model of Weitzman, 1974) from unknown marginal abatement cost curves.

If the government sells the emission permits to polluters, this also generates additional public revenues that can – like environmental tax income – be used to reduce distorting taxes. In the ideal case, emission trading will bring about the same double dividend as an environmental tax. In this respect, however, differences between both instruments may occur if risk aspects are taken into account since revenue may become more uncertain for the government under emission trading when permit prices are fluctuating strongly. Furthermore, since the demand for permits heavily depends on the polluters' expectations, a precise calculation of the revenues from selling the permits becomes rather difficult.

4.6.1.2 Allocating Permits to Polluters Free of Charge

The auctioning of emission permits is not the only option for establishing a permit market. An alternative to auctioning is the allocation of emission permits to polluters, e.g., in proportion to their original emissions, free of charge. Polluters can then use these given permits either by themselves, or they can sell them to other polluters instead.

There are different methods that can be used to allocate emission allowances to polluters free of charge: in the *grandfathering* scheme, permits are allocated to polluting firms on the basis of historical data in particular on emissions or fuel use (see, e.g., Böhringer & Lange, 2005: 2042). In the *benchmarking* scheme, firms instead receive permits according to external criteria that reflect technological standards for abatement measures that polluters should be able to fulfill.

Definition

When emission allowances are given free of charge:
- A *grandfathering procedure* is applied if a polluter's permit allocation is positively correlated with the polluter's past emissions.
- A *benchmarking procedure* is applied if a polluter's permit allocation depends on an external standard as set, e.g., by a best available technology.

To describe the functioning of the permit market in a second scenario where permits are given away to polluters free of charge, the amount of allowances that is allocated to polluter i before trading starts is denoted by $\overline{e}_i$. If the total emission target is $\overline{E}$ as before, clearly $\sum_{i=1}^{n} \overline{e}_i = \overline{E}$ must hold.

The functions $e_i(z)$ as specified already above (in the first scenario) now do not exclusively specify the *demand* of the ith polluter, but they might as well describe his *supply* of permits.

Whether a polluter will demand or supply permits depends on the permit price z. The *critical value* of the permit price $\overline{z}_i$, at which polluter i, endowed with an amount of permits $\overline{e}_i$, is indifferent between buying and selling, is given by $\overline{z}_i = R_i'\left(e_i^0 - \overline{e}_i\right)$ if $\overline{e}_i < e_i^0$, i.e. if the assigned number of permits $\overline{e}_i$ is smaller than the polluter's initial emission level e_i^0. If the permit price deviates from this critical value, polluter i will behave as follows:
- If $z < \overline{z}_i$, then emitter i purchases permits of the quantity $e_i\left(z\right) - \overline{e}_i$.
- If $z > \overline{z}_i$, then emitter i sells permits of the quantity $\overline{e}_i - e_i\left(z\right)$.

Control Question

On which characteristics of a polluter does it depend whether he wants to be a buyer or seller of emission permits?

In the competitive market equilibrium, aggregate supply of permits must be equal to aggregate demand. The formal condition for the equilibrium market price $\check{z}$ thus is

$$\sum_{i \in purchaser} \left(e_i\left(\check{z}\right) - \overline{e}_i\right) = \sum_{i \in supplier} \left(\overline{e}_i - e_i\left(\check{z}\right)\right).$$

Adding $\sum_{i \in supplier} \left(e_i\left(\check{z}\right) - \overline{e}_i\right)$ on both sides yields

$$\sum_{i=1}^{n} \left(e_i\left(\check{z}\right) - \overline{e}_i\right) = 0.$$

As $\sum_{i=1}^{n} \overline{e}_i = \overline{E}$, this is equivalent to:

$$\sum_{i=1}^{n} e_i\left(\check{z}\right) = \overline{E}.$$

After this transformation, the condition for the market equilibrium in the second scenario is identical to that in the first scenario, i.e. $E^d(\hat{z}) = \overline{E}$, where $E^d(\hat{z}) = \sum_{i=1}^{n} e_i(\hat{z})$. Therefore, market prices in the two scenarios are equal, i.e. $\check{z} = \hat{z}$. In both scenarios, a polluter chooses the same abatement levels $e_i^0 - e_i(\hat{z}) = e_i^0 - e_i(\check{z})$, for which marginal abatement costs are equal to the equilibrium permit prices. Hence, abatement efforts of each polluter are identical in both systems.

With regard to the *allocative effects*, i.e. with a view to activities of the economic agents and the resulting welfare effects, in the framework of our simple model it is therefore not relevant in which way the permits are assigned to polluters. It can easily be shown that this kind of neutrality also holds for mixed systems in which permits are partly allocated free of charge and partly auctioned.

However, the two approaches of permit allocation discussed above clearly differ with respect to their *distributive effects*. Every emitter who receives an initial endowment of permits is better off when he gets the permits for free than when he has to buy them. The extent of this advantage depends on the amount of permits being made available by the government and the precise criteria for their distribution. On the one hand, the financial relief of polluters through a free allocation of permits can be regarded as an advantage as the double burden for polluters, and the associated welfare risks, can at least be attenuated. On the other hand, however, no revenues for the government are generated so that no double dividend can be achieved. Moreover, in a grandfathering scheme, the free allocation of permits tends to create a fairness problem since it rewards polluters for bad behavior, in particular those with high emissions in the past.

While under ideal conditions the method by which permits are allocated to polluters does not play any role for emission levels and aggregate welfare, this is not necessarily the case in reality, e.g., when polluters anticipate a grandfathering procedure, they may deliberately increase their pollution in order to obtain more allowances later on. Moreover, polluters that have obtained permits free of charge might want to hoard part of them to hedge against higher allowance prices in the future by driving their abatement activities to non-optimally high levels. Such effects prevent the permit market from attaining an efficient outcome.

? Control Question

What additional effects that are caused by the method of free permit allocation may thwart the efficiency of emission trading?

4.6.2 Dynamic Efficiency of Permit Schemes

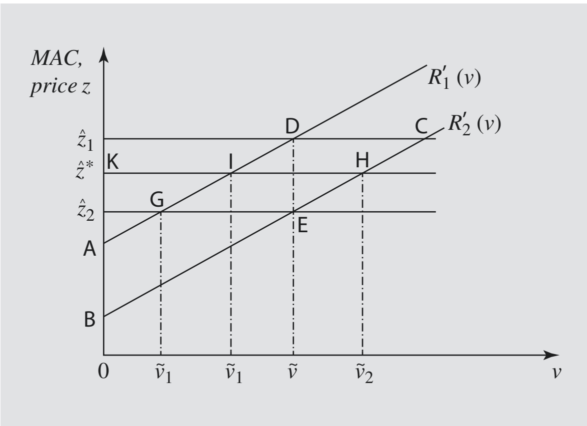
With respect to the promotion of environmentally sound technological progress, there is a significant difference between environmental taxes and emissions trading schemes. On the emission permit market, the price of residual emissions is variable and depends – from an individual polluter's point of view – also on the abatement activities of the other polluters. If emitters switch to an abatement technology with low marginal abatement costs, then the demand for permits will decrease, and, consequently, the permit price will decline. Thereby, the gain that a polluter may get from switching technologies will shrink when other polluters have already accomplished the technology change. A polluter may even regret an early technology shift – that was profitable at the initial permit price – because after many other polluters have also changed technology, it is not profitable anymore. In order to make these considerations precise, we extend the model on dynamic incentive effects developed above in ► Sect. 4.3.2 to the case of a permit market (also see Requate & Unold, 2003).

To this end, let us assume that there is a large number n of identical polluters with initial emissions of e_0, respectively, so that aggregate initial emissions are $E_0 = ne_0$. Every polluter has the possibility to switch from his initial abatement technology (the "old" technology) with marginal abatement costs $R_1'(v)$ to an abatement technology with lower marginal abatement costs $R_2'(v)$ (the "new" technology). The transition, however, incurs (additional) fixed costs F.

The government strives for a reduction of aggregate emissions to the level $\bar{E} < E_0$ and, consequently, auctions emission allowances to the same extent. Since the polluters have to abate the aggregate amount of emissions $\bar{V} = E_0 - \bar{E}$, in a symmetric solution each polluter has to contribute $\bar{v} = \bar{V} / n$ to that. If all polluters apply the first abatement technology, then the equilibrium price is $\hat{z}_1 = R_1'(\bar{v})$, i.e. each polluter attains point D in ◘ Fig. 4.19. If all polluters apply the second abatement technology instead, then we get the equilibrium price $\hat{z} = R_2'(\bar{v})$, and polluters are in point E.

Under certain conditions, $\hat{z}_1$ or $\hat{z}_2$ already fully describe the equilibrium solution for the permit market:

- If $F > ABCD$, then the equilibrium price is $\hat{z}_1$, since in this case no polluter has an incentive to change the abatement technology.



◘ **Fig. 4.19** Equilibrium on the permit market when two abatement technologies are available

- If $F < ABEG$, then the equilibrium price is $\hat{z}_2$, since in this case all polluters prefer to use the new abatement technology.

The remaining case is more intricate: if $ABCD > F > ABEG$, polluters have an incentive to change technology starting from the initial state in which all polluters apply the original abatement technology and the permit price is $\hat{z}_1$. However, they would regret the technology shift after all polluters have chosen the new technology so that the permit price has declined to $\hat{z}_2$. Therefore, neither the permit price $\hat{z}_1$ nor the permit price $\hat{z}_2$ can be equilibrium prices in this case.

What makes the determination of the market equilibrium difficult in this case can be explained by a heuristic consideration: let us assume that an individual polluter speculates that all other emitters will replace their initial technologies. Then the polluter will expect that he will enjoy the lower permit price $\hat{z}_2$ without having to bear the costs for switching technologies. Such a strategically thinking polluter will aspire to take a "free ride" on the lower permit price at the expense of the other emitters who have switched technology. If, however, all emitters would act in this manner, no one would shift to the new technology – and everything would remain as it was.

Due to the complexity of the strategic interaction in the case of $ABCD > F > ABEG$, it hence remains unclear for the time being, which equilibrium we can expect on the permit market.

To pave the way to the determination of the equilibrium outcome, we assume that the emitters will only successively switch from the old to the new technology. This might be due to the circumstance that the adoption of the second technology involves learning and imitation processes, which require some time.

Because of $ABCD > F$, some emitters will – given the permit price $\hat{z}_1$ – switch technologies immediately. These emitters will abate more afterward and will demand a smaller amount of permits, which in turn will cause the permit price to fall somewhat below $\hat{z}_1$. Because the permit price does not decline too much initially, there remains an incentive for other polluters to switch technologies, too. When further polluters adopt the new technology, the permit price continues to fall until it reaches the level $\hat{z}^*$ where both technologies are equally profitable. Since $F > ABEG$, clearly $\hat{z}^* > \hat{z}_2$ results. At price $\hat{z}^*$, polluters will be indifferent between both technologies. This critical price level $\hat{z}^*$ (see ◘ Fig. 4.19) is characterized by the condition that the reduction of running abatement cost due to the technology change must equal the fixed costs of the transition, i.e. $ABHI = F$ must hold.

At the permit price $\hat{z}^*$, no further polluter has an incentive to switch technology, and no polluter who has already adopted the new technology regrets his decision. With regard to the technology choice, at $\hat{z}^*$ we indeed face a (Nash) equilibrium on the permit market, which in its general form is defined as follows:

Definition

A *Nash equilibrium* prevails if – given the actions of all other agents – no agent has an incentive to change his own action.

Control Question

Why do polluters have an incentive to change the abatement technology if the permit price z is above $\hat{z}^*$, but have no such incentive if z is below $\hat{z}^*$? Explain!

In choosing their individual abatement level, agents using the old and agents applying the new technology adjust to the equilibrium price $\hat{z}^*$ in different ways. Users of the old technology choose abatement level $\hat{v}_1$ (as defined by $R_1'(\hat{v}_1) = \hat{z}^*$), while the abatement level of the polluters applying the new technology is $\hat{v}_2$ (as defined by $R_2'(\hat{v}_2) = \hat{z}^*$). The equilibrium on the permit market hence is *asymmetric,* which means that polluters do not act identically, neither with respect to the technology choice nor with respect to the choice of the individual abatement level.

4

The number $\hat{m}$ of polluters that in this equilibrium have chosen the new technology is determined by the following condition,

$$\left(n-\hat{m}\right)\hat{v}_1 + \hat{m}\hat{v}_2 = \bar{V}$$

as the aggregate abatement level $\bar{V}$ has to be achieved by the abatement efforts of *all* emitters, i.e. by those applying the old and those applying the new technology. Therefore, we get:

$$\hat{m} = \frac{\bar{V} - n\hat{v}_1}{\hat{v}_2 - \hat{v}_1} = n\frac{\bar{v} - \hat{v}_1}{\hat{v}_2 - \hat{v}_1},$$

Having explored what the equilibrium on the permit market looks like when two different abatement technologies are available, in a next step we will take a closer look at the welfare properties of the equilibrium outcome. In doing so, we minimize the aggregate abatement costs of all emitters where the number m of the users of the new technology is initially not determined. To facilitate the exposition, the number m is regarded as a continuous variable (and not as an integer). Yet, given a large number of n emitters, the solution determined in this way does not deviate by very much from the actual optimum with an integer m.

To maximize social welfare, we have to minimize total abatement costs

$$\left(n-m\right)R_1\left(v_1\right)+m\left(R_2\left(v_2\right)+F\right)$$

so that

$$\left(n-m\right)v_1 + mv_2 = \bar{V},$$

by choosing v_1, v_2, and m. This yields the Lagrangian function

$$L = \left(n-m\right)R_1\left(v_1\right)+m\left(R_2\left(v_2\right)+F\right)-\lambda\left(\left(n-m\right)v_1 + mv_2 - \bar{V}\right),$$

which we have to differentiate for v_1, v_2 and m. The first-order conditions for the social optimum as characterized by v_1^*, v_2^* and m^* are:

(*i*) $\left(n-m^*\right)R_1'\left(v_1^*\right)-\lambda^*\left(n-m^*\right)=0$

(*ii*) $m^*R_2'\left(v_2^*\right)-\lambda^*m^*=0$

(*iii*) $-R_1\left(v_1^*\right)+R_2\left(v_2^*\right)+F+\lambda^*\left(v_1^*-v_2^*\right)=0$

From (*i*) and (*ii*), we directly get

$$\lambda^*=R_1'\left(v_1^*\right)=R_2'\left(v_2^*\right),$$

and from (*iii*) we obtain

$$F=R_1\left(v_1^*\right)-R_2\left(v_2^*\right)+\lambda^*\left(v_2^*-v_1^*\right)=\lambda^*v_2^*-R_2\left(v_2^*\right)-\left(\lambda^*v_1^*-R_1\left(v_1^*\right)\right).$$

This condition is fulfilled if $\lambda^*=\hat{z}^*$, $v_1^*=\hat{v}_1$ and $v_2^*=\hat{v}_2$. This can be seen from ◘ Fig. 4.19 where

$$\lambda^*v_2^*-R_2\left(v_2^*\right)=\hat{z}^*\hat{v}_2-R_2(\hat{v}_2)=BHK$$

and

$$\lambda^*v_1^*-R_1\left(v_1^*\right)=\hat{z}^*\hat{v}_1-R_1(\hat{v}_1)=AIK.$$

Since $ABHI = BHK - AIK$ and – according to the definition of $\hat{z}^*$ – it holds that $F = ABHI$, it is confirmed that the asymmetric permit market equilibrium entails a minimization of aggregate abatement costs and – in this sense – provides a socially optimal solution. From this we can draw the conclusion that the adjustment of the permit price:

- Mitigates the incentives to support climate friendly technological progress (what is bad news)
- Leads to a cost-effective attainment of the abatement target (what is good news)

Control Question

Which welfare loss in comparison to the first-best outcome would result under an emission tax with either the tax rate $\hat{t}_1=\hat{z}_1$ or the tax rate $\hat{t}_2=\hat{z}_2$? When is it possible that the first-best solution is implemented through an emission tax system?

4.6.3 Problems with Emission Trading

Only under certain conditions emission trading will work properly and bring about the desired outcome. If these conditions are not present, serious problems arise that impede the applicability of this environmental instrument and threaten its fundamental advantages. The problems that are associated with emission trading will now be addressed.

4.6.3.1 Lack of Perfect Competition on the Market for Emission Permits

The minimization of total abatement costs represents the key advantage of the trading scheme for emission permits. Yet, to attain a cost-effective outcome, there must be perfect competition on the emission permit market, i.e. many agents that all act as price-takers must participate in emission trading. In a system where polluters obtain permits from the government free of charge, this requirement has to be fulfilled both on the demand and on the supply side. If in this scenario, however, there are only few agents on one market side, a cap-and-trade system no longer leads to cost-effectiveness of abatement (see Hahn, 1984, for the basic theoretical analysis and Hintermann, 2017, for an empirical study of market power effects in the European emissions trading system). We illustrate this by an extreme example in which there is a single (old) firm M on the supply side that initially has been the only polluter and hence is in possession of all emission certificates $\overline{E}$. The firm's marginal abatement cost function is $R_M'(v_M)$, and its initial emission level is e_0^M. On the demand side, there is a large number n of small polluting firms that enter the permit market for the first time. For the sake of simplicity, they all have the same marginal abatement cost curve $R'(v)$ and the same initial emission levels e_0.

We now consider how the market equilibrium looks like if firm M as a monopolist strives for the maximization of its own profit: if M sells the amount $E \le \overline{E}$ of its permits to the small new firms, in a symmetric market equilibrium each of these polluters purchases the amount $\frac{E}{n}$ of permits and thus has to abate $e_0 - \frac{E}{n}$ emissions (where $ne_0 > \overline{E}$ is assumed). Then the marginal abatement costs of every small polluter are $R'\left(e_0 - \frac{E}{n}\right)$. As we assumed that many of these price-taking polluters exist, the permit price $z(E)$ in the market outcome must equal these marginal abatement costs, i.e.

$$z(E) = R'\left(e_0 - \frac{E}{n}\right).$$

Selling E permits, M retains $\overline{E} - E$ permits, which it will use for its own emissions. Then it has to abate the amount $e_0^M - \left(\overline{E} - E\right)$ of emissions so that the monopolist's abatement costs are

$$R_M\left(e_0^M - \left(\overline{E} - E\right)\right) = R_M\left(E + e_0^M - \overline{E}\right),$$

which gives the monopolist's net profit

$$z(E)E - R_M\left(E + e_0^M - \overline{E}\right).$$

The first-order condition, which characterizes the amount $\hat{E}$ of permits whose sale maximizes M's profits, hence is

$$R_M'\left(\hat{E} + e_0^M - \overline{E}\right) = z\left(\hat{E}\right) + z'\left(\hat{E}\right)\hat{E}.$$

Observing $z'(E) = -\frac{1}{n} R''\left(e_0 - \frac{E}{n}\right)$, the first-order condition above becomes equivalent to

$$R'_M\left(\hat{E} + e_0^M - \bar{E}\right) = R'\left(e_0 - \frac{\hat{E}}{n}\right) - R''\left(e_0 - \frac{\hat{E}}{n}\right)\frac{\hat{E}}{n}.$$

Given marginal abatement costs that are increasing with the abatement level, i.e. $R''(v) > 0$, we then have

$$R'_M\left(\hat{E} + e_0^M - \bar{E}\right) < R'\left(e_0 - \frac{\hat{E}}{n}\right).$$

As a consequence, a minimization of total abatement costs is not attained since this would require an equalization of the marginal abatement costs of *all* polluters. Here, however, the marginal abatement costs of the monopolistic supplier of permits are lower than the marginal abatement costs of the purchasers. Because the monopolist M shortens the supply of permits to increase his profits, he abates too little as compared to the least-cost outcome. In contrast, the other polluters who now get relatively fewer permits will abate too much.

Box 4.4: Unused Permits in the Monopolistic Case
In the monopolistic scenario, we have assumed that the initial emissions of the monopolist are so high that he will have to abate emissions. Then, the total amount $\bar{E}$ of permits allocated by the government are used. However, if $e_0^M < \bar{E}$, it becomes possible that the monopolist's profit-maximizing amount of permit sales is smaller than $\bar{E} - e_0^M$. Then, after selling permits, the monopolist still owns more permits than he could use himself, and these permits will remain unused so that the objective of the price-standard approach is violated.

A more even distribution of permits among polluters prevents the risk of monopolistic behavior and, henceforth, the risk of the related welfare losses. Yet, when there is only a small number of polluters in the initial state, it might be relatively easy for them to collude and to form a cartel, which mainly is detrimental for new entrants to the permit market who do not have an initial endowment of permits. To avoid the concomitant barriers of entry, special regulations for newcomers are required when permits are supplied for free. By auctioning off at least part of the permits, the government can mitigate problems arising from a market dominance of specific polluters since in such a system old and new polluters are treated more equally. Completely equal treatment of all actual and potential polluters is achieved if all allowances are auctioned.

? Control Question
How would the model treated above be changed if the small polluters were initially given part of the emission cap $\bar{E}$?

4.6.3.2 Potential Abuse of the Permit Market to Eliminate Competitors

If an auctioning procedure is applied, another related risk may occur. At least in principle it is conceivable that a single polluter purchases all issued permits $\bar{E}$ in order to eliminate his competitors on markets for goods and production factors. If competitors are excluded from the use of permits, they must cease to produce or at least reduce their production. Then, the sole owner of permits gets high market power either as monopolist on a goods market or as monopsonist on a factor market (e.g., hoarding of permits by a single firm tends to reduce the demand on the labor market and thus to decline the wage rate).

To make such a strategy successful for a firm, it must compare its monopoly or monopsony rents being generated in this way with the cost of ousting the competitors from the market. Given the total amount $\bar{E}$ of permits and $\bar{z}$ as the permit price at which the competitors are forced to cease their production, these costs amount to $\bar{z} \cdot \bar{E}$. They are the lower, the smaller $\bar{E}$, and therefore, the smaller the extent of the permit market is. In a large nationwide or even international emissions trading system, strategies to eliminate competitors therefore do not seem to be promising. In the case of small local permit markets, much depends on how easily the production of polluting firms threatened of being driven away can be relocated to other regions and how mobile the relevant production factors are. If there is complete mobility of, e.g., the factor labor, then there exists a uniform wage rate nationwide, so that attempts to manipulate regional wage rates clearly are doomed to failure. The same applies if polluters can rapidly develop and use cheap abatement options making the purchase of permits largely unnecessary. In general, the potential of using the permit market as a tool for eliminating competitors depends on the flexibility of polluters, i.e. how much their production depends on emission allowances and how easily they can evade an increase of the permit price.

Concerning the empirical relevance of this argument, also note that in many cases a dominant market position might be attained more easily by the purchase of patents for a group of goods (e.g., medications) or by the poaching of highly specified experts (e.g., researchers). These methods for evicting other firms will usually be of much higher importance than the squeezing out of competitors via the abuse of emission trading.

? Control Question

What measures can be employed to avoid that polluting firms strategically use an emissions trading system to keep other firms out of a goods or factor market?

4.6.3.3 Price Risks on Small Markets

Small permit markets with a small number of participants and few transactions are to a large extent subject to random influences, which tends to cause a high price volatility. If the total number of polluters is 10, an idiosyncratic shock on the demand for permits caused by a single polluter clearly has a much stronger influence on the permit price than if there were 100 polluters present at the market. If the risks underlying these shocks are stochastically independent, they will balance out on a large permit market with many participants, so that the price risk is reduced. On small permit markets, one instead has to be aware of the risk that in extreme situations the permit price might go to infinity, which means that no permits are available on the market anymore and firms lacking permits would have to stop their production.

In the case of high price risks, polluters become reluctant to invest in the production sites included in a small emissions trading system or have an incentive to relocate their production to other regions, which both will cause job losses at least at the regional level. Alternatively, polluters may expand their abatement efforts to make themselves independent from the purchase of emission permits and thus to be on the safe side. Then, the permits issued by the government will possibly not be fully used. While this effect would be beneficial from a purely ecological perspective, the financial burden for polluting firms may, however, increase significantly by following such a hedging strategy. This not only impairs the international competitiveness of these firms, but the precise objectives both of the price-standard approach (abatement of a specific amount of emissions) and of the Pigouvian approach (attainment of a social optimum) are exceeded and thus missed. Moreover, the higher investments in abatement technologies confine the firms' possibilities for other investments in expanding their production activities, in innovative projects, and in restructuring and cost-saving measures. This problem can only partly be alleviated by raising a firm's borrowing as – due to a higher bankruptcy risk – interest rates are rising with an increased debt ratio.

The considerations above demonstrate that functional shortcomings of emissions trading systems will be more likely when permit markets are small. Then the economic advantages that permit markets are promising cannot fully be exploited. The most obvious approach to prevent these deficiencies would be to expand the scope of a permit market, which can be done in different ways, i.e. by:

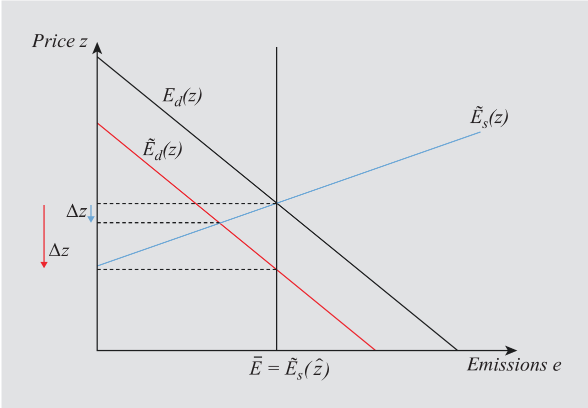
- A spatial expansion, i.e. by establishing large supra-regional or international permit markets.
- A temporal expansion, i.e. in a certain period of time (e.g., the year 2020) not only the permits issued specifically for this period are available on the market, but also yet unused permits which have been issued in the past or will be issued in the future. The use of these "older" permits is possible if the regulator allows for banking and borrowing of permits, i.e. if the polluters are given the right to carry unused permits forward and backward between different compliance periods. Banking and borrowing implies that the supply of permits *in a certain period* is no longer completely price inelastic as assumed before (see ◘ Fig. 4.18). Rather, due to the increased flexibility granted by banking and borrowing, an increasing supply curve $E_s(z)$ for a single trading period is obtained. As seen from ◘ Fig. 4.20, a certain change of permit demand will then change the equilibrium price of permits to a lesser extent than in the case of a fixed permit supply.

  Following this argument, it can be expected that introducing an option for banking and borrowing of permits brings about a steadier development of permit prices and consequently a risk reduction for polluters. Yet, under specific conditions the temporal expansion may also raise price risks because the banking option facilitates hoarding of permits by individual market participants.

Control Question

What factors could determine the slope of the supply curve for emission allowances in ◘ Fig. 4.20?

Fig. 4.20 A decline in permit demand



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- The expansion of the number of participants, so that not only polluters but also other agents like environmental groups are allowed to act on the permit market. These new market participants could purchase permits with the intention to prevent their use, which would reduce emissions below the level aimed at by the government – and thus would violate the price-standard approach. The new participants could also purchase permits in order to resell them on better terms to environmentally friendly firms and hence implicitly subsidize their production. Since the behavior of such private groups can hardly be predicted, the number of permits that is actually available for polluters becomes more uncertain. Their investment risk is increased, which may impair economic development, but would clearly have positive effects on environmental quality.

An expansion of the permit market in either way bears substantial risks. A spatial as well as a temporal expansion may bring about a concentration of emissions at specific locations or in specific short time intervals. Spatial and temporal "hot spots" would arise. The extent of the resulting ecological problems depends largely on the type of pollutants to which emission trading applies. If a pollutant does not dissipate in the environmental medium so that high concentrations can result, and if there are nonlinear damage functions, then there is a major threat that such hot spots will seriously harm the local environment even in the short term. However, if the pollutant dissipates in the environment and damaging effects are caused in the long-run by the accumulated stock of the pollutant (like in the case of CO_2), then the establishment of a large permit market with banking and borrowing options can hardly be objected on ecological grounds.

Another possibility to avoid too strong fluctuations of the permit price and the concomitant high risk for polluters is the explicit introduction of price floors and price ceilings, whose compliance is ensured by market interventions of the government or a regulatory authority. Then the cap-and-trade system is complemented by an emission tax component, so that a *hybrid system* for pricing emissions results.

4.6.4 Cap-and-Trade Systems in Practice

Despite the widely recognized theoretical palatability of emission trading, there has been a lot of resistance against the application of this approach in reality. The opponents had quite different motives: on the one hand, partisans of the green movement considered trading of emission allowances as immoral since buying permits grants the right to do something that is essentially "wrong," i.e. to cause harm to others. On the other hand, especially owners of polluting firms were afraid that they would have to pay twice under a cap-and-trade system, i.e. not only for abatement measures but also for the emission rights, that would further reduce their competitiveness. In the meantime, these objections have become less important in the political debate because both parties did acknowledge the big economic advantages that emission trading could have, i.e. to get a better environmental quality at lower cost. (Yet, note that the famous philosopher Michael Sandel (see, e.g., Sandel, 2005) still today expresses ethically grounded reservations against emission trading.) This change of mind was supported by positive experiences that have been made with cap-and-trade systems and which showed that emission trading is not only an appealing theoretical idea but can also work in practice.

A major breakthrough of emission trading as an instrument of practical environmental policy came when in the 1990s the US federal government introduced a trading system for sulfur dioxide (SO_2) emissions. This first large-scale emissions trading system represented the central part of the "acid rain program", which aimed at reducing the acidification of aquatic ecosystems especially in the northeastern states. This program was successful as it helped to attain the environmental targets at aggregate abatement costs that were much lower than previously expected. The estimates of the cost savings that emission trading has achieved over counterfactual CAC scenarios vary greatly and lie – depending on which CAC instrument has been used for the comparison – between 225 million $ p.a. (Schmalensee, Joskow, Ellerman, Montero, & Bailey, 1998) and 1 billion $ p.a. (Stavins, 1998). Notwithstanding these differences, the conclusion is justified "that the SO_2 allowance-trading system provided a compelling demonstration of the cost advantages of a market-based approach" (Schmalensee & Stavins, 2013: 20).

In the past 20 years, climate change policy has become the most promising field of application for emission trading. The use of permit markets for combating global warming seems to be particularly attractive since greenhouse gases are well mixed in the atmosphere so that – unlike in the case of SO_2 – the market can be chosen arbitrarily large without any danger of causing hot spots (of particularly high atmospheric concentrations). Already the Kyoto Protocol of 1997, aiming at achieving international cooperation on greenhouse gas abatement, contained elements of emissions trading where this notion had been (in Article 17 of the Protocol) reserved for exchanges of emission reductions between the industrialized Annex-B countries (that were obliged to abate greenhouse gas emissions by the Kyoto Protocol). Beyond that the Clean Development Mechanism CDM (according to Article 12 of the Protocol) made it possible for polluters in the Annex-B countries to fulfil their commitments through investing in additional abatement projects in developing countries, while Joint Implementation JI (Article 6) allowed an exchange of emission reductions between projects in different Annex-B countries.

In the meantime, the European emissions trading system EU-ETS has, besides some smaller emission trading schemes (like the Regional Greenhouse Gas Initiative of nine northeastern US states and the Californian AB-32 Cap-and-Trade System), become the most prominent example of a cap-and-trade system in the climate change context – and at the moment it is the largest permit market in the world. The EU-ETS, which mainly covers the carbon dioxide emissions (plus emissions of nitrous oxide since 2008 and perfluorocarbons from aluminium production since 2013) in the 28 EU member states (and Norway, Iceland and Lichtenstein) of all power plants and of some industrial sectors (like steel, cement and pulp and paper production), started in 2005 with a short pilot phase that ended in 2007. Two further phases, one lasting from 2008 to 2012 and the present one lasting from 2013 until 2020, have followed, and the planning for the fourth period that will start in 2021 is already well-advanced. Remarkably, the rules underlying the EU-ETS have been revised drastically between its distinct phases, e.g., with respect to the allocation process for allowances, the possibilities for banking, the acceptance of emission offsets from outside the EU (especially as part of the Clean Development Mechanism), and precautions for stabilizing permit prices and thus the polluters' expectations. In ► Box 4.4 we briefly summarize the major design features of the EU-ETS and the concomitant problems in the different trading periods.

Box 4.5: The Development of the EU-ETS

Phase 1 (2005–2007): The caps were fixed independently by the member states through intricate "National Allocation Plans" NAPs, which lead to inconsistent and – due to massive lobbying by polluting firms – even strategic behavior by the various governments. Permits have been emitted through a grandfathering procedure (for about 90 percent of all included emissions), which created huge windfall profits for some polluters, particularly big power plants. Due to an overallocation of permits and the absence of a banking option to transfer permits to the subsequent trading period, the allowance price fell to almost zero in 2007.

Phase 2 (2008–2012): Permit allocation through grandfathering was continued. The reduced emission caps for this trading period could be easily attained since Certified Emission Reductions from CDM and Emission Reduction Units from JI were increasingly used (although in a legally limited way), while abatement efforts within the EU were not significantly increased. Due to the import of emission credits from outside the EU and the economic crisis that started in 2008, the permit price fell from over 20 Euro/t to about 6 Euro/t in 2012. Yet, the price did not drop to zero at the end of this second trading period since banking for use in the third period was now allowed. In 2012 aviation emissions from flights within the EU were included after much controversy with the USA and China, which made it impossible to include flights to destinations outside the EU.

Phase 3 (2013–2020): This trading period is characterized by a deep structural change of the EU-ETS design – NAPs are abandoned and replaced by an EU-wide cap that annually decreases by 1.74 percent till 2020. An auctioning system supplants the former grandfathering system completely for power plants (already in 2013 but with exceptions for poor EU countries in the East) and partially for manufacturers (with a continual decrease of the share of freely granted permits that are now allocated using a benchmark system). The lion's share (i.e. 88 percent) of the entire auctioning revenue is redistributed to the countries according to their CO_2 emissions in 2005. In 2013, permit prices fell even below 3 Euro and hence completely lost their signaling function. Therefore, first of all ad hoc "backloading" measures were taken in 2013 by which the volume of available allowances was reduced in the years 2014–2016. To stabilize permit prices in the long run later on (in 2015), a "market stability reserve" MSR was introduced to provide a rule-based mechanism by which – beginning in 2019 – the supply of allowances shall be balanced and primarily a price-depressing oversupply of permits shall be avoided. In the meantime permit prices have increased drastically.

Phase 4 (2020–2030): To increase scarcity of allowances and thus to stabilize their price, the linear reduction factor will (from 2021 on) be increased from 1.74 percent to 2.2 percent. To accelerate the reduction of the existing excess supply of permits, the annual quantity of allowances that is fed into the MSR is doubled (from 12 to 24 percent). Additionally, the volume of the MSR for some year is limited to the amount of allowances that have been auctioned in the year before. All permits above this level are definitely cancelled. To avoid a "waterbed effect," i.e. a shift of emissions to other countries, each member state is given the right to neutralize permits that have been set free through its additional unilateral abatement efforts. Only industries whose international competitiveness is endangered by carbon leakage will furthermore get permits free of charge (and may also get electricity at a lower price). For other industries which do not satisfy the criteria to be included in the "carbon leakage list", the share of free allowances will be continually reduced to zero. A relatively small part of the auctioning revenue flows into an innovation fund and into a modernization fund that shall assist poor EU member states with the decarbonization of their energy supply. These countries also retain the right to give allowances free of charge to power plants.

Starting from some pilot markets in seven regions, a cap-and-trade system that will be bigger than that in the EU is currently evolving in China.

What can be learnt from the practical experiences that have been made with emission trading especially in the EU-ETS so far? An answer to this question reads as follows:

(i) A uniform and clear-cut blueprint for a cap-and-trade system does not exist. Rather, the design of an emissions trading scheme for a specific application makes it necessary to have a closer look at the particular features of the situation. The protracted development of the EU-ETS (and recent experiences in China) shows that many difficulties of implementing an emissions trading scheme cannot be anticipated exactly at the start, but instead some learning-by-doing is inevitable. This may come into conflict with the need to have a quick solution for an urgent environmental problem.

(ii) The process of institutional learning will be especially difficult when various jurisdictions with different interests have to be brought together. This clearly is the case in the EU where the member states' dependence on fossil fuels (as well as their social and economic level and their environmental preferences) vary greatly. Such disparities might also jeopardize stability of an emissions trading system and thus make expectations of the participating agents less certain. This may be a danger for the future of the EU-ETS.

(iii) Regularly an emissions trading system will have to be equipped with provisions that address two problems: on the one hand, the volatility of permit prices, which has been rather high in existing cap-and-trade schemes, and on the other hand, the danger of carbon leakage effects. Coping with these problems, however, makes the shaping of emissions trading systems more complex and may reduce some of their advantages, which are vastly due to the simplicity of this market-based instrument. Moreover, some adaptation of the cap-and-trade mechanism will be needed when the involved jurisdictions still pursue complementary environmental policies by their own and concerns for distributional equity between regions have to be taken into account. These complicating factors can also be observed in the EU-ETS where specific provisions and privileges have been introduced in Phase 3 and will also be included in Phase 4 to address these problems.

(iv) Allocating permits through a grandfathering procedure has been very helpful to get acceptance of emission trading by the affected firms at the initial phase of the program. Yet, giving allowances free of charge causes serious fairness problems since polluters may be rewarded for their high emissions in the past. Therefore, political pressure will arise to replace grandfathering by the auctioning of permits, which is also desirable from the perspective of the double dividend argument. The history of the EU-ETS shows that such a transition of the allocation procedure is possible even in a political system in which the influence of industrial lobbies is strong.

(v) Trading permits means that differences in marginal abatement costs are exploited to reduce aggregate abatement costs. Yet, more basically, emission trading allows for more flexibility in choosing abatement measures since only total emissions of a polluter matter. This flexibility option which automatically is provided by a cap-and-trade system can lead to lower abatement costs even if trading activities do not occur very frequently (see Hanemann, 2010). The possibility for banking allowances represents a special inter-temporal flexibility option, which in an assessment of various actual emissions trading systems has been considered to be an important efficiency-enhancing ingredient of cap-and-trade schemes (Schmalensee & Stavins, 2017). So it is a clear advantage that banking has become part of the EU-ETS.

(vi) Overall abatement cost-effectiveness would require all emissions of the same pollutant to have an identical price, which means that a cap-and-trade system should cover all relevant sectors. In the EU-ETS this is not the case, but the scheme only covers 45% of emissions (ICAP, 2017: 28). Transportation, housing, and agriculture are not included in the emissions trading system but are addressed by other measures, which to a significant part are not coordinated between the EU states. Realistically, it cannot be expected that this will change in the foreseeable future and that the EU-ETS will be extended to these sources of emissions. As a consequence, the marginal abatement costs in the EU diverge vastly between sectors so that total greenhouse gas abatement costs are not minimized. That the EU-ETS is only patchy is often attributed to the high transaction costs that would be caused if millions of car drivers and house owners had to purchase emission allowances. This problem, however, could be easily avoided if an upstream system instead of a downstream system for permit allocation were established in which the suppliers of fossil fuels (i.e. importers of oil and natural gas) are obliged to hold permits. The incompleteness of the EU-ETS can rather be traced back to distributional reasons, i.e. the wish to avoid an overburdening of low-income households. Moreover, it would not be possible without major difficulties and distributional repercussions to integrate emission trading with the complex and diverse systems of fuel taxation that already exist in all EU countries and might be extended and complemented by CO_2 taxes that already exist in some EU states as Sweden and France.

Conclusion

You have learnt in this section that:

- Trading of emission permits can be used as an alternative price-based instrument of environmental policy through which the price of emissions is formed endogenously.
- Emission allowances can be allocated to the polluters through an auctioning process or free of charge.
- Ideally the allocation procedure has no effect on abatement activities and aggregate welfare, but this neutrality no longer holds under real-world conditions.
- Although the falling permit price caused by the transition of some polluters to a more cost-effective abatement technology makes the technology shift less attractive for the other polluters in the price-standard scenario, an optimal solution nevertheless may result.
- The functioning of cap-and-trade schemes will be impaired by factors like monopoly power and limited size of permit markets.
- In reality the development of emissions trading systems requires a time-consuming learning process in which the cap-and-trade mechanism is adjusted to the specific conditions of each case.

4.7 The Limits of Emission Pricing: An Integrated View of Instrument Choice in Environmental Policy

The pricing of emissions, either through taxes or through emission trading, has many advantages and should therefore be a cornerstone of any economically sensible environmental policy. In this sense the famous German climate economist Ottmar Edenhofer forcefully states that "what penicillin is for medicine, a CO_2 price is for climate policy". Yet, to pursue a wise environmental policy one should at the same time have in mind that emission pricing is not a magic bullet on which one can rely blindly and without further provisions (see, e.g., Goulder & Parry, 2008, or Hanemann, 2010, for a comprehensive discussion of instrument choice in environmental policy). Some of the problems that are associated with price-based instruments of environmental policy have already been considered before. We will now discuss some additional challenges which are associated with price-based approaches and which may justify the use of complementary measures.

(i) Behavioral economics as a by now flourishing branch of economics has confirmed through many experiments and field studies that people often do not behave as rationally as required for an ideal working of an emission pricing scheme (see Croson & Treich, 2014, for a survey on behavioral approaches in environmental economics). They will rather hold on to familiar behavior and neglect alternatives that could help them reduce their abatement costs. This inherent behavioral inertia can partly be explained by a *status quo bias* (see, e.g., Kahneman, Knetsch, & Thaler, 1991). Polluters even might not be interested in gathering and capable of processing the information that is needed for understanding the effects of the price signal and the available options for reacting to them. Moreover, individual time preference rates of many people are very high so that they underestimate the long-term benefits of abatement measures. The obstacles that impede the steering function of the price-based instruments are particularly important if these instruments are newly introduced and thus unfamiliar to polluters. A learning phase in which also specialized consulting firms establish will be required, which however may be too long if an environmental problem necessitates immediate action. CAC instruments in contrast have the advantage of setting direct and easily understandable signals that incentivize clear reactions without demanding too many onerous search activities from the polluters. To put it differently, flexibility of adaptation that is granted by the price-based instruments has a price that in some situations may seem too high. This explains why fuel standards and emission limits for motor vehicles as well as thermal protection regulations are a central part of environmental policy in many countries. The danger is that these complementary instruments can be used by politicians for purely symbolic actions to demonstrate the voters that environmental problems are taken serious.

(ii) Many environmental problems, as paradigmatically the climate change problem, require a far-reaching change of technology, which entails high costs and high risks and may take a long time. Governments may be able to cope better with these challenges than private firms, which have to be profitable already in the short-term and which face the danger of bankruptcy. Moreover, the development of new technologies partly has the features of a public good since the use of newly created knowledge is basically possible without incurring additional costs. Due to its positive external effects, the incentives for doing research and development will be too low for private firms especially if the protection of their intellectual property that is provided by patent laws is deficient. Moreover, the government may have an interest that a new "green" technology is applied rapidly by many polluters if the solution of an environmental problem seems to be urgent. For these reasons, there is a broad consensus that subsidization of research and development in the field of environmentally friendly technologies is advisable. Yet, even after an innovation has been made, the new technology will only become competitive if its application is widely spread so that its costs are falling both due to learning-by-doing effects and economies of scale in producing the innovative systems. In order to push them into the market and to provide for them a level playing field with conventional technologies, subsidies for the application of green technologies (and not only for research and development) may be appropriate, too. Subsidization also helps to overcome the liquidity constraints that often hinder the implementation of new technological devices. This explains why the generation of renewable energy from sources like wind, solar, and biomass is subsidized in many countries. In this context, however, a big and empirically quite relevant danger is that due to successful lobbying by the beneficiaries, the subsidies remain at a too high level for a too long period of time.

(iii) If dealing with an environmental problem must be based on a fundamental change of technology, there is a need of behavioral adjustments not only by the polluters that have to pay the price for emissions. Consequently, the actions of a large number of indirectly affected agents have to be coordinated. Coping with this challenge probably demands too much from an emission pricing scheme when it serves as the sole instrument of environmental policy. Rather, emission pricing has to be accompanied by a cleverly designed "green industrial policy" (see Rodrik, 2014) through which communication of all relevant agents is improved and their expectations are aligned and stabilized. Coordination activities by the government are inescapable anyway if abatement measures must entail massive adaptations of infrastructure as required by the transition to a decarbonized energy system. The danger here is that this coordination task overstrains the government and that something like *regulatory capture* occurs, i.e. that in the coordination process firms become able to use their better information to their own advantage.

(iv) To trigger the needed abatement measures, emission prices must be sufficiently high. Then, however, the distributional effects of emission pricing might be so enormous and appear to be so unfair that massive political resistance will arise. Clearly, some compensation for the actual and potential losers can be financed from the revenue of the emission pricing scheme, which reduces the regressive distributional effect of carbon pricing. In a recent study (see Fremstad & Paul, 2017), it has been shown that while taxing CO_2 at a tax rate of about \$50 would make 75% of the bottom half of Americans worse off; this ratio could be reduced to 11% if the tax revenue were rebated in a lump-sum way. Yet, the exact design of such redistributive measures usually is a demanding task for governments especially as the exact distributional effects of emission pricing are unknown beforehand. Hence, a genuine conflict between effectiveness and political acceptability of emission pricing mechanisms exists, i.e. if the emission price were high enough to induce the desired abatement measures, it would not find acceptance in the political process. In this respect, CAC instruments may have an advantage, as their effects are more transparent and less uncertain. Yet, there are some promising approaches how to defuse this conflict (e.g. Klenert et al., 2018) and to make price-based instruments work.

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International Environmental Problems

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5

Objectives of This Chapter

In this chapter, students should learn:

- Which types of international environmental problems exist
- How in case of reciprocal environmental spillovers the basic strategic interactions between countries can be described in a simple binary game model in which different game types may occur
- How the type of the game is transformed if some parameters, e.g., the abatement costs, are changing or fairness motivations become relevant for the countries
- How threat strategies can stabilize cooperation in repeated games
- Which factors are favorable for making international cooperation on environmental problems successful

5.1 Introduction

In the last decades, international environmental problems where transboundary externalities occur, i.e. emissions in one country affect environmental quality in other countries, became increasingly important and got much attention, both in environmental economics and in practical environmental policy. The theoretical analysis of these problems requires a shift of perspective as compared to the approaches of the previous sections of this book. Since there is no supranational institution that has the authority to implement environmental regulations in autonomous states, the Pigouvian approach, for which governmental ability to exert power and to set binding rules is essential, cannot be directly applied to international environmental problems. At least as a benchmark, one rather has to consider the outcomes that result when the autonomous states choose their environmental policies independently. In this context, we observe strategic behavior of governments, which leads us into the spheres of non-cooperative game theory.

Different types of international environmental problems have to be distinguished. On the one hand, there are unilateral environmental spillovers between countries (as the pollution of the river Rhine caused by French potash mines, which affected the downstream countries Germany and The Netherlands and lasted for decades (see, e.g., Dieperink, 2011), or Chinese sulfur dioxide emissions causing acid rain in Japan (see, e.g., Nagase & Silva, 2007)) where the polluters and the impaired victims are situated in different jurisdictions. In this case, the laissez-faire allocation as described in our treatment of Coasean bargaining can be regarded as the outcome of independent behavior, which is not efficient and triggers negotiations as we have shown in ► Chap. 2. With side-payments, the victim can "bribe" the polluter to reduce his emissions and thus to internalize the externality. This leads to a Pareto improvement relative to the non-cooperative outcome but raises questions of distributional equity – which are pertinent for almost all international environmental problems. On the other hand, there are multilateral environmental spillovers where pollution in one country does not only harm people in other countries but also its own citizens so that, to a higher or lower degree, polluters and victims become identical. By now, the most important example for this type of international environmental problem is climate change, which mainly originates from burning fossil fuels, and which is expected to have dramatic consequences for humankind as a whole (see, e.g., Nordhaus, 2013, or Stern, 2015, for extensive treatments of the climate change problem from the economist's viewpoint). Other examples of reciprocal externalities that have been treated extensively in the literature are acidifications of the environment by sulfur dioxide emissions from coal-fired power plants and ozone layer depletion caused by ozone-depleting substances like halocarbon refrigerants and chlorofluorocarbons. In the extreme, reciprocal externalities turn into a global public bad, which means that all countries are affected by emissions to the same extent irrespective of the location of their source. Conversely, the reduction of total emissions has the character of an international or even global public good whose benefits accrue to all countries – and no one can be excluded from enjoying these benefits.

Control Question

How can international environmental problems be classified? Give some examples for each class of environmental problems.

The structure of this chapter will be as follows:

First of all, in ► Sect. 5.2, we present a model of a one-shot game with a reciprocal externality, in which – for the sake of simplification – there are only two countries and actions are taken simultaneously. In this model we describe how, dependent on the relation between the benefits of improved environmental quality and country-specific abatement costs, different game types (e.g., the prisoners' dilemma or the chicken game) may arise when the countries act non-cooperatively. In ► Sect. 5.3, we then examine how an originally given one-shot game may be transformed into another one, e.g., by changes of the countries' abatement costs (e.g., induced by environmentally friendly technological progress) or extensions of preferences (e.g., by adding a fairness component to the countries' material payoffs). Since the Nash equilibria of the non-cooperative game are inefficient, we particularly analyze in this context how policy measures (e.g., subsidizing ("matching")) can increase the countries' abatement activities. Yet, the outcome can also be deteriorated through game transformations, which is a possibility addressed in ► Sect. 5.3, too. In ► Sect. 5.4, we consider how cooperation between the countries leading to efficient outcomes can be brought about when the original one-shot game is repeated so that threat strategies can apply. In ► Sect. 5.5, we conclude by briefly discussing the significant factors that are crucial for a successful internalization of multilateral transboundary externalities and a more efficient provision of environmental global public goods.

5.2 A Two-Country Game with a Reciprocal Externality: Payoff Structure and Nash Equilibria

5.2.1 The Model

There are two countries 1 and 2 whose abatement activities are denoted by g_1 and g_2, respectively. To simplify the exposition we assume that marginal abatement costs c_i are constant in both countries $i = 1, 2$, so that abatement costs in country i are $c_i g_i$. Given g_1 and g_2, country i's environmental benefit (= reduced environmental damage through the abatement activities) is $B_i(\alpha_{ii}g_i + \alpha_{ji}g_j)$ ($i, j = 1, 2, i \neq j$), where B_i is a twice differentiable and monotone increasing function with $B_i(0) = 0$ for $i = 1, 2$. The parameter α_{ji} describes the intensity of the spillover effect, which an abatement activity in country j has on environmental quality in country i, while α_{ii} measures the domestic effect of country i's own abatement. Specifically, if $\alpha_{11} = \alpha_{22} = \alpha_{12} = 0$ but $\alpha_{21} > 0$, we are in the standard situation of a unilateral externality with country 2 as the polluter and country 1 as the victim. To concentrate on multilateral externalities in the following, we, however, assume that (by normalization) $\alpha_{11} = \alpha_{22} = 1$ but $\alpha_{12} > 0$ and $\alpha_{21} > 0$ so that there are reciprocal spillovers. If, in addition, $\alpha_{12} = \alpha_{21} = 1$, each country is affected by the other country's abatement effort in the same way as by its own so that emission reduction becomes a pure public good. This general setting includes the case where B_i is linear so that, for some constant $b_i > 0$, we have $B_i(\alpha_{ii}g_i + \alpha_{ji}g_j) = b_i(\alpha_{ii}g_i + \alpha_{ji}g_j)$.

5

In this setting, each country shall have a binary choice over its abatement activities (see, e.g., Finus, 2001; Arce & Sandler, 2005; DeCanio & Fremstad, 2013; and Peinhardt & Sandler, 2015, for applications of binary games to international environmental problems). This means that each country either chooses $g_i = 0$ (non-abatement = non-cooperation) or $g_i = \bar{g}_i = 1 > 0$ (abatement = cooperation). To characterize the corresponding one-shot game and to determine the resulting Nash equilibria, we must consider for each country the ordering of its four possible payoff levels, respectively, which depend on country i's own action (which appears as the first argument of u_i) and the action of the other country j (which appears as the second argument of u_i).

(i) $u_i(0,0) = B_i(0) - c_i \cdot 0 = 0$, i.e. neither country i nor country j abates.
(ii) $u_i(1,1) = B_i(1 + \alpha_{ji}) - c_i$, i.e. both countries abate.
(iii) $u_i(1,0) = B_i(1) - c_i$, i.e. country i abates but country j does not, so that in a certain sense, country i is exploited by country j.
(iv) $u_i(0,1) = B_i(\alpha_{ji})$, i.e. country j abates but country i does not, so that country i is a free rider on the abatement activities of country j.

5.2.2 Payoff Rankings and Country Types

Clearly, $u_i(0,1) > u_i(0,0)$ and $u_i(1,1) > u_i(1,0)$, which simply means that each country wants the other country to abate. If $\alpha_{ji} \geq 1$ (or α_{ji} only is a little smaller than 1) so that the spillover effect from country j to country i is relatively strong, which is assumed for the time being, we have $u_i(0,1) > u_i(1,0)$, too. We additionally assume that $u_i(1,1) > u_i(0,0)$ since otherwise common action with country j would not be in the interest of country i so that, at least from the perspective of this country, no cooperation problem would exist. Combining all these pairwise rankings, four orderings of country i's payoffs are still possible, which are characteristic for the most important types of binary cooperation games between two agents.

(i) $u_i(1,0) < u_i(0,0) < u_i(1,1) < u_i(0,1)$
(ii) $u_i(0,0) < u_i(1,0) < u_i(1,1) < u_i(0,1)$
(iii) $u_i(1,0) < u_i(0,0) < u_i(0,1) < u_i(1,1)$
(iv) $u_i(0,0) < u_i(1,0) < u_i(0,1) < u_i(1,1)$

When the spillover effect is weak, i.e. α_{ji} is small, but all other assumptions hold, another ordering occurs:

(v) $u_i(0,0) < u_i(0,1) < u_i(1,0) < u_i(1,1)$

Given these payoff orderings, it is straightforward to determine country *i*'s optimal reactions to the action of country *j*:

- In Case (i), the best country *i* can do is to refrain from abatement activities irrespective of whether country *j* does abate or not. Thus, no abatement is the dominant strategy of country *i* as in the famous prisoners' dilemma (PD). Therefore, we will say that country *i* is a PD-type in this case.
- In Case (ii), country *i* will abate if country *j* does not abate while it will not abate if country *j* does. Since such asymmetric reactions are the characteristic feature of the chicken game, we say that country *i* is a CH-type in this case.
- In Case (iii), country *i* will abate if and only if country *j* does, so that country *i*'s payoff ranking corresponds to that of the assurance game and country *i* is called to be an AS-type in this case.
- In Case (iv), country *i* will always abate irrespective of whether country *j* abates or not. Hence, as in the harmony game, it is thus a dominant strategy for country *i* to abate and country *i* is labeled an HA-type in this case. The same outcome results when the payoff ranking is given by Case (v).

5.2.3 Explanation of Country Types

Before we explore which Nash equilibria emerge when identical or different types of countries meet and enter the non-cooperative Nash game, we discuss how the different country-specific payoff structures are implied by properties of the environmental benefit function B_i, the levels of the cost parameter c_i and the spillover parameter α_{ji}.

- A PD-type results if $B(1) < c_i$ and $B_i(1 + \alpha_{ji}) - B(\alpha_{ji}) < c_i$, which clearly is more likely if the abatement costs c_i are high. If the benefit function is concave or linear, the first condition $B_i(1) < c_i$ implies the second one $B_i(1 + \alpha_{ji}) - B(\alpha_{ji}) < c_i$ for all levels of the spillover parameter α_{ji}. This means that if it does not pay for country *i* to abate alone, it also does not pay for it to abate when country *j* has made an abatement effort.
- A CH-type results if $B(1) > c_i$ and $B_i(1 + \alpha_{ji}) - B(\alpha_{ji}) < c_i$, i.e. when the unilateral abatement effort of country *i* induces a high environmental benefit for this country, e.g., it helps to avoid an environmental catastrophe. Adding a further abatement effort to that of country *j* does not provide enough benefits for country *i* to cover its abatement cost, which is more likely if the benefit function B_i is concave and strongly curved and the spillover parameter α_{ji} is large. Among all cost parameters that satisfy $B_i(1) > c_i$, a sufficiently high c_i is required to satisfy the second condition for a CH-type.
- An AS-type results if $B(1) < c_i$ and $B_i(1 + \alpha_{ji}) - B(\alpha_{ji}) > c_i$. In this case, a unilateral abatement effort by country *i* is not worthwhile for this country because it is not

sufficient to avoid an environmental catastrophe. For achieving this, common action by both countries is needed, and country i is voluntarily ready to contribute its part if the other country j does. To fulfill the two conditions for an AS-type, the benefit function cannot be concave or linear and the spillover parameter must be large enough. For a sufficiently convex benefit function, an AS-type always results.

- An HA-type occurs if $B(1) > c_i$ and $B_i(1 + \alpha_{ji}) - B(\alpha_{ji}) > c_i$, which clearly is more likely if the abatement costs c_i are low. If the benefit function is convex or linear, the first condition $B_i(1) > c_i$ implies the second one $B_i(1 + \alpha_{ji}) - B(\alpha_{ji}) > c_i$ for all levels of the spillover parameter α_{ji}. This means that if it pays for country i to make the first abatement effort, it also pays for it to make the second one when country j has already made its contribution.

5

Control Question

Which types of countries can be distinguished in our game-theoretic framework? Give an analytical and verbal explanation of the factors on which the occurrence of each type depends.

5.2.4 Nash Equilibria

In order to determine Nash equilibria, we plot the payoffs of the two countries into a normal form representation and mark the optimal reactions by an asterisk. In Fig. 5.1, this is done for the case where both countries have the same environmental benefit function B, the same abatement costs c and $\alpha_{11} = \alpha_{22} = \alpha_{12} = \alpha_{21} = 1$ holds. If both countries are of the PD-type, the optimal reactions are marked in red, and in green when both countries are of the CH-type. A Nash equilibrium is found where both red and

Fig. 5.1 The symmetric PD-game and CH-game in their normal form

country i / country j	not abate	abate
not abate	0^{*} 0^{*}	$B(1) - c^{*}$ $B(1)^{**}$
abate	$B(1)^{**}$ $B(1) - c^{*}$	$B(2) - c$ $B(2) - c$

both green stars meet respectively, i.e. where the optimal reactions of both countries are consistent. Then no country has an incentive to change its action, and thus each country is satisfied with the outcome.

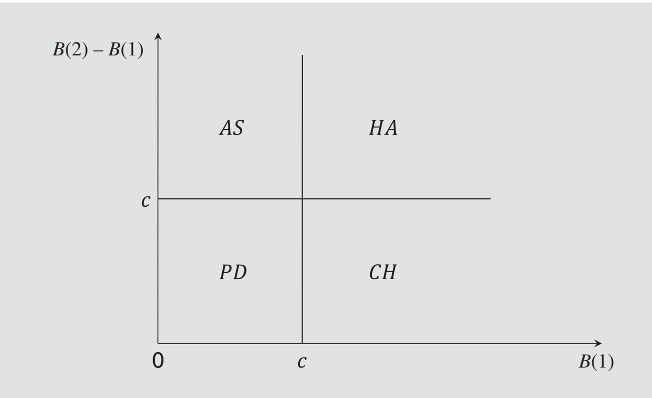
For these two most prominent games, we thus get the following results on Nash equilibria: If both countries are of the PD-type, there is a unique Nash equilibrium in the dominant strategy "not abate." If both countries, however, are of CH-type, there are two asymmetric Nash equilibria in which just one country chooses "abate" but the other one does not.

It is a straightforward exercise to include the other two game types in Fig. 5.1. Then it is easily seen that if both countries are of the AS-type, there are two Nash equilibria as in the CH-game. However, in contrast to the CH-game, the two Nash equilibria are symmetric, i.e. both countries either do abate or do not. In situations with multiple equilibria, a problem of equilibrium selection occurs, which we will discuss separately later on. If, finally, both countries are of the HA-type, there is – as in the PD-game – a unique Nash equilibrium in dominant strategies. In the HA-game, the dominant strategy for both countries is to abate.

Again, for the special case in which both countries have the same environmental benefit function B and the same abatement cost c, Fig. 5.2 visualizes how, for a given c, the emergence of the four game types depends on $B(2) - B(1)$ and $B(1)$, i.e. on the changes of environmental benefits that a country gets through its own abatement efforts when the other country does abate or not.

Things get a little more complicated when countries of different types are paired, which is rarely treated in the literature although it is empirically important. Consider, e.g., the case when a PD-country i meets some country j, which is of a CH-type or an HA-type. Then, there is a unique Nash equilibrium in which the PD-country clearly does not abate while the other country does. Such a situation may occur when country j is large so that its abatement, e.g., of greenhouse gas (GHG) emissions has a strong impact on the global climate, while the PD-country i is small with only insignificant influence on the earth's temperature. This provides some simple version of the famous thesis that in public good provision the large contributors are "exploited" by the small ones. If a PD-country i, however, meets an AS-country j, non-abatement on both sides results in the unique Nash equilibrium. Such a situation may emerge when country i

Fig. 5.2 The ranges of the four game types



still is smaller than country *j* but country *j* is not large enough to prevent an environmental catastrophe alone.

Three further combinations of different country types remain to be explored, two of which are very easy to handle: If country *i* is of the CH-type and country *j* is of the HA-type, there is a unique Nash equilibrium where country *j* abates and country *i* does not. Country *j* has so much interest in the improvement of environmental quality that it, literally speaking, spares country *i* from pursuing own abatement activities. If instead country *i* is of the AS-type and country *j* is of the HA-type, both countries are taking abatement measures in the unique Nash equilibrium.

The most demanding situation occurs when country *i* is of the CH-type and country *j* is of the AS-type. Then, surprisingly, no Nash equilibrium exists since country *i* always wants to do the opposite as country *j*, while country *j* always wants to do the same as country *i*. To get such a situation, the environmental benefit functions of both countries must be completely different, i.e. B_i must be strongly concave while B_j must be convex. In the context of climate change, such a constellation might be present when a catastrophic outcome for country *i* (in the North with a low basic temperature) could already be avoided if the rise of global temperature was limited to, say, 5 °C which country *i* is able to bring about by unilateral abatement activities. Some further limitation of global temperature is not of great interest for country *i* because it can cope with some moderate rise of temperature in a relatively cheap way by means of adaptation measures (in other words, adaptation efforts as a means for additional damage prevention are less costly than the abatement of GHGs). Country *j* (in the South with a high basic temperature) would also be able to restrict global temperature to this 5 °C limit through own abatement measures which would save abatement costs for country *i*. Taken together, this implies that country *i* will only engage in abatement activities when country *j* does not, which makes country *i* of the CH-type.

Country *j*, however, would not benefit too much from a limitation of the temperature rise to 5 °C, since compliance with this relatively high threshold might still render country *j* uninhabitable and thus puts its sheer existence in jeopardy. Hence, country *j* is not willing to take unilateral action to reduce its GHG emissions to attain the 5 °C-target. Country *j* only is on the safe side if the rise of temperature is kept below 2 °C, which it cannot achieve alone but only with common action in both countries. Thus, taking own abatement measures is profitable for country *j* if and only if country *i* also reduces emissions, which implies that country *j* is of the AS-type.

Control Question

Use the normal-form representation to determine the Nash equilibria of the binary game when the countries' types are different.

5.3 Transformations of the One-Shot Game

5.3.1 Changes of Abatement Costs

In our analysis of game transformations (see Pittel & Rübbelke, 2012, on game transformations in the context of international environmental problems), we first of all examine how the payoff ordering of a country *i*, which originally is a PD-type, changes when its abatement costs fall from the initial level c to a lower level $\tilde{c} < c$. Furthermore, we assume that $B_i(2) - B_i(1) < B_i(1)$, which follows if the environmental benefit function B_i is concave. If then $B_i(2) - B_i(1) < \tilde{c} < B_i(1)$, country i becomes a CH-type while for still smaller levels of $\tilde{c}$ for which $\tilde{c} < B_i(2) - B_i(1) < B_i(1)$ holds it becomes an HA-type. If, however, we have $B_i(2) - B_i(1) > B_i(1)$, which follows if the environmental benefit function is convex, country i turns into an AS-type as long as the decrease of abatement cost is not too strong, i.e. if $B_i(2) - B_i(1) > \tilde{c} > B_i(1)$, and again into an HA-type if $\tilde{c} < B_i(1) < B_i(2) - B_i(1)$. In an analogous way, we realize that a decrease of its abatement cost may transform both a CH-type and an AS-type into an HA-type. If instead country i is initially an HA-type, no change will result.

If the abatement costs of the other country also fall by a sufficiently large amount, it immediately follows from these considerations that an HA-game is obtained in whose Nash equilibrium both countries choose to abate. A Pareto improvement then results in a twofold way: On the one hand, simply by making abatement cheaper, and on the other by creating incentives for the countries to take abatement measures at all in the non-cooperative game.

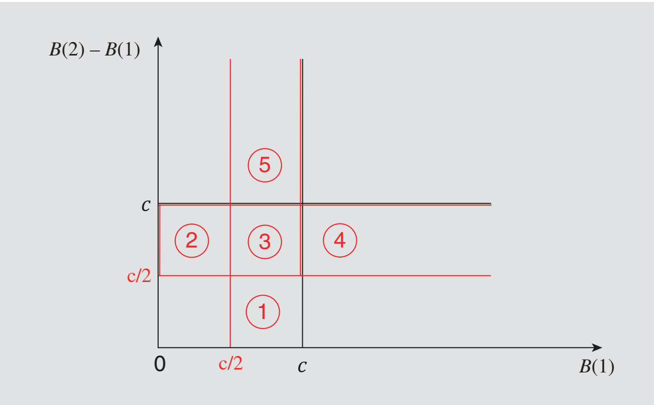
For the case of two completely identical countries (with the same benefit function B and initially the same abatement costs c), the transformation of the game type through abatement cost reductions is visualized in ◘ Fig. 5.3, where it is specifically assumed that the abatement costs of both countries fall from c to $\tilde{c} = \frac{c}{2}$.

The various game transformations that result from this common reduction of abatement costs are represented by different regions of payoff combinations in ◘ Fig. 5.3 (see Buchholz, Peters, & Ufert, 2014): In regions 1, 2, and 3, there is a transformation of the initial PD-game, i.e. in region 1 into a CH-game, in region 2 into an AS-game, and in region 3 into an HA-game. In region 4 CH-games and in region 5 AS-games are transformed into an HA-game. In total, the range of PD-games is shrinking, and the range of HA-games is enlarged.

Control Question

Show in an appropriate figure that is analogous to ◘ Fig. 5.3 which game transformation occurs if abatement costs of both countries are reduced to $\frac{c}{2}$.

◘ **Fig. 5.3** Game transformation through abatement cost reduction



In the situation as described by ◘ Fig. 5.3 all possible game transformations reduce neither aggregate abatement nor any country's payoff so that it unequivocally can be deemed beneficial. Yet, if there is some asymmetry between the two countries, it becomes possible that the emergence of an improved abatement technology with lower abatement costs may not improve but worsen the outcome, i.e. it may lead to lower levels of environmental quality and aggregate welfare. This is shown by a simple example in which both countries have an identical environmental benefit function B and initially also identical abatement costs c (and all other assumptions are the same as before), but country 2 generates a higher level of public good contribution $g_2 > 1 = g_1$ than country 1 when it spends c on abatement activities.

5

Example

Scenario 1: We have $c = 9$ for the common abatement cost parameter and now, unlike in the other parts of this section, $g_2 = 2$. For the relevant levels of environmental benefits, we assume $B(0) = 0$, $B(1) = 6$, $B(2) = 11$, and $B(3) = B(1 + 2) = 14$, which is compatible with a concave environmental benefit function B. As $B(2) - c = 11 - 9 = 2 > 0$, country 2 will choose to abate when country 1 does not abate, while $B(3) - B(2) = 3 < 9 = c$ implies that country 1 reacts by not abating when country 2 abates. Hence, the combination of non-abatement by country 1 and abatement by country 2 is a Nash equilibrium of this game. It is the unique one since $B(1) - c = 6 - 9 = -3 < 0$ entails that non-abatement is the dominant strategy of country 1. In this Nash equilibrium, total abatement efforts amount to $g_2 = 2$ and aggregate welfare is $2B(2) - c = 22 - 9 = 13$. This would also be the welfare-maximizing outcome in this scenario.
Scenario 2: We assume that abatement costs of country 1 decrease to $\tilde{c} = 0$, i.e. abatement becomes costless for this country, while nothing else changes. As now $B(1) - \tilde{c} = 6 - 0 = 6 > 0$, country 1 will abate when country 2 refrains from abatement. But if country 1 abates, country 2 will lose its incentive to abate since $B(3) - B(1) = 14 - 6 = 8 < 9 = c$. Therefore, a Nash equilibrium results in which, in contrast to Scenario 1, country 1 abates but country 2 does not. Aggregate abatement efforts now are only $g_1 = 1$ and thus smaller than in Scenario 1. Aggregate welfare in Scenario 2 is $2B(1) - \tilde{c} = 2 \cdot 6 - 0 = 12$, which is below aggregate welfare in the Nash equilibrium of Scenario 1. The Nash equilibrium again is unique since $B(3) = 14 > 11 - 0 = B(2) - \tilde{c}$ implies that abatement becomes the dominant strategy of country 1.

In this example, a *technological paradox* occurs: Even though environmental protection becomes cheaper in one country, it happens in the strategic context of the non-cooperative game that environmentally friendly "green" technological progress unexpectedly leads to lower aggregate abatement efforts and thus to lower overall environmental quality. Aggregate welfare is also reduced, which in the specific situation even means that a welfare-maximizing solution is abandoned.

Control Question

What is meant by a "technological paradox" in the context of international environmental problems? Try to give an intuitive explanation for its occurrence.

For this comparison, it has been assumed that the technological progress in country 1 occurs automatically, e.g., due to independent R&D efforts by firms that lie outside the control of the government in country 1. Yet if, alternatively, it is assumed that the government in country 1 is able to choose the level of its abatement costs, then the outcome changes considerably: Because $B(2) = 11 > 6 - 0 = B(1) - \tilde{c}$ country 1 – anticipating the Nash equilibrium in Scenario 2 – would prefer to stay with the old abatement technology where it has positive abatement costs. Then country 1 can achieve to remain a free rider on the abatement efforts of country 2 and avoids to end up in the disadvantageous position as the sole contributor to the public good being exploited by country 2. In the non-cooperative framework considered here, this blocking of technological progress is of advantage since it prevents a decline of environmental quality and aggregate welfare.

If the countries cooperate to maximize aggregate welfare instead (by distributing the gains of cooperation so that each country becomes better off), the application of the cheaper abatement technology is clearly desirable since it allows to increase aggregate welfare from $2B(2) - c = 13$ to $2B(3) - c = 28 - 9 = 19$. The simple reason is that given $\tilde{c} = 0$ the increase of total abatement from 2 to 3 is costless. Anticipating the cooperative solution, country 1 would also have an incentive to make use of the improved technology so that everything would be in order. However, if there is some chance that negotiations fail (e.g., because the countries cannot reach an agreement on the distribution of the benefits) the willingness of country 1 to shift over to the new abatement technology may nevertheless be thwarted.

The effects of preference changes may have similar effects as changes of the abatement costs and can be examined in a completely analogous way. The resulting transformation of the game type is of particular interest when a possibility for *strategic delegation* exists.

This means that the citizens in a country can choose a representative who is put in charge of making the decisions in the abatement game according to his own preferences. By deliberately selecting an agent whose preferences are different from their true preferences, the citizens of a country may improve their welfare in the strategic context of the non-cooperative game.

? Control Question

Provide – in the framework of our elementary game-theoretic model – an example in which a country applies strategic delegation to improve its utility in the Nash equilibrium.

A special case for a country-specific reduction of abatement costs results when each country subsidizes ("matches") the abatement costs of the other country. Matching (also see Guttman, 1978, Danziger & Schnytzer, 1991, and Althammer & Buchholz, 1993) means that if country i takes an abatement activity whose cost is c the other country j commits itself to bearing the s-th share of country i's abatement costs. Hence, the net abatement costs of country i, which determine its decisions, are reduced to $(1 - s)c$. The decisions of the actively matching country j, however, are not affected by its matching activities, but it pays for abatement in a twofold way: Directly like in the scenario without matching, and indirectly through the subsidization of the abatement measure by country i.

If only one country matches the abatement efforts of the other one, the matching mechanism is called unilateral. But it is possible that both countries match each other's abatement so that matching becomes reciprocal, which leads to a reduction of direct abatement costs in both countries and thus to cost-sharing. How the different game types are transformed through such mutual matching with a common matching rate $s = \frac{1}{2}$ can be seen from ◘ Fig. 5.3, where matching reduces the direct abatement costs of each country from c to $\tilde{c} = \left(1 - \frac{1}{2}\right)c = \frac{1}{2}c$.

5

5.3.2 Extended Preferences

People do not only care about their material well-being but also appreciate distributional equity of outcomes and fair behavior by others (see Posner & Weisbach, 2010, for a general discussion on this in the context of climate change). As has been shown also by many studies in experimental economics (see, e.g., Dannenberg, Sturm, & Vogt, 2010) such psychological preference components have much impact on the willingness to cooperate and on the stability of cooperative solutions in climate policy (see, e.g., Lange & Vogt, 2003; Lange, 2006).

An important example especially for the relevance of fairness attitudes in international climate policy is given by the Byrd-Hagel resolution, which the US Senate passed by a unanimous vote in 1997 and which in the end prevented the American participation in the Kyoto Protocol. In this resolution, it was stated that "the United States should not be a signatory to any protocol … which would … mandate new commitments to limit or reduce greenhouse gas emissions for the Annex I Parties [i.e. *the industrialized countries, W.B. and D.R.*], unless the protocol … also mandates new specific scheduled commitments to limit or reduce greenhouse gas emissions for Developing Country Parties … ." This declaration is a reflection of a demand for fairness in a cooperative arrangement and of intrinsic reciprocity as an additional element of preferences (see Buchholz & Peters, 2005; Cramton, MacKay, Ockenfels, & Stoft, 2017; and Nyborg, 2018, for the importance of reciprocity in climate change policy). We now infer how the presence of such an additional psychological preference component can transform the original game in which material payoffs are given in the same way as before. (In a general setting Buchholz, Peters, & Ufert, 2018, consider game transformations caused by reciprocal preferences and their implications for stable cooperative outcomes.)

In our static model, where we again assume $g_1 = g_2 = 1$, a country i's preference for reciprocity is captured by assuming that (like in the seminal paper by Rabin, 1993) country i's utility is reduced by $r_i(1,0) > 1$, if it takes abatement measures itself but the other country does not. Moreover, $r_i(0,1) \geq 0$ holds in the opposite case when only the other country j abates so that country i exhibits a preference for homogeneous behavior. These psychological valuations can be explained as follows: On the one hand, if it is only country i that abates so that it is the sole contributor to the public good, it will feel as a "sucker" being unfairly exploited by the other free-riding country j. This creates a feeling of annoyance and discomfort with country i, which reduces its overall utility. On the other hand, if country i refrains from abating and free rides on the abatement activities of country j, negative feelings of shame and guilt may be stirred for country i. It is a plausible assumption that anger about non-cooperativeness of others is weighing more than bad conscience, i.e. $r_i(1,0) > r_i(0,1)$

holds, which is also supported by experimental studies. When the behavior of both countries is homogeneous two-sided non-abatement, it may be presumed that this is perceived as emotionally neutral, i.e. $r_i(0,0) = 0$. Being happy about successful cooperation, two-sided abatement may instead cause positive emotions, i.e. $r_i(1,1) \geq 0$.

When these psychological preference elements are taken into account and $\alpha_{ji} = 1$ is assumed for the spillover parameter, the new payoffs of country i read as follows:

(i) $\tilde{u}_i(0,0) = B_i(0) = 0$, if neither country i nor country j abates.
(ii) $\tilde{u}_i(1,1) = B_i(2) - c_i + r_i(1,1)$, if both countries abate.
(iii) $\tilde{u}_i(1,0) = B_i(1) - c_i - r_i(1,0)$, if country i abates but country j does not.
(iv) $\tilde{u}_i(0,1) = B_i(1) - r_i(0,1)$, if country j abates but country i does not.

We now analyze how the type of country i may change due to fairness concerns.

- If country i originally is a PD-type, the payoff of country i as a sole contributor is further reduced through its feeling of anger so that non-abatement is again answered by non-abatement. But if the fairness valuations $r_i(0,1)$ and $r_i(1,1)$ are sufficiently strong, the ranking of country i's payoffs in the case where country j abates will be reversed, i.e. $\tilde{u}_i(1,1) > \tilde{u}_1(0,1)$ may result and – due to its desire to avoid a bad conscience and its feeling of joy about mutual cooperation – country i abates if country j does. A PD-type then is transformed into an AS-type.
- If country i originally is a CH-type, several cases have to be considered, whose occurrence also heavily depends on the values of $B_i(1)$, $B_i(2)$, and c_i.
  - $r_i(0,1)$ or $r_i(1,1)$ are large while $r_i(1,0)$ is not too large. Now country i's reaction to abatement by country j is changed to "abate" so that country i becomes an HA-type.
  - $r_i(0,1)$ or $r_i(1,1)$ and, simultaneously, $r_i(1,0)$ all are so large that they affect the decision of country i so that it changes into an AS-type.
  - $r_i(1,0)$ is large while $r_i(0,1)$ and $r_i(1,1)$ are small, i.e. country i's anger about free-riding by country j dominates its feeling of shame if it were the free rider itself and its joy about successful cooperation. In this case, fairness preferences make country i less willing to take abatement efforts.

Control Question

How may the inclusion of fairness preferences change a country's type if it originally is an AS-type or an HA-type?

The analysis of all possible cases shows that fairness preferences in general tend to reduce country i's willingness to take unilateral abatement measures while they increase its willingness to follow suit when the other country j abates.

How these changes of country-specific cooperation incentives affect the Nash equilibria is shown for the case that there are two completely identical countries with common material payoff variables $B(1)$, $B(2)$, and c and common fairness terms $r(1,0)$, $r(0,1)$, and $r(1,1)$. Then each country is willing to undertake unilateral action if $B(1) - c - r(1,0) > 0$ or, equivalently, if $B(1) > c + r(1,0)$. Country i is willing to top the abatement activities of country j if $B(2) - c + r(1,1) > B(1) - r(0,1)$ or, equivalently, if $B(2) - B(1) > c - r(0,1) - r(1,1)$. This is visualized in Fig. 5.4 where we (analogously to Figs. 5.2 and 5.3) depict the regions of the game types PD, CH, AS, and HA, when abatement costs c and now additionally the three fairness terms are given.

5

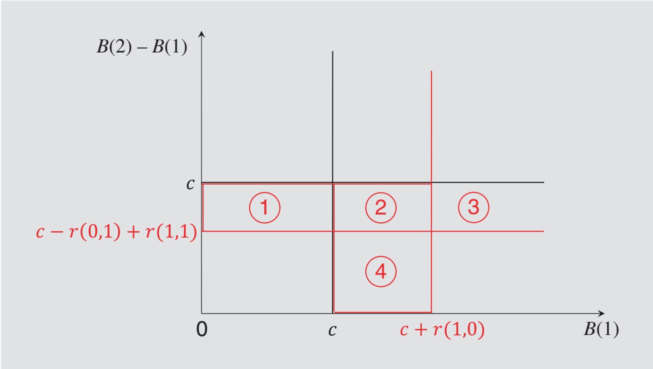


Fig. 5.4 Game transformation through fairness preferences

In region 1 of **Fig. 5.4**, the PD-game is transformed into an AS-game, while the other three regions describe transformations of an initial CH-game, i.e. in region 2 into an AS-game and in region 3 into an HA-game, but in region 4 into a PD-game where aggregate abatement is lower. The regions of AS- and HA-games are expanded as compared to the situation without fairness preferences, while the range of CH-games is shrinking, partly in favor of an enlargement of the PD-game zone. This clearly shows that fairness preferences are not necessarily beneficial for environmental quality, since abatement efforts in the Nash equilibrium of the CH-game naturally are higher than in the Nash equilibrium of the PD-game.

Control Question
Why is it possible that fairness preferences lead to a lower level of environmental quality? Give an intuitive explanation.

5.4 Repeated Games

Countries are long-standing institutions that do not only meet once but entertain continual relationships. This creates the possibility to apply threat strategies through which other countries can be incited to cooperate, i.e. to take abatement measures and thus to tackle the international environmental problem. Usually cooperation on international environmental problems is codified in formal agreements, which are the outcome of negotiations between the countries involved. Such international environmental agreements, however, are prone to opportunistic behavior by the signatories. In a static one-shot scenario countries then have the incentive to breach the contract and to choose non-abatement even if they have agreed to common abatement measures before. International environmental agreement thus are exposed to the big risk to be (internally) unstable which is a major topic in the literature on international environmental economics (e.g. Finus, 2001). Against this background, the role of threat strategies can be perceived as a tool for stabilizing international environmental agreements and thus to render international environmental policy effective in the long-run.

In our game-theoretic framework, this means that the one-shot abatement game is repeated, and each country threats to stop abatement in later rounds of the repeated game if the other country has defected and chosen non-abatement in some round. In this context, different threat strategies exist, the most prominent ones being the unforgivable *grim strategy* and the *tit-for-tat strategy*, which allows a return to cooperation after some defection has occurred (see, e.g., Dixit, Skeath, & Reiley Jr., 2015). To facilitate the exposition, we will concentrate our treatment of repeated games on the case of a PD-game between two completely identical countries, which is repeated over infinitely many periods $t = 1, 2, 3, \ldots$. Each country is, as before, characterized by its environmental benefit function B and its abatement costs c, and can choose between abatement (= 1) or nonabatement (= 0). Depending on how many countries are abating, the relevant payoffs of a country thus are $u(0,0) = 0$, $u(0,1) = B(1)$, $u(1,0) = B(1) - c$, and $u(1,1) = B(2) - c$ with the PD-ranking $u(1,0) < u(0,0) < u(1,1) < u(0,1)$. But in the repeated game scenario now an additional parameter comes into play that, as we will see, plays a crucial role for the cooperation success: The discount factor δ indicates how much a payoff in some period $t + 1$ is worth from the perspective of the previous period t. The standard assumption is that $\delta < 1$, i.e. that later payoffs count less than present ones so that countries exhibit a preference for the present. If, analogously to the interest rate, a positive time discount *rate* $r > 0$ is given, the time discount *factor* is obtained as $\delta = \dfrac{1}{1+r} < 1$.

The most simple threat strategy is the *grim strategy* by which country i threats to stop abatement forever if the other country j defects in a single period by choosing non-abatement there. We now explore under which conditions this threat is effective, i.e. motivates country j to abate. To that end, we assume that country j is thinking about whether it should defect in some period, say period T. Until period $T - 1$, both countries have successfully cooperated by making joint abatement activities, and in period T, country i is still doing so. Then country j gets the free-rider utility $u(0,1) = B(1)$ in period T but – as a rational player with perfect foresight – anticipates the punishment that is inflicted on him in all future periods by country i, which, by applying the grim strategy, will never be ready to abate again. The best that country j then can do is to also refrain from abatement in the infinitely many periods t after period T. The aggregate present payoff value that country j thus can attain through defection in period T is $B(1) + \delta \cdot 0 + \delta^2 \cdot 0 + \ldots = B(1)$. The alternative is not to defect in period T (and then neither in any subsequent period), which leads to the permanent flow of the cooperative payoff $B(2) - c$ and thus, by the standard formula for geometric series, to the aggregate present payoff value $\sum_{t=0}^{\infty} \delta^t \cdot \left(B(2) - c\right) = \dfrac{B(2)-1}{1-\delta}$. Consequently, country j does not benefit from defecting so that the grim strategy fulfils its purpose as long as $\dfrac{B(2)-1}{1-\delta} \geq B(1)$. Concerning the level of the discount factor δ, which is required in this case to prevent defection, this condition implies

$$\delta \geq \delta_G^* = 1 - \frac{B(2)-c}{B(1)} = 1 - \frac{u(1,1)}{u(0,1)}. \tag{5.1}$$

This means that a stable cooperation leading to common abatement activities is more likely if both the time discount factor δ and the cooperation utility $B(2) - c$ are large (so that the negative effects of punishment would affect country j's aggregate payoff gravely), while the defection utility is small (so that country j cannot benefit too much from defection). Since $B(1) > B(2) - c > 0$ by assumption $0 < \delta_G^* < 1$ is ensured, i.e. for our PD-game there always exist discount factors δ smaller than one, for which the grim strategy becomes effective.

Control Question

Why are a high discount factor δ, low abatement costs c, and a high cooperation utility $B(2)$ conducive for stabilizing cooperation by means of the grim strategy? Give an intuitive explanation!

The threat underlying the grim strategy "I will never abate again after one single defection", however, is so serious that it does not seem realistic, since at least in the very long run self-interest would motivate countries to try to resume cooperation and thus to abandon the grim strategy. Therefore, it is important to realize that there are also milder and thus more realistic threat strategies by which enduring cooperation can be brought about in the repeated PD-abatement game. The common element of these strategies is that they allow for a restart of cooperation after one country has defected, but now demonstrates goodwill as it is willing to resume its abatement efforts again.

The most prominent of these more forgiving threat strategies is the *tit-for-tat strategy*, which directly translates the idea of reciprocity into the repeated game framework: In period $t \geq 2$, a country chooses abatement if and only if the other country has chosen abatement in the previous period, i.e. cooperation is responded by cooperation in the subsequent period and non-cooperation by non-cooperation. We now examine what choices country j will make knowing that the other country i follows the tit-for-tat strategy. Similar as before in the case of the grim strategy we assume that country j is contemplating whether it should defect in some period, say again period T. Then, we have to consider three possible decisions on defection country j can make in period T.

Case 1 Country j decides to defect not only in period T but also in all subsequent periods. As before it gets its free-rider payoff $B(1)$ in period T and, as country i – by applying the tit-for-tat strategy – will choose non-abatement in all future periods, the non-cooperation payoff 0 in periods $t = T + 1, T + 2, \ldots$. Looking again at the respective aggregate present payoff values shows that the tit-for-tat strategy keeps country j from permanent defection if $B(1) < \dfrac{B(2) - c}{1 - \delta}$, which leads to the same critical value for the discount factor δ as in (5.1), i.e.

$$\delta_{T1}^* = \delta_G^* = \frac{c - \left(B(2) - B(1)\right)}{B(1)}. \tag{5.2}$$

Case 2 Country j decides to defect in period T but then abates again in period $T + 1$ to initiate cooperation again. In period T, country j's payoff then is $B(1)$ and in period $T + 1$ it is $B(1) - c$, which gives the aggregate (discounted) payoff $B(1) + \delta(B(1) - c)$ over these two periods. To assess the profitability of this option for country j, this aggregate payoff has to be compared with the payoff that results if both countries cooperate in periods T and $T+1$, which is $(1+\delta)(B(2)-c)$. Thus, defection is not profitable for country j in this case if $(1+\delta)B(1)-\delta c<(1+\delta)(B(2)-c)$. This leads to a second critical value for the discount factor δ, which is

$$\delta_{T2}^{*}=\frac{c-\big(B(2)-B(1)\big)}{B(2)-B(1)}=\frac{c}{B(2)-B(1)}-1. \tag{5.3}$$

Case 3 Country j decides to defect in period T and also in a finite number $\tau\geq 2$ of periods after period T before it resumes cooperation and abates again in period $T+\tau+1$. We now infer under which conditions country j can improve on that by reducing the defection period to $\tau-1$ periods before returning to cooperation. Then only its payoffs in the periods $T+\tau$ and $T+\tau+1$ are affected, which change from 0 (from nonabatement by both countries in period $T+\tau$) and $B(1)-c$ (from unilateral abatement by country j in period $T+\tau+1$) to $B(1)-c$ (from unilateral abatement by country j now in the earlier period $T+\tau$) and $B(2)-c$ (from abatement by both countries in period $T+\tau+1$ that is implied by tit-for-tat behavior by country i), respectively. Shortening the defection period hence pays for country j if $B(1)-c+\delta(B(2)-c)>0+\delta(B(1)-c)$, which gives a third critical value for the time discount factor. This critical value reads:

$$\delta_{T3}^{*}=\frac{c-B(1)}{B(2)-B(1)}. \tag{5.4}$$

Yet, as abatement costs and benefits are the same in all periods of time, it is then also profitable for country j to shorten the defection period further and to start cooperation again already in period $T+\tau-1$ given that $\delta>\delta_{T3}^{*}$. If we continue this reasoning, then by backward induction, it follows that defection over more than one period does not pay for country j if $\delta>\delta_{T3}^{*}$ so that we fall back to Case 2.

Combining Case 2 and Case 3 shows that country j cannot improve its aggregate payoff by defecting in any finite number of periods if $\delta>\max\left\{\delta_{T2}^{*},\delta_{T3}^{*}\right\}$. Taking also Case 1 into account in which the option of an infinite defection has been considered, we observe that defection never pays for country j if the other country follows the tit-for-tat strategy and $\delta>\left\{\delta_{T1}^{*},\delta_{T2}^{*},\delta_{T3}^{*}\right\}$ holds for country j's discount factor. Then country i's tit-for-tat strategy is effective as it incites country j to cooperate/abate forever. Consequently, if country j also applies the tit-for-tat strategy, it is possible to implement permanent abatement by this threat strategy.

Comparing (5.2), (5.3), and (5.4) immediately shows that the condition on the countries' discount factor δ that ensures effectiveness of tit-for-tat can be greatly simplified if an additional assumption is imposed on the environmental benefit function B, i.e. if it is strictly concave so that $B(2)-B(1)<B(1)$ holds. In this case, on the one hand, the denominator of (5.3) is smaller than the denominator of (5.2), while the numerators of (5.2) and (5.3) are identical, which implies $\delta_{T2}^{*}>\delta_{T1}^{*}$. On the other hand, the numerator of (5.4) then is smaller than the numerator in (5.3), while (5.3) and (5.4) have the same denominator so that $\delta_{T2}^{*}>\delta_{T3}^{*}$. Hence, if the environmental benefit function is concave, the critical level δ_{T2}^{*} is binding, i.e. the tit-for-tat strategy is effective for all discount factors $\delta>\delta_{T2}^{*}$ being shared by both countries. According to the last term in (5.3), δ_{T2}^{*}

becomes small and thus prospects for entertaining continual cooperation are favorable when the abatement costs c are small and the additional gain $B(2) - B(1)$ in environmental benefits from mutual cooperation is large.

Since $\delta_{T1}^* = \delta_G^*$ we also have $\delta_{T2}^* > \delta_G^*$, which – loosely speaking – means that the grim strategy is more likely to be effective than tit-for-tat given a strictly concave environmental benefit function.

Control Question

Show that the critical values for the discount factor coincide, i.e. $\delta_G^* = \delta_{T1}^* = \delta_{T2}^* = \delta_{T3}^*$, if the environmental benefit function is linear so that $B(1) = b$ and $B(2) = 2b$ holds.

In the case of tit-for-tat, it may even happen that the relevant critical value δ_{T2}^* exceeds one so that the tit-for-tat strategy unfortunately cannot be effective, irrespective of the level of the discount factor δ. To get a range of discount factors $\delta < 1$ for which tit-for-tat works, we, in addition, have to assume that the increase of environmental benefits through mutual cooperation is not too small, i.e. that $B(2) - B(1) > \frac{c}{2}$. This also follows directly from (5.3).

In our presentation of the repeated game scenario, we assumed that the one-shot game is repeated infinitely often. Specifically, it must be ensured that there is no definite end of the repeated game but that, with some positive probability, always a further round might occur. This assumption is indispensable for getting effective threat strategies when agents are fully rational and perfectly far-sighted. Otherwise, a threat in the ultimate period $\bar{T}$ could not have any effect since – due to the absence of further periods – no punishment for defection in period $\bar{T}$ could be exerted. Given this failure of threat, two utility-maximizing PD-players then would choose defection in $\bar{T}$. But when the outcome of the PD-game being played in period $\bar{T}$ is determined at the outset, no possibility exists to make actions in this round dependent on what has happened in the penultimate period $\bar{T}-1$, i.e. threats of future non-cooperation also come to nothing in period $\bar{T}-1$. Iterating this argument yields that threats will also be ineffective in period $\bar{T}-2$, then in period $\bar{T}-3$, and so on until one arrives in period 1. Thus, it follows from backward induction that threats are completely worthless in this case and the outcome is non-abatement by both countries in each period of the only finitely repeated PD-game. Yet, despite the logical validity of the induction theory (backward induction) it may not in every case describe the most plausible behavior of agents (see, e.g., Selten, 1978). Therefore, there should be caution regarding the results based on backward induction.

Control Question

Explain why – from a theoretical viewpoint – infinitely many repetitions of the PD-game are required to make the grim strategy and the tit-for-tat strategy effective. How much importance do you attach to this argument in practice?

Moreover, carrying out the threat of non-cooperation against country j always is in the interest of country i, i.e. its payoff in a period where punishment is appropriate is higher if it does not abate than if it abates in this period and thus refrains from making its threat true. Hence, the threat is *credible*, which – in addition to effectiveness as demonstrated before – is also needed to enforce permanent cooperation by means of the tit-for-tat strategy. Credibility of threats is a direct consequence of the payoff ranking in the PD-game: If country j does not abate in some period t, it follows from $u(0,1) = B(1) - c < 0 = B(0) - 0 = u(0,0)$ that the best country i can do in this period is to choose non-abatement, too. This makes the threat credible in Case 1 as considered above. But if, as in Cases 2 and 3, country j resumes abatement in a period t after it did not abate in period $t-1$ it is also beneficial for country i to carry out its threat and to choose non-abatement in period t since $u(0,1) = B(1) > B(2) - c = u(1,1)$. This ensures credibility of the threat also in these two cases.

In order to keep the presentation simple, we have in our theoretical analysis only considered threats in the same field of action, i.e. in the termination of abatement efforts. In reality, however, threats may also refer to other fields of interaction between countries as collective defense, research policy, and, above all, foreign trade as, e.g., suggested by the 2018 Nobel laureate Nordhaus (2015) in his climate club approach. In this vein, trade sanctions such as import tariffs have often been proposed as instruments to prevent leakage effects and free-riding behavior by countries and thus to foster cooperation on international environmental problems, especially on climate change. As it extends the scope of sanctioning options such *issue linkage* (see, e.g., Maggi, 2016) improves the chances for successful international cooperation. Other factors that may be crucial for the success of cooperation will be briefly discussed in the next section.

Control Question

When is a threat credible? What is meant by "issue linkage"?

5.5 Success Factors of International Environmental Cooperation: Two Examples

In reality, there is quite a mixed picture concerning the effectiveness of international cooperation on environmental issues (see Barrett, 2003, and Peinhardt & Sandler, 2015, for extensive treatments on this issue). That international collective action may be more or less successful is clearly shown by two examples in which environmental protection even is a global public good whose supply affects welfare in all countries in the world. On the one hand, some international environmental agreements as the *Montreal Protocol* (of 1987 and its amendments in the subsequent years) on protecting the global ozone layer are considered to be successful, while on the other hand the *Kyoto Protocol* (of 1997) on climate change is considered to be a failure. Basic factors that may account for this difference already become clear from the preceding analysis, especially of the parameters that determine the success of cooperation in repeated games:

- The costs for the abatement of CFC gases that are responsible for the depletion of the ozone layer have been relatively low since relatively cheap substitution technologies had already been available at the time when the Montreal Protocol was concluded and no strong commercial interests were resisting CFC regulation. An effective mitigation of greenhouse gases and especially carbon dioxide, however, will require the application of new not yet fully developed technologies for energy production (e.g., renewables) and energy use (e.g., e-mobility or geothermal heating). The fundamental structural change of the economy, which characterizes the process of decarbonization, i.e. the process of replacing fossil fuels as the major source of greenhouse gases, will cause substantial costs, which cannot be reliably calculated at the moment.

- The damages from ozone shield depletion (e.g., skin cancer through increased incoming ultraviolet radiation) and thus the benefits from CFC abatement were predicted to be enormous and certain. For the USA as the leading country in CFC production, the mitigation costs even were estimated to be substantially lower than the avoided health and environmental damages. Moreover, the damages caused by the thinning of the ozone layer were expected to occur rather soon, while the lions' share of the climate change damages will lie in the more distant future and mainly affect future generations and the younger people living today. Hence, from the perspective of today, the benefits of greenhouse gas abatement may be perceived to be not so significant when the time discount rate for evaluation of benefits and costs across generations is high.

Other factors that are not directly reflected in our model make it understandable in addition why the Montreal Protocol has been a success story (see Murdoch & Sandler, 1997).

- The number of countries that produced a significant amount of CFCs was rather small, which reduced the transaction costs for international cooperation and facilitated control. The limited scope of CFC production and use moreover made it easier to implement trade restrictions on CFC products. In the case of climate change, things, however, are quite different since carbon dioxide emissions are pervasive: Energy supply in most countries is still heavily based on fossil fuels, and – related to that – the implicit carbon content of almost all manufactured goods is high. Its precise assessment moreover is virtually impossible, which thwarts the application of well-targeted restrictions on trade with carbon-intensive goods.
- As fossil fuels are – unlike CFC gases – still an essential basis of economic prosperity, poor countries are much afraid of being permanently and unfairly excluded from economic progress and welfare increases when, as a consequence of an ambitious climate policy, the use of these fuels would be strongly restricted or even prohibited. In the Kyoto Protocol, developing and emerging economies were therefore exempted from any specific obligations for greenhouse gas abatement. As described earlier, the US Senate, however, considered this as a flagrant violation of the reciprocity principle providing for the exempted countries an undue advantage in international trade. Therefore, the US Senate refused to ratify the Kyoto Protocol, which considerably reduced the Protocol's effectiveness from the start. This conflict of interest clearly highlights how different notions of fairness (right for economic development on the one hand, and reciprocity on the other) collide and thus hamper an effective global climate policy.

- The contractual framework of the Montreal Protocol was designed in a way that allowed for flexible adjustments, which helped to develop the agreement step by step by including new pollutants and attracting additional countries. Such elements have been missing in the Kyoto Protocol, which in retrospect appears as a much-too-static approach for a truly dynamic problem.

  In the *Paris Agreement*, which was concluded at the Conference of the Parties COP 21 in Paris in December 2015, this deficiency of the Kyoto Protocol has been overcome since global climate policy should now be based on a continual re-evaluation of the countries' efforts to curb greenhouse gas emissions coined as *Nationally Determined Contributions (NDCs)*. In the Paris Agreement, moreover the distinction between rich countries, which are committed to abatement measures, on the one hand, and poor countries, which are not, on the other no longer exists. Therefore, also developing countries and emerging economies with their rapidly increasing share in global greenhouse gas emissions have, at least in principle, to do their part in combating climate change, but monetary and technological transfers from the rich countries should help them to do so. There are, however, severe doubts as to whether the *pledge-and-review mechanism* as the cornerstone of the Paris Agreement will work because it mainly relies on goodwill of the countries and their fear of losing international reputation when they act completely selfishly and do not make enough efforts to reduce their greenhouse gas emissions. Hence, a naming and shaming approach is applied in the Paris Agreement, which thus is invoking psychological preference components. Yet, as we have seen above, the success of international environmental agreements is not necessarily improved through such nonmaterial preferences.

? Control Questions

What is meant by a "pledge-and-review mechanism"? Can it be expected to be effective? Try to give a judgment.

The comparison between the two examples of global environmental problems (see Sandler, 2017; Sunstein, 2007) clearly shows that factors, which we discussed in our theoretical analysis (as the avoided environmental damage, the level of abatement costs, and the discount rate), matter for the outcome of international environmental cooperation in reality. Beyond these general factors, however, it depends on the specific features of the environmental problem at stake (e.g., the number of countries involved) whether cooperation can be expected to be successful or not.

? Control Questions

Why is it difficult to achieve successful global collective action in the fight against climate change? Summarize the arguments given above and try to provide some additional ones.

Conclusion

You have learnt in this chapter that:

- There are different types of international environmental problems (unilateral vs. multilateral) and how spillover parameters account for these differences.
- Depending on the environmental benefit function and the abatement costs, countries may be of different types in a binary game: prisoner-dilemma type, chicken type, assurance type, or harmony type.
- According to the combination of country types in the binary one-shot game, different Nash equilibria will result.
- Neither a reduction of abatement costs nor an inclusion of fairness preferences always entails more abatement efforts in the non-cooperative Nash equilibrium.
- Threat strategies (the grim and the more realistic tit-for-tat strategy) can ensure permanent cooperation when a prisoners' dilemma game is repeated infinitely often.
- The success of these threat strategies crucially depends on the level of the discount factor as well as on the environmental benefit function and the abatement costs.
- International cooperation on ozone shield protection is working while global collective action on climate protection is hard to come by.

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Supplementary Information

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