Water Quality Trading Systems:

An Integrated Economic Analysis of Theoretical and Practical Approaches

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PREFACE

This dissertation has been written while I was a research and teaching assistant at the Chair of Prof. Dr. Juergen B. Donges at the University of Cologne.

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LIST OF ABBREVIATIONS

ANZECC	Australian and New Zealand Environment and Conservation Council
ARMCANZ	Agriculture and Resource Management Council of Australia and New Zealand
ATV	Abwassertechnische Vereinigung (German Association of Wastewater Technique)
AUD	Australian Dollar
BMBF	Bundesministerium für Bildung und Forschung (Federal Ministry of Education and Research, Germany)
BOD	Biochemical Oxygen Demand
CO_2	Carbon Dioxide
COD	Chemical Oxygen Demand
DAFF	Department of Agriculture, Fisheries and Forestry, Australia
DEH	Department of the Environment and Heritage, Australia
EC	Electrical Conductivity
EEA	European Environment Agency
EU	European Union
HELCOM	Helsinki Commission – Baltic Marine Environment Protection Commission
HITS	Hunter Integrated Telemetry System
ICPDR	International Commission for the Protection of the Danube River
IEPA	Illinois Environmental Protection Agency
IWQWM	Institute for Water Quality and Waste Management, Vienna, Austria
MDBC	Murray-Darling Basin Commission, Australia
NC	North Carolina, US
NC DEHNR-	North Carolina, Department of Environment, Health and Natural Resources
NC DEM	North Carolina Division of Environmental Management
NC DENR	North Carolina Department of Environment and Natural Resources
NC DWQ	North Carolina Division of Water Quality
NC EMC	North Carolina Environmental Management Commission
NPC	National People's Congress of the People's Republic of China
NSW	New South Wales, Australia
NSW DEC	New South Wales Department of Environment and Conservation
NSW DLWC	New South Wales Department of Land and Water Conservation
NSW EPA	New South Wales Environment Protection Authority
OECD	Organisation for Economic Cooperation and Development
OFWAT	Water Services Regulation Authority England and Wales
UBA	Umweltbundesamt (Federal Environment Agency, Germany)
UNECE	United Nations Economic Commission for Europe
UNEP	United Nation Environment Programme
US	United States of America
US EPA	US Environmental Protection Agency
WSDE	Washington State Department of Ecology

1 Introduction

The efficient and optimal use of environmental resources has emerged as an area of great concern during the last few decades. In this context, permit trading systems are often the centre of current interest and activity in the creation of market-based environmental instruments as it is assumed that these systems can offer significant advantages over conventional approaches to pollution control.

The idea behind the instrument of permit trading is simple: sources of pollution are only allowed to emit pollutants that are covered by permits. The total number of permits[1], as set by the regulatory authority, defines the maximum amount of pollution. After allocation, permits can be traded on the market. If a source of pollution wishes to increase its emissions, further permits must be purchased on the market. If a source is able to reduce emissions due to abatement measures, 'surplus' permits can be sold on the market. On paper a permit trading system is ecologically effective and economically efficient and is therefore often recommended for practical application.[2]

Economic investigation of permit trading systems often focuses on case studies for air pollution control. In the Unites States (US), in particular, permit trading systems have been introduced for different types of air pollutants.[3] Recently, the European Union has also started an emissions trading system in order to reduce CO_2 emissions.

Such permit trading systems also exist (in theory and practice) for water pollution control, in form of water quality trading systems.[4] Particularly in the United States, but also in Australia, different types of tradeable discharge permits have been introduced in the last thirty years. The fact that pressure is being put on the quality of water resources in many countries resulting in serious problems means that there is a need to analyse the application of this instrument to control water pollution of rivers in more detail in order to identify its potential.

Within this context, it is important to distinguish the different types of permit trading systems for the medium of water. The focus of this study lies on water quality trading systems[5] for rivers (surface water).[6] Permits are defined in terms of water quality. Sources of pollution must hold permits in order to cover their discharges.[7] These systems seek to reduce water pollution, or in other words to increase water quality. Not discussed in this study are those permit trading systems which enable the abstraction of water (water quantity trading) or permit trading systems which regulate water-borne resources, e.g. applying to fishing processes.[8]

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[1] Further synonyms of the term 'permits' can be found in the literature, such as licences, credits, tradeable permits or discharge permits.
[2] For background literature on environmental economics with external effects etc., see exemplary Perman *et al.* (2003) or Sterner (2003).
[3] For an overview, see Hansjürgens (2005).
[4] Even as early as 1968, Dales (1968) developed the general basis for permit trading systems using water as an example.
[5] Many synonyms of the term water quality trading can be found in the literature, such as effluent trading, pollutant trading, pollution-credit trading, nutrient trading, watershed-based trading.
[6] Coastal waters or other inland waters such as lakes are not covered by the scope of this study. Because they have different characteristics, an individual discussion of environmental instruments is required. Groundwater is not explicitly treated in this paper either. Individual and additional instruments need to be introduced for these types of bodies of water. For specific characteristics of groundwater, see Hecht and Werbeck (2006, pp. 83-84).
[7] This study only analyses cap-and-trade systems. For differences between cap-and-trade and baseline-and-credit trading systems, see Dewees (2001).
[8] For water (quantity) trading systems, see Bauer (2005), for example, who focuses on Chile, or Bjørnlund (2005) with a focus on Australia. For permit trading systems in the fishing sector, see Anderson (1999).

The economic analyses of permit trading systems for air pollution control are countless. Standard works discuss the main elements of permit trading systems and their application.[9] In addition, numerous research works examine the more specific aspects in great detail.[10]

In comparison, corresponding literature for water quality trading systems is rare. Few studies on water quality trading systems exist; but: they only describe the specific trading schemes without evaluating them, or they analyse very specific aspects for individual applications.[11] No basis piece of research work exists which determines – in the same way as for air pollution control – the general design requirements for the application to water.

This is the starting point for this study. It tries to fill the gap in the literature in several respects.

Firstly, this study builds a basic work on water quality trading systems. A comprehensive analysis shows the particularities of the application of permit trading systems to water, and to rivers, in particular. Ecological and economic criteria are redefined with respect to the specific characteristics of rivers. A comprehensive and standardised evaluation system is developed that allows a comparable examination of the trading systems.

Secondly, this study evaluates the existing theoretical and practical approaches of water quality trading systems with respect to these same developed criteria. For the first time, an integrated, comprehensive and comparable analysis of different trading systems applies. Few economic evaluations of the theoretical models exist. Practical approaches have rarely been analysed from an economic point of view. If at all, individual studies analyse the specific aspects of one water quality trading system or another.

Thirdly, results of the evaluations are consolidated by linking theoretical and practical approaches. Until now, these approaches have been studied side by side without being connected. The standardised analysis allowed the definition of the critical design elements of water quality trading systems. On the one hand, we will see, that the suitability of one system or another depends heavily on the conditions of the specific river to be managed. On the other hand, combinations of different approaches may be reasonable in order to fulfil specific local requirements.

Fourthly, this study develops general propositions concerning the area of implementation. Whereas the identification of the adequate design of a water quality trading scheme forms a main prerequisite for a successful implementation, it is also of great importance to identify the adequate area of implementation. This study specifies conditions that will determine whether the introduction of water quality trading systems would be reasonable in a specific country. Two regions, Eastern European Countries and China, provide examples which are examined in more detail in order to give an initial idea where the application of new water quality trading systems may be suitable.

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[9] See, for example, Tietenberg (2006, 2001, 1985) or Bader (1999); for the standard work on the integration of the transport sector, see Deuber (2002) and Cames and Deuber (2004).

[10] For a comprehensive and actual overview of the literature on specific problems in tradeable permits for air (but also for all other types), see the 'Tradable Permit Bibliography' on Tietenberg's homepage (`http://www.colby.edu/~thtieten/`, November 2006).

[11] For general description of specific water quality trading schemes, see Breetz *et al.* (2004), Environomics (1999) or NSW EPA (2003). For the analysis of specific trading schemes, see, for example, Hoag and Hughes-Popp (1997) or Kerr *et al.* (2000). At the same time, many studies analyse the specific problem of nonpoint sources and their integration into water quality trading systems (Horan and Shortle, 2001; Jarvie and Salomon, 1998; Malik *et al.*, 1993). This group of pollutants is not integrated in the analysis of this study (for specific reasons, see Section 2.2).

This study seeks to identify water-specific aspects and their impact on the design of different water quality trading approaches. The focus of the analysis is thus water-specific in every respect. More general aspects relevant to any permit trading system implemented, e.g. the general debate on the initial allocation (grandfathering versus auction), validity periods of permits or similar, have already been discussed extensively in the literature and are therefore excluded from the discussion.[12]

Furthermore, this study only discusses the instrument of a permit trading system. This work seeks to identify the cases for which the introduction of water quality trading systems is reasonable and how they should be designed in order to be effective and efficient even in the rather complex world of rivers. No comparisons with other environmental policy instruments are discussed.[13]

Structure of the Study

Chapter 2 deals with the relevant and specific characteristics of rivers. Particularities of the sources of pollution, pollutants and the river itself significantly influence the design requirements of water quality trading systems. The precise understanding of the determining factors and the relationship between these elements are crucial factors in determining an effective and efficient trading system design.

Chapter 3 develops the normative framework. Section 3.1 deals with general principles stipulated in the international Law. This chapter is further based on the key criteria of ecological effectiveness and economic efficiency. The analysis of ecological effectiveness, in particular, clarifies that the definition of the ecological objective is very complex and must be analysed in a more differentiated fashion (Section 3.2). Unlike CO_2 pollution control (air), it is not sufficient to define a maximum volume of emissions in order to avoid hot spots. Further ecological dimensions must be defined.

The analysis of economic efficiency refers to cost-effectiveness, dynamic incentives, level of transaction costs and aspects of competition, practicability and enforcement (Section 3.3). Some of these criteria need to be 'redefined' with respect to the application of the trading system to rivers. The term of transaction costs, in particular, calls for specification. Different water quality trading systems cause design-specific transaction costs which need to be identified.

Chapter 4 is the first chapter to present and evaluate permit trading approaches. While the analysis of models originally developed for air pollution control forms a crucial prerequisite (Section 4.2), the focus of this chapter lies on the analysis of the river-specific water quality trading approaches (Section 4.3). All sub-chapters have the same structure: a short description of the theoretical approach followed by an analysis with respect to the criteria developed in Chapter 3. Section 4.4 summarises the results for all theoretical approaches by already providing more general conclusions.

The structure of Chapter 5 is the same as for the previous chapter. Two practical approaches of water quality trading systems, one in the United States and one in Australia, are analysed. However, the presentation and analysis of each programme is embedded in an examination of the statutory framework in each country (Paragraph 5.2.1 and 5.3.1). In addition, these case studies are put into the perspective of other trading systems that apply in these countries (Paragraph 5.2.2 and 5.3.2). The chapter finishes with a presentation of generated results regarding design options (Section 5.4).

[12] See, for example, Tietenberg (2006, 2001, 1985) or Bader (1999).

[13] For studies on the comparison of different instruments, see Sterner (2003), Perman *et al.* (2003, Chapter 7) or Wicke (1993).

Chapter 6 links the results of the theoretical and practical approaches. Section 6.1 summarises the comparable results that can be drawn from the analysis of both, theoretical and practical approaches. Furthermore, this section formulates design-specific proposals. As well as the implementation of water quality trading approaches in the original design, the combination of design elements may be appropriate. Section 6.2 provides some proposals with respect to the country of application.

Chapter 7 summarises the main results of the study and draws the conclusion.

2 Characterisation of Rivers

2.1 Basic Facts

When discussing the implementation of a water quality trading system and its appropriate design one should be aware of the wide range of specific characteristics pertaining to an individual body of water and its pollutants. The characteristics of the medium water and its pollutants in part differ considerably from the medium air and its pollutants. Permit trading system design needs to adequately reflect these differences in order to allow for an effective and efficient solution to water pollution control. It is thus very important to carry out a detailed analysis of the specific characteristics of the body of water (river) and its pollutants.

Parallels can be drawn with permit trading systems for air pollution control; nevertheless, the medium water is affected by specific conditions. Experiences gained from observing existing permit trading systems in air pollution control cannot be transferred on a one-to-one basis to the implementation of this instrument in water pollution control.

This chapter will discuss the relevant properties of a river and its pollutants. These characteristics are also of importance when discussing and analysing theoretical and practical approaches to permit trading systems (Chapters 4 and 5). Section 4.3 will illustrate in more detail how theoretical models, originally applied to air pollution, could be adopted for water pollution control problems.

2.2 Sources of Pollution

2.2.1 Point Sources

Definition

When managing river basins, two potential sources of pollution must be taken into account: point sources and nonpoint sources.[14] Point sources are, for example, industrial facilities or sewage treatment plants.[15] The US Environmental Protection Agency (US EPA) defines that "A point source is any discernible, confined, and discrete conveyance, including but not limited to, any pipe, ditch, channel, tunnel, conduit, well, discrete fissure, container, rolling stock, concentrated animal feeding operation, landfill leachate collection system, or vessel or other floating craft from which pollutants are or maybe discharged" (US EPA, 1996a, Chapter 2, p. 2). Point sources discharge their pollutants at a fixed and well identifiable point into the river.[16] The discharges made by point sources can therefore be precisely measured: this makes it possible to assign individual accountability for the pollution. This is an important prerequisite for the introduction of a permit trading system. Permits to be traded between

[14] Already Dales (1968, p. 79) distinguished these two types of sources.
[15] For an schematised overview of point and nonpoint sources, see Figure A-1 (Appendix).
[16] We assume that relevant point sources are stationary. In the case of air pollution mobile pollution sources are also of great importance, see Commonwealth of Australia (1992, p. 26).

sources can only be allocated on an individual basis if responsibility for the pollution can be individually defined.

When analysing the proportion of river pollution caused by point sources one has to take into account at least two aspects: on the one hand, in a country-by-country comparison results will vary significantly because they reflect the history of regulations in each country. Figure A-2 and A-3 (Appendix) show, using Germany as an example, that point sources in particular reduced their nutrient discharges (nitrogen and phosphorus) significantly in the period from 1975 to 2000.[17] Nonpoint sources, particularly agriculture (see Paragraph 2.2.2), are still responsible for more than 50 percent of nutrient pollution.[18] The structure of pollution is different in other countries. The example of eastern European countries shows that, in particular, in the case of phosphorus, the contribution of urban systems (nonpoint sources) was high in the past. The installation of new sewage treatment plants (point sources) led to a significant reduction of this proportion. Figure A-4 (Appendix) illustrates this for the case of Poland.

On the other hand, the level of responsibility of point sources for the pollution they cause will depend heavily on the type of pollutant concerned (e.g. nitrogen or phosphorus). Particularly in the case of nutrient discharges, nonpoint sources (like the agriculture) cause high percentages of the total amount of nitrogen (Figure A-2 to A-4; Appendix). While nonpoint sources cause up to 80 percent and more of nitrogen pollution, they are responsible for (only) approximately 60 to 70 percent of the phosphorus pollution.[19]

In the case of salinity in Australian rivers, for example, point sources are the relevant polluters of the water (NSW EPA, 2003, p. 3; Paragraph 5.3.3). Coal mines and power stations introduce saline water into the river, thus increasing the salinity.

Direct versus Indirect Dischargers

Within the group of point sources, we can furthermore differentiate between direct and indirect dischargers. A direct discharger characterises a point source that releases its pollution directly into the water,[20] indirect dischargers release their pollutants into a waste-treatment system (direct discharger) and their discharges flow indirectly into the body of water.[21]

Pollution discharged by direct dischargers thus are released directly into the body of water. Discharges from indirect sources have no direct impact on the water quality; they flow through closed pipes until they reach the waste-treatment system. The quality of water in the river will be influenced by the activities and measures taken by a sewage treatment plant (direct discharger) which receives water from an indirect discharger.[22] As a result of differences mentioned above, the conditions under which these

[17] Figure A-2 to A-4 (Appendix) show that even a river-by-river comparison would find slightly different results.

[18] See Kraemer *et al.* (2003, p. 25). For general reduction potentials at the point source level, see Interwies *et al.* (2004, pp. 15-21) or Petry *et al.* (2005.)

[19] See also Behrendt *et al.* (2003a), HELCOM (2005) or Interwies *et al.* (2004).

[20] Definition of the US Environmental Protection Agency, see `http://iaspub.epa.gov/trs/trs_proc_qry.navigate_term?p_term_id=15164&p_term_cd=TERMDIS`, January 2007. For further definitions, see also van Mark *et al.* (1992) or UBA (1993, pp. 34-37).

[21] Definition of the US Environmental Protection Agency, see `http://www.epa.gov/OCEPAterms/iterms.html`, January 2007. For further definitions, see also van Mark *et al.* (1992) or UBA (1993, pp. 34-37).

[22] Cleaning technologies and costs within the treatment plant determines a certain maximum level of pollution for the water it receives from indirect dischargers. If a group of indirect dischargers exceeds the limit, the final discharge into the river might be lower even if the treatment plant uses the same technology.

two types of point sources (indirect versus direct) may be integrated into a water quality trading system will vary considerably.

The specific characteristics of indirect dischargers may simplify the introduction of a permit trading system amongst a group of indirect dischargers.[23] One could assume that the impact of discharges within the discharge system is not affected by the location of the source; this would dispense with the requirement of homogeneity in impacts[24] (van Mark *et al.*, 1992, p. 52). In reality, however, within the discharge system, receptor points may exist that are very sensitive to different discharge characteristics. The mixing of different types of effluents can lead to chemical reactions with negative effects. Furthermore, the concentration of waste water may in some cases become too high for the receiving sewage treatment plants to cope with (Gawel, 1992, p. 529). Thus the location and timing (as well as quantity) of discharges released by indirect dischargers into the system remain important factors.[25]

Trading conditions between indirect dischargers and receiving sewage treatment plants are similar; specific conditions and special features of sewage treatment plants must be included in the system.[26]

The integration of indirect dischargers in a permit trading system needs to be decided on a case by case basis. The number of sources and the type of waste water, for example, affects whether a permit trading system can reasonably be implemented. This study focuses on direct dischargers; this discussion and the results could, however, be adapted to address the case of indirect dischargers.

This study addresses the integration of (direct) point sources in a water quality trading system: their emissions can be measured accurately, and permits can be allocated individually. This is not the case for nonpoint sources (Paragraph 2.2.2) which makes it difficult to introduce a pure permit trading system for this group of sources. The next paragraph explains the main characteristics of nonpoint sources.

2.2.2 Nonpoint Sources

Nonpoint sources, for example, agriculture, do not discharge pollutants at a precise point.[27] "Emissions from nonpoint sources are […] diffuse. They are exemplified by emissions from mobile sources, leaching and runoff of pollutants from farm fields, and runoff from parking lots or streets. The diffuse nature of nonpoint emissions makes routine observation and accurate metering of them prohibitively costly" (Shortle and Horan 2001, p. 256).[28] It is not possible to assign individual accountability for the

[23] A description of indirect dischargers in very detail can be found in UBA (1993).
[24] According to the definition in Paragraph 3.2.1.
[25] For more details, see van Mark *et al.* (1992, pp 52). In some countries, such as England, water and sewerage charges for indirect discharges are defined with specific respect to changing intake capacities at the direct discharger level; see, for example, OFWAT (2006).
[26] Van Mark *et al.* (1992) discusses for which pollutants the integration of sewage treatment plants in an offset system could be reasonable. Additionally, existing regulation instruments within the group of indirect dischargers have to be taken into account. In Germany, for example, §7a, 4 WHG *(Waserhaushaltsgesetz,* national implementation of the *Indirekteinleiterverordnung)* requires that emission loads of indirect dischargers do not exceed the emission load that would result from the application of BAT (Best Available Technology). The coexistence of BAT regulation and a permit trading is not without conflicts, see Paragraph 3.3.5. Amendments of the existing law could become necessary.
[27] Agriculture is one of the most important polluters within the group of nonpoint sources. Nevertheless, the focus in this study lies on agriculture. For a schematised overview of point and nonpoint sources, see Figure A-1, Appendix.
[28] Due to very complex fate and transport processes historical emissions are also relevant; the time lag between discharges at the source level and the attainment of the body of water often vary a lot depending, for example, on soil and weather conditions (see also Hecht and Werbeck, 2006, pp. 244-248).

resulting pollution.[29] The combination of nonpoint emissions that cannot be measured as well as the large number of contributors results in a significant moral hazard problem.[30] In consequence, it is not possible to implement a 'purely' permit trading-based system; this would demand for the individual allocation of permits, i.e. individual assignment of pollution. It is very difficult to follow the polluter-pays-principle since the polluter can rarely be identified.

The motivation to nevertheless include nonpoint sources in a water quality trading system is based on the fact that (marginal) abatement costs at the nonpoint source level for some pollutants, e.g. nutrients, are much lower than those for point sources.[31] However, due to the measurement constraints, "trading between point sources and nonpoint sources entails a fundamental departure from text book tradeable discharge markets" (Shortle and Horan, 2001, p. 274). Trading activities involving nonpoint sources cannot be based on actual emissions due to the problems related to measurement. Another, more appropriate basis needs to be defined in order to measure nonpoint source performance. Shortle and Horan (2001) propose and evaluate alternative parameters on which to base the trading activities: emissions proxies (1) and inputs or practices (2); each of these is characterised by their own specific implementation problems.

A trading between point and nonpoint sources based on emission proxies (nonpoint sources) is usually supplemented by applying an 'emission uncertainty trading ratio'. This trading ratio reflects the uncertainty regarding the impact of reduction measures at the nonpoint source level on the ambient pollution level. It thus gives the number of units of nonpoint loadings reduction that must be purchased by a point source in order to avoid reducing its own loadings by one unit (Malik *et al.*, 1993, p. 963). A trading ratio equal to one, reflects indifference between abatement measures at different source levels. A ratio above one, lets the pollution control costs at the nonpoint source level increase in relation to control in point sources and thus reflects the uncertainty in the result of nonpoint source abatement measures.

Since the exact impacts of nonpoint source pollution and abatement measures are unknown, it is very difficult to define the appropriate level of the trading ratio. A trading ratio set too high would hamper trading activities as trading becomes very costly; but a trading ratio set too low, places the ecological effectiveness of the system at risk.

An input-based system would define the nonpoint source permits in terms of inputs. These inputs are assumed to be related to pollution flow. It is, however, challenging to define the relevant inputs and the impact of their (non-)use on the quality of the water. The ecological effectiveness of the system very much depends on the correct definition of traded input entities. Again, trading ratios would be applied in order to incorporate uncertainties of input changes and their impact on water quality.[32]

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[29] Already Dales (1968, p. 79) underlined the difficulties of measuring pollution from nonpoint sources and of integrating nonpoint sources in a stringent environmental policy.

[30] Additionally, discharges cannot be easily intercepted and neutralised through 'end-of-pipe' measures (Dosi and Zeitouni, 2000, p. 1). For further information, see also Ribaudo *et al.* (1999, p. 21).

[31] For the case of the Tar Pamlico River, North Carolina, US the marginal abatement costs at the nonpoint source level are assumed to be 7 times lower than at the point source level (Paragraph 5.2.3). For abatement potential, see Interwies *et al.* (2004, pp. 15-21) or Petry *et al.* (2005).

[32] For more information, see Shortle and Horan (2001).

The units to be traded (emissions proxies or inputs) need to be chosen carefully to guarantee the effectiveness and efficiency of the trading system.[33]

2.2.3 Implications

In the following, the implementation of a permit trading system will only be discussed for the group of point sources. The implementation of a 'pure' permit trading system for nonpoint sources seems to be impossible. Even if modern technologies can precisely predict the impact of different activities, e.g. agriculture, on the ambient pollution level of the river concerned,[34] it is impossible to make a precise individual assignment of a defined share of the resulting ambient pollution at a certain point in time. A permit trading system cannot be introduced on an individual level for nonpoint sources.[35]

However, this does not mean that nonpoint sources should be excluded from any environmental regulation. On the contrary: considering the high proportion of pollution and the relatively low activities of abatement in some countries and for some pollutants,[36] specific and stringent economic incentives are required in order for nonpoint sources to increase their abatement activities.[37]

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[33] Due to the measurement problems, authors often propose other economic instruments to regulate nonpoint sources. Shortle and Horan (2001) give an extended survey on the economics of nonpoint source pollution control. Actually, experimental studies also evaluate the application of different economic instruments. See Camacho and Requate (2004).

[34] The European Union currently invests heavily in research projects in this area. Particular emphasis should be given on the project Aquaterra (`www.eu-aquaterra.de`, January 2007). This project is the first to follow an all-embracing approach by including the transportation processes of substances in the earth, groundwater and surface water as well as in sediments. The objective of this project is to construct a computer system, which can precisely predict the circulation processes of substances under different conditions. It should thus be able to predict the impact of changes in methods of cultivation in agriculture. The project started in 2004 and should be finished in 2009. The website of this project lists other EU-projects dealing with this subject.

[35] As the pollution of groundwater is almost exclusively caused by nonpoint sources, one can conclude from the same arguments that permit trading cannot be the right instrument for the protection of groundwater. For more information, see Hecht and Werbeck (2006, pp. 83-84).

[36] See Paragraph 2.2.1. In many European countries the emissions from point sources have been strongly regulated for some time; as a result these sources have already noticeably reduced their discharges; for nonpoint sources this does not yet hold true (for an overview for Europe, see Kraemer *et al.* (2003, p. 25)). Nevertheless, due to the high proportion of pollution held by the nonpoint sources (for the European Union, see `http://europa.eu.int/scadplus/leg/en/lvb/l28013.htm`, January 2007, for Germany Interwies *et al.* (2004, p. 6)), it is essential to regulate these sources more consistently.

[37] The literature discusses different economic regulation instruments for nonpoint sources and its problems. See, for example, Shortle and Horan 2001, Segerson and Wu (2006), Dosi and Zeitouni (2000), Ribaudo *et al.* (1999) or Karp (2006).

2.3 Relevant Pollutants

Discharged pollutants are very different depending on the dispersion and diffusion within the receiving body of water and thus the impact on the ambient pollution level of the medium.[38] These different impacts of the pollutants are of great importance when determining the environmental policy objective within an ecological framework. The choice of an adequate design cannot be done without well-founded knowledge of the pollutant-specific characteristics.

2.3.1 Characterisation

General Aspects

One general condition determines whether a specific pollutant can adequately be incorporated into a water quality trading system or not. Only those substances can be integrated, for which a maximum level of concentration can be defined, which adequately limits the environmental damage. Substances, for which a complete cessation of emissions is required by law, cannot be regulated by a permit trading system. In this case, prohibition of those emissions is the only instrument to be applied. But for pollutants which are harmless up to a certain level of concentration, a permit trading system may be an efficient instrument of regulation.

Furthermore, it may be considered to incorporate some (groups of) pollutants in a single trading system. Permits defined in standardised units would guarantee homogeneity of traded entities.[39] A main prerequisite for such an integrated trading system is that it does not matter how much of a single pollutant is abated; only the total amount of these pollutants is relevant, not the exact composition. In the special case of river basin management, we will see that it is already difficult to create homogeneity in an ambient-based trading system for a single pollutant. An integrated trading system for diverse substances might not be feasible in all cases. The decision on how to proceed should always be preceded by a careful analysis of interdependencies between pollutants.

Assimilative versus Accumulative Pollutants

The initial difference between the substances is whether they are assimilative or accumulative. Assimilative pollutants can be assimilated, absorbed or recycled in the water, provided that sufficient time is available. They thus have a certain threshold, below which there is no environmental effect. Only above this threshold (maximum) do these substances become harmful. Nutrients represent a typical example for assimilative pollutants (Paragraph 2.3.2).

The assimilative capacity for these substances may be different in space and time, for example, for regional or local specifications of the water medium. Thus if they surpass a certain amount, assimilative

[38] Dispersion means the distribution within the water medium due to movements in the water (active). Diffusion is a passive process and means the uniform distribution within the water due to molecular characteristics (van Mark *et al.*, 1992, pp. 47-48). For more detailed definitions, see Schnoor (1996, p. 35) or Eiswirth (2003). In this study the term 'dispersion' is used as generic term.

[39] Kemper (1993, pp. 55-56). The EU emission trading system (2005) incorporates 6 different greenhouse gases: carbon dioxide (CO_2), methane (CH_4), nitrous oxide (N_2O), hydrofluorocarbon (HFCs), perfluorocarbon (PFCs), and sulphur hexafluoride (SF_6). Permits are traded in terms of CO_2 equivalents.

substances may take on the properties of accumulative substances and should than be treated as such (Kemper, 1993, p. 71).[40]

Accumulative substances are not assimilated in the water;[41] additional quantities introduced into the water accumulate. The receiving water may be able to take in a certain amount of specific accumulative pollutants without serious damage occurring.[42] This could be a minimum amount which exists naturally. There are, however, limits which should not be exceeded. Paragraph 2.3.2 shows that salt is an accumulative substance that often exists in rivers.

Uniformly versus Non-uniformly Mixed Pollutants

Pollutants can further be classified by their degree of mixing in the medium. Uniformly mixed pollutants spread quickly and uniformly throughout the medium. No concentration issues (hot spots) must be taken into account; different meteorological conditions do not change the characteristics. An example of a uniformly-mixed pollutant in the air is CO_2. The integration in a permit trading system is relatively simple: it does not matter where and when emissions take place. Only the absolute amount of emissions is relevant for the resulting pollution.[43]

Non-uniformly mixed pollutants, on the other hand, do not disperse quickly and (thus) not uniformly.[44] This is relevant when controlling water pollution. Pollutants, such as nutrients or salt, do not disperse uniformly within the water; thus local concentrations (hot spots) are possible. As a result, environmental damage emerges locally, too.

Uniformly mixed pollutants are of global relevance, yet non-uniformly mixed pollutants are of more local relevance (risks of hot spot formation).[45] While the extent and spatial pattern of damage caused by uniformly mixed pollutants (assimilative or accumulative) depends solely on the level of emissions, this does not hold for non-uniformly mixed pollutants. Here, the extent and spatial pattern of damages are affected by the level of emissions, but also by location, time and dispersion characteristics of the emissions, i.e. the distribution in the medium.[46] This must be taken into account when designing a water quality trading system.[47]

Emissions versus Immission

As a result of the two preceding sections regarding the properties of pollutants, the basic terms 'emission' and 'immission' need to be defined. The term 'emission' (also effluents or discharges) means the discharge a source introduces into the water; the term 'immission' means the resulting ambient pollution level (see Figure 2-1).[48] Depending on the characteristics of the discharged substances

[40] CO_2, for example, can be assimilated by the air medium up to a certain amount. However, the actual level of CO_2 in the air exceeds the assimilative capacity of the air. CO_2 thus takes over the typical characteristics of accumulative substances.
[41] …or are assimilated very slowly.
[42] See Kemper (1993, p. 71).
[43] Permits can thus be defined in emissions, no conversion into immission values is necessary.
[44] See Tietenberg (2006) or Commonwealth of Australia (1992, p. 22).
[45] Kemper (1993, p. 72) follows a similar definition; he distinguishes between 'local' and 'regional' pollutants depending on the distance they disperse. Local pollutants can cause local concentrations, while the risk for hot spots is lower for regional pollutants.
[46] Tietenberg (2006, pp. 33-38).
[47] Perman *et al.* (2003, p. 267) conclude that in the case of non-uniformly mixed pollutants a permit trading systems is the only marketable instrument that will be able to achieve the predetermined objective as cost-effectively as possible in the presence of asymmetric information about the sources' cost curves.
[48] See Hecht and Werbeck (2006, p. 235). It is worth being mentioned that already Dales (1968, pp. 77-81) pointed out the importance of distinguishing between emissions and immission.

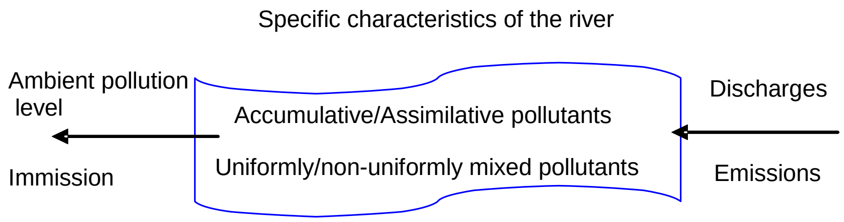

Figure 2-1: Emissions versus Immission

discussed in the two sections above (assimilative versus accumulative and (non-)uniformly mixed), effluents (discharges) can have very different impacts on the immission level, i.e. the quality in terms of concentration.[49] Due to assimilative and accumulative processes, only part of the discharges lead to immission.[50] Different dispersion characteristics (uniformly versus non-uniformly) lead to different contributions of immission.

The properties of the receiving medium influence also the relationship between emission and immission levels (see Section 2.4 for more details). Finally, the impact of the discharge of the same substance at the same location can be very different depending on the characteristics of the sources: the location of the source is relevant as well as certain parameters of the discharge (temperature of discharges, speed…).

Dispersion Function and Water Quality Models

All these parameters are reflected by a dispersion function and the resulting dispersion coefficients. These coefficients show the link between specific discharges (emissions) at a specific point in time from a specific location (source) and the impact on the ambient pollution level (immission) at a certain point in time and at a specific receptor point. The immission standard thus becomes operable. The dispersion function is of great relevance for most of the theoretical and practical water quality trading approaches and will therefore be discussed in more detail later (Chapters 4 and 5).

Mathematical water quality models are used to predict changes in the ambient pollution levels (immission) due to modifications of the discharge behaviour of the sources (emissions). They seek to determine the exact dispersion function and their coefficients for specific cases. Consequently, these models enable the derivation of the emission cap for a given immission goal; they are an indispensable basis for the efficient and effective design of a water quality trading system.

Water quality models can be classified by their complexity, the type of receiving water, and the water quality parameters (nutrients, salt etc.). A more complex model tends to give a better, more detailed, prediction on the impacts of discharges; the application, however, becomes more cost-intensive. World Bank Group (1998, p. 101) mentions four factors that directly influence the complexity of a water quality model: number and type of water quality indicators, level of spatial detail, level of temporal detail, and the complexity of the body of water.

There are a large number of water quality models with different specifications. In many cases, water quality models need to be adjusted in order to account for particular watersheds or project-specific

[49] The ambient pollution level, i.e. the immission load, can decrease due to lower emission loads or due to increasing water flows *ceteris paribus*. This characteristic is relevant for the design of water quality trading systems, see Section 2.4.
[50] See also Karl (1998, p. 83).

analyses.[51] Other models, however, have been applied to a wide variety of water management analyses. The World Bank Group (1998) refers to 5 water quality models[52] and shows the differences in complexity and in their application potentials (rivers vs. lakes; static vs. dynamic perspective etc). A different type of model must be selected to suit the problem to be solved.

Again, there is an important difference between water and air pollution control. Particularly the fact, that rivers are flowing bodies of water which transport pollutants uni-directionally simplifies water quality modelling enormously compared to air quality modelling. Paragraph 4.3 and Chapter 5 show in more detail, how water quality trading approaches can benefit from these characteristics.

Furthermore, the accuracy of the model projections depends severely on the quality and quantity of the data available. This can be a problem in practice. This is particularly important in countries that, until now, did not have a consistent environmental policy and thus for which very little data is available. There may also be differences for the varying type of pollutants. Scientific interest in these data has so far been relatively low, because the previous environmental regulations did not require such differentiated predictions and monitoring activities in many countries.

2.3.2 Selected Substances: Nutrients and Salt (Salinity)

Due to the case studies presented in Chapter 5 (Hunter River, Australia: salinity; Tar Pamlico River, US: nutrients) this study focuses on the pollutants salt and nutrients (i.e. phosphorus and nitrogen).[53] Irrespective of these two case studies, these substances are some of the most relevant with respect to the proportion in water pollution in many regions. The majority of the implemented water quality trading systems regulates the emission of nutrients or salinity. In the European Union and the United States, rivers in large part suffer from too high nutrient levels, whereas some Australian river basins are characterised by high salinity levels.[54]

Nutrients

The discharge of pollutants causes a negative effect but this is not true in every instance and not necessarily right from the first beginning: a certain amount of pollutant can be absorbed by the medium (assimilative) and this may even be desirable. This is the case for nutrients (nitrogen and phosphorus).[55] Nutrients themselves are not harmful to water or water-based ecosystems.[56] They are even necessary components of plant life. Indeed a certain amount of these substances in the water is desirable or even necessary. However, if the amount is too high, it encourages algal growth and can thus provoke eutrophication.[57]

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51 Some rivers, for example, are frozen for several months of the year; the effect on pollution and transportation processes of pollutants must be integrated into an adequate water quality model. The thawing of snow becomes relevant in these countries. In other countries, water quality models would be more likely to include long periods of drought and extreme low flow.

52 WQAM, QUAL2E, WASP, CE-QUAL-RIV1 and HEC-5Q.

53 Further relevant substances in the medium water in general: temperature, biochemical oxygen demand (BOD) and chemical oxygen demand (COD), pH-value or heavy metals (van Mark *et al.*, 1992, p. 44). Most water quality trading systems are implemented for nutrients (see also Paragraph 5.2.2); but also for BOD and even for temperature water quality trading systems and/or water quality models exist (Environomics, 1999).

54 See Interwies *et al.* (2004) and National Action Plan for Salinity and Water Quality (Commonwealth of Australia, 2005).

55 Phosphorus and nitrogen from point sources mainly originate from sewage, and phosphate detergents. Nonpoint sources introduce phosphorus through the use of industrial and/or organic fertilizers (manure).

56 Kraemer and Banholzer (1999, p. 94) or Bahnsen (2002, p. 91).

57 For more information on eutrophication processes, see Smith *et al.* (1999). This term is most commonly applied to freshwater lakes and reservoirs. It can also be applied to flowing waters (Smith *et al.*, 1999, p.181).

Nutrients are assimilative[58] and are non-uniformly mixed in a river[59]: a certain proportion of the nutrients is absorbed by the water; the dispersion of nutrients in a body of water is non-uniform.[60] The properties of a given river, different flow speeds and so on will further influence the dispersion processes. Rapidly flowing water has a high assimilative capacity for nutrients; they are thus rendered harmless. At the same time, higher water flow rate also washes an increased amount of nutrients out of the soil. No general result can be determined; each individual case must be analysed carefully. The problem of eutrophication caused by excess amounts of nutrients is particularly relevant for bodies of standing water or those with very slow water movements (van Mark *et al.*, 1992, pp. 70-72).

The measurement and estimation of (future) nutrient amounts in the water are less complex processes even though fate and transport properties are relatively complex (Behrendt and Opitz 2000). Researchers generally conclude that nutrients are principally appropriate to be regulated in a water quality trading system.[61] Due to the discussed characteristics, it would be reasonable to regulate nutrients in an ambient-based system; different impacts of discharges on water quality need to be reflected by the system. Furthermore, nutrients, i.e. nitrogen and phosphorus, can be (and often are) integrated into a single trading system; in this case, permits need to be standardised.

Salt (Salinity)

Salinity describes the concentration of salt in the water. The behaviour of salt in a river also depends on the properties of the specific body of water. Different flows (quantity and velocity) as well as different soil conditions influence the distribution of salt in the water. While salt *per se* is a uniformly mixed substance in water in general, salinity mixes non-uniformly in a river.[62] Salt accumulates in the water and is non-degradable.[63]

If the salinity of the water is too high, it affects the suitability of water for many purposes, such as irrigation and drinking water supply.[64] Relatively sensitive ecosystems can also be negatively affected by high levels of salinity.[65]

It is relatively easy to precisely measure the level of salinity: the electrical conductivity of the water tells us about the concentration of salts in the water body or in the discharges (Kraemer and Banholzer, 1999, p. 90). Salt is thus assumed to be appropriate to be regulated by a water quality trading system. In some river basins natural salt levels are already quite high, e.g. in Australia (Section 5.3).[66] When introducing a water quality trading system, the effects of natural salinity need to be separated from impacts caused by industry or other users. Again, the incorporation of salinity problems in a water quality trading system requires (in most cases) an ambient-based system to integrate the relation between discharges and water quality levels.

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58 WSDE (1999).
59 Kemper (1993, p. 74).
60 Additionally, nutrients are in part degradable.
61 Van Mark *et al.* (1992, pp. 67-68) distinguish different forms of nitrogen. Some of them are easier to integrate in a water quality trading system. When incorporating a trading system for phosphorus, the fact that phosphorus could be the limiting factor for eutrophication need to be taken into account.
62 See Commonwealth of Australia (1992, p. 22).
63 A certain quantity of salt can be held in the water (dissolved). If additional quantities of salt are added, salt will not be dissolved by the water anymore; the 'surplus' of salt remains undissolved in the water medium.
64 See Kraemer and Banholzer (1999, p. 90).
65 For more information, see NSW DLWC (2000b, p. 57).
66 Additionally, sewage treatment plants and power stations are potential sources for the introduction of saline water.

2.4 The River Itself

General Characteristics

Unlike lakes, rivers are flowing bodies of water. This should be taken into account when considering the introduction of a permit trading system.[67] Pollutants are transported within the body of water. Emissions from single sources can have very different impacts on the immission load, i.e. the water quality: the properties of the pollutant, but also the physical characteristics and intake capacities of the receiving body of water will influence the resulting quality (Kemper 1993, p. 67). The problem of hot spots has to be addressed carefully.[68]

Generally, different types of rivers have to be considered; each of them representing different characteristics (mountain river vs. lowland river; rivers affected by tides etc.; ATV, 1989, p. 755).[69] A single river may display a number of different river characteristics. These river forms will in turn influence the fate and transport processes of substances and thus their harmfulness.

The hydromorphology of a river will also influence the fate and transport processes and thus the resulting water quality.[70] Artificial straightening of rivers as well as the installation of dams significantly influence flow conditions (quantity and velocity).[71] In an extreme situation this can cause a flow rate of zero, which is followed by decreasing intake capacities. River regulation may also lead to higher flow velocities. This may increase the absorption of some substances; but on the other hand, important retention areas for substances are destroyed by this. In the case of nutrients, in particular, flow conditions (velocity) influence the damage caused by the pollutants. Changes in the hydromorphology of the river are more relevant in the medium or long term. These modifications could thus be integrated step by step (adjustment of the caps etc.) in a trading system.[72]

Moreover, stochastic components like precipitation and other weather events influence the properties of a river – regardless of their physical type – resulting in changing water levels and flow rates[73] or different temperatures over time and in space. Consequently, the intake capacities for pollutants vary.

Depending on the type of the river and the riverbed (geographical location), the natural (or background) level of certain pollutants can vary significantly. Background pollution means the pollution which already exists in the water. Background pollution lowers the remaining emission intake capacity and thus influences the definition of the emission cap. Positive background pollution consequently reduces the capacity for additional discharges, given a certain immission load objective. Sources can only be held responsible for the proportion of the pollution they themselves caused by releasing

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[67] Again, differences with respect to the CO_2 permit trading with uniformly mixed pollutants must be taken into consideration.
[68] Hot spots could be caused both by pollutant and/or river characteristics. Both types of hot spots have to be adequately integrated in a water quality trading system.
[69] For an example of river classification, see Rumm *et al.* (2006).
[70] See also Petry *et al.* (2005).
[71] See Graw (2004).
[72] The general question addressing under which situations hydromorphological changes are reasonable, is not discussed in this study, e.g. by integrating cost-benefit-analysis. Hydromorphological aspects are becoming more and more respected as important parameters in international law. The Water Framework Directive, for example, sets specific hydromorphological objectives (Article 4 WFD); water status is also defined by hydromorphological aspects.
[73] The impact of changing water levels on the water quality can be very important. A separated analysis shows how this aspect can be integrated into the design of a permit trading system (Paragraphs 4.3.2 and 5.3.3).

discharges. It is thus important to separate the background pollution[74] of specific pollutants in the water from the amounts introduced by participating sources.[75] Although the separation of these two aspects might not always be simple,[76] it is an important basis for an efficient and effective permit trading system.

Finally, rivers do not always flow within national borders; often they are transboundary, flowing through different countries.[77] This fact requires close cooperation between all countries concerned (Section 3.1 and Paragraph 3.2.2).

Water Quality versus Water Quantity

When regulating water quality in a river basin, water quantity aspects are also of great relevance. The concentration level (immission) in the body of water not only depends on the absolute amount of emissions (load) but also on the level of water quantity. The same emission load (in absolute terms) would cause different levels of concentration in a river, depending on the water level. In periods of high water levels, the emission would cause lower immission than in periods of low water levels.

Different aspects can decrease water levels: on the one hand, natural and stochastic (weather) variation influences the water flow level; on the other hand, industry, agriculture and other users divert water for production use (cooling water, irrigation…). While the former can hardly be influenced (at least in the short term)[78] and are thus exogenous, the latter can be directly influenced by individual behaviour and regulations and are thus endogenous. Individual responsibility can usually be assigned. This aspect is, in particular, important for some of the water quality trading approaches (Paragraphs 4.3.2 and 5.3.3).

Since water levels directly and in the short term influence the water quality, it may be reasonable to combine regulations for water quantity aspects and water quality aspects in a single system. The resulting problem for an ambient-based water quality trading system can be illustrated as follows (Figure 2-2). The value of the permits has to be defined in terms of emissions although the caps are formulated in terms of immission (see Section 2.5). The absolute value of permits (emission load) in an ambient-based system should be lower in low flow periods to hold a constant immission cap (concentration); if water quantity levels are low, effluents cause higher immission levels. In high flow periods, the absolute value of ambient-based permits (in terms of emissions) could be increased: the water medium has higher intake capacities.[79]

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74 The terms 'natural' and 'background' pollution are used as synonymously (see also Novotny, 2003, p. 27).
75 See Section 5.3 for Australia. The background salinity in the Hunter River is quite high.
76 See Bahnsen (2002, pp. 90-97) for the case of the Baltic Sea.
77 An extreme case is the Danube River Basin as the river flows through 19 countries (see also Paragraph 6.2.2).
78 Natural changes in weather conditions cannot be directly influenced. Long-term weather conditions, however, may be influenced by human activities. Studies on Climate Change show that changes in the temperature would lead to decreasing water levels in a part of European river basins. A general decrease in water quantity levels would aggravate existing water quality problems (see BMBF 2005).
79 If water quantity aspects are relevant and are nevertheless not integrated in such detail into the design of the trading system, the most stringent emission cap (in absolute terms) should apply to guarantee ecological effectiveness.

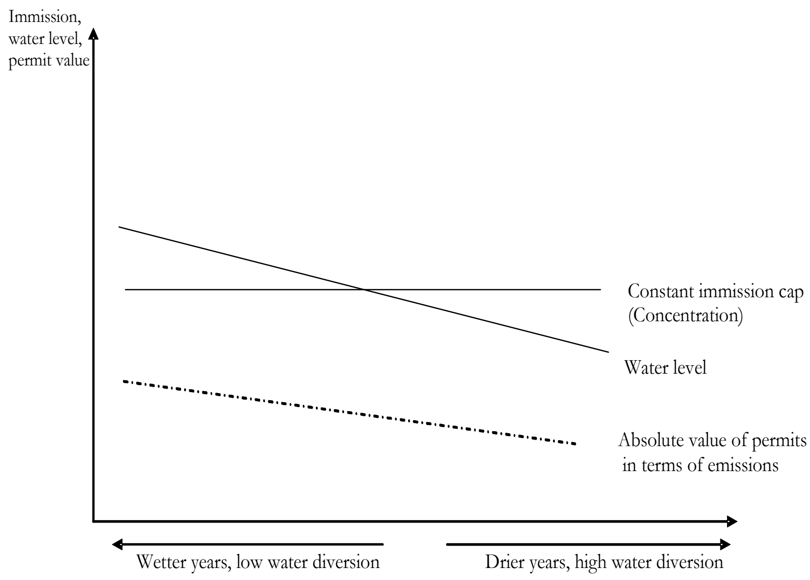

Figure 2-2: Water Level and Water Quality

In the event that water flow is constant over time, the influence of water quantity levels on the water quality can be ignored; an emission-based trading system might be appropriate.[80]

Upstream-Downstream Problem

As a river is a flowing body of water, the same volume of water passes different sources located on the riverbanks at different points in time. Changes in the discharge behaviour of an upstream source influences the water quality for all sources located downstream. Trading activities between non-adjacent sources can cause third-party effects on sources located in between. This is a crucial difference to the CO_2 trading situation, where trading activities between two sources do not influence the emission situation for other sources. CO_2 disperses uniformly in the air; the total amount of emissions is relevant for air quality, independent of time and location of emission.

The upstream-downstream problem is a relevant basis for the understanding of specific river characteristics. (Almost) All permit trading approaches (theoretical and practical), which will be presented in Chapters 4 and 5, are influenced by these considerations. Trading approaches find very specific ways to integrate these special characteristics of pollution processes in a river. The following Figures and Tables illustrate the upstream-downstream problem in more detail.[81]

[80] Characteristics of the pollutants could nevertheless require an ambient-based system.
[81] This presentation follows the discussion of van Mark *et al.* (1992). For an analogue illustration, see Hecht and Werbeck (2006, pp. 80-83) or Durth (1996a, Chapter 3).

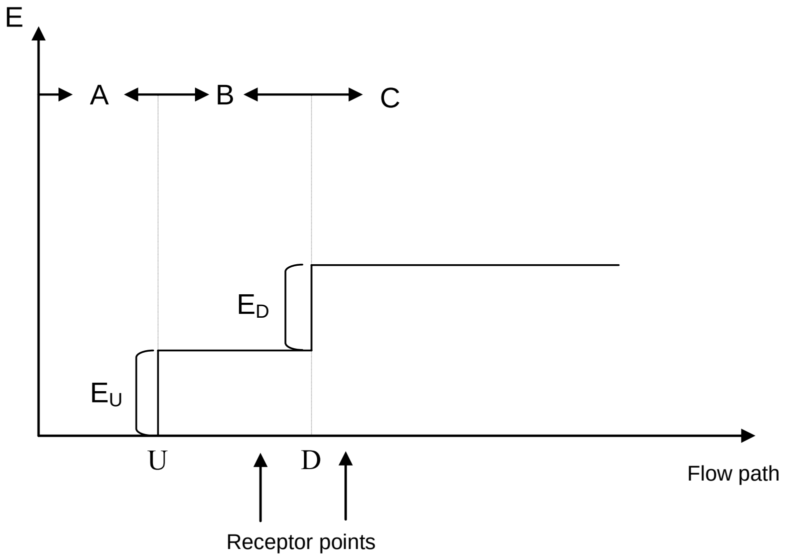

Figure 2-3: Upstream-downstream Problem (without load reduction)
According to van Mark *et al.* (1992, p. 56)

Permit transfer		Discharge load	Resulting load	
			Upstream of the discharge point D	Downstream of the discharge point D
Initial allocation		$E_U=E_D$	C_U	C_D
After trading	U ➔ D	$E_U<E_D$	$< C_U$	C_D
	D ➔ U	$E_U>E_D$	$> C_U$	C_D

Table 2-1: Results, Upstream-downstream Problem (without load reduction)
According to van Mark *et al.* (1992, p. 56).

Van Mark *et al.* (1992) assume that pollutants are introduced into the river at two different discharge points (U : upstream source; D: downstream source). In the first case, the pollutants in the river are not reduced due to degradation or sedimentation (assimilation); pollutants completely accumulate ('without load reduction'). Figure 2-3 shows the resulting loads at two receptor points located upstream and downstream from the downstream discharge point D and measuring the pollution. Table 2-1 illustrates the general development of resulting loads if discharge activities are transferred, for example, by trading, from upstream to downstream or the other way around.[82]

Figure 2-3 shows three zones (A, B, and C) affected by the pollution. The upstream discharger introduces effluents E_U at the discharge point U into the water.[83] At point D the downstream source introduces additional pollutants (E_D); the resulting load increases by E_D.

[82] The upstream-downstream model assumes that discharges are uniformly distributed in time. This assumption does not hold for every case in reality; discharge processes are often discontinued. The time profile of discharges may be different between sources. This should be taken into account. Additionally, the relevance of peak loads against continuous load of lower concentration is different for specific types of pollutants.

[83] The fact that the emission load is not reduced by assimilation or similar is reflected by the horizontal line.

The contents of Table 2-1 can be discussed as follows. Emission transfers, e.g. by permit transfers, between the two sources would not affect the water quality in zone A and C. The cumulative total load in C remains the same independent of the allocation. From an ecological point of view the water quality in zone B becomes relevant: while emission transfers from upstream source to downstream source would increase water quality in zone B, water quality in B would decrease, if the upstream source purchases additional emissions allowances. Zone B is thus affected by transfer activities between source U and source D. A source located in zone B would thus be concerned by transfer activities between other sources (third-party effects).

Van Mark *et al.* (1992, pp. 57-59) analyse the same situation in the event that emission load decreases from upstream to downstream due to degradation, sedimentation or similar effects (assimilation).[84] Figure 2-4 assumes that the emission load degrades with a constant rate and does not completely accumulate. In this case both zones B *and* C are affected by changes in the permit allocation (Table 2-2). This situation cannot be evaluated in a definitive way, results will depend significantly on the degradation rates. Additional discharges from the upstream source could be reduced significantly by assimilative processes when reaching the receptor point. This would reduce the negative effect of the emission transfer on zone B (original increase in E_U). As the downstream source would reduce its emissions according to the emission transfer, the overall effect would be a positive one. Other sources located in these zones could be positively affected by the emission transfer processes. But again, these results depend on the degree of assimilative processes.

An emission transfer from upstream to downstream would lower the emission load in zone B, the emission load in C would, however, increase. The total effect strongly depends on the assimilative processes and can thus not be identified without further analyses. Again, (negative) third-party effects evolve.

The model of van Mark *et al.* (1992) shows that depending on the transport and fate processes within the body of water trading activities have different impacts on the water quality of specific zones; third-party effects (positive and negative) are possible. On the one hand, one could assume that impacts of discharges from upstream sources on the water quality at the receiving water is lower than that of discharges from downstream and are thus preferable. On the other hand, discharges from upstream sources lower the discharge opportunities for downstream sources along the river. The further upstream a reduction of discharges takes place, the more sources downstream can profit. A well-designed water quality trading system would automatically integrate these aspects by means of different (implicit) values of permits in order to avoid hot spots and inefficiencies caused by third-party effects.[85]

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[84] This is approximately true for nutrients, see Bräuer and Marggraf (2004) and Graw (2004, pp. 19-22).

[85] Furthermore, this model can be used to evaluate the appropriate geographical scope of a water quality trading system (see Paragraph 3.2.2).

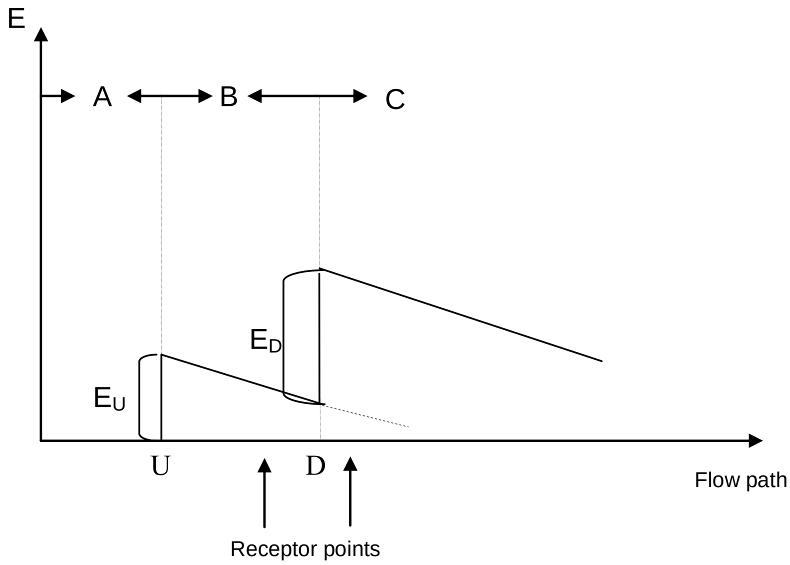

Figure 2-4: Upstream-downstream Problem (with load reduction)
According to van Mark *et al.* (1992, p. 58)

Permit transfer		Discharge load	Resulting load	
			Upstream of the discharge point D	Downstream of the discharge point D
Initial allocation		$E_U=E_D$	C_U	C_D
After trading	U ➔ D	$E_U<E_D$	$< C_U$	$> C_D$
	D ➔ U	$E_U>E_D$	$> C_U$	$< C_D$

Table 2-2: Results, Upstream-downstream Problem (with load reduction)
According to van Mark *et al.* (1992, p. 58)

Within a permit trading system, the flow conditions of the river cause another type of upstream-downstream problem. The first source located upstream cannot profit from upstream reductions as no upstream source exist anymore that could abate discharges in order to increase water quality at the subsequent location. At the same time, the last source located downstream can reduce discharges, but it cannot sell the resulting surplus permit, as no further source is located downstream. Surplus permits cannot be sold to upstream sources, as the reduction downstream has no impact on the water quality upstream of this location. The analysis of theoretical and practical approaches (Chapters 4 and 5) will demonstrate, that some of these models are able to avoid this asymmetry between sources.

2.5 First Technical Implications

The specific characteristics of the pollutants concerned, especially uniformly versus non-uniformly mixed, demands for the differentiation between emissions and immission. Depending on location and time of discharge, impacts on the water quality can differ significantly. Dispersion functions and coefficients are often used to describe the relation between discharges and the resulting water quality.

In the case of rivers, the water quantity level also influences the water quality directly. The same discharge load leads to a higher (lower) water quality for a higher (lower) water flow. A well-designed water quality trading system needs to take this link into account. Furthermore, the flowing characteristics of the river create the upstream-downstream problem. Trading activities can generate third-party effects that must be considered in the water quality trading design. Experiences from air pollution control, even for non-uniformly mixed pollutants, can not be directly transferred to the case of water quality trading systems.

Emission-based versus Ambient-based

The properties of the pollutants as well as of the river itself must be adequately integrated into a water quality trading system. Two general approaches developed for such permit trading systems: emission-based versus ambient-based.[86] An emission-based system defines the permit in terms of emissions independent of the location of the source or the time of discharge. The ambient pollution level is thus not directly incorporated in the ecological objective. Differences between discharges and the resulting immission are ignored. The control of emissions alone cannot guarantee specific water quality standards and thus homogeneity in terms of water quality.[87] Emission-based trading systems are thus only relevant if the relation between emissions and immission can be ignored.

Permits in an ambient-based trading system are defined in terms of immission loads (ambient pollution level). Water quality standards define a certain concentration of substances which is not allowed to be exceeded at a given location at a given point in time (Tietenberg 1980, p. 482). The determination of immission caps guarantees the achievement of a predetermined water quality. However, as, on the one hand, the ambient pollution level can only be influenced by changes in the emission loads, and as, on the other hand, the application of emission caps to individual sources is much more operable,[88] the ambient cap needs to be converted into emission caps. These caps, in terms of emission loads, will be derived from the determined immission objective through the application of dispersion functions.

For the most part, non-uniformly-mixed pollutants are relevant for river management (Section 2.3). While permit trading systems for uniformly-mixed pollutants, e.g. CO_2, can ignore the relation between emissions and immission (location and time of discharge do not influence the resulting quality) without any impact on the homogeneity of traded entities and thus on the effectiveness of the trading system, the regulation of non-uniformly mixed pollutants requires a more detailed examination. Consequently, emission-based trading systems would rarely be appropriate. Ambient-based systems are rather adequate to reflect fate and transport processes of pollutants.

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[86] See, for example, Commonwealth of Australia (1992, p. 23). These approaches should not be confounded with two general philosophies in environmental economics: The 'emission standard philosophy' and the '(air) quality standards philosophy' (Kemper, 1993, pp. 68-69, Bonus, 1984, pp. 66-70 and de Nevers, 1977). The former philosophy requires that each source controls his emissions to the maximum degree (de Nevers, 1977, p. 198 and Bonus, 1983, pp. 330-332). Therefore, representatives of this philosophy require that every installation emitting pollutants uses abatement technologies corresponding to the Best Available Technology (BAT), which is assumed to be the maximum of pollution control. No emission rate or emission cap is specified in this type of regulation; emissions are influenced indirectly by setting the technological standard. The 'quality standards philosophy' has the immission load (ambient pollution level) as a starting point, not the emissions. The environmental quality is of crucial relevance not the emission of individual sources.). It is – more or less – of no importance where and how the emissions are reduced but it is important, that the immission standard is fulfilled everywhere and every time. It thus is also not important, whether the Best Available Technology is implemented everywhere.

[87] Exceptions are pollutants for which emission levels equal immission results (Kemper, 1993, p. 66, fn 3).

[88] Sources can only be held responsible for the damage they definitely cause, i.e. for the emissions.

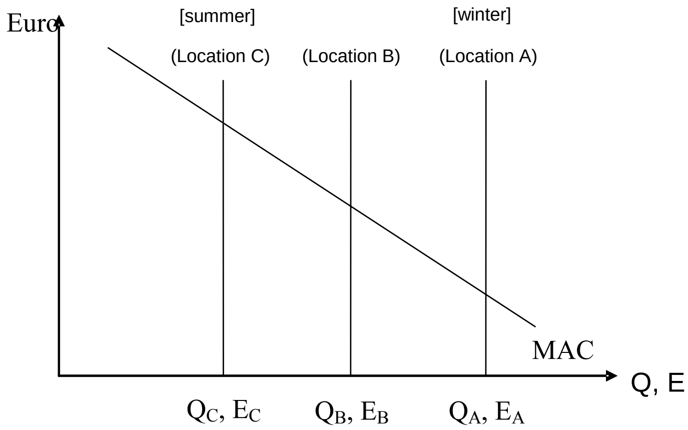

Figure 2-5: Differentiation of the Caps
According to Kemper (1993, p. 89)

In order to reach this specified water quality in an ambient-based permit trading system, a careful conversion to the emission cap is of great importance to avoid hot spots and to guarantee ecological effectiveness through homogeneity.[89] The determination of the emission cap must reflect the specific characteristics of the river and the pollutants concerned, because they strongly influence the relation between discharges (emissions) and the resulting ambient pollution level.

In general, the required maximum emission load E_i may need to be differentiated in space (for example, E_A, E_B, E_C for different locations A, B, and C, Figure 2-5) and/or time (summer, winter, Figure 2-5) in order to reach the same water quality standard Q at different locations $(Q_A = Q_B = Q_C)$ and at different times. In this example, the emission cap E_C in location C needs to be defined more strictly than in the other regions in order to reach the same ambient pollution level (water quality). The same emission amount causes a higher immission load at location C than at location B and A. At the same time, the marginal abatement costs (MAC) are rising with increasing emission reductions.

The definition of the emission cap also depends on a time component. For example, in some cases a stronger emission cap would be necessary in summer months than in winter months, in order to achieve the same water quality standard, because of the different water temperatures.[90] But a variation in water quantity levels can also significantly influence the impact of discharges (Section 2.4).[91]

(Impact) Trading Ratios

Another specific element of ambient-based permit trading systems for non-uniformly mixed pollutants must be considered in more detail. Again, depending, for example, on the location of the sources, the emission discharge of different sources may have different impacts on the immission, i.e. the environmental quality of the medium at a certain point. If this holds true, a one-to-one trade of permits between these sources could affect the ecological effectiveness of the system in a negative way. To

[89] The result relies heavily on the conversion mechanism. For an integrated international river basin management scheme the conversion should follow identical methods. Different conversion methods would lead to varying emission caps and thus cause inefficiencies or even risk ecological effectiveness. Legal restrictions that cause a differentiation in emission caps in space through different transfer mechanisms need to be avoided (cooperation principle).
[90] Van Mark *et al.* (1992, pp. 71-72) discusses the case of nutrient effluents in summer and winter time and the impacts on the water quality. Not only seasonal, but also daily variations in the assimilative capacity for emissions may exist for a constant immission load goal.
[91] If necessary, an emission cap of zero can be formulated for different parts of the river if the current immission load already equals the maximum or the designed water quality goal.

avoid this, one could introduce trading ratios for all transactions between sources.[92] These trading ratios determine by how much source 1 must decrease its emissions if source 2 wants to increase its emissions by one unit, purchasing a permit from source 1. For example, a $2:1$ trading ratio means that source 1 must abate two units if source 2 increases its emissions by one unit. This would be the case, if of one unit of source's 1 discharge has a lower impact on the environmental quality at a given point than one unit discharged by source 2. The system can thus guarantee a constant environmental quality level, i.e. immission load. Different impacts of emissions on the immission are thus taken into account. A permit is thus standardised in its value (homogeneity, Paragraph 3.2.1).

It is often criticised that too high trading ratios would hamper trading activities by changing the price and making abatement measures too cost-intensive. A trading ratio of 2:1, for example, would raise the price by 200 percent and thus increase the costs of abatement. Implementing reduction measures within the own plant could thus become cheaper.[93] This is a direct consequence of the price intervention that is, however, necessary in order to take into account different impacts of abatement measures at different locations or at different points in time. The trading ratio guarantees homogeneity of traded entities in terms of water quality. The trading ratio thus reflects the adjusted price of the permit in terms of water quality.

One could further criticise that it is not possible to determine the trading ratio adequately, i.e. a trading ratio that reflects exactly the given impact relations. Incorrect trading ratios would bias the system and hinder the achievement of ecological effectiveness. It is, however assumed, that trading ratios set in order to balance different locations of sources and the conditions of the environment can be determined rather precisely by using water quality models (see Section 2.3, Hall and Howett, 1994). The risk of significant bias effects is thus low.[94]

Trading ratios can be determined exogenously or endogenously. The exogenous definition allows the determination of the specific value one-time and *ex ante*. However, frequently changing conditions that influence the relationship between discharges and the impacts on the water quality are not integrated. An exogenous definition might thus only be reasonable in cases where changes in the relation between discharges and impacts are not significant. Homogeneity is thus only guaranteed for (almost) constant relations between discharges and their impacts on water quality.

An endogenous definition of trading ratios would avoid this problem. Variations in the relation between emissions and immission are permanently integrated into the determination process. The definition of homogeneity is thus 'deeper'. However, time lags between the observation of changing conditions and the adjustment of the trading ratios must be avoided. Only timely re-adjustments of the trading ratio can guarantee ecological effectiveness. The endogenous definition of trading ratios causes higher transaction costs for both sources and the environmental authority as they need to permanently adopt to changing conditions (see Chapters 4 and 5).

The trading ratio described above is an *impact* trading ratio. It reflects different impacts of discharges on the ambient pollution level. Note that impact trading ratios are only relevant in the case of ambient-

92 See also Farrow *et al.* (2005).
93 See, for example, King and Kuch (2003, p. 10357). He analyses the case of uncertainty trading ratios. The discussion is transferred to the case of impact trading ratios.
94 Uncertainty trading ratios set to integrate the risk in the effectiveness of the measure are more complicated to determine. If the impact of reduction measures were clear, no uncertainty trading ratio would apply. As the impacts are, however, uncertain, the exact determination of the trading ratio cannot be derived from a model, like a water quality model. The trading ratio is more of an approximation. If the determination does not reflect the real conditions, results would be biased.

based models. An emission-based model does not account for the resulting immission; different impacts of emissions on the water quality between trading sources do not raise a problem.

This type of trading ratio should not be confounded with the *uncertainty* trading ratio that applies if nonpoint sources are included in the trading system (Paragraphs 2.2.2 and 5.2.3). The impact of emission reductions on the water quality cannot be measured exactly. To avoid the violation of the ecological effectiveness, the trading ratio requires that a nonpoint source abates more than it sells in permits to another source.[95]

Trading ratios are thus very important in facilitating ecological effectiveness. The analysis of theoretical and practical approaches shows that trading ratios can be set endogenously or exogenously which, in turn, influences the level of transaction costs. Different types of ambient-based water quality trading systems and of trading ratios will be analysed in more detail in the theoretical and practical approaches in Chapters 4 and 5.

[95] In some of the trading programmes in the United States trading ratios vary according to the agricultural practice applied by nonpoint sources. In the case of the Kalamazoo River Water Quality Trading Demonstration Project, the application of new pollution control measures guarantees a 2:1 trading ratio; if the actual practice is far away from the accepted agricultural management practice then a higher trading ratio (4:1) applies (NIRAS, 2004, p. 29).

3 Ecological and Economic Requirements

The analysis in Chapter 2 clarified that water quality trading systems must be carefully designed by taking care of specific characteristics of rivers and pollutants. Chapter 3 will now develop the normative framework for the implementation of a water quality trading system. This chapter is based on the key criteria of ecological effectiveness and economic efficiency. However, these criteria must be adapted and specified for the case of water pollution control (Sections 3.2 and 3.3). This chapter starts with a brief discussion of principles defined by International Law which also influence the evaluation of environmental instruments (Section 3.1).

3.1 General Principles

International Law determines the general principles to be adhered to by any environmental policy introduced: the precautionary principle, the polluter-pays-principle, and the cooperation principle (Bahnsen, 2002, p. 59). Within this framework, international conventions and directives regulate the international environmental water policy.

Precautionary Principle

The precautionary principle was originally referred to in German domestic law in the 1970s and 1980s. It was subsequently integrated into a number of environmental agreements in Europe and also became common in international treaties and agreements,[96] for example, in the Rio Declaration on Environment and Development (1992). The binding character of this principle has changed over time; Marchant and Mossman state that "In every jurisdiction in which it has been adopted to date, the precautionary principle has evolved from policy guidance to a binding legal rule" (Marchant and Mossman, 2005, p. 7).

A final and standardised definition of the precautionary principle does not exist. This principle has an anticipatory and preventive character and becomes relevant based on scientific uncertainties (Bahnsen, 2002, pp. 59-61).[97] In the Agenda 21 (35, 3) the United Nations Conference on Environment and Development (UNCED) defined the precautionary principle as follows: "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".

The precautionary principle requires an *ex ante* evaluation of all activities that could potentially damage the environment. The probability of damage is relevant; not the evidence. This signifies a reversal of the burden of proof: it is assumed that all activities or substances are harmful until proved harmless (Bahnsen, 2002, p. 60).[98]

One way to incorporate the precautionary principle in permit trading systems could be the integration of safety margins: in air pollution control, the US Environmental Protection Agency (US EPA) incorporates an additional margin of safety when setting the quality standards "to account for health

[96] The European Union incorporated the precautionary principle into common law in the 1992 Maastricht amendments to the European Treaty. In this treaty, no definition or requirements of the precautionary principle exists. In 2000 the European commission published a 'communication' on the precautionary principle with a very detailed description of the principle (Marchant and Mossman, 2005, p. 25).
[97] See also Wicke 1993.
[98] Further aspects of the precautionary principle are discussed in Raffensperger and Tickner (1999).

effects that science has not yet discovered" (Marchant and Mossman, 2005, p. 4). The adaptation to water quality trading systems would mean that the water quality standard would be determined more stringently than estimations and evaluations require, in order to ensure that the standard would not be exceeded even in the event of something unexpected happening.

While the precautionary principle seems to be intuitive, it faces two basic problems: firstly, the precautionary principle may result in too strict constraints being forced on the production process, this could hamper certain activities or further technical developments. Secondly, the determination of risk and uncertainties related to certain activities is very complex and costly. Section 3.3 clarifies the relevance of this principle for the economic evaluation.

Polluter-Pays-Principle

The polluter-pays-principle has often been interpreted as complementary to the precautionary principle. While the precautionary principle seeks to avoid damages *ex ante*, the polluter-pays-principle defines who caused existing damages and who is responsible for abatement (Bahnsen, 2002, p. 61). The polluter-pays-principle can be traced back to the *Guiding Principles Concerning the International Economic Aspects of Environmental Policies,* developed by the OECD (OECD, 1972).[99] A shorter definition is given by the European Environment Agency (EEA): "The principle that those causing pollution should meet the costs to which it gives rise."[100]

The definition seems to be intuitive and clear. In reality, the implementation of this principle can be very complex and costly depending, for example, on the number of polluters (Heister *et al.*, 1991, p. 37). High numbers of polluters and difficulties in assigning individual responsibility for pollution may prevent the polluter-pays-principle being implemented in its 'truest' form. It is not always clear whether the producer or the consumer, e.g. in the act of consuming the product, should be responsible for the pollution and thus for the abatement.[101] As a result, the polluter-pays-principle sometimes does not burden the polluter directly but those who are the easiest to charge.

Furthermore, the scope of the polluter-pays-principle can be very different;[102] direct as well as indirect costs (administrative costs, preventive, consequential costs…) may be integrated and burdened to the polluter.

Bahnsen shows that the polluter-pays-principle applies mostly to intragovernmental environmental problems; often, it does not apply to the internalisation of international external effects (Bahnsen, 2002, p. 63). This would, however, be a consistent and necessary extension of the polluter-pays-principle. The case of water management for international rivers shows the relevance of the application of the polluter-pays-principle; this, in turn, requires a stringent international cooperation.

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[99] "The principle to be used for allocating costs of pollution prevention and control measures to encourage rational use of scarce environmental resources and to avoid distortions in international trade and investment is the so-called 'polluter-pays-principle'. This principle means that the polluter should bear the expenses of carrying out the above mentioned measures decided by public authorities to ensure that the environment is in an acceptable state. In other words, the cost of these measures should be reflected in the cost of goods and services which cause pollution in production and/or consumption. Such measures should not be accompanied by subsidies that would create significant distortions in international trade and investment" (OECD, 1972, 1.A.a),4.).

[100] See `http://glossary.eea.eu.int/EEAGlossary/P/polluter_pays_principle` (January 2007).

[101] More general, Coase argues that both the polluter as well as the person injured by his activities can be responsible for the externality (Coase, 1960). He assumes that both parties compete for scarce resources. Coase thus diverges from the common interpretation of the polluter-pays-principle.

[102] The International Environmental Law does not define the scope of the polluter-pays-principle.

Cooperation Principle

The cooperation principle stipulates cooperation not only between governments but also at a lower level (Bahnsen, 2002, p. 63). Members of the industry, Non-Governmental Organisations and the general public should be involved in environmental concerns. The cooperation principle comes as a result of the conviction that international environmental resources need to be protected together, not at the cost of each other. At the same time, governments must avoid national activities that lead to environmental damage outside their own territory (Durth, 1996a, pp. 90-93). This particularly applies to the use of common resources, like transboundary waters, which need to be coordinated in order to accommodate different user interests.[103]

The cooperation principle requires the exchange of data and information, scientific knowledge and practical expertise in administrative, and legislative and technical matters, which are crucial to cooperation.

[103] Again the linkage between the cooperation principle and the polluter-pays-principle.

3.2 Ecological Effectiveness

3.2.1 The Criterion

The criterion of ecological effectiveness in its original sense asks whether an instrument is able to achieve the predetermined ecological objective and to avoid hot spots.[104] The definition process used to find the objective is not of concern, only the compliance. Permit trading systems are in general assumed to be characterised by ecological effectiveness: the predetermined objective is fixed by the total number of allocated permits and cannot be exceeded.

Permit trading systems can only be ecologically effective if traded permit units are homogeneous in their value (in time and space for specific river and pollutant characteristics). The system should guarantee that the increase in immission by a purchasing source is equal to the decrease of immission through the reduction of the selling source in order to avoid hot spots. This is only the case if traded units are homogeneous.

The 'depth' of the term homogeneity in turn depends on the specific ecological conditions. For uniformly-mixed pollutants in the air, e.g. CO_2, permits with constant absolute emission values would be sufficient to guarantee homogeneity of traded entities; equivalence of traded entities is thus realised. Permits in water quality trading systems are generally defined in terms of water quality (immission). As a result, traded entities must be homogenous in their impact on the water quality. Very different combinations of discharges (location, time) can be found behind the same result in terms of immission. It is thus more complex to guarantee homogeneity of traded entities. The following paragraph shows that the terms of water quality and immission cap need to be determined carefully in order to achieve homogeneity.

We will see that the complexity of the homogeneity requirements varies with the level of differentiation, i.e. the dimensions, of the ecological objectives. If ecological objectives lead to a highly differentiated definition of the water quality standard, traded units need to be defined with respect to this differentiation in order to guarantee the complete homogeneity. The next paragraphs will evaluate central aspects defining the determination of ecological objectives (and thus the requirements for the fulfilment of homogeneity).

3.2.2 The Objective: Ecological Dimensions

Definition of the Geographical Scope

A water quality trading system for river basins may be organised at different levels (level of organisation). Such a system could, for example, include all concerned entities within the borders of a national state (single state level). The instrument also could apply only to parts of a national state, i.e. to

[104] Michaelis (1996, pp. 35-36) introduces an additional aspect of ecological effectiveness. Not only the achievement of the predetermined objective is relevant, but also the pace of the ecological impact. It is assumed that the implementation of a permit trading system in general in the short term fulfils the ecological objective if sanction mechanism ensure the compliance of all sources.

regions or other administrative entities (intrastate level).[105] The instrument could apply to a group of states, for example, to the European Union (interstate level). In the special case of river management, however, the organisation should tend to adhere to other aspects rather than administrative ones.

The geographical or natural structures are of great importance for rivers, in particular, as the sustainable and effective environmental management of a river can only be guaranteed by subjecting the entire river to the management. From an ecological point of view a harmonised river management would be desirable. Differing ecological goals between countries which could hamper the achievement of specific ecological objectives could thus be avoided.

From an economic point of view, separated river management risks provoke free rider behaviour and thus lose effectiveness and efficiency.[106] In the case of national organisation the advantages of a permit trading system cannot be exploited. Positive results from the river management in one country could be overcompensated by the mismanagement of other parties. Only a (transnational) permit trading system for the entire river would hinder a second state in preventing the effectiveness of a national permit trading by insufficient river management on their territory. A strict and well coordinated cooperation between all countries involved would lead to a sustainable and adequate river management according to the polluter-pays-principle.[107]

Europe is a good example and shows why transboundary river basins are of such great relevance. The Netherlands represent an extreme case: 88 percent of the available water resources originate in neighbouring countries (Strosser *et al.*, 1999). The water quality of these bodies of water in the Netherlands is thus highly dependent on the environmental activities in other states. But even in Portugal, Luxembourg or Germany, inflows account for more than 40 percent of the available water resources. These data illustrate that water management should be oriented at entire river basins; regulations at the regional or national administrative level cannot reflect these interdependencies.

Furthermore, the criterion of cost-effectiveness becomes relevant.[108] Hecht and Werbeck (2006, pp. 195-201) supplement damages and abatement costs in the upstream-downstream model illustrated by van Mark *et al.* (1992) (Section 2.4). A cost-effective solution is achieved when the given water quality standard is reached at least cost. A comparison of marginal abatement costs becomes thus relevant. Hecht and Werbeck (2006) additionally introduce the case of a transboundary river with a national border between the discharge points U and D (Figure 2-3). To achieve cost-effectiveness, marginal abatement costs need to be compared between sources, even if they are located in different countries. Sources with lower marginal abatement costs should reduce their emissions first. This could be achieved by trading activities. To exploit cost saving potentials, cooperation between both countries is necessary. At the same time, a permit trading for entire rivers, i.e. the inclusion of all relevant participants, allows a maximum of participants, which can result in a higher efficiency. A permit trading system for the entire river basin would realise the highest cost-savings: all sources located at the river would be integrated in the system. Abatement activities would take place at the source with the lowest abatement costs.

[105] In Germany this would refer to the Federal States ('Länder').
[106] The economic criteria will be discussed in more detail in Section 3.3.
[107] Cooperation should be guaranteed by setting the right incentives and/or sanction mechanisms. Bahnsen (2002, pp. 136-169) shows in which cases countries may have an incentive not to cooperate, although they agreed on cooperation in a contract. In some cases a sort of exogenous shock provokes international cooperation. This has been the case for the Rhine River. Too high transboundary pollution has forced the countries to cooperate (Durth, 1996a).
[108] The criterion is discussed in more detail in Paragraph 3.3.2.

Although cooperation was not explicitly required before, international commissions have already been implemented to organise the water management of rivers in an integrated fashion. As a result, international commissions already exist for most of the transboundary rivers in the European Union and cooperation between countries is already high.[109]

Instream Flow Need versus Endpoint Constraint

An additional specification of the ecological objective must be made. The objective can be defined either for the river itself (instream flow needs) or for the body of water into which the river flows (endpoint), or for both.

A policy designed for at the river itself (instream flow needs) can determine the water quality level with respect to the river only. Traded entities would be homogenous with respect to the impact on the river itself. In this case, the load transferred from the river into the receiving body of water and the resulting immission in this body of water is ignored. However, it does not seem logical why one should formulate a certain ecological goal for rivers and not for the receiving body of water. At the same time, the ambient pollution level in the receiving bodies of water depends heavily on the ambient pollution level of the rivers flowing into it.[110] Furthermore, the emission load of rivers can – even if the quality standard is not violated – endanger the quality of the receiving body of water. Even if the concentrations in the river are definitely lower than allowed, the emission load (per year) reaching the sea might be quite high. Furthermore, when passing from the river into the sea the water is no longer a flowing body of water in the original sense. In consequence, these loads could cause higher concentrations due to lower streaming activities.

If an endpoint-oriented objective applies, the ecological goal for the river must be defined in connection with the standards for the receiving body of water. In some cases, this may require stronger emission caps for the river than under a pure river-oriented (instream flow needs) approach. Homogeneity would then be defined with respect to endpoint conditions.

A combined approach would be the adequate solution. As a first step, the water quality standard for the receiving body of water must be defined. This standard should be transformed into a total emission load for the receiving sea. As a last step, this emission load has to be transformed into corresponding caps for the river concerned. Additionally, specific water quality requirements for the river (in space or over time) must be taken into account. Water quality standards need to adequately reflect the ecological objective (instream flow needs versus endpoint constraint) in order to allow for the homogeneity of traded entities.

In this context, Kroiss *et al.* (2003) analyse the example of the Danube River Basin: high pollution loads transferred from the river cause eutrophication in the Black Sea. Due to high water levels even relatively low levels of nutrient concentration (immission) can cause quite high absolute nutrient loads

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[109] International Commission for the Protection of the Danube River (ICPDR, `www.icpdr.org`, December 2006); International Commission for the Protection of the Rhine (ICPR, `www.iksr.org`, December 2006) ; Internationale Kommission zum Schutz der Elbe (IKSE, `www.ikse.de`, December 2006). For the Danube River Basin, even non-EU countries actively participate in the water management.

[110] International conventions for the Protection of the Marine Environment on the European continent (OSPAR Convention for the protection of the North-East-Atlantic (OSPAR Commission, 2001); Helsinki Convention for the Baltic Sea (HELCOM, 2004); Barcelona Convention for the Mediterranean (UNEP, 1995), and the Bucharest Convention for the Black Sea (Black Sea Commission, 1992)) underline the interdependencies between the quality of the receiving body of water and the quality of the river flowing into it. For further information on the development of these Conventions, see Aubine and Varone (2004).

per year. Even good water quality in the Danube River can thus cause eutrophication in the Black Sea.[111] Therefore, Kroiss *et al.* (2003) explicitly postulates an endpoint oriented approach.

Also the Final Report of the *daNUbs* project shows that the emission load of the Danube River causes most of the eutrophication in the Black Sea. Therefore, the report also requires a cap to be set for the Danube River in connection with the ecological goal for the Black Sea (IWQWM, 2005). Studies carried out by the *Danube Regional Project*[112] about pollution control instruments for the Danube River Basin also demand that the ecological objective is formulated with orientation to the Black Sea (NIRAS, 2004).

'Water Quality' and the Differentiation of the Immission Cap

One could formulate a single, identical immission cap for the whole river basin; but this might not be reasonable in most cases. The following discussion shows that in a more realistic world, different immission levels in space and/or over time can be required due to several aspects.[113] In consequence, the assessment of the fulfilment of the ecological effectiveness criteria is more complicated as well, because the 'depth' of homogeneity increases.

It is crucial for the effectiveness of a water quality trading system that the traded entities are equal in their value, i.e. homogeneity of traded entities.[114] To evaluate the homogeneity of traded entities, it is important to define the term 'water quality', the traded unit in an ambient-based system. Consequently, the trading system must ensure that traded entities are equivalent in their impact on the quality of the body of water (Michaelis, 1996, pp. 128-131).[115] Thus, one unit of a permit can, depending on the pollutants' and the river's characteristics (emissions versus immission), represent different amounts of emissions in order to guarantee the same ambient pollution level at different points in space or time (Chapter 2). If one source purchases a permit valued at one unit, the system must ensure that the additional discharge allowed to the purchasing source and the discharge avoided by the selling source are equivalent in the impact on the water quality (over time and in space) even if sources introduce their discharges at different locations and at different points in time.

To evaluate compliance with the homogeneity requirement, the term 'water quality' needs to be defined in a standardised and comparable way. The literature comes to the conclusion that 'water quality' can only be defined in connection with the water use. The 'water status' builds the basis for the definition of water quality. While the term 'water status' is a neutral one which describes the concentration of certain pollutants, physical, chemical, biological and other characteristics of a specific body of water, the term water quality reflects the use of the water and thus judges the water status depending on its suitability for a given use.[116] Indicators are used to define the adequate status of the water for each use, i.e. the water quality. The same water status can signify a higher or lower water quality depending on the intended water use. To reach the same water quality, water designed for drinking water extraction would ask for a higher water status than water designed for bathing use. Water quality is thus a relative term which depends on the defined use of a body of water.[117]

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[111] For a detailed analysis of nutrient discharges into the Danube River with different water quality models, see Kroiss *et al.* (2003).
[112] For further information on the project and its aims, see `http://www.undp-drp.org/drp/index.html` (January 2007).
[113] Paragraph 3.3.2 shows that this differentiation is also required for economic reasons.
[114] See also Huckestein (1996) and Michaelis (1996, pp. 128-131).
[115] Water Quality Models identify individual dispersion functions and dispersion coefficients; they build the basis for the determination of the permits to be equivalent in the impact on the water quality (see Section 2.3).
[116] For the determination of the water status (biological versus chemical methods), see Baur (1998).
[117] For more details, see Doetsch and Pöppinghaus (1985).

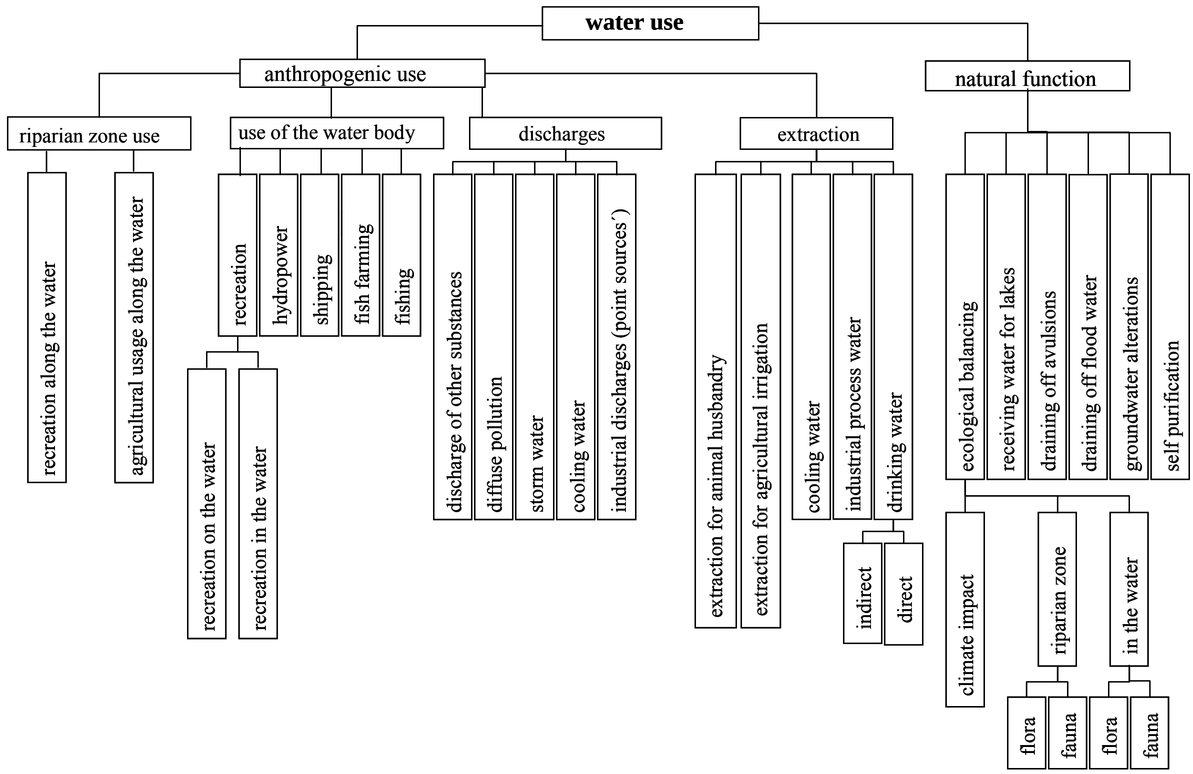

Figure 3-1: Different Types of Water Use
Doetsch and Pöppinghaus (1985, p. 297, own translation)

Forms of water use are manifold (Figure 3-1). They range from the one end of the spectrum, drinking water extraction, to the other end, shipping which permits lower water status without being adversely affected. Uses such as cooling water abstraction or recreation areas lie between these two extremes. Besides this anthropogenic use of water, the natural function of water becomes relevant. Biological processes and/or flora and fauna living within a body of water might demand for different maximum immission loads at different locations thus influencing the water quality requirements.[118] Each type of use requires its own indicator system to quantify the needed water status in order to reach a certain water quality.[119]

Furthermore, different water use aspects may ask for an adjustment of the immission cap, i.e. the predetermined water quality, for specific periods of time. Natural functions, for example, could rely on specific water quality levels for different periods of time: due to biological processes, a particular level of pollution should not be exceeded for a certain period of time. This may be relevant for periods of fish ascents or spawning processes in certain regions or for other reasons as a result of the water use. If relevant, specific water quality standards need to be defined for a certain amount of time. Characteristics of the river and its pollutants would also influence this decision (Chapter 2).[120] It might

118 This is a central difference between air and water pollution control. For non-uniformly-mixed pollutants in the air it would not be possible to define different standards for different zones. Dispersion of emissions cannot be controlled with such precision.
119 The resulting water quality is, of course, influenced by the degree of completeness and the accuracy of the indicators (van Mark *et al.*, 1992, p. 44).
120 Generally, it would be possible to allow a weaker water quality level in winter than in summer, if, for example, water use in winter differs from water use in summer. In winter, the lack of bathing activities might be relevant; thus a lower water quality would not matter. Lags in the impact of discharges on water quality need, however, to be taken into account; even if the water quality is lower in winter, the higher water quality needs to be ensured in summer.

also be required by law or by other, more ecological, reasons to reduce the immission cap over time. In this case, the emission caps must, of course, to be adapted accordingly.[121]

In reality, the definition of perfect equivalence of traded units for the case of rivers and their pollutants is often difficult to achieve. However, thanks to modern technologies and methods, estimations (approximations) are a solid basis on which to define (practically) equivalent trading units. Theoretical and practical approaches (Chapters 4 and 5) illustrate, which elements can be introduced in a permit trading system in order to guarantee homogeneity in the impact of trading units.

Flexible Adjustment of the Emission Cap

Under an ambient-based permit trading system immission caps need to be converted into emission caps in order to make the system operable. On the one hand, even for a constant immission cap (space and time) specific conditions might ask for a permanent adjustment of the emission caps. Different water levels or temperatures over time could ask for changing emission caps, other things being equal. The relation between discharges and the resulting water quality would thus change (Chapter 2). On the other hand, changing immission caps in space or over time would require the adequate and flexible adjustment of the emission cap.

Only a well-defined design of a permit trading system enables these adjustments and thus enables the specific 'depth' of the homogeneity. Specific approaches to integrate these are discussed for different trading models (Chapters 4 and 5).

Water Quantity Aspects

Section 2.4 has shown that water quantity aspects (exogenous or endogenous) may significantly influence water quality levels. It is thus important that a water quality trading system adequately considers the water quantity aspects wherever necessary in order to guarantee the homogeneity of traded entities. In cases where alteration of water quantity is not significant or does not significantly influence water quality, no integration in the trading system is necessary. While this has to be examined for any specific trading system to be implemented, the analysis of theoretical and practical models shows how trading systems could incorporate changes in water quantity.

[121] This could be done by reducing the value of the permit over time (in percent or in absolute terms) or by distributing a lower amount of permits in a new period.

3.3 Economic Criteria

The economic criteria offer the opportunity to compare different instrument designs, each of them fulfilling the ecological objectives. This study uses the following common criteria: cost-effectiveness, dynamic incentives, and level of transaction costs in order to judge the economic efficiency. On paper a permit trading fulfils all of these common criteria. But we will see that a permit trading system needs to be carefully constructed if it is to fulfil these criteria in the 'real world' with all the characteristics discussed in Chapter 2.

Additional aspects are relevant. The design of an efficient and effective trading system is useless, if the instrument cannot be implemented due to reasons of competitive distortions, practicability and/or enforceability. These aspects are particularly relevant for the practical approaches.

This chapter builds the basis for the evaluation of theoretical and practical approaches (Chapters 4 and 5). We explicitly abstain from formulating conditions that have to be fulfilled for any permit trading system to work.[122] In this chapter we rather want to formulate the requirements of a permit trading system particularly relevant for the application for rivers.

3.3.1 Efficiency

Section 3.1 analysed parameters influencing the setting of ecological objectives. The question arises which water quality level needs to be determined in order to be efficient from an economic point of view. An economist would not recommend a pollution-free environment as an ecological goal, because this would mean not to use environmental resources at all. The question on the determination of an optimal (efficient) level of pollution (or quality) arises. The neoclassical welfare defines the optimum as follows: the pareto-optimal level of pollution is attained when the marginal environmental damages (MD) of the pollution equal the marginal abatement costs (MAC), both depending on the emission level E (Figure 3-2).[123] At this point, the sum of damage costs and abatement costs is at the minimum ($E*$).

[122] For this, see David (2003) and Kraemer *et al.* (2003, p. 25).

[123] For more details, see Kemper (1993, pp. 10-15), Hecht and Werbeck (2006, pp. 71-80) or Tietenberg (2003, pp. 338-342).

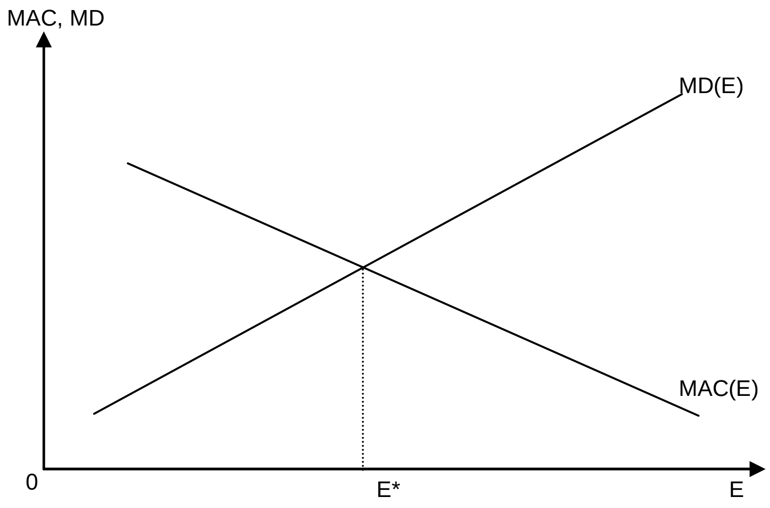

Figure 3-2: Optimal Level of Pollution

This theoretical approach, however, shows some deficits for practical implementation. Firstly, the determination of the marginal damages (MD) is still complex and problematic and thus makes the definition of the pareto-optimum difficult. Also the determination of the marginal abatement costs curve (MAC) is nearly impossible: this would demand for information about all adoption and abatement potentials and this for all relevant types of pollution.[124] While the static view is already very complex, the dynamic measurement over time is not feasible at all (Tietenberg, 1980, pp. 481-482). Technological progress, for example, would change the conditions and the marginal cost curves.

Due to these measurement and information problems the exact definition of the pareto-optimum (i.e. $MD = MAC$) is not feasible. Therefore, in the majority of cases, the definition of the quality standard is a political one. Politicians should, of course, consider available information about costs (damages) and benefits of different pollution control levels.[125] According to the discussion above, the rather arbitrary setting of the ecological goal by politicians will only by coincidence lead to an optimal pollution control level in the neoclassical sense of a pareto-optimum.[126]

3.3.2 Cost-effectiveness

Tietenberg (2006, pp. 66-68) refers to different studies that show for the case of air pollution that the potentials for cost saving in permit trading systems are generally high.[127] One goal of the introduction of a market instrument is to save costs with respect to other instruments through a cost-effective solution. The definition of the term of cost-effectiveness follows.

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124 Additionally, asymmetric information concerning the cost curves can occur and influence the results; see Fees (1998, pp. 269-276).

125 For more details about cost-benefit-analysis for environmental policy and its limits, see, for example, Wicke (1993) Perman *et al.* (1999) or Feess (1998, Chapter 12).

126 The instrument of a permit trading would thus be chosen to reach a given quality standard in a cost-effective manner. It does not by itself determine the ecological objective by itself. Although this instrument is a marketable one, a political determination of the ecological objective is necessary.

127 See also Bonus (1984, pp. 131-140) and Hahn and Hester (1989, pp. 391-396).

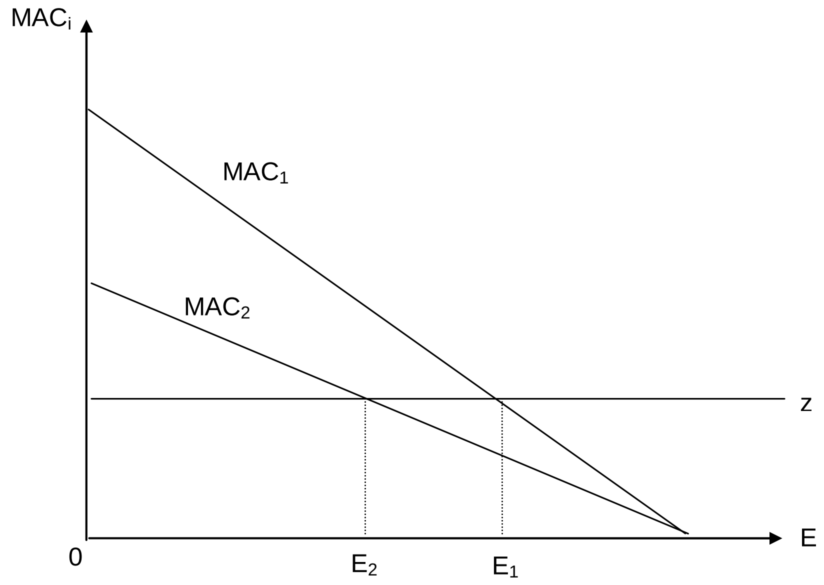

Figure 3-3: Cost-effectiveness

Definition and Specification

Paragraph 3.3.1 has shown that, the definition of the (pareto-) optimum is not often feasible in reality. For these reasons the criterion of cost-effectiveness has been accepted instead. The cost-effectiveness criterion requires that a predetermined ecological objective will be achieved at least cost. Only if the ecological objective set by the politicians happens to minimise the sum of marginal abatement costs and marginal damages, will the cost-effective solution also be efficient.[128]

Figure 3-3 shows that in theory a permit trading system automatically guarantees cost-effectiveness.[129] Each emitter ($i = 1,2$) compares his marginal abatement costs MAC_i and the permit price z. As long as the permit price exceeds the marginal abatement costs, the emitter would abate emissions. If the marginal abatement costs exceeds the permit price z, the emitter would cover his emissions with permits which he has to purchase on the market. The emitter thus produces at the point where $MAC_i = z$. This calculus holds true for all emitters; all emitters would thus produce with $MAC_i = z$. The absolute levels of emissions are different for source 1 and 2 (E_1; E_2). The equivalence of the marginal abatement costs between sources indicates the cost-effectiveness of an instrument.[130] Moreover, one can show that cost savings through trading activities increase with higher differences in the marginal abatement costs curves.

The original definition of cost-effectiveness, in the sense that marginal abatement costs are balanced only holds true for the uniformly mixed pollutants for which the impact on the ambient pollution level is independent of time, space and the source of emissions. For non-uniformly mixed pollutants the impact of emissions on the ambient pollution level varies with the location (and point in time) of discharge (Chapter 2). Marginal abatement costs for the reduction of emissions are thus equal after

[128] See also Hung and Shaw (2005, pp. 85-86) or Tietenberg (1995).
[129] For an alternative illustration, see Tietenberg (1985, p. 20).
[130] It is assumed that no institutional or other circumstances prevent the system from coming to the optimum.

having been weighted using the dispersion coefficients; the 'marginal *immission* abatement costs' with respect to a specific point are ideally the same (Cansier, 1996, pp. 71-74).[131]

Since each source of pollution makes an individual decision about whether to purchase permits, or to abate and sell permits and since the achievement of the predetermined objectives is obligatory, no cost-benefit analysis is necessary. Each source compares the individual abatement costs and the available price for permits and makes an autonomous decision regarding implemented measures. This approach can thus avoid the information problem (Paragraph 3.3.1): The environmental authority is not dependant on knowing the individual marginal abatement cost curves anymore.[132]

Turning to cross-country cost-effectiveness, the discussion of the upstream-downstream model in Section 2.4 clarified that close cooperation between different nations pertaining to a transboundary river is indispensable. A common river management scheme enables the exploitation of the differences in the marginal abatement costs in different countries and achievement of the quality standard at least cost. Additionally, close cooperation in transboundary river management avoids free rider behaviour of single states located along the river which would prevent the system from being cost-effective in the end.

The upstream-downstream model further emphasised that asymmetric trading positions between sources can exist (Section 2.4). Depending on the location, trading potentials may be different for sources. The first source in the river (upstream) cannot profit from any reductions upstream. At the same time, no source within the river benefits from abatement measures at the last source (downstream). This reduces for some sources the number of potential trading partners and prevents the system from exhausting all cost savings by means of trading activities. This aspect will be discussed in more detail within the specific model analyses. Specific design elements can be integrated to avoid this asymmetry.

The criterion of equalised marginal abatement costs is especially suitable for the evaluation of theoretical approaches (Chapter 4). However, for practical approaches it is difficult or not feasible to check equalisation of marginal abatement costs.[133] It is thus reasonable to use additional indicators to evaluate the functioning of a permit trading system in practice. Common criteria are the volume of trading activities and the price of permits.[134]

Definition of the Immission Cap

It should be noted that the differentiation of the immission cap over time and/or in space is not only very important for the definition of the ecological objectives (Section 3.1) but also from an economic point of view.[135] A non-differentiated immission objective would stipulate a very strict determination of the ecological objective. It must take account of the characteristics of all regions and at all possible

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[131] See also Førsund and Nævdal (1998, pp. 406-407).

[132] See also Tietenberg (2006, p. 27). This holds for point sources, but not unrestricted for nonpoint sources. Due to measurement problems for nonpoint sources it is not possible to verify the achievement of a predetermined immission/emission goal or to individually allocate responsibility. If participation in a permit trading system is assumed not to be possible, the environmental authority can carry out cost-benefit analyses to identify the best measures, i.e. the less cost-intensive measures. This analysis will, however, be accompanied by information and transaction cost problems that are avoided in a well-functioning permit trading system.

[133] For limitations of methods that seek to determine cost-effectiveness, see also Tietenberg (2006, Chapter 3).

[134] In theory, actual trading activities should be compared with potential trading volumes. Again, the determination is characterised by a lot of measurement problems (see Fromm and Hansjürgens, 1998, pp. 155-157).

[135] See also Kemper (1993, 88).

points in time; thus the most strict constraint decides the definition of the objective that applies, without further differentiation over time or in space, to the whole system (E_C or Q_C, Figure 2-5, p. 22). Such an objective should, for example, integrate the requirements of the region with the water use that demands the lowest ambient pollution level, e.g. drinking water, for the whole river in order to guarantee that the objective is not violated anywhere. This would, however, cause too high marginal abatement costs, because the control requirement would be higher than necessary in many parts of the river or in some periods of time. The efficiency of the system is affected in a negative way.

Differentiated instruments set differentiated incentives for the licensees. The consideration of different water quality objectives in time and space influences the production decisions as they influence the abatement costs in time and in space. Weaker water quality standards or emission caps would effectively reduce abatement costs. Depending on the water use, water quality standards could be lower thus allowing for higher emission caps and lowering abatement costs; in winter months, intake capacities and thus the permitted amount of emissions could be higher than in summer months, even if the same water quality standard holds true, abatement costs are lower. Sources would choose a production location where abatement costs are lower or vary the production processes in time in order to profit from lower abatement costs.[136] Polluters are able to adapt their decisions about emissions to the design of the instrument and thus reduce their abatement costs. New firms would prefer to settle in areas or emit in periods of time with weaker emission caps to avoid abatement costs. Environmental aspects are thus integrated in firm decisions.

One of the economic objectives of the differentiation is thus to bring abatement activities to the place and the time where it is less costly in order to achieve the least-cost solution (cost-effectiveness). Without this differentiation, costs would be higher than necessary.[137]

Several studies come to the conclusion that a stronger spatial differentiation in emission trading systems would lead to higher savings in total marginal abatement costs.[138] However, a trade-off exists between the decrease of the total marginal abatement costs and the increase in transaction costs.

Non-degradation Principle and Non-binding Caps

Not all types of spatial or local differentiation will lead to a cost-effective solution; the application of the non-degradation principle would, for example, affect the efficiency of the trading system in a negative way. In this case, trading potentials cannot be exploited; marginal abatement costs are fixed at a too high level.

The non-degradation principle does not permit a reduction in water quality in any part of the river, even if this quality level is higher than the water quality standard would ask for. Even in the event that the quality of the water is better than required at a specific point, so that additional emissions would not contravene the quality standard, no increase in concentration is allowed. The application of this principle may affect the cost-effectiveness of the system in a negative way (see below). Setting the safety margins in water quality standards at too high level would have a similar effect. They do not allow

[136] See Kemper (1993, p. 91).
[137] See also Atkinson and Tietenberg (1982).
[138] The comparison of cost savings for different types of instruments is not a subject of this study. Empirical studies mostly confirm theoretical results. Cost savings are higher for marketable instruments like a permit trading than for command-and-control instruments (see, for example, Tietenberg (1985, Chapter 3)). Faeth (2000) shows how cost-effectiveness can be compared for different systems (Faeth, 2000, pp. 31-37). NIRAS (2004, pp. 59-61) discusses the methodical problems of empirical studies on the comparison of costs savings for different instruments.

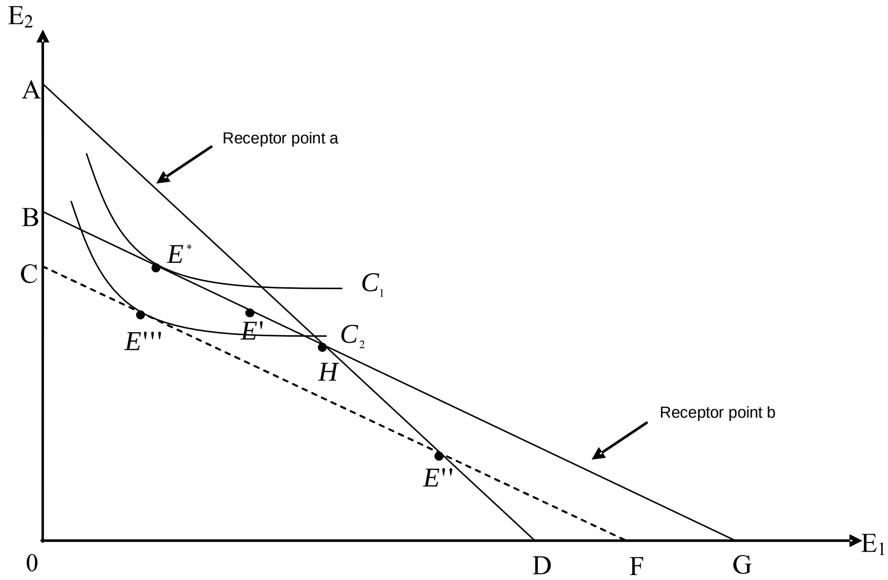

Figure 3-4: The Non-degradation Principle
According to Krupnick *et al.* (1983, p. 239)

exploitation of all cost saving potentials as they define a higher quality standard than actually predetermined.

Figure 3-4 illustrates this fact: the axes measure the emission E_1 and E_2 of sources 1 and 2. C_1 and C_2 are two isocost curves for pollution abatement costs. Note that a higher curve coincides with lower total pollution abatement costs.[139] The lines AD and BG indicate the pollution constraint associated with receptor points a and b. The points on these lines correspond to combinations of E_1 and E_2 fulfilling the formulated constraint of the receptor points. The slope of each line equals the ratio of dispersion coefficients; it indicates by how much source 1 can increase its emissions when source 2 decreases its emissions by one unit with no change in the concentration (immission) at receptor point a (or b). The area $0BHD$ indicates the combinations for E_1 and E_2 for which the pollution constraints are fulfilled at both receptor points. $E*$ equals the least-cost solution; this is the point where the ratio of marginal abatement costs coincide with the slope of the BG curve, i.e. the ratio of the transfer coefficients. $E*$ therefore indicates the cost-effective solution.

Imagine that the environmental authority sets an initial allocation of permits E'. Following the slope of the constraint curve and the isocost curve, source 2 would find it profitable to purchase permits from source 1. As the constraint of the receptor point b is the binding one in E', the slope BG will indicate the exchange rate. Both sources profit from the trade until $E*$ is achieved. For this case, i.e. with the starting point E', we find that the market equilibrium equals the least-cost solution.

This would be different if the environmental authority confers in the initial allocation permits corresponding to E''. Source 2 would again find it profitable to purchase permits from source 1 in order to increase its emissions. However, taking the non-degradation principle into account, the constraints are now fulfilled within the area $0CE''D$. The dotted line CF thus determines the exchange rate between sources. Trading leads the sources to E''' which is inferior to $E*$. This new market equilibrium causes higher abatement costs than the cost-minimum solution in $E*$ would do.

139 A sufficient (but not necessary) condition for the isocost curves to have the desired curvature in Figure 3-4 is that both firms face a schedule of rising marginal abatement costs.

The non-degradation principle prevents movement to $E*$; trading potentials and thus cost saving potentials are not exploited.

The achievement of the cost-minimum solution thus depends on the initial allocation of permits. The environmental authority must have information on these curves when deciding about the initial allocation.[140]

This information problem becomes even more important when more than two receptor points become relevant (Krupnick *et al.*, 1983, pp. 240-241). This would introduce a third line in Figure 3-4 which by coincidence only would cross both of the existing lines at point H. If the third line does not pass through H, there is no initial allocation which would be binding for every receptor point, regardless of the non-degradation principle. In this case, the environmental authority must know $E*$, and therefore the marginal abatement cost functions of the sources, in order to then allocate the permits in such a way that the least-cost solution will be achieved.

3.3.3 Dynamic Efficiency

A crucial economic concern is the question whether a given environmental instrument sets an incentive for participating sources to install new, less polluting technologies (dynamic efficiency). New technologies can promote a more efficient use of the production factor environment or better abatement measures. Technological progress can thus lower costs of pollution control, in which abatement costs per unit of discharge decrease.

The incentive to invest in new technologies can be characterised by the cost savings resulting for innovative emitters.[141] In general, the incentive to invest in new technologies is assumed to be implemented for permit trading systems. Emission reductions lead to a surplus of permits; these can be sold on the market and thus overcompensate the expenses for the abatement technology.

The advantage of innovation is illustrated in Figure 3-5. The new technology reduces the marginal abatement costs of the concerned source (shift from MAC_0 to MAC_1). Again, the emitter compares the permit price z_0 with the MAC_i and produces at the point where MAC_i are equal to the permit price z_0. Before the technical innovation, the emitter thus reduces emissions and holds permits for the emissions from 0 to E' (point B). The innovation results in lower MAC_1. The same emission level E' is achieved at lower abatement costs (cost savings ABC). The emitter has an incentive to abate more: the equivalent of the permit price z_0 and the MAC_1 is realised at point D. The surplus of permits (E'' to E') can be sold on the market. The financial benefit gained by selling surplus permits ($E''DBE'$) is higher than the additional abatement costs ($E''DCE'$) by DBC. The total incentive to abate is thus indicated by $CDBA$.

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140 A similar case could result from different legal conditions in participating countries. Principally, all participants should trade with the same conditions. If, for legal reasons, the objective is set more strictly in some parts of the river basin, this would influence the scarcity of the good water quality which would be reflected in the prices. The efficiency of the trading system is hampered. Otherwise, the strictest interpretation of legal conditions in the countries concerned may apply. This, however, causes an artificial scarcity for the entire river and not only for some parts.

141 Again, it is not the subject of this study to compare dynamic efficiency for different instruments. For more information on this, see Kemper (1993, p. 165).

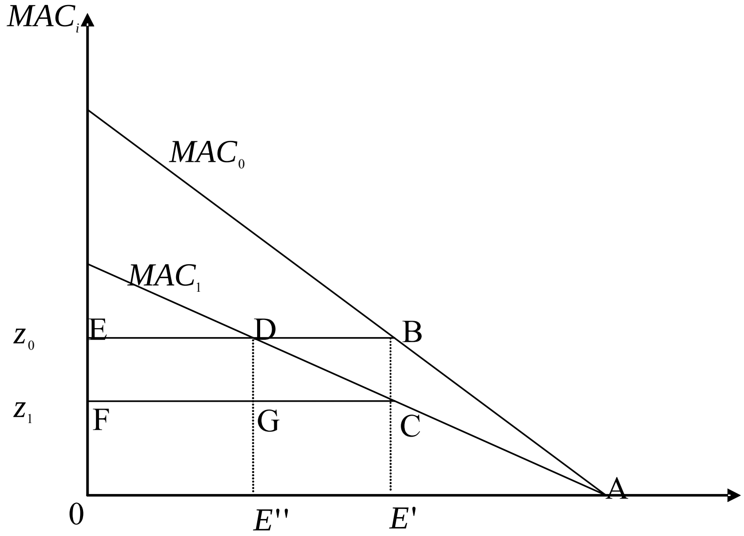

Figure 3-5: Dynamic Efficiency

If all sources adopt new technologies, the demand for additional permits and thus the price z_0 may decrease (z_1) as the supply increases. This would result in a lower, but still positive, incentive to invest in new technologies. If the number of emitters is high, the innovation of a 'small' emitter would not influence the price noticeable (Kemper, 1993, p. 170). Additionally, this effect might be lowered or even overcompensated by the effects of economic growth or of an increasing number in emitters.[142] Finally, the total amount of permits and thus the amount of discharges, may be decreasing over time as a result of political or ecological reasons. In the long term, this would also create the incentive to innovate as a result of higher prices.

The precautionary principle may prevent a permit trading system from coming to an efficient solution. New abatement technologies must be evaluated *ex ante* and run the risk of not being allowed on the market due to a 'lack of full scientific certainty' (Section 3.1). If the *ex ante* evaluation is too costly and/or the risk of rejection is too high, e.g. due to too strict constraints, the original incentive of the permit trading system to innovate could be hampered. Additionally, a decreasing cap guided by Best Available Technology (BAT) values may lower the incentive to innovate.[143] If the immission cap is adopted according to better technologies implemented by some firms, innovative activities could be hampered, as emitters do not want to accelerate the decreasing of the cap.[144]

[142] Sometimes, governmental intervention is required assuming that additional purchases by a governmental organisation or the shortage of the amount of permits could solve the problem by increasing prices. Governmental interventions should, however, be weighed up very carefully to avoid further price bias.

[143] Integrating the Best Available Technology in the definition of a benchmark in order to define the emission or immission caps in a comprehensive cost-benefit-analysis is not necessarily inefficient. Under such a defined cap, firms would be free to decide how they would reach the predetermined objective. However, a Best Available Technology regulation in form of a standard can set 'wrong' incentives. The incentive of sources to invest in innovative technologies is biased as any improvement would be obligatory for all firms in future. Sources have no incentive to use other technologies or to abate more than required by the Best Available Technology. Neither cost-effectiveness nor dynamic efficiency can be achieved. For a comprehensive discussion, see Perman *et al.* (2003, pp. 213-216). Further discussion shows how this argumentation can be transferred to the case of nonpoint sources (Paragraph 5.2.3).

[144] Furthermore, the incentive to invest in new technologies depends on the relative stability of the instrument (and its design). If emitters are confronted with frequently changing conditions and thus uncertainties, incentives for innovations are reduced. Regulations that are expected to be in place over time will create stronger incentives *ceteris paribus* (Field, B.C. and Field, M.K., 2002, p. 187).

Again, it is easier to evaluate the dynamic efficiency of theoretical rather than of practical approaches.[145] The evaluation in Chapters 4 and 5 tries to identify specific elements in water quality trading systems which hamper the system in fulfilling the criterion of dynamic efficiency.

3.3.4 Transaction Costs

Transaction costs (TAC), the crucial relevance of which initially was explained by Coase (1937)[146], are defined broadly in this study, similar to Commons (1990). Not only the costs for the concrete trading action, but also the costs for the installation and the maintaining of the trading system are subsumed under the term transaction costs.[147]

Dales (1968, p. 97) already recommended the use of permit trading systems for pollution control by referring to the low level of transaction costs. He states that the "administrative simplicity of the system is certainly one of its main attractions". The environmental authority should set the framework for a trading system; market functions ensure the optimal solution. Transaction costs are assumed to be low.

However, studies of implemented trading systems underline that too high transaction costs often hamper trading activities and thus the efficiency of the system. Stavins (1995), for example, reports that different studies have discovered that low or even zero trading activities in permit trading systems are, in most cases, caused by too high transaction costs. Gangadharan (2000, p. 601) also underlines the importance of low transaction costs. For the permit trading system RECLAIM (air pollution, oxides of nitrogen and sulphur) Gangadharan finds out that the "presence of transaction costs reduces the probability of trading by about 32%".

The relevance of transaction costs on the efficiency of permit trading systems must be analysed in more detail and with respect to specific conditions of the application for rivers. The level and structure of transaction costs influence the general functioning, effectiveness and efficiency of a permit trading system. The specific design of the trading system, in turn, influences the level and structure of transaction costs. This connection needs to be recognised when implementing a permit trading system. As a first step, the influence of transaction costs on the efficiency of a permit trading system will be analysed.

Relevance of Transaction Costs in a Permit Trading System

A basic model of a permit trading system with transaction costs has been developed by Stavins (1995).[148] Transaction costs are defined broadly: the margin between the buying and selling price of a commodity in a given market. In this model, the environmental authority allocates a total of

145 There are only a few studies on dynamic incentives of water quality trading systems. A general comparison of policies in Europe and in the United States by Harrington *et al.* (2004), finds out that market-based instruments provide greater incentives. Popp (2003), for example, examines the impact of the Sulfur Allowance Program in the United States on innovation.

146 The concept of transaction costs was introduced by Coase (1937), in the theory of the firm as well as its extensions to other forms of institutional governance. However, Coase was not very explicit about what he meant by transaction cost. For the further development of the term transaction costs in the literature, see Crals and Vereeck (2005), Betz (2003) or Furubotn and Richter (2000, Chapter 2).

147 Tietenberg (2006, p. 42), for example, distinguishes between transaction costs and administrative costs. No such differentiation applies in this study.

148 Stavins originally developed this model for the case of uniformly-mixed pollutants. The general question regarding the influence of transaction costs on the trading equilibrium remains the same for non-uniformly-mixed pollutants.

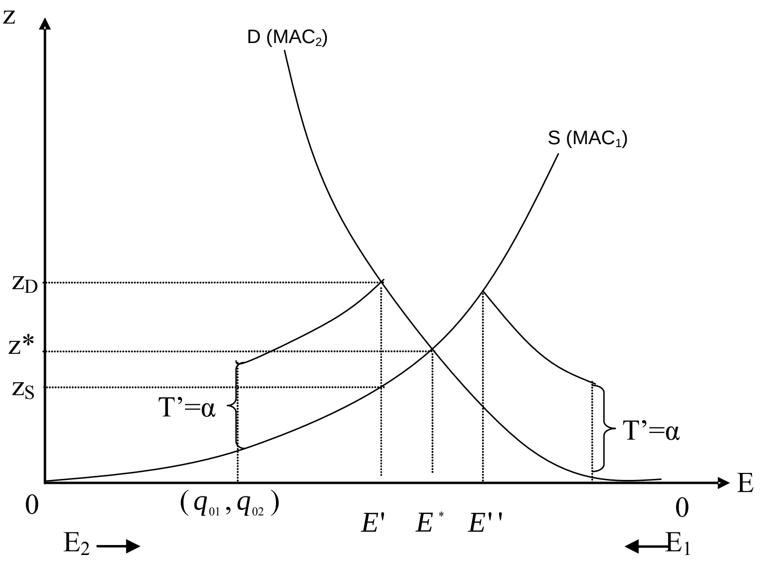

Figure 3-6: Transaction Costs in a Permit Trading System
Stavins (1995, p. 139)

emissions $\overline{E}$ [149] in form of permits, q_{0i} to each firm $(i = 1,...,n)$.[150] Permits can be traded amongst firms. E_i indicates the emission per source. Stavins defines a common transaction cost function, $T(t_i)$, with $T'(t_i) > 0$ and with $T''(t_i)$ positive, negative or zero-valued.[151] Transaction costs are thus rising with an increasing number of permits traded.[152]

Figure 3-6 illustrates the situation described in a two-source scenario. Marginal transaction costs are assumed to be constant and these costs are paid directly by the permit seller; S (MAC_1) indicates the marginal abatement costs of the seller.[153] Marginal transaction costs are added on the marginal abatement costs (T'). Trading activities lead to the pollution control allocation E' ($MAC_2 = MAC_1 + T'$). Without transaction costs the system would come to the allocation E^* ($MAC_2 = MAC_1$).

If permits are allocated to the right of the posttrading equilibrium by initial allocation, source 2 becomes the seller and thus experiences the transaction costs. The outcome of trading is the allocation E''. In conclusion, the result of trading depends on the initial allocation of permits, i.e. on seller and buyer identities.

The analysis of Figure 3-6 shows that the environmental constraint is fulfilled. Nevertheless, the existence of transaction costs changes the result: rather than balancing marginal abatement costs,

149 It is assumed that aggregate emission E is simply the sum of emissions E_i, from n individual sources. In the case of non-uniformly mixed pollutants the link between emissions and immission must be integrated additionally.

150 Only transaction costs paid by the sources are of relevance. Further analysis explicitly determines the transaction costs that the environmental authority incur. Transaction costs of the environmental authority could, however, be passed through to the sources.

151 It is assumed that $T(t_i)$ is known with certainty.

152 Only transaction costs caused by the trading activity itself is introduced in this model. General transaction costs, that are not directly linked to trading activities, but that are necessary to make the system run, are not integrated in the model.

153 Who becomes the seller or buyer depends on the initial allocation (q_{01}, q_{02}). For the initial allocation (q_{01}, q_{02}) in Figure 3-6, marginal abatement costs plus transactions costs are lower for source 1 than for source 2; source 1 would therefore abate emissions and sell permits at the market (seller); source 2 would purchase permits (buyer).

trading activities balance the sum of marginal abatement *and* marginal transaction costs. The level of transaction costs lowers permit trading activities.[154] Trading potentials are not exploited; the cost-effective solution is not achieved.[155]

Finally, one can conclude that the presence of transaction costs influences the trading results. Depending on the slope of transaction costs curves and the initial allocation of permits the final allocation will differ more or less from the optimum E^*. Total control costs are higher than under the optimum. Cason and Gangadharan (2003) confirm the results of Stavins (1995) in an experimental study. They use laboratory double auction markets in order to examine the impact of transaction costs on market outcomes when initial permit endowments differ. The results of this experimental study are consistent with Stavins' model.[156]

Identifying Transaction Costs

There are different classifications and different types of transaction costs for permit trading systems in general (Betz, 2003). When comparing the level of transaction costs for different water quality trading system designs, only those transaction costs become relevant that are different for these approaches, i.e. influenced directly by the design of the trading system. Neither transaction costs that occur for any permit trading system to be implemented, such as general planning and decision costs, nor the costs for the installation of an appropriate statutory framework or sanction costs, depend on the design. In the approach-specific analyses, the relevant transaction costs need to be identified. Additionally, transaction costs will be differentiated according to who incurs them: the participants (sources) or the environmental authority.

The goal of this study is not to identify the exact transaction costs for different trading systems. This would be an individual and complex task and one that must be implemented separately for any trading system. This study rather seeks to identify general blocks of transaction costs that arise for specific permit trading approaches. Whether transaction costs for one element or another would be higher or lower in reality will depend on specific conditions. Some specific transaction costs are broadly presented here in order to give an initial indication of what the relevant transaction costs could be like.

Independent of the allocation modus (grandfathering versus auction), the design of a water quality trading systems may, for example, significantly influence the costs of initial allocation. A trading system could seek to integrate specific characteristics of the river and its pollutants (emissions versus immission) within the initial allocation by means of the application of water quality models. This would, of course, cause higher transaction costs than a 'simple' initial allocation (Paragraph 4.3.1).

Paragraph 4.2.2 indicates that the specific design of a permit trading system may ask for water quality simulations before any potential modification of discharges is made, in order to forecast the impacts.

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154 Similar to the tax incidence result, the decreased price of the seller and the increased price for the buyer result in a financial burden. The distribution of the burden between the buyer and seller depends on the relative elasticity of the control cost curve. The burden will be greater for relatively high-cost controllers (with steeper control costs curves). This holds true, regardless of the question of who actually pays direct transaction costs.

155 Furthermore, Stavins (1995) analyses the case of increasing as well as of decreasing transaction costs. Under increasing transaction costs the initial allocation affects the result of trading activities: allocating a higher amount of permits to a source (reducing its initial abatement responsibility) would reduce the total abatement level, shifting it away from the optimum, while pushing up total control costs (Figure A-5, Appendix). In the case of decreasing marginal abatement costs, trading activities result in an allocation that is closer than otherwise to the cost-effective equilibrium, if the initial allocation moves away from the cost-effective equilibrium, (Figure A-6, Appendix).

156 Even if the level of transaction costs is an important factor for the efficiency of a trading system, Chapters 4 and 5 will show that besides too high transaction costs other elements of the design may prevent the system from being efficient.

This method is very costly. On the other hand, models could rely on real-time data (Section 5.3). Monitoring systems need to run permanently.

Chapters 4 and 5 will show that (impact) trading ratios are often essential in order to guarantee homogeneity of traded entities under an ambient-based trading system. The determination of trading ratios causes additional transaction costs for the environmental authority as well as for sources, these may be higher or lower depending on the exact definition.

Section 3.2 came to the conclusion that a differentiation of the immission cap in space and/or over time may be necessary for specific cases. This would, in turn, influence transaction costs for both the environmental authority and the sources.

Under an ambient-based trading system, water quality standards (immission caps) need to be converted into emission caps in order to make the system operable (Section 2.5). Again, transaction costs for the determination of the emission caps will vary according to the specific design.

Examination of theoretical and practical approaches (Chapters 4 and 5) will make obvious that information costs (including the costs of informing sources) depend on the specific design of the water quality trading system and are rising with an increasing complexity of the instrumental design.[157]

Generally, the environmental authority may pass transaction costs (partly or fully) on to the participants of the trading system. If this holds true, these costs would also practically be paid for by the sources. Nevertheless, transaction costs are discussed separately for the authority and sources in the analyses of theoretical and practical approaches (Chapters 4 and 5) in order to structure them in a comprehensive way.

The individual analysis of transaction costs (environmental authority and sources) for each individual model presented below also distinguishes between transaction costs that occur once in order to establish the trading system and transaction costs that arise permanently in order to keep the trading system working. Furthermore, the level of transaction costs can be directly linked to (trade-dependent) or uncoupled from (trade-independent) the actual number of realised trading activities. While one-time establishment costs are trade-independent and are not relevant for the further functioning of the trading system, permanent maintenance costs may be trade-independent or trade-dependent (Chapters 4 and 5). This differentiation allows to identify transaction costs that are relevant according to the model by Stavins (1995) which illustrates the role of trade-dependent transaction costs on the efficiency of the trading system.

For the final integration of a water quality trading system for a specific river, the existing statutory framework influences the level of arising transaction costs. In countries with a high density of water-specific regulations, specific conditions such as the intensive monitoring or similar features may already be fulfilled. The implementation of a water quality trading system would thus not cause additional transaction costs for monitoring. In countries that as yet do not have an intensive environmental policy and which are introducing the instrument from scratch, all transaction costs become relevant. The incidence of transaction costs thus needs to be defined with respect to the *status quo*. At the same time,

[157] The costs of finding a trading partner depend heavily on the market structure (exchange, clearinghouse or bilateral negotiations) (Woodward *et al.*, 2002). The adequate market structure, in turn, depends on the traded entities, number and types of participants etc. The question of the optimal market structure has to be solved for any permit trading system regardless of the specific river conditions. We thus abstain from analysing this aspect in more detail.

other elements influence the decision about where to introduce a trading system. In highly regulated countries with already high water qualities it may not be proper to introduce a new instrument (Chapter 6). Rather high transaction costs for the introduction of a water quality instrument in countries with weak regulations would not necessarily be an argument against the implementation. Any instrument implemented for water pollution control would cause additional costs.

Liability and Transaction Costs

Strictly speaking, the assignment of liability to the buyer or the seller is not a design-specific problem; rather, the question of liability rules, is relevant for all permit trading systems.[158] Liability rules, however, strongly influence the trading behaviour and thus the efficiency of the system, in addition to the direct effects of increasing or decreasing transaction costs for the sources.

A market for water quality permits differs from a market in the usual sense.[159] Trading activities on the water quality market shift the responsibility for improving water quality from the seller to the buyer of permits. Unlike on the conventional markets, the buyer of a water quality permit has no incentive to ensure the achievement of the environmental goal; it would thus remain the task of the seller to guarantee a reduction in pollution according to the amount of permits sold on the market. The seller, however, could have an incentive to over-sell permits.[160] The question then arises who is liable for non-compliance by the seller.

One generally distinguishes between buyer and seller liability.[161] Buyer liability means that it is the task of the purchasing source to guarantee adequate reduction (according to the permits sold) by the seller. The buyer is thus responsible for the validity of the permit and is obliged to control compliance and to detect over-selling. If sellers are found out to be in non-compliance, the purchased permits become invalidated.

While buyer liability might be reasonable for permit trading systems between states,[162] it is assumed that buyer liability is not practicable in the case of water quality trading systems. Buyers, i.e. individual sources, are not able to control sellers' activities.[163] High uncertainties at the source level about the control costs and the risk of invalidation could hamper trading activities and thus lower the efficiency

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158 It could even be interpreted as a design element itself.

159 "While the purchaser of most goods can relatively easily verify that a product of satisfactory quality has been delivered, the same cannot be said for WQT [water quality trading] markets" (Woodward *et al.*, 2002, pp. 971-972).

160 Over-selling could occur for different reasons: unintentionally, wilfully or inadvertently. Unintentionally over-selling, which happens due to unexpected events or similar, could allow for exceptions in the compliance rules, if it can be identified as such. No exceptions should be permitted for wilfully and inadvertently over-selling (see also Zhan, 2001, p. 501).

161 For an alternative discussion of liability rules, see Haites and Missfeldt (2001). Combined seller/buyer liability rules are also possible.

162 Under an interstate permit trading system, like the emission trading under the Kyoto Protocol, it is assumed that buyer liability creates considerable incentives for risk-averse buyers to monitor the sellers' activities. An increase in buyers transaction costs may be justified on the grounds that "the information is needed in any emission trading system, and [that] it is better to let the market sort it out […] than to rely on a […] international institution to do the job" (Victor, 2001, p. 71). Woodward *et al.* (2002) and Victor (2001) assume that the group of buyers would, for the most part, be represented by developed countries with good institutional conditions. It would be easier for them to monitor the sellers (developing countries) than it would be for the sellers themselves. These arguments cannot be transferred one-to-one in the case of water quality trading systems.

163 Furthermore, the incentive for the buyer to control the sellers' activities only exists if effective sanction mechanisms are in place. The buyer has no interest in a devaluation of his permits. He could thus have the incentive to keep quiet about non-compliance of a seller in order to hold the actual value of his permits. An additional sanction system to avoid this behaviour requires, in turn, a monitoring activity by the environmental authority. High transaction costs for the environmental authority are thus not avoided. In addition to the transaction costs and the monitoring processes at the source level, the environmental authority incurs the monitoring burden and the transaction costs.

of the system. In addition, asymmetric information problems would make it difficult for the buyer to monitor the seller effectively. The seller might not be willing to lay open his activities. Long trading periods might strengthen this problem: the longer the trading period, the less likely it is that an offender will be caught during the current period. Transaction costs for the buyer are high.

In the case of a water quality trading system, seller liability seems to be the appropriate method: the seller is responsible for the fulfilment of the reduction requirements.[164] Any permits purchased by the buyer are valid regardless of whether the seller is in compliance or not. While the buyer has no monitoring responsibilities, i.e. no additional transaction costs, it would be the task of the environmental authority to control the abatement measures of the seller. Under seller liability, contracts are discrete and the buyer is under no further obligations to monitor fulfilment over time. Buyers thus bear no risk and are more likely to participate in market activities, which is important for the efficiency of the system. Seller liability would reduce transaction costs for buyers, not necessarily for the environmental authority (see also Woodward *et al.* 2002, p. 971).[165] The environmental authority could, however, benefit from one single and uniform control system and thus from economies of scale.[166]

3.3.5 Competition, Practicability, and Enforcement

Competition

Reasons for distortion of competition or the rise of market power are often caused by general, i.e. design-independent, aspects of permit trading systems and are thus relevant for any trading system to be introduced. Nevertheless, specific design conditions of water quality trading systems could give additional leeway to participants to behave in a non-competitive way. If, for example, design elements lead to small effective numbers of participants and thus to lower liquidity at the market, competition bias could be provoked.[167] The question of market power becomes particularly relevant for the analysis of the practical approaches and this discussion can be seen in Chapter 5. Nevertheless, some general aspects of market power and competitive distortions on a permit trading market are discusses briefly already here.

Market power can prevent the trading system from coming to the least-cost solution (Hahn, 1984).[168] Concentrations of permits at one source and/or price fixing agreements between sources could, in principle, lower the efficiency of the market because the marginal abatement costs are not equalised.[169] Kemper (1993, pp. 152-161) discusses both cases in more detail for permit trading systems in general, for both auctioning and grandfathering systems, and relativises the risk of market power. He concludes,

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164 Zhang (2001, p. 502) argues that seller-liability works well for (intra-)national trading systems; he assumes that compliance rules (sanction mechanisms) are effective and make non-compliance too expensive. This might be especially true if the number of participants is relatively small. However, the emission trading system under the Kyoto Protocol shows that the efficacy of the seller liability in international trading systems depends strongly on the enforcement regime.

165 If the authority does, however, pass these costs on to the firms, the result in terms of financial burden does not really change .

166 In addition, each emission trading system requires the data reporting of each individual source of pollution. These data need to be verified anyway which is the task of the environmental authority. The authority could even use these data to control compliance in a seller-liability system. While the environmental authority controls seller compliance indirectly in a buyer-liability system through the control of buyers' control activities, sellers are controlled directly in a seller-liability system. Transaction costs (and asymmetric information problems) are assumed to be lower, if sellers can be controlled directly.

167 Strategic behaviour is easier to enforce, if the number of participants is rather small (Betz, 2003, p. 298).

168 See also Weimann (1998) or Sartzetakis and McFetrige (1999).

169 For an overview, see Kemper (1993, pp. 152-161), Endres (1985, pp. 75-85) or Tietenberg (2006, Chapter 7).

that (theoretical) market power problems are of little importance in practice.[170] Some studies confirm that by showing that market power is not a high risk in reality or that it can be avoided by the inclusion of general design elements.[171] Further studies show how trading systems can be enforced even if (endogenous or exogenous) market power exists (Chavez and Stranlund, 2003; van Egteren and Weber, 1996).[172]

The economic analysis, especially for the criterion of cost-effectiveness, showed that trading potentials between sources can be asymmetric (Paragraph 3.3.2). On the one hand, this lowers the number of potential trading partners for some sources; on the other hand, the asymmetric position could bias competitive conditions by discriminating against individual sources. The discussion of this aspect is integrated in the economic analysis of specific approaches.

For international trading systems, e.g. for transboundary rivers, the initial allocation of permits influences the competitive position between sources. Betz (2003) refers to an international permit trading system with different allocation modes (auction versus grandfathering) in different countries; sources acting in different countries would consequently act at different competition conditions. Even if permits are grandfathered in all countries, the competition conditions can be influenced (intentionally or not). For example, different regulations for newcomers compared to established sources would have different effects on their competitive abilities.[173] Again, different regulations in different countries integrated in the same trading system would also cause distortion of competition between sources located in different nations.[174]

A large share of the sources' transaction costs might be not proportional to the company size; consequently small companies can be faced with the same amount of transaction costs as bigger companies. This could lead to a disadvantage which can make it more difficult for small companies participate in trading processes. It may be reasonable to consider a simplification of processes for smaller firms in order to reduce this overproportional burden and thus lower the level of transaction costs. One proposition is to exclude very small firms from the trading system, if it is assumed that transaction costs would cause a too high financial burden (Tietenberg, 1980, p. 485). The trade-off between disburdening smaller facilities and lowering the efficiency of the system through a small number of participants must be taken into account.

Practicability and Political Feasibility

The aspect of practicability and political feasibility is also relevant for all types of permit trading systems. It is for the most part independent of the specific design. This aspect is, however, of crucial relevance for practical approaches and will be thus discussed in more detail in Chapter 5. Again, only general important aspects of the practicability problem are singled out at this stage.[175]

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[170] See also Weimann (1998). He shows that most of the arguments in favour of the risk of market power can be disproved.
[171] Carlén (2003) shows in experimental studies that market power is not as strong as is often predicted. Godby *et al.* (1999), for example, adopt a more critical position concerning the relevance of market power. Sturm (2006, Chapter 4) gives an overview over the experimental studies concerning market power; he discusses in more detail the potential of a double auction to prevent market power behaviour.
[172] Commonwealth of Australia(1992, pp. 36-40) discusses different options to reduce the risk of competition distortions and market power.
[173] See also Kemper (1993, pp. 158-161) or Liski and Montero (2004).
[174] In the case of the CO_2 trading system in the European Union allocation processes are regulated in the 'national allocation plans'. Allocation modes vary a lot between Member States; participating sources are thus confronted with differing competitive conditions. See also Michaelowa (2004).
[175] The perspective of the political economy regarding the implementation of permit trading systems in general has been excluded from the discussion as this would be similar for all trading systems. For a general discussion, see Sterner (2003,

Firstly, the conditions of integrating the instrument in the legal framework should be examined. Specific design options may ask for certain adjustments of the actual legal work.[176] Existing legal conditions, e.g. regulations regarding property ownership, may, in turn, influence the design options.[177] International law also becomes relevant. Cooperation between states is strongly required (cooperation principle, see Section 3.1). Within an existing cooperative system (like the European Union) cooperation would be easier to achieve than for a group of individual countries (Durth 1996a).

Secondly, it is important to examine the regulation history and density in a country. If water quality regulations are already quite dense and good water quality has already been reached, it is not necessarily reasonable to introduce a new instrument with all its administrative and information related consequences. In a world (nearly) without any regulations, where decisions are about completely new introduction of instruments, discussions are open for all kinds of instruments. The implementation of a permit trading might thus be easier (Section 6.2).

Thirdly, existing instruments already regulating or influencing relevant resources must be taken into account. A permit trading is not compatible with all types of instruments. In the case of the introduction of a permit trading system, existing instruments need to be identified and eventually adjusted or abolished.[178] A purely Best Available Technology (BAT)-based regulation, for example, would not allow for combination with a permit trading system. While the Best Available Technology requires the use of specific technologies, the central idea behind a permit trading system consists in the fact that sources are free in their decisions regarding production processes and technologies.[179] On the other hand, existing instruments may form a solid basis, e.g. from an administrative or transaction costs point of view, for the introduction of a water quality trading system.[180]

Fourthly, a functioning institutional and administrative framework is crucial in order to implement an effective and efficient permit trading system.

Furthermore, planning reliability should be guaranteed for all participants. A permit trading system is only practicable and efficient, if all sources can plan without uncertainties caused by the design. Sources should be informed about their right to emit and about all relevant elements of the trading system; regulations should be consistent over time.[181]

Chapter 16), Russell (2003) or Zöckler (2004). For the acceptance of permit trading systems within different interests groups in general, see Kemper (1993, pp. 93-102).

176 For the general discussion of the legitimacy of a permit trading in contrast to command-and-control instruments, see Michaelis (1996, pp. 138-153).

177 The German (and EU) legal framework, for example, does not always allow for (complete) permit auctions, with respect to the existence of property ownership regulations (Endres 2000, pp. 127-130). For a general discussion on property rights under the introduction of a permit trading system (air), see Bader (1999, pp. 251-283), Graichen and Requate (2003) or Wackerbauer (2003).

178 Smith (1999), Kraemer and Banholzer (1999, p. 100) or van Mark *et al.* (1992).

179 Kraemer and Banholzer (1999, p. 101) nevertheless propose the combination of Best Available Technology and tradeable permits. This proposal illustrates rather a specific implementation process of a required Best Available Technology programme than an integration of a permit trading system in its original sense into a Best Available Technology policy programme. Free choice of technology is not possible under this approach which would, however, be a central element of a permit trading.

180 Smith (1999, p. 205) mentions that initial permit values could be derived from existing regulatory emissions limits.

181 A two-phase introduction could simplify the installation of the instrument while reducing uncertainties. A pilot phase precedes the definite start of a permit trading. Participants can become accustomed to the instrument; improvements can be introduced (NIRAS, 2004, p. 5).

Scepticism with regard to permit trading systems is often high. Jacobs (1997, p. 98) analyses reasons for the low acceptance of permit trading systems within different groups of persons concerned (sources of pollution, publicity…).[182] The introduction of a water quality trading system should thus be accompanied by information processes to inform all participants from the advantages and necessity of the implementation. This would facilitate the acceptance and thus the feasibility of the trading system.[183]

Enforcement

Enforcement is a crucial factor of any permit trading system regardless of the design. Only effective compliance rules can guarantee the effectiveness and efficiency of the system. Again, this aspect is particularly relevant for the analysis of practical approaches (Chapter 5).

Enforcement depends on the technical ability to detect violations and the legal ability to deal with these violations (Tietenberg, 2006, p. 165).[184] Among other things, the basic foundation for a sanction system is a precise monitoring system at the source level which helps identify infringements. Furthermore, enforcement of permit trading systems is only guaranteed if the application of sanctions is credible. Implementation of sanctions must be automatic and the sanctions need to be set high enough to create the incentive for compliance. Penalties need to be higher than the financial benefit from non-compliance (over-selling,…) to create the incentive to fulfil the reduction requirements (Betz, 2003, pp. 58-60). However, penalties that are set too high are not reasonable either. On the one hand, they are not credible; on the other hand, courts may be reluctant to apply them. Practical experience has shown that sanctioning is often a complicated and time intensive process.[185] This might lead to sanction systems that are only applied in some cases or applied less stringently than the law calls for. In some cases, the environmental authorities try to achieve voluntary compliance to avoid penalties.

In addition, enforcement can only be guaranteed if leakage and substitution processes are effectively avoided. Leakage takes place when "pressure on the regulated resource is diverted to an unregulated or lesser regulated resource" (Tietenberg, 2006, p. 64). Polluters may move their facilities to countries with lower environmental standards or use other non-regulated substances which would have an effect on the discharges. Both processes lead to inefficiencies and have the potential to disturb effectiveness. It depends, of course, on the sector of industry and on the production processes whether leakage and substitution processes are possible or not.

Paragraph 3.3.4 has already referred to the complexity of enforcement and the dependence on the specific design and the river and pollutant characteristics. Discharges of some pollutants are easier to measure than others. In addition, the trading unit of the permit (emission versus immission) influences the complexity of the control and enforcement mechanism.

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182 See also Kemper (1993, pp. 93-102), Gawel (1998), and Cansier (1998).

183 General problems of implementation and of switching to a new policy instrument cannot be discussed in this study. For more information, see Rudolph *et al.* (2005) or Thompson (2000).

184 Tietenberg (1985, Chapter 8) defines different steps of the enforcement mechanism (detecting the violation, notifying the source, negotiating a compliance schedule, applying sanctions for non-compliance when appropriate). See also Field, B.C. and Field, M.K. (2002, p. 189) for enforcement problems in US Environmental Protection Agency programmes.

185 See Field, B.C. and Field, M.K. (2002, pp. 189-190). Experimental studies show that, in some cases, penalties can also cause effects opposite to those imposed. For moral considerations influencing the enforceability of an instrument, see Field, B.C. and Field, M.K. (2002, p. 191).

Sterner (2003, p. 197) states that "if agents are particularly concerned to be "caught", then frequent and efficient monitoring and enforcement may be the most important aspects of a good policy. If the size of the maximum penalty influences the behavior of the agents, then the penalty structure might be the most important design factor". The behaviour is, of course, influenced by risk behaviour of firms and thus on the expected risk to be monitored.

3.4 The Result: A 'Matrix' of Criteria

Based on the analysis in Chapters 2 and 3, we can develop ecological dimensions in order to evaluate the potential of a model to fulfil different ecological objectives and thus to be able to reflect different homogeneity conditions. The economic requirements of a water quality trading system can be systematised, too. The concrete analysis of theoretical and practical approaches with respect to these criteria will result in some further specifications of even these criteria.

Ecological Dimensions

Depending on the ecological objectives (desired or ecologically necessary) a water quality trading system is obliged to be characterised by a more or less differentiated design. Pollutants that are characterised by equal emission and immission levels can be more easily integrated in a permit trading system; the design of the model can be simplified. This is not the case for other pollutants. Moreover, the politically required preciseness of the trading system specification may be higher or lower. The requirement of a detailed differentiation of the immission cap in time and/or space due to different water uses or similar would, of course, demand for a trading system that is able to reflect these specific characteristics.

The following table shows relevant dimensions of the design that characterise the potential ecological design differentiations (Table 3-1). Note again that not all aspects need to be fulfilled by a single trading system; the specific design depends rather on ecological settings and conditions. Table 3-1 is directly based on the results of the preceding chapters. A few remarks should make clear how the table is to be interpreted.

Each water quality trading model is characterised by two general design elements. Firstly, the *cap type* of a model indicates whether it is emission-based or ambient-based (Section 2.5). Secondly, depending on the design of the water quality trading system, the integration of (impact) *trading ratios* could become necessary in order to guarantee the ecological effectiveness of trading activities between sources.[186] Trading ratios can be determined endogenously or exogenously.

Moreover, the ecological dimensions are relevant. Before implementing a water quality trading system, one should decide on the *geographical scope*. Paragraph 3.2.2 has shown that, in the case of rivers, the integration of the entire river basin is the most appropriate approach to guarantee the effectiveness of the instrument.

Models could follow an endpoint constraint or instream flow need constraint (*orientation*). Whether such a differentiation becomes necessary or not will depend on the specific ecological settings. Not all models can reflect this differentiation. The ecological goal thus influences the choice of the design.

[186] Again, two types of trading ratios exist: the impact trading ratio, which reflects different impacts on the immission of identical discharges from different sources, and the uncertainty trading ratio, which considers uncertainties about the resulting immission level from discharge reduction at different sources. This type of trading ratio applies especially for the case of integrated nonpoint sources.

General Design Elements	
Cap type	- Emission-based versus ambient-based
Trading ratio	- Exogenous versus endogenous
Ecological Dimensions	
Geographical scope	- National/administrative - Geographical/natural
Orientation	- Endpoint versus instream flow need constraints
Differentiation of the immission cap	- Space - Time
Adjustment of the emission cap	- Space - Time
Water quantity aspects	- Integrated (exogenous, endogenous) - Not integrated

Table 3-1: Ecological Dimensions

Whether the *immission cap* needs to be differentiated in space and/or over time depends on the specific definition of the ecological goal which, in turn, also depends on the specific river and pollutant characteristics (Paragraph 3.2.2). It is thus important to know *ex ante* whether a specific model is able to reflect such a differentiation.

For ambient-based permit trading systems, the immission cap must be converted into an *emission cap* in order to make the system operable. The precise definition of the emission cap is influenced by the properties of the pollutants and the river itself (Chapter 2). Not all approaches which will be discussed in Chapters 4 and 5 are able to adjust the emission cap to changing immission caps in space and/or over time. The decision regarding the adequate design is thus influenced by the individual properties of pollutants and rivers.

Section 2.4 has shown that water quality aspects are strongly linked to changes in *water quantity* levels. For some river basins, variation in water quantity levels might be negligible. No integration of (endogenous or exogenous) changes in the water flow would be necessary. Other river basins, however, ask for a differentiated integration of water quantity aspects into the model. Again, the actual conditions dictate which model type is most appropriate and can reflect the specific characteristics and requirements.

The discussion of theoretical and practical approaches will demonstrate that any model can avoid hot spots under specific conditions, i.e. ecological objectives and river conditions. Note again, that neither one model or the other is better. Whether or not the model is appropriate will depend on the specific requirement.

Economic Criteria

As shown in Section 3.3, the common economic criteria for the evaluation of an environmental instrument are relevant for water quality trading systems. Table 3-2 summarises the results. While the presentation in this chapter is quite general, the economic analysis will be adapted to specific conditions of a water quality trading system in the following chapters. Again, we shall illustrate how the table should be read.

The criteria of *cost-effectiveness* demands that the ecological goal is achieved at the least cost. In theoretical models, the condition for cost minimisation is the equalisation of the marginal abatement costs in all

Economic Criteria

Cost-effectiveness	- Equalisation marginal abatement costs = cost minimisation
Dynamic efficiency	- Incentives, innovations
Transaction costs	- Sources versus environmental authority - One-time versus permanent - Trade-dependent versus trade-independent
Competition, practicability and enforcement	- Practical approaches

Table 3-2: Economic Criteria

sources (Paragraph 3.3.2). A model should set the incentive for sources to exploit differences in the marginal abatement costs in time and space. Additional indicators (prices, volume of trade) may be necessary in order to evaluate practical approaches. The application of a non-degradation principle should be avoided in order to prevent inefficiencies.

Dynamic efficiency requires that the model sets an incentive to innovate and to find new abatement technologies (Paragraph 3.3.3). Surplus permits created by the innovation process can be sold on the market; a financial incentive does thus guarantee dynamic efficiency in the original permit trading model. Too strict implementation of the precautionary principle could hamper dynamic efficiency. The analysis of theoretical and practical approaches will show which elements can additionally hamper systems from achieving dynamic efficiency.

The level of *transaction costs* is also important for the functioning and the efficiency of a permit trading system. If the transaction costs for individual sources are too high, then trading activities may be hampered thus decreasing the efficiency of the trading system. Also the transaction costs at the level of the environmental authority become relevant (Chapters 4 and 5).

The aspects of *market power, practicability and enforcement* will be discussed for the practical approaches only. It is assumed that they are similar for any permit trading system to be introduced. An in depth discussion would thus go beyond of the scope of this study. Specific conditions that are obvious for the selected water quality trading systems (practical approaches) will nevertheless be discussed.

4 Design Options: Theoretical Approaches

4.1 General Remarks

Different theoretical and practical approaches to water quality trading systems exist. However, the interrelation between these models is often ignored by the literature although the understanding of different approaches is important for the introduction of adequate water quality trading systems. Depending on specific conditions very different designs of permit trading systems may be required. It is thus necessary to be informed about the potentials of different design elements.

In order to fill this gap, the following section starts to evaluate theoretical approaches with respect to the standardised and comparable criteria developed in Chapter 3. Theoretical approaches are presented and analysed in order to identify typical design elements for water quality trading systems. The structure is the same for all models: a compact presentation of the model and its design elements is followed by an evaluation. Similarly, Chapter 5 analyses the practical approaches, i.e. case studies, by following the same method (presentation, evaluation).

The following analyses discuss which specific ecological dimensions the concerned theoretical approach can integrate. It is not always necessary that a water quality trading system fulfils all of the potential criteria. When introducing a water quality trading system it is very important to be informed about the potential ecological performance of the model. The definition of the ecological objective and the characteristics of the river influence the decision regarding the optimal trading design. As to the economic criteria, they all need to be attained by a water quality trading system in order to guarantee efficiency of the system.

4.2 Starting Point: Theoretical Approaches for the Medium Air

Dales (1968) and Crocker (1966) were the first to introduce the system of tradeable permits in the literature. While Dales developed the trading system for water pollution control, Crocker analysed the case of air pollution. Although these systems are rather descriptive, they form the main basis for the development of theoretical and practical water quality trading systems.[187]

The following paragraphs introduce theoretical permit trading approaches for non-uniformly mixed (assimilative) pollutants with different design options. Most of these models originally applied to air pollution. They can, however, be applied to water pollution problems. River-specific approaches will even show that the application to water pollution problems in practice would in some cases be easier than to air pollution: dispersion characteristics of non-uniformly mixed pollutants in a river can be more precisely documented in quality models.

All these models are based on the central mechanism of a permit trading: each source is obliged to hold permits for the emissions it introduces into the environmental medium (air or water). When reducing its discharges, a source is able to sell the surplus of permits; when increasing its discharges, a source has

[187] Dales does not develop a specific model; it instead describes the general mechanisms of a permit trading system with the example of water pollution control. Therefore, this study sets the further analysis of Dales' 'model' aside.

to purchase permits from other sources in order to cover these additional emissions.[188] The total amount of emissions allowed is determined by the number of permits distributed. Trading activities can only take place within a single river; exchanges between different rivers are not permitted.

4.2.1 Ambient Permit System

4.2.1.1 The Model

Montgomery (1972) was the first to develop a theoretical approach to permit trading for the control of non-uniformly mixed assimilative pollutants by explicitly reflecting spatial considerations.[189] Under the *Ambient Permit System (APS)* each source must cover its discharges with permits that can be traded amongst sources. The total amount of permits is defined by the required quality standard. A discharge permit allows emitting pollutants (emissions) at a rate that will not impair the predetermined quality level (immission) at a specified point, the receptor point. The approach is thus ambient-based.[190]

In most cases, changes in the discharge activities of one source will impact the pollution level at more than one receptor point. When increasing discharges, a source needs to obtain permits for each receptor point that is affected by the change in discharges. Each source needs to hold a portfolio of permits in order to cover all receptor points influenced by its discharges. Separate permits are therefore issued for each receptor point. Consequently, each receptor point functions as a separate permit market, all operating simultaneously. A source that modifies its discharge behaviour needs to find trading partners at any receptor point it influences by its activity. All third-party effects are thus considered.

The Ambient Permit System has n industrial (point) sources of pollution and m receptor points in the river. Sources are fixed in location and owned by independent, profit-maximising firms. An environmental authority defines quality standards of one pollutant concerned at various receptor points, denoted as a vector

$Q^* = (q_1^*, ..., q_m^*)$, where q_j^* $(j = 1, ..., m)$

is the determined quality standard at receptor point j.[191] Each of the sources $(i = 1, ..., n)$ emits a single pollutant at the rate e_i, which can be replicated by an emission vector

188 All approaches discussed in this study follow the cap-and-trade mechanism. A total amount of permits is allocated to individual sources and can then be traded. No baseline-and-credit design applies; only reductions beyond a determined baseline create credits that can be sold on the market. For a discussion of these approaches, see Dewees (2001).

189 Although the Ambient Permit System is mainly formulated for air pollution control, Montgomery underlines that it is also valuable for water pollution control. Originally, Montgomery mentions a smoke plume from an elevated source emitting at a constant rate with a wind of constant direction and speed. The simplification of a wind, constant in direction and speed, might be adequately transferred to a river situation, where the water transports the pollutants in a constant direction with constant speed.

190 Montgomery also proposes an *Emission Permit System (EPS)* with emission-based permits. Because the Emission Permit System has both theoretical and practical problems, it will not be considered in the following analysis. For further discussion of the Emission Permit System, see Krupnick *et al.* (1983) or Hung and Shaw (2005, p. 84).

191 For the case of air, q_j^* could be an annual average immission load of sulphur dioxide at the receptor point j in an air basin. For water pollution control, q_j^* might be a measure of annual average immission load of nutrients at the receptor point j on a river.

$E = (e_1,...,e_n)$.

As the emissions of the sources have, e.g. depending on their location, different impacts on the quality (immission) of the receiving medium, this emission vector will be mapped into quality levels (immission loads) by a matrix H, so that

$E \cdot H = Q$.

The matrix H thus shows how immission is affected by the emissions of different sources. More precisely, the matrix

$$H = \begin{pmatrix} & \vdots & \\ \dots & h_{ij} & \dots \\ & \vdots & \end{pmatrix}$$

shows, how one unit of emissions from source i affects the quality level at receptor point j.[192] The elements of the matrix h_{ij} are thus called 'dispersion coefficients'.[193] They are results of the dispersion functions that are necessary to reflect the relation between emissions and immission (Section 2.3). The quality standard Q^*, set by the environmental authority, defines the immission constraint which should not be exceeded, i.e.

$E \cdot H \leq Q^*$.

The ratio of dispersion coefficients of the matrix H defines the trading ratio for transactions between sources exogenously. The trading ratio reflects the fact that emission discharges of different sources may affect the quality of the receiving medium in a different way and defines to what degree a source can increase its emissions when purchasing a permit valued at one unit from another source (impact trading ratio, Section 2.5).

In addition, a non-degradation principle applies under the Ambient Permit System. In the case of water pollution control, this does not permit a reduction of water quality in any part of the river, even if this quality level is higher than the water quality standard requires.[194]

4.2.1.2 Evaluation

Although the Ambient Permit System has already been judged before,[195] the model is now evaluated in a more standardised way. Results for the Ambient Permit System can thus be compared with the results

[192] The Ambient Permit System supposes only one relevant pollutant. This is suitable as long as the desired quality of the medium in terms of one relevant pollutant is independent of the desired quality for any other pollutant. The model could be generalised by adding constraints representing emission vectors that achieve desired levels of many pollutants and joint production of pollution. This problem then is to be solved in the same way as the one-pollutant system developed here. Furthermore, it is assumed that all prices (except those associated with pollution) are unaffected by measures undertaken to control pollution which is a common assumption in economic analysis of environmental problems.

[193] Concentrations are modelled as a linear function of emissions. Other functional relations could also be reflected by the model; they would require more complex dispersion functions, which are not dealt with in this study.

[194] See Paragraph 3.3.2. This element will be criticised within the economic evaluation of this approach.

[195] See Krupnick *et al.* (1983) or Hung and Shaw (2005).

of other relevant trading models discussed below. Recommendations can be formulated on a more complete and standardised basis.

Ecological Dimensions

As the Ambient Permit System follows a clear ambient-based approach (*cap type*), it corresponds to the requirements determined in Paragraph 3.2.2. The portfolio of permits held by the sources reflects the impact of discharges on the water quality level at the receptor points. Permits are purchased and sold according to changes in the immission impact, not the emission impacts. If for some pollutants the differentiation between immission and emission is not of relevance, the model can be simplified; the definition of dispersion coefficients and of trading ratio would no longer be necessary.

The *trading ratio* (ratio of dispersion coefficients) in this model is defined exogenously and fixed for a certain period of time. While the exogenous and fixed definition causes lower transaction costs, this could prevent the system from reaching the ecologically effective situation. Trading ratios reflect different impacts on the ambient pollution level of discharges from different sources. If conditions alter (water quantity or temperature), dispersion coefficients change as well. If the trading ratio is defined as ratio of dispersion coefficients, it can only be ecologically effective, if dispersion coefficients are determined precisely and correctly. If dispersion coefficients are not accurately adjusted over time, trading ratios would not reflect the actual situation. The risk of hot spots arises. A definition that is too severe could, however, cause excessive costs compared to the ecological benefit it generates. The mechanism of trading ratio adjustment should thus be chosen carefully with respect to the specific conditions in the regulated river (Section 2.5).

The Ambient Permit System can apply to different *geographical scopes*. As we have seen in Paragraph 3.2.2 the most reasonable scope for a water quality trading system is the entire river basin. Under the Ambient Permit System, the mesh of receptor points should be adapted to the application area of the trading system.

The trading system can be configured in order to meet an endpoint or instream flow need constraint (*orientation*). Only the determination of the water quality standards would be influenced by this decision. Depending on the orientation, water quality models would define stringent or less stringent immission caps for the receptor points. This would not influence the operability of the system.

Receptor points may require different maximum *immission caps* (water quality standards) at different locations or at different periods in time due to water use conditions (Paragraph 3.2.2). A spatial differentiation could be achieved by a separated standard setting for any single receptor point.[196] This would be directly reflected by the initial allocation of permits and thus be fixed over time. Any change of the immission cap over time needs to be reflected by an adjustment of the permit allocation over time. The total number of permits for a single receptor point reflects the immission cap for this same point. Modifications of the immission cap over time must be followed by an adjustment of the total amount of permits and thus of the allocation. One solution would be to allocate permits as percentages of the total immission cap. If the total immission cap changes over time for certain receptor points, the immission value of individual permits would vary proportionally.[197]

[196] This would not be possible without restrictions for air pollution control. Dispersion of emissions cannot be controlled with such precision.
[197] This option will be discussed in more detail in Section 6.1.

A differentiation of the immission cap (time or space) needs in any case to be followed by similar changes to the *emission caps*. Even if immission caps are fixed in space and over time, emission caps need to be defined carefully. Changing conditions (water level, temperature…) may influence the impact of discharges on the water quality; emission caps must be adjusted for the entire portfolio. Under the Ambient Permit System, the individually permitted immission caps need to be converted into an emission cap using the dispersion coefficients as determined by the environmental authority. Generally, the adjustment of the emission cap in space and over time is assumed to be possible.

As stated, the Ambient Permit System mainly applies to air pollution. Therefore, the integration of *water quantity aspects* is not mentioned explicitly in the model. When applying this model to water pollution control, dispersion coefficients and trading ratios may integrate the impact of water quantity aspects if they significantly influence the water quality. Changes in water quantity can emerge for exogenous and/or endogenous reasons (Section 2.4). Such an integration of water quantity aspects would, however, demand for further adjustments of the model. The value of permits in terms of emissions, and thus the dispersion coefficients, would need to be adjusted with any variation in the water flow levels and for the entire portfolio. In addition, the *ex ante* exogenous definition of trading ratios is not compatible with a continuous integration of water quantity aspects. Alteration of the water flow has an impact on the dispersion coefficients and thus on the trading ratio. If changes in water flow are constantly considered, the trading ratio will also need to be adjusted permanently.

The integration of water quantity aspects into the model thus seems possible. However, necessary adaptations of the model result in a more complex structure. The individual situation (river, pollutants etc.) decides on the necessity and adequacy of such an extension.

If one assumes that water quantity aspects are not as important for a specific river basin and permanent adjustments of dispersion coefficients and trading ratios is not necessary, too high concentrations (hot spots) at receptor points cannot occur. One might assume that discharges of two proximate sources that are both important in terms of emissions could cause hot spots independent of the water level development. But: any source holds permits in terms of immission. Sources cannot cause higher immission at the receptor point than they are allowed by the permits. The location effect is included through the application of dispersion coefficients and trading ratios. Trading ratios and dispersion coefficients, if determined accurately, integrate the relation between emissions and immission and avoid hot spots.

To guarantee the water quality for the whole river, it may be necessary to install a fine mesh of receptor points. Requirements on the mesh of receptor points depend on the exact definition of 'hot spot'. The achievement of predetermined water quality standards can be required for each receptor point or for any point in the river, i.e. also for points in between two receptor points. This would, of course demand for a higher density of receptor points. The adequate installation of receptors is thus a basic condition for the avoidance of hot spots.[198] Table 4-1 (see below) summarises the results concerning the ecological part of the analysis.

Economic Criteria

The evaluation of the economic characteristics of the system comes to the following result. A competitive equilibrium exists and corresponds to the cost-minimum attainment of a set of predetermined water quality standards (*cost-effectiveness*, equalisation of marginal abatement costs).

[198] This, of course, influences the transaction costs, see below.

Ambient Permit System (APS)

General Design Elements

Cap type	- Ambient-based
Trading ratio	- Exogenous

Ecological Dimensions

Geographical scope	- Entire river basin possible
Orientation	- Endpoint and/or instream possible
Differentiation of the immission cap in	
Space	- Possible
Time	- Possible
Adjustment of the emission cap in	
Space	- Possible
Time	- Possible
Water quantity aspects	- If at all: implicitly in the dispersion coefficient

Table 4-1: APS, Ecological Dimensions

However, Krupnick *et al.* (1983, p. 240) demonstrate that the market equilibrium only coincides with the cost-minimum solution, if the initial allocation of permits makes the water quality standard, i.e. a pollution constraint, binding at all receptor points. The water quality standard at any receptor point j is not binding if the actual emission behaviour of the sources as set by the environmental authority (initial allocation of permits) results in a higher water quality, i.e. a lower immission load, than the allowable level set by the water quality standard.

If the existing water quality is higher than the standard requires, the water quality standard is not binding. The least-cost solution, however, can only be reached if the initial allocation of permits is in such a way that the water quality standard is binding at all receptor points; only when this is the case will all trading potentials be exhausted (Krupnick *et al.*, 1983).[199] Hung and Shaw (2005, p. 84) argue that it is usually unattainable to make all standards binding, which would mean that the cost-minimum solution is not necessarily achieved.[200] The analysis of Figure 3-4 (p. 39) also illustrates that it is very difficult to make all caps binding. It is, however, assumed that costs savings can be realised independently of the initial allocation, but not to the full extent. Due to the negative affects of the non-degradation principle on the cost-effectiveness, the exclusion of this principle is required from an economic point of view.

Furthermore, the existence of single trading markets for each receptor point raises the question whether the number of participants in each individual market will be high enough to guarantee efficient trading.[201] Only if the trading source is able to find a trading partner at all the affected receptor points, can changes in discharges and efficiency gains be realised.

The free riding problem, arising in other models (in particular Paragraph 4.2.2), is avoided: all discharge permits are allocated to individual sources. A source can only increase its discharges if it purchases an adequate amount of additional permits from another source that has reduced its effluents. The portfolio of permits covers all receptor points and all impacts of changing discharges are integrated. If

[199] Paragraph 3.3.2 already showed that the non-degradation principle is restrictive in an economic sense, thus causing inefficiencies.
[200] International convention and directives often stipulate the application of the non-degradation principle. Thus setting of binding caps is biased by international law. However, if the application of the non-degradation principle is required by law anyway and for any trading system to be implemented, this criterion cannot further be used to distinguish between different models.
[201] See also Michaelis (1996, p. 129).

Ambient Permit System (APS)	
Cost-effectiveness	- Depending on initial allocation - No free rider
Dynamic efficiency	- Yes, but possibly biased
Transaction costs	- See discussion of Table 4-3 and Table 4-4 (p. 60-64)

Table 4-2: APS, Economic Criteria

no potential for free rider behaviour exists, it is assumed that all trading opportunities are exhausted. Cost-effectiveness is thus not hampered.

Even if the cost-effectiveness of the system cannot be guaranteed in every case, the Ambient Permit System can be characterised by *dynamic efficiency*. Cost saving potentials exist nevertheless and the properties of the permit trading system itself remain independent of the equilibrium attained. Surplus permits can be sold if emissions have been abated; the source can realise a benefit (Paragraph 3.3.3). As each unit of effluents must be covered by a permit, the reduction of effluents is profitable anyway. The level of benefits can, however, be impaired in reaching the optimum due to a lack of cost-effectiveness because of distorted market prices. If caps are not binding, cost-effectiveness as well as dynamic incentives are biased. Not all cost saving potentials would be exhausted; the incentive to invest in new technologies would be lower than under 'full' cost-effectiveness.

Given a flowing river, an asymmetric situation with respect to the trading opportunities of sources might arise (Paragraph 3.3.2). A reduction in emission loads at the last source (downstream) does not create additional discharge windows for upstream sources. There is thus no demand for the created permit; the incentive to reduce emission loads is biased. The first source (upstream) cannot buy permits which will allow it to increase the discharge possibilities at this location. Starting positions for sources are different; competition is distorted.[202] Nevertheless, sources affecting the same receptor point are able to trade with one another. The number of potential trading partners is, however, lower as trading activities are restricted to the specific area. The weight of these competition aspects must be determined for any specific application. Note that this problem does not occur in the case of air pollution where sources are located within an area without a definable first or last receptor point. Table 4-2 (see above) summarises the results.

The level of *transaction costs* cannot be determined in absolute terms. This chapter rather seeks to specify blocks of transaction costs relative to other models. As a first step, the relevant transaction costs for the environmental authority are discussed; as a second step, the transaction costs paid by the sources are defined. Transaction costs are furthermore identified as permanent, trade-dependent or trade-independent costs. The final level of transaction costs has to be defined with specification to the conditions *in situ*. The analysis of the *status quo* reveals which part of the transaction costs arises additionally to already existing transaction costs.

Transaction costs (one-off) caused by the *'installation' of receptor points* are relatively low.[203] The number of receptor points would, however, influence the transaction costs as it influences the number of trading

[202] In the case of newcomers, trading potentials might influence the decision of location.
[203] Krupnick *et al.* (1983, p. 234) specify that "by receptor points, we are not referring to the location of monitors". They rather assume a concentration gradient for an area that is constructed, for example, in conjunction with a (water) quality

ratios to be defined. A large number of receptor points and thus markets would be necessary to achieve the predetermined environmental quality at every location (high monitoring density). The exact number is among other things, influenced by the geographical scope of the system and needs to be individually determined for any specific trading system.

Under the Ambient Permit System, *water quality models* are necessary to define the dispersion coefficients (and the trading ratios). However, they do not go before any *potential* change in discharges, like in other models (Pollution Offset System, Paragraph 4.2.2). The environmental authority defines the water quality standards for the river. Furthermore the environmental authority defines dispersion coefficients for each individual source and their impacts on each individual receptor point. Sources use this information in order to convert the immission caps specified in the permits into emission caps.

The Ambient Permit System does not rely on *real-time data*, like other models do (Paragraph 5.3.3); all information is taken from water quality models. The costs of implementing water quality models emerge at the environmental authority level and are independent of actual trading activities.

The *initial allocation* of permits in terms of immission does not take into account any relations between emissions and immission. As a result, the initial allocation of permits does not require the application of an additional water quality model as it will be the case for other models discussed below (Trading Ratio System, Paragraph 4.3.1).

The Ambient Permit System uses a trading ratio which corresponds to the ratio of *dispersion coefficients* and thus reflects different impacts of effluents on the ambient water quality level. The exogenous, i.e. advance, determination of the *trading ratios* causes additional, one-off, transaction costs for the environmental authority which are lower than in other models where the trading ratios have to be determined endogenously. The number of trading ratios to be determined and thus the transaction costs (environmental authority) increase with the number of receptor points and the number of participating sources. The costs of defining the trading ratio are independent of the actual trading activities.

The application of differentiated *immission caps* would result in increasing transaction costs for the environmental authority. For a spatial differentiation, immission caps must be determined for each individual receptor point; this would cause one-off transaction costs necessary to establish the trading system. Dynamic aspects, i.e. a differentiation of the immission cap over time, eventually become relevant as well (Paragraph 3.2.2). The differentiation causes higher (more permanent and trade-independent) transaction costs for the environmental authority, including higher monitoring costs. These transaction costs could be justified by a higher ecological accuracy of the trading system. However, the differentiation over time, in particular, would cause additional (trade-independent) transaction costs for the sources. Any modification of the immission cap would require the re-definition of the *emission caps* (Section 3.1). Even if this is not done by the sources themselves, additional transaction costs in form of information and decision costs emerge as the trading conditions change.

Under the Ambient Permit System it is not specified who bears the costs of converting the cap (*determination of the emission cap*). We assume that the emission load cap resulting from the permit (defined in terms of immission) is determined by the sources according to the dispersion coefficients determined

model. Points on that gradient would be selected to be receptor points. They underline that "there need not be any actual monitors at these locations".

Ambient Permit System (APS)	
Transaction Costs (TAC), Environmental Authority	
Determination of receptor points, zones or blocks	
- Higher TAC	
- Lower TAC	X
Running Water Quality Model (WQM) before any *potential* change in discharges	
- Yes, higher TAC	
- No, lower TAC	X
Real-time data	
- Yes, higher TAC	
- No, lower TAC	X
Determination of the initial allocation of permits	
- Including Water Quality Model, higher TAC	
- Not including Water Quality Model, lower TAC	X
Definition of the dispersion coefficient (Water Quality Model)	
- Yes, higher TAC	X
- No, lower TAC	
Definition of the trading ratio	
- Endogenous, higher TAC	
- Exogenous, lower TAC	X
Differentiation of the immission cap in space	
- Higher TAC	
- Lower TAC	X
Differentiation of the immission cap over time	
- Higher TAC	X
- Lower TAC	
Determination of the emission cap	
- Environmental authority, higher TAC	
- Sources, no TAC environmental authority	X
Information requirements	
- Higher	
- Lower	X

Table 4-3: APS, Transaction Costs, Environmental Authority

by the environmental authority.[204] In this case, no additional transaction costs would occur to the environmental authority for the conversion of the immission cap into emission caps.

The *information requirements* for the environmental authority to inform sources about the running process are relatively low. This will become obvious in later chapters when the information requirements of other systems will be discussed. The environmental authority defines the water quality standards and allocates permits; sources should be informed about the value of permits, i.e. the dispersion coefficients and the trading ratios. The *ex ante* definition of dispersion coefficients and trading ratios makes these costs independent of actual trading activities. Table 4-3 shows the transaction costs that the environmental authority has to pay under the Ambient Permit System.

Transaction costs caused for sources can be identified as follows (Table 4-4). Permits under the Ambient Permit System are defined in terms of immission. It is assumed that sources are responsible for transforming these goals to emission caps using the dispersion coefficients determined by the environmental authority (*determination of the emission cap*). The advantage lies in the fact that the authority can use a single model to define the dispersion coefficients. Sources thus use the same basis to define their emission caps. Defining the dispersion coefficients and thus the emission caps at the source level would lead to high transaction costs and, furthermore, include the risk of integrating different transforming mechanisms thus resulting in biased solutions.

[204] The volume of water discharged into the river can be very different for the same emission load. Sources need to define the individual volume discharge cap.

Ambient Permit System (APS)

Transaction Costs (TAC), Sources	
Determination of the emission cap	
- Yes, higher TAC - No, lower TAC	**X**
Portfolio of permits	
- Yes, higher TAC - No, lower TAC	**X**
Constant adjustment of the permit value	
- Yes, higher TAC - No, lower TAC	Depending on the final design
Trading ratio	
- Endogenous, higher TAC - Exogenous, lower TAC	**X**

Table 4-4: APS, Transaction Costs, Sources

The existence of receptor points requires that a source obtains or sells permits for each receptor point affected when increasing or decreasing discharges. Thus, for each receptor point a single market establishes; sources must hold and manage a *portfolio of permits* from each of the receptor points that are affected by its discharges. An increase in emissions could require trading activities with several sources.[205] This causes higher permanent transaction costs at the source level in the form of information, bargaining and decision costs.[206] It is assumed that a proportion of these costs is trade-independent; sources seek to find trading potentials. If trading activities do actually take place, then additional trade-dependent transaction costs arise.

Whether the value of the permit, in terms of immission, is fixed or changes over time and/or in space will depend on the design of the system (*adjustment of the permit value*). Permanent modifications of the immission goal would cause greater efforts to define the corresponding emission cap. If the immission cap changes over time, the entire portfolio needs to be adapted (Paragraph 4.2.1.1). This would in turn cause high (trade-independent) transaction costs for sources: decisions need to be constantly adjusted to new permit values. Monitoring and reporting would be more complex and thus more cost-intensive. Moreover, a constant immission cap might require permanent adjustments of the emission cap due to river and pollutant characteristics. Finally, the number of receptor points influences the transaction costs of sources. Additional receptor points make the trading activities more complex. The volume of the portfolio increases; more trading partners could become relevant. The necessity of constantly adjusting the value of permits cannot be addressed without more information on the specific design of the trading approach.

Trading ratios are defined exogenously by the environmental authority; this causes lower (trade-independent) transaction costs for sources than endogenously defined trading ratios. Exogenous trading ratios are known *ex ante* thus creating a degree of planning reliability for sources and lowering information needs (Section 2.5). Transaction costs are nevertheless assumed to occur permanently as the trading ratios for different receptor points are constantly integrated in decision processes.

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205 It is assumed that a singe pollutant is traded in this case. The integration of further pollutants would, of course, lead to more complex portfolio decisions.

206 See Klaassen (1996, pp. 50-51) for more details. An increasing number of receptor points due to an expansion of the system to the entire river basin would not necessarily increase the number of permits sources need to hold. The discharge of a firm certainly influences the immission load of receptor points nearby. The impact on receptor points that are far away will be lower or even negligible. An expansion of the system to the entire river basin would thus not change the portfolio for all sources.

One can conclude for the Ambient Permit System, that the transaction costs for the environmental authority are relatively low, while transaction costs for sources are relatively high, particularly due to the necessity of managing an entire portfolio of permits. Furthermore, the proportion of trade-independent transaction costs is relatively high. In turn, additional trade-dependent transaction costs that would directly influence the efficiency of the trading system according to Stavins (1995) are relatively low.[207]

Results of the ecological and economic evaluation are summarised in Paragraph 4.2.4 with the results of the other models. Generally, one could state that the Ambient Permit System performs quite well with respect to the ecological criteria and can be introduced for different ecological objectives. However, some economic criteria, particularly cost-effectiveness, are not fulfilled automatically.

[207] We discuss only the transaction costs that are different for water quality trading models. Transaction costs that occur for any permit trading system to be implemented are ignored. The influence of trade-dependent or trade-independent transaction costs is thus only discussed for these specific transaction costs. This argumentation holds true for the analysis of all following models.

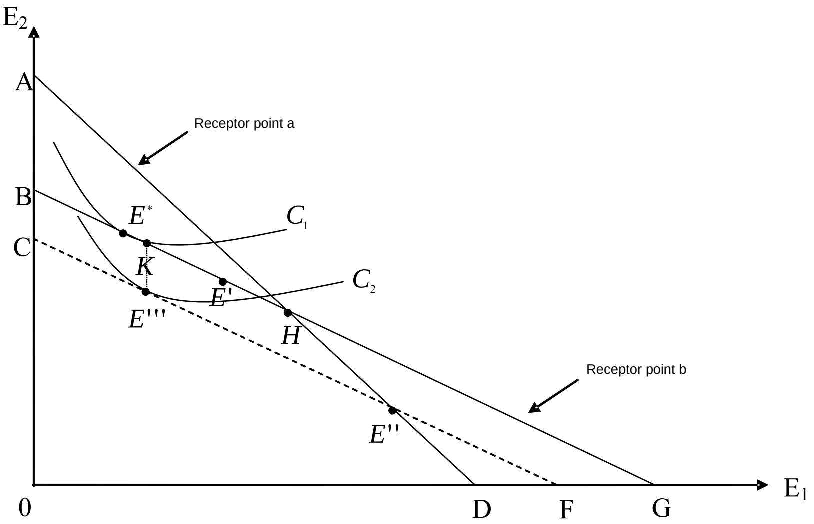

Figure 4-1: Pollution Offset System (POS)
According to Krupnick *et al.* (1983, p. 239)

4.2.2 Pollution Offset System

4.2.2.1 The Model

Based on their own criticism of Montgomery's Ambient Permit System Krupnick *et al.* (1983) developed the *Pollution Offset System (POS)*, originally designed for air pollution control. The permits held by the sources contain the right to discharge emissions. Sources can obtain permits from the environmental authority for free as long as a prior environmental model simulation shows that the proposed transaction will not infringe the quality standard at any receptor point.[208] The model is thus ambient-based, too. If the simulation shows that a change in discharges, would cause an infringement of the predefined standard at any receptor point, i.e. the receptor point becomes binding, the sources must trade while applying a trading ratio which is equal to the ratio of the two source's dispersion coefficients, indicating, how the quality at the receptor point is influenced by the discharge of pollutants of the concerned sources (Section 2.5). These trading ratios are endogenous and given by the simulation model (Hung and Shaw, 2005, p. 85).

This approach is illustrated in Figure 4-1. The Pollution Offset System would disqualify E''' as a point of market equilibrium: the quality standard of receptor b is not binding in E''', firms will thus obtain additional permits free of charge. Suppose that source 2 obtains additional permits thereby moving the vector of emissions from E''' to K. K lies on the line BG; exhausting the trading potentials will lead the two sources to the point of E^*. E^*, the least-cost solution, would thus be an equilibrium.

To avoid the inefficiencies of the Ambient Permit System (Paragraph 4.2.1), no non-degradation principle applies under the Pollution Offset System. The model thus allows increasing the discharges at any point as long as the simulation model shows that the quality standard will not be exceeded at any

[208] It is assumed that these permits are allocated on a 'first come, first served' basis. As long as permits are available free of charge newcomers are also able to obtain them. If additional discharges by newcomers would violate the water quality standard at any receptor point, newcomers should also obtain permits through trading with other sources.

Pollution Offset System (POS)		
General Design Elements		
Cap type	-	Ambient-based
Trading ratio	-	Endogenous
Ecological Dimensions		
Geographical scope	-	Entire river basin possible
Orientation	-	Endpoint and/or instream possible
Differentiation of the immission cap in		
Space	-	Possible
Time	-	Restricted
Adjustment of the emission cap in		
Space	-	Restricted
Time	-	Restricted
Water quantity aspects	-	Integrated in the simulation model, not constantly

Table 4-5: POS, Ecological Dimensions

receptor point, and this, even if this lowers the prior quality at any receptor point. Consequently, the quality standards are not all and not always binding.[209]

4.2.2.2 Evaluation

Ecological Dimensions

The results of the ecological analysis are summarised in Table 4-5. While individual permits are defined in emissions, the caps for the receptor points are defined in terms of immission, i.e. water quality standards. The system is thus ambient-based (*cap type*) and fulfils the requirements formulated in Paragraph 3.2.2.

Trading ratios apply only, if trading activities are necessary in order to offset increasing discharges. They are defined case by case and thus endogenously. The trading ratio would thus guarantee the ecological effectiveness of trading activities in contrast to fixed trading ratios. This only holds true, however, if there are no delays between the definition of the trading ratio and the trading activity itself, which could otherwise distort the effectiveness of the trading ratio. At the same time, it causes, higher transaction costs (Section 2.5).

The Pollution Offset System can apply to different *geographical areas*. When adjusting the number of receptors and the simulation model, this system can, without problems span the entire river basin.

The Pollution Offset System can, depending on the ecological objectives, follow an endpoint as well as an instream flow need oriented approach (*orientation*). Depending on the chosen strategy, water quality goals for the receptor points and thus the discharge permits (emissions) need to be subject to a one-off adaptation referring to water quality models.

[209] In the special case where it does not matter where the pollutant is emitted, because it has the same impact on the concentration at all receptor points, i.e. for uniformly mixed assimilative pollutants, the two systems – Ambient Permit System and Pollution Offset System – degenerate into one. Only one receptor point is needed and the dispersion coefficient is identical for each source; this leads to an Ambient Permit System with one single market, where the permits are traded at a one-to-one basis. The potential of trades increases. Also the Pollution Offset System becomes a single market system with a single trading ratio. Carbon dioxide is an example of a uniformly mixed assimilative pollutant (see Krupnick *et al.*, 1983, pp. 242-243).

Immission caps for the receptor points can be differentiated in space without any problems if this is required by the defined objectives of the environmental authority. Differentiated immission caps for different receptor points (space) would be integrated in the simulation model. A differentiation of the immission caps over time is more complicated. Changes in the water quality goal over time can temporarily transform a non-binding receptor point into a binding one and *vice versa*. The latter case would not be a problem; sources could purchase additional permits for free to fill the gap. The former case is more complicated as the absolute immission cap is decreasing. As a result, emission loads and the value of permits must be adjusted. Again, a determination of the permits in term of percentages could be a solution; permitted amounts would be adapted proportionally (Section 6.1). In any case, the simulation model must be adjusted.

The definition of the *emission cap* per receptor point is directly related to the immission cap and thus reflects spatial and temporary aspects of the immission cap. Impacts are calculated individually for each potential change in discharge (time) and for each concerned receptor point (space). Maximum emission caps are thus determined with close reference to the immission load. This only holds true, however, for the case of potential modifications of discharges. It is possible that the same absolute emission discharge will cause a higher or lower ambient pollution level if, for example, water levels or temperature vary. If sources do not change their discharge behaviour, no simulation model would control the impacts of their emissions over time. Nevertheless, in order to guarantee ecological effectiveness, safety margins or similar should apply in order to avoid hot spots. However, safety margins should reflect the worst case; for many situations the margin would thus be set too strictly. Trading potentials would be lost in this case.

Water quantity aspects are incorporated in the water simulation model. It is assumed, that the ambient pollution level resulting from additional discharges is implicitly defined with respect to the actual water level. Changes in the water level, also those caused by extraction by the sources themselves, are exogenously integrated in the system. Again, this only holds true for actual plans on modifications of discharges. Only in discrete cases, i.e. when a source decides to change discharge behaviour, will simulation models apply. In all other cases, actual alteration of water levels is not integrated, although it could influence the water quality.

The prior simulation of all trading activities seeks to ensure a system without hot spots. Individually set water quality caps for the receptor points cannot be exceeded. This assumes, of course, that the simulation model is precise and simultaneous and so that no relevant time-lag between simulation and discharge respectively trading activities takes place. In addition, the analysis has shown that the impacts of discharges are only simulated if changes are planned. But in some cases fixed emission amounts per source could impair the water quality standard if conditions (water flow or temperature) are changing. This is not reflected by the model; the risk of hot spots is thus only avoided if the relation between emissions and immissions is assumed to be constant over time.

Economic Criteria

The economic performance of the Pollution Offset System is quite ambivalent. Krupnick *et al.* (1983) demonstrate that this model comes to a least-cost solution (*cost-effectiveness*) independently of the initial allocation of permits. In contrast, McGartland (1988) criticises that the cost-minimum solution under the Pollution Offset System will be achieved only if all potential gains from trade are exhausted.

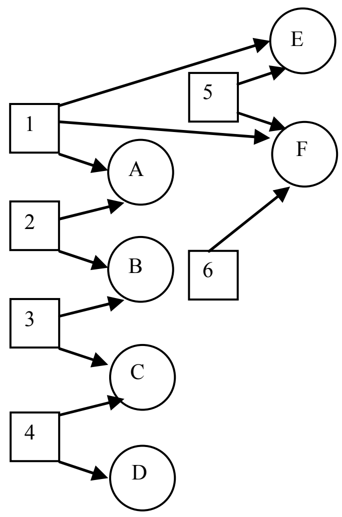

Figure 4-2: Pollution Offset System (POS), Dispersion Characteristics According to McGartland (1988, p. 39)

Principally, this holds true for any trading system.[210] But, in the special case of the Pollution Offset System, this problem becomes particularly relevant as non-binding receptor points are integrated explicitly in the system. Only if all receptor points are binding, will all cost savings be exhausted and cost-effectiveness is given.

The main problem of the Pollution Offset System is related to cost-effectiveness issues and the occurrence of free rider effects. These effects appear due to the fact that sources can always obtain additional discharge permits free of charge as long as they do not violate the quality standard.[211] It is assumed that discharges emitted by source 1 affect the quality at the receptor point A, E, and F (Figure 4-2). It is then supposed that source 1 purchases permits from source 2 (affecting common receptor point A) and from source 5 (affecting common receptor point E and F).[212] As a result, source 2 and 5 must reduce their emissions. The mentioned trade will improve the quality at receptor point B, due to the abatement of source 2, even though this receptor point is not directly affected by source 1. Other sources affecting receptor point B, e.g. source 3, will benefit: they are able to increase their discharges contributing to the quality at receptor point B at no cost as long as the quality standard is not infringed; they are free riding.[213] Free rider behaviour prevents the system from coming to the least-cost solution. To put it in another way: not all trading potentials are exhausted.[214]

210 Uncertainties or other institutional issues could prevent the market from coming to the optimum.
211 This was not the case for the Ambient Permit System. The polluter could keep their surplus of permits in the event that they do not find a buyer.
212 Trading and trading ratios only become obligatory, if a receptor point is a binding one. Otherwise, sources can obtain permits free of charge from the environmental authority. Thus, the number of relevant, i.e. binding receptors, is kept to a minimum.
213 McGartland (1988, pp. 39-41); Klaassen (1996, p. 55) and Hung and Shaw (2005, p. 99). It is assumed that sources are informed about the additional discharge opportunities and thus know the possibility of free riding.
214 Krupnick *et al.* (1983) argue that the efficient solution would be to integrate the sources concerned (affecting B) in the bargaining process. McGartland (1988, p. 239) shows different situations in which bargaining could be profitable for firm 3. One could, of course, include all affected firms in the bargaining process. This leads, however, to a complex system. The advantage of lower transaction costs would be lost.

Pollution Offset System (POS)	
Cost-effectiveness	- Yes, if receptors are binding - Free rider
Dynamic efficiency	- Yes, if caps are binding
Transaction costs	- See discussion of Table 4-7 and Table 4-8 (p. 69-72).

Table 4-6: POS, Economic Criteria

The *dynamic efficiency* also depends on whether or not binding immission caps can be set. As long as additional permits can be obtained free of charge, there is no incentive to implement technical innovation and to increase abatement. Only if binding caps are set for all receptor points and consistently over time, can dynamic incentives function adequately. Again, this is relevant for any permit trading system to be implemented. However, in the case of the Pollution Offset System, non-binding caps are implemented explicitly.

If applied to water issues, also under the Pollution Offset System, an asymmetric position of sources arises. Sources located at the source or at the estuary of the river cannot without restrictions sell or buy permits in order to maximise their benefits. The situation is similar to the results under the Ambient Permit System (Paragraph 4.2.1). The design of the permit trading system cannot solve this problem adequately. Table 4-6 summarises the results of the economic analysis.

Let us now analyse the role of *transaction costs* under the Pollution Offset System. Again, it is not task of this chapter to quantify the exact value of transaction costs. The level of transaction costs are rather to be specified relative to other models.

The determination of the *receptor points* does not cause higher (one-off) transaction costs than for other models, they are assumed to be relatively low and trade-independent.

The *prior simulation* for any potential modification of discharges causes high permanent trade-independent transaction and monitoring costs incurred to the environmental authority; the impact on all other receptor points and thus sources must be analysed before any potential change in discharges.[215] If trading becomes necessary, the trading ratio must be defined (trade-dependent transaction costs). Water quality models thus apply case by case. Dispersion coefficients are implicitly calculated within the simulation in order to evaluate the impacts of single discharge variation. An explicit and complete definition of all dispersion coefficients and trading ratios (each source, each receptor point) as under the Ambient Permit System becomes unnecessary.

A permanent determination of the *real-time data* of emission and immission loads becomes unnecessary in order to make the system work; no additional transaction costs arise.

The *initial allocation* of permits does not integrate relations between emissions and immission. It can thus be carried out without running prior water quality models. This reduces the transaction costs in comparison with models that require the initial allocation of permits to already considers the interdependencies between different parts of the body of water (Paragraph 4.3.1).

[215] We assume that the simulation model can adequately and in time predict violations at any receptor point; this is, of course, one main condition for the practicability of the Pollution Offset System.

The trading ratio, i.e. the ratio of *dispersion coefficients*, is only defined if, in the result of the simulation, trading activities are necessary in order to remain in compliance with the water quality goals, and not for any potential trade. The determination of the *trading ratio* is trade-dependent and endogenous and valid for this single trade. This causes higher, more permanent, transaction costs to the environmental authority as the determination of a fixed trading ratio (exogenous) would do. Førsund and Nævdal (1998, p.404) underline that the endogenous definition of the trading ratio is very complex and causes high transaction costs.

Under the Pollution Offset System, a differentiation of the *immission cap* in space would not cause significant additional transaction costs: different water qualities can be defined for individual receptor points without causing too high additional transaction costs. These spatially differentiated values should then be introduced into the simulation model, which does not appear so costly.

A differentiation of the immission cap over time could causes higher permanent transaction costs if the standard has not been binding before and is then changed to a more stringent one. If the absolute immission cap is lower than actual permits would allow or lower than the actual immission cap, the value of permits must be adjusted.

The determination of the *emission cap* in accordance with the immission cap is assumed to be the task of the environmental authority since permits are defined in terms of emission loads.[216] This causes additional permanent and trade-independent transaction costs. As Paragraph 4.2.1.2 has shown, the total amount of transaction costs may be lower if the environmental authority defines the emission caps with a single and standardised methodology; results are thus comparable.

The requirement to keep the sources informed is already higher than it has been for the Ambient Permit System and causes thus higher permanent (trade-independent) transaction costs for the environmental authority (*information requirements*). Simulations must be carried out and sources need to be informed about the results and trading conditions (free purchase or trading with trading ratios) for each potential change in discharge behaviour. On the other hand, sources would be interested in information about discharge potentials at different receptor points over time; they could benefit from modifications of the discharge behaviour of other sources. Table 4-7 (see below) summarises the results for the analysis of transaction costs at the level of the environmental authority.

[216] The volume of discharge might be different for the same emission load depending on the current concentration of discharged water.

Pollution Offset System (POS)

Transaction Costs (TAC), Environmental Authority	
Determination of receptor points, zones or blocks	
- Higher TAC - Lower TAC	 **X**
Running Water Quality Model (WQM) before any *potential* change in discharges	
- Yes, higher TAC - No, lower TAC	**X**
Real-time data	
- Yes, higher TAC - No, lower TAC	 **X**
Determination of the initial allocation of permits	
- Including Water Quality Model, higher TAC - Not including Water Quality Model, lower TAC	 **X**
Definition of the dispersion coefficient (Water Quality Model)	
- Yes, higher TAC - No, lower TAC	**(X)**
Definition of the trading ratio	
- Endogenous, higher TAC - Exogenous, lower TAC	**X**
Differentiation of the immission cap in space	
- Higher TAC - Lower TAC	 **X**
Differentiation of the immission cap over time	
- Higher TAC - Lower TAC	**X**
Determination of the emission cap	
- Environmental authority, higher TAC - Sources, no TAC environmental authority	**X**
Information requirements	
- Higher - Lower	**X**

Table 4-7: POS, Transaction Costs, Environmental Authority

Transaction costs for sources under the Pollution Offset System are discussed in Table 4-8 (see below). The simulation model analyses whether a certain additional amount of emissions would exceed the cap at any receptor point. The environmental authority thus defines the transformation between emission and immission values (*determination of the emission cap*). The sources do not incur any additional transaction costs in order to calculate or use the dispersion coefficients themselves.

Sources are only concerned with those receptor points whose quality would violate the quality standard as a result of an increase in emissions.[217] This lowers the transaction costs compared to the Ambient Permit System, under which sources must hold a *portfolio of permits*, because each receptor point forms a single market.[218] The costs of permit purchasing do, however, vary: while permits for non-binding caps can be purchased for free, permits for binding receptor points must be purchased on the market by trading. In this case, the value in terms of resulting emission loads after the trading ratio is defined for any single trade.

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[217] McGartland (1988, pp.42-43) demonstrates in an empirical study that the number of relevant, i.e. binding receptors can be very low. If this holds true, this would lead to a higher relevance of the Pollution Offset System.

[218] While this is true for actual trading activities, the discussion above showed that receptor points becoming non-binding through changes in discharge behaviour of other sources could also be relevant. Thus, the information about non-binding receptor points over time is also of high relevance for sources; the transaction costs in form of information costs increase.

Pollution Offset System (POS)

Transaction Costs (TAC), Sources	
Determination of the emission cap	
- Yes, higher TAC - No, lower TAC	X
Portfolio of permits	
- Yes, higher TAC - No, lower TAC	X
Constant adjustment of the permit value	
- Yes, higher TAC - No, lower TAC	X
Trading ratio	
- Endogenous, higher TAC - Exogenous, lower TAC	X

Table 4-8: POS, Transaction Costs, Sources

Permits are already defined in terms of emissions and fixed for a certain period; no adjustment of the value takes place (*adjustment of the permit value*). Monitoring is not very complex. No additional transaction costs arise.

Trading ratios are defined endogenously for each potential trade. They are not fixed and not predetermined in advance. This creates permanent and trade-independent transaction costs for sources: the trading ratio is not known as long as the sources are not obliged to trade which creates uncertainty about the required abatement activity and the price of the permits.

Although the Pollution Offset System explicitly integrates the relationship between emissions and immission, ecological effectiveness cannot be achieved for all initial ecological settings. The Pollution Offset System is not able to adjust emission caps to meet changing conditions; depending on the specific river conditions, this could impair the ecological effectiveness of the system when water quantity aspects are significant.

Furthermore, the Pollution Offset System fulfils the economic criteria only under specific conditions (binding caps). The free rider problematic and the non-binding caps, in particular, prevent the system from coming to the cost-effective and dynamically efficient solution.

Transaction costs incurred by the environmental authority are lower than under the Ambient Permit System. Some transaction costs, mainly those incurred by the environmental authority and for the determination of the trading ratio, are dependent on the number of (potential) trading activities. If these transaction costs are transferred to the sources, they could hinder active trading activities and thus lower the efficiency of the system (Paragraph 3.3.4).[219] However, the level of transaction costs does not appear to be the main disadvantage of the Pollution Offset System.

[219] Additional trade-dependent transaction costs that occur for any permit trading system may become relevant.

4.2.3 Exchange Rate System

4.2.3.1 The Model

A third model, developed by Førsund and Nævdal (1998) and Klaassen *et al.* (1994), was originally developed for the abatement of SO_2, i.e. air pollution control:[220] the *Exchange Rate System (ERS)*, is a more ambient-based approach.[221] Permits allow a given amount of emission. It is assumed that there is a set of environmental constraints, i.e. different receptor points with specific environmental constraints. A fixed and exogenously set 'exchange rate' applies to any transaction between sources and this reflects the different impacts of discharges. Under the Exchange Rate System, no *ex ante* testing of violation at the receptor points through simulations is necessary: trades are allowed as long as the exchange rate applies. The exchange rate is equal to the ratio of the sources' marginal abatement costs in the optimum, i.e. the least-cost solution. The authors assume that the marginal abatement costs differ for the sources depending: depending on the impact that discharges of a certain source have on the quality (immission load), the abatement activity required to achieve a better environmental quality will vary, thus causing different marginal abatement costs.

The Exchange Rate System is presented in this study for the sake of completeness. However, the analysis shows that the Exchange Rate System follows no 'pure' ambient-based system. The ecological effectiveness and the efficiency of this system are low. Only the main criticism is presented here. The model is not integrated in further reflections and discussions.

4.2.3.2 Evaluation

Ecological Dimensions

The definition and application of exchange rates for transactions between sources pretend to follow an ambient-based view; however, exchange rates are defined in terms of the relation between the sources' marginal abatement costs in the optimum. They thus indirectly reflect different cost structures, which are caused by different dispersion conditions. They do not directly reflect the relation between discharged effluents and the impact on the immission level. This approach cannot guarantee the avoidance of hot spots. Therefore, this approach after all cannot be unambiguously classified as ambient-based.

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[220] The model was originally formulated for trades between countries.

[221] For further explanations, see the evaluation part, Paragraph 4.2.3.2.

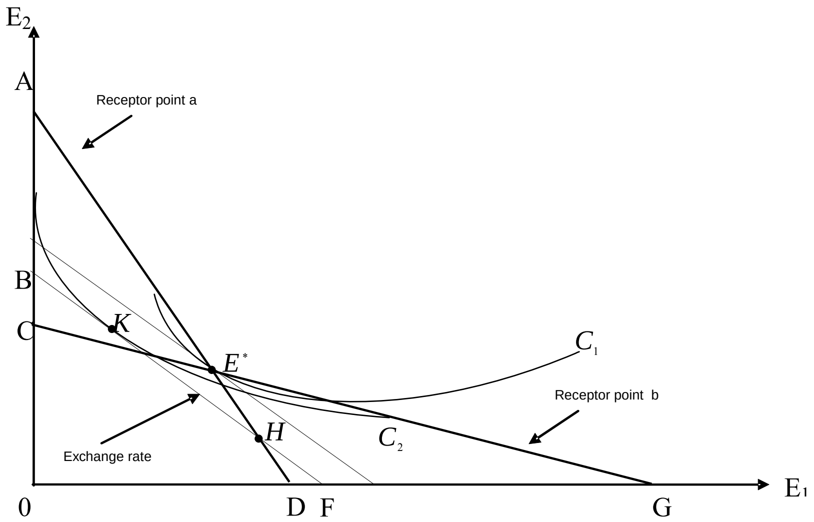

Figure 4-3: Exchange Rate System (ERS), Risk of Hot Spot According to Klaassen (1996, p. 220)

Figure 4-3 illustrates the hot spot problem of the Exchange Rate System. Even if the initial allocation of permits is situated within the receptor constraints ($0CE^*D$), trading activities may result in an allocation outside this area depending on the configuration of the isocost-curves. Assume that trading starts in point H. Trading subject to the exchange rate (BF) could lead to the point K. While the constraint for receptor point a is met in point K, the constraint from receptor point b is violated (Klaassen *et al.*, 1994, p. 221). Exchange rate trading can thus result in a reduction of the ecological effectiveness.

An effective differentiation of the immission cap is not possible. The application of the exchange rates reflecting the marginal abatement costs cannot control the water quality in a differentiated way (Klaassen, 1996, pp. 237-238). Even the differentiation between endpoint or instream flow need constraints cannot be reflected by the model; the element of the exchange rate does not allow the precise achievement of any predetermined objective.

Economic Criteria

The first economic disadvantage of this model is formulated by Førsund and Nævdal (1998, p.409) themselves: they note that only for specific initial allocations can the exchange rate be defined in such a way that it guarantees an efficient result (*cost-effectiveness*). Figure 4-4 (see below) shows this for the case of two sources and two receptors. Suppose that trading starts in point H. Cost savings are possible; trading activities are allowed along the exchange rate line BF. Starting from point H the point E''' would be the attainable least-cost solution with the given exchange rate. E''' differs from E^*, the optimum. Abatement costs C_2 are thus higher than in the cost minimum (C_1). Not all trading potentials can be exhausted. Only if the starting point lies on the line LM, can the least-cost solution in E^* be achieved.[222]

[222] All points on the line LM, except E^* violate one of the constraints set by the receptor points.

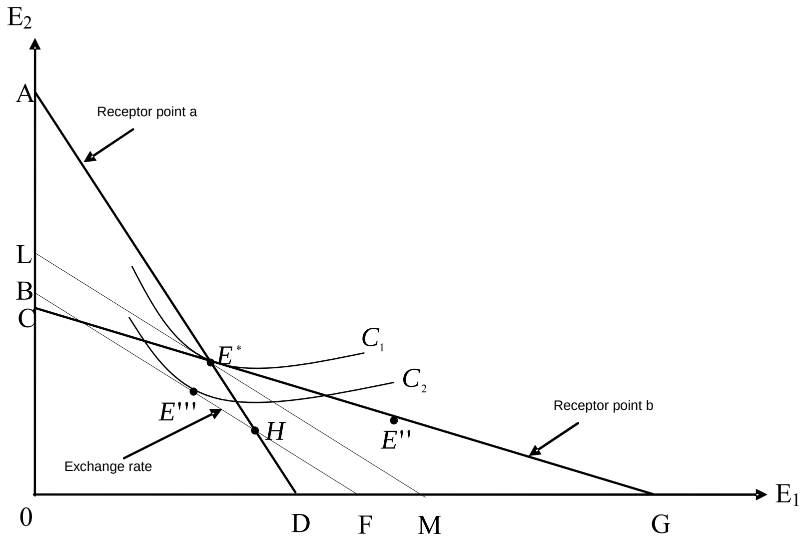

Figure 4-4: Exchange Rate System (ERS), the Criterion of Cost-effectiveness According to Klaassen (1996, p. 222)

Dynamic efficiency is generally given under the Exchange Rate System, even though the incentive to innovate may be biased. The central elements of a permit trading system apply: surplus permits can be sold on the market thus generating benefits. The level of exchange rates, of course, influences trading activities and thus the prices of permits. If exchange rates are not set adequately, i.e. do not reflect the real impact differences, trading prices will be biased. The incentive to innovate is in turn influenced by the market price and would thus also be affected.

The previous analysis has shown that the Exchange Rate System is neither ecologically effective nor economically efficient. Therefore the evaluation of *transaction costs* is abbreviated; only some specific aspects are described below.

The Exchange Rate System uses an exchange rate that is applied when sources are trading with each other. The exogenously fixed exchange rate would principally lower transaction costs for the environmental authority. But, this exchange rate should not be confounded with the (impact) trading ratio. While the trading ratio reflects different impacts of sources' discharges on the ambient pollution level, the exchange rate is defined as the ratio of the marginal abatement costs of two sources in the least-cost solution. Consequently, the cost-minimum solution and the individual marginal abatement costs should be known in advance to calculate the exchange rates. Information costs for the environmental authority under this system are extremely high.[223]

The fact that marginal abatement costs need to be known in order to allow trading at specific rates contradicts the original idea of a permit trading system. The idea of a permit trading is, among others, that the cost-effective solution is achieved even if the environmental authority has no information about individual cost curves. Individual sources would instead use their information about their

[223] Klaassen *et al.* (1994, p. 310) state, that for the case of SO_2 this information on individual abatement costs is available. However, this is not the case for other substances and other environmental media. Differences in river and pollutant characteristics would make it difficult to determine individual marginal abatement costs adequately and precisely. See also Hung and Shaw (2005, p. 98).

individual cost curves to come to the cost-minimum solution by comparing abatement costs and the price of permits. One of the main advantages of the permit trading system in general is thus lost.

The Exchange Rate System is characterised by its low flexibility in integrating specific ecological concerns and the risk of hot spot; the economic performance of the system is low. The element of the exchange rate appears to nullify the advantages of a permit trading system. This model cannot be recommended and is thus not further discussed.

4.2.4 Summary

A first reflection on the theoretical approaches shows that the presented models for non-uniformly mixed pollutants are different in the ecological and the economic evaluation. While the ecological part can only be finally evaluated with respect to the specific ecological settings, e.g. the requirements of differentiation, the economic analysis already shows that economic efficiency is not achieved in any case.

The Ambient Permit System and the Pollution Offset System are ambient-based (*cap type*). They thus fulfil one crucial condition formulated in Section 3.1. Ambient-based models can also be used when the immission orientation is not necessary, for example, if the impact of emissions on immission is one to one. But these models have the general flexibility to differentiate between emissions and immission which would be crucial in most cases. The following results show that the Ambient Permit System is able to reflect a higher differentiation level for the immission cap.

Trading ratios are defined exogenously under the Ambient Permit System and endogenously under the Pollution Offset System. While the former runs the risk of not being ecologically effective by the non-integration of actual conditions, the latter runs the risk of hampering ecological effectiveness by time lags, apart from being very cost-intensive (Section 2.5).

Both models, the Ambient Permit System and the Pollution Offset System, can span the entire river basin (*geographical scope*). It is an ecological or political decision whether the model should be based on the water quality of the sea (endpoint need) or of the river (instream flow need, *orientation*). Again, both, the Ambient Permit System and the Pollution Offset System can be adapted for a sea and/or river orientation.

The Ambient Permit System can integrate *immission caps* that are differentiated over time and/or in space. Depending on the conditions this can be of great importance. This differentiation ensures the avoidance of hot spots. Under the Pollution Offset System, a differentiation of the water quality standard in space appears to be possible: different immission goals can be defined for different receptor points. The differentiation over time appears to be more complicated, at least if the immission cap has already been binding in the past and if the ecological goal should be formulated more stringently than in the past. The application of the Pollution Offset System would thus be proposed if no differentiation over time is required. Under both systems, the *emission caps* can usually be adjusted to differentiated immission caps.

	Ambient Permit System (APS)	Pollution Offset System (POS)
General Design Elements		
Cap type	- Ambient-based	- Ambient-based
Trading ratio	- Exogenous	- Endogenous
Ecological Dimensions		
Geographical scope	- Entire river basin possible	- Entire river basin possible
Orientation	- Endpoint and/or instream possible	- Endpoint and/or instream possible
Differentiation of the immission cap in		
Space	- Possible	- Possible
Time	- Possible	- Restricted
Adjustment of the emission cap in		
Space	- Possible	- Restricted
Time	- Possible	- Restricted
Water quantity aspects	- If at all: implicitly in the dispersion coefficient	- Integrated in the simulation model, not constantly

Table 4-9: APS and POS, Ecological Dimensions

Both models consider *water quantity aspects*, but only once and in an exogenous way. Permanent alteration of the water quantity levels is not considered. Endogenous changes due to diversion activities by the sources are not integrated in these models. Sources would have no incentive to combine their water quantity and quality decisions. Both models cannot adequately reflect permanent changes in water quantity and the impacts on the water quality. They can thus only apply if the variation in the water quantity is not significant or does not significantly influence water quality aspects.

The Ambient Permit System is more flexible with respect to ecological differentiations than the Pollution Offset System. Hot spots can be avoided even for very differentiated trading systems. The Pollution Offset System avoids hot spots only for specific cases; it is not capable of reflecting complex ecological differentiations. Results are summarised in Table 4-9 (see above).

The economic analysis (Table 4-10) of these approaches comes to the following conclusion: for both models, the *cost-effectiveness* depends on the initial allocation. The question of practicability arises. The environmental authority relies on very detailed information; the adequate initial allocation will be met rather by chance. It is assumed that both models generate cost-savings; the optimum can only be reached under certain conditions. The cost-effectiveness of the Pollution Offset System is also hindered by the free rider problem. If free rider behaviour occurs, not all trading potentials can be exhausted; cost-effectiveness is hampered. Under the Ambient Permit System no free rider problem emerges.

The Ambient Permit System corresponds to a permit trading system in its original form and thus generally guarantees *dynamic efficiency*; even if a lack of cost-effectiveness could bias the results. This is not the case for the Pollution Offset System where it heavily depends on the specific conditions whether sources have an incentive to innovate or not.

	Ambient Permit System (APS)	Pollution Offset System (POS)
Cost-effectiveness	- Depending on initial allocation - No free rider	- Yes, if receptors are binding - Free rider
Dynamic efficiency	- Yes, possibly biased	- Yes, if caps are binding
Transaction costs	- See discussion of Table 4-11 and Table 4-12 (p. 79-81).	

Table 4-10: APS and POS, Economic Criteria

The analysis shows that *transaction costs* are different in these models. A comparison without concrete data is difficult; final values need to be defined individually for the specific case. However, the following analysis of the transaction costs gives an overview of general differences in transaction costs structures for the Ambient Permit System and the Pollution Offset System.

Transaction costs for the environmental authority are generally higher under the Pollution Offset System than under the Ambient Permit System (Table 4-11, see below). The *simulation model* before any potential change in emissions under the Pollution Offset System, in particular, causes high transaction costs that do not occur under the Ambient Permit System.

Both systems use dispersion coefficients. While under the Ambient Permit System, *dispersion coefficients* need to be defined for each sources and each receptor point *ex ante*, the Pollution Offset System determines dispersion coefficients case-by-case. Only with potential changes in discharges, do dispersion coefficients apply within the simulation model. Transaction costs for the definition of dispersion coefficients are thus assumed to be lower under the Pollution Offset System.

Trading ratios under the Pollution Offset System are defined endogenously, thus causing higher transaction costs than the Ambient Permit System with exogenous trading ratios. Again, trading ratios under the Pollution Offset System need only to be determined for actual trading activities. Nevertheless, the endogenous determination is cost-intensive.

Under the Pollution Offset System the environmental authority defines the emission cap resulting from specific immission standards (*determination of the emission cap*). Under the Ambient Permit System, sources are assumed to define their emission caps according to the dispersion coefficients determined by the authority; sources thus pay these transaction costs. If the environmental authority is able to pass these transaction costs on to sources, this difference between the models would be nullified.

Information requirements under the Pollution Offset System are higher than under the Ambient Permit System. As many elements are defined endogenously under the Pollution Offset System, sources need to be permanently informed. *Ex ante* and fixed determinations allow for lower information needs. Most transaction costs for the environmental authority are trade-independent.

	Ambient Permit System (APS)	Pollution Offset System (POS)
Transaction Costs (TAC), Environmental Authority		
Determination of receptor points, zones or blocks		
- Higher TAC		
- Lower TAC	X	X
Running Water Quality Trading Model (WQM) before any *potential* change in discharges		
- Yes, higher TAC		X
- No, lower TAC	X	
Real-time data		
- Yes, higher TAC		
- No, lower TAC	X	X
Determination of the initial allocation of permits		
- Including Water Quality Model, higher TAC		
- Not including Water Quality Model, lower TAC	X	X
Definition of the dispersion coefficient (Water Quality Model)		
- Yes, higher TAC	X	(X)
- No, lower TAC		
Definition of the trading ratio		
- Endogenous, higher TAC		X
- Exogenous, lower TAC	X	
Differentiation of the immission cap in space		
- Higher TAC		
- Lower TAC	X	X
Differentiation of the immission cap over time		
- Higher TAC	X	X
- Lower TAC		
Determination of the emission cap		
- Environmental authority, higher TAC		X
- Sources, no TAC environmental authority	X	
Information requirements		
- Higher		X
- Lower	X	

Table 4-11: APS and POS, Transaction Costs, Environmental Authority

Results for the sources' transaction costs are as follows are summarised in Table 4-12 (see below). Sources are not responsible for the *determination of the emission cap* under the Pollution Offset System as is the case for the Ambient Permit System. Furthermore, they are not obliged to manage an entire *portfolio of permits* (Pollution Offset System). Whether the *permit value* changes over time under the Ambient Permit System depends on the final design. Under the Pollution Offset System sources do not need to adapt to changing permit values. The endogenous definition of the *trading ratio* under the Pollution Offset System, however, leads to higher transaction costs for sources (information costs). Most transaction costs for sources are trade-independent under both systems, Ambient Permit System and Pollution Offset System.

The analysis has shown that most of the transaction costs are trade-independent. The model of Stavins (1995) (Paragraph 3.3.4) assumed that the trade-dependent transaction costs, in particular, hinder the system in coming to an efficient solution. Too high costs related to trading activities may prevent sources from trading. Transaction costs that must be paid even if no trade takes place do not directly influence the effectiveness of the system. If, however, total transaction costs caused by the introduction of the trading system are significantly higher than the benefit, the implementation is not advisable.

	Ambient Permit System (APS)	Pollution Offset System (POS)
Transaction Costs (TAC), Sources		
Determination of the emission cap		
- Yes, higher TAC	X	
- No, lower TAC		X
Portfolio of permits		
- Yes, higher TAC	X	
- No, lower TAC		X
Constant adjustment of the permit value		
- Yes, higher TAC	Depending on the final design	
- No, lower TAC		X
Trading ratio		
- Endogenous, higher TAC		X
- Exogenous, lower TAC	X	

Table 4-12: APS and POS, Transaction Costs, Sources

These results lead to the assumption that the Ambient Permit System has a higher potential to adapt to differentiated ecological settings than the Pollution Offset System. At the same time, the Ambient Permit System is confronted with smaller economic problems. Transaction costs for the environmental authority are assumed to be lower under the Ambient Permit System than under the Pollution Offset System. Transaction costs for sources appear to be higher under the Ambient Permit System. They may, however, be justified by a better performance of the system.

In the following section, the Pollution Offset System by Krupnick *et al.* (1983) is excluded from general discussion. The analysis demonstrated that cost-effectiveness depends heavily on the initial allocation of permits; only binding caps would guarantee cost-effectiveness and dynamic efficiency. At the same time, the Pollution Offset System is characterised by a free rider problem that hampers efficiency. Ecological flexibility is restricted. Immission caps could be adjusted in space but not in time; the adjustment of the emission caps is restricted. The Pollution Offset System would thus only be relevant for cases that do not require these flexibilities.

4.3 River Specific Approaches

The models presented in Section 4.2 were originally developed for air pollution control even though they can be converted for the application in water pollution control. These models have been formulated for non-uniformly mixed pollutants; this type of pollutants is most relevant for river pollution. The following river specific permit trading models are directly based on the mechanisms of the theoretical air quality trading systems.

Within the case of non-uniformly mixed pollutants there are at least two further characteristics that must be integrated in a water quality model in order to adequately reflect the specific properties of a river. On the one hand, a river is a flowing body of water thus transporting the pollutants within the water uni-directionally to the lowest level (Section 2.4). This is not the case for the air medium where non-uniformly mixed pollutants disperse in all directions. This property simplifies water quality modelling of non-uniformly mixed pollutants compared to air quality modelling (Paragraph 4.3.1).[224] On the other hand, the link between quantity and quality aspects, discussed in Section 2.4, signifies a specific property of water that does not have the same relevance for the air. Two relevant river specific models are presented in the following which illustrate how these water specific aspects can be incorporated in the design of a water quality trading system.

4.3.1 Trading Ratio System

4.3.1.1 The Model

The *Trading Ratio System (TRS)* of Hung and Shaw (2005) represents a zonal approach: the river basin is divided into n zones ($n \in N^{+}$) using the river bank as basis (Figure 4-5, see below).[225] The dispersion characteristics of discharges of sources within a zone are assumed to be closely related.[226] For transactions between these zones, a (n impact) trading ratio applies, which is exogenously determined by the dispersion coefficients and thus reflects the impact of individual discharges on the water quality level. This trading ratio is promulgated *ex ante*.

The environmental authority sets a 'zonal total (emission) load standard' E_j for each zone.[227] The determined emission load is derived from a given water quality standard (ambient-based). The environmental authority defines zonal emission caps one by one from the upstream to the downstream zones "such that the zonal emission cap is equal to the zonal total load standard minus the emission load transferred from the upstream zones" (Hung and Shaw, 2005, p. 87). In the following, the caps are converted into their equivalent amounts of 'zonal tradeable discharge permits' ($\overline{T}_j$), which are defined in terms of their original zonal location j. The first zone obtains discharge permits equal to the zonal

[224] Results would be different for uniformly mixed pollutants (Paragraph 2.3.1)

[225] For simplification the zones are ordered by location and only one representative discharger is located in each zone. The most upstream zone is indexed by 1, the most downstream by n. This symbolic simplification will not lose any of the generality. Zones can be varied in size in order to reflect dispersion conditions correctly.

[226] Zones can be different in size in order to reflect similar dispersion processes within zones.

[227] These standards are derived from the Water Quality Standards determined for the river basin. Hung and Shaw (2005, p. 86, fn. 6): "It is not necessary for the water quality standard and the total load standard to be the same for each zone and all the time. The environmental authority could, for example, prescribe more stringent water quality standards and total load standards in densely populated areas and protection areas of drinking water sources, or more lenient total load standards during seasons in which the assimilative capacity of the river is stronger".

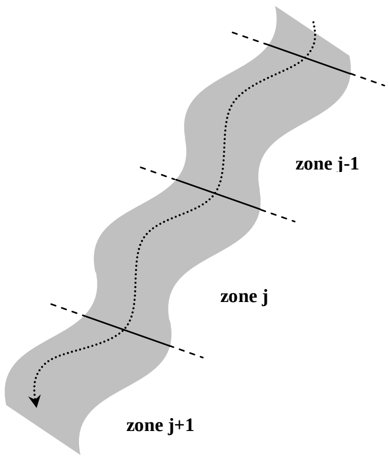

Figure 4-5: Trading Ratio System (TRS), Zonal Approach

total load standard, $\overline{T_1} = E_1$. Zones following downstream obtain permits depending on the zonal total load standard E_j and the weighted upstream emission load transferred:

$$\overline{T}_j = E_j - \sum_{k=1}^{j-1} t_{kj} \overline{T}_k \,.[228]$$

t_{kj} is the dispersion coefficient, indicating the contribution that one unit of discharge from zone k makes to the total load in zone j. At the same time, t_{kj} equals the trading ratio, indicating by how much a source in zone j can increase its discharges if he purchases one unit of $\overline{T_k}$ from any other discharger.[229]

In some cases the downstream zone may be a 'critical zone'; in this case the pollution load transferred from the upstream zone is higher than the zonal total load standard E_j for this zone. Thus, the cap in the upstream zone must be set more stringently. Analytically, this means that, if

$$t_{(j-1)j} E_{j-1} > E_j \,,$$

the zonal tradeable discharge permit for the critical zone j will be fixed at $\overline{T}_j = 0$, i.e. no discharge is allowed. For the upstream zone $j-1$ zonal tradeable discharge permits correspond to

$$\overline{T}_{j-1} = {E_j}\big/{t_{(j-1)j}} - \sum_{k=1}^{j-2} t_{kj} \overline{T}_k$$

and are thus set more stringent in order to guarantee for the water quality standard in zone j.

Compliance is controlled by monitoring the emissions e_j of each source. Exact measuring of the emission on the individual source level is regarded as possible. Emissions discharged into the river

228 Assume that $k < j$; i.e. zone k is an upstream zone to zone j.

229 This model also integrates the assumptions of the upstream-downstream model according to van Mark *et al.* (1992) in Section 2.4.

should be equal to or lower than the zonal tradeable discharge permit plus the tradeable discharge permits sold to downstream zones. The Trading Ratio System thus postulates

$$e_i \le \overline{T}_i + \sum_{k=1}^{i-1} t_{ki} T_{ki} - \sum_{k=i+1}^{n} T_{ik} \, .$$

4.3.1.2 Evaluation

Ecological Dimensions

The Trading Ratio System avoids some problems of the trading approaches discussed above, as can be seen in Table 4-13 (see below). This model is an ambient-based approach (*cap type*): zonal total load standards are determined in terms of emission loads; the allocation of permits and the trading ratio guarantee the achievement of a previously determined water quality standard, i.e. immission load.

Trading ratios are defined exogenously by means of the dispersion coefficients. While this reduces the transaction costs, such a determination of trading ratios could affect the ecological effectiveness in a negative way; changing dispersion processes are not reflected (Section 2.5).

This approach is designated to span entire river basins (*geographical scope*). The emission loads are set with respect to the maximum emission load of other zones located upstream and downstream; the Trading Ratio System thus follows instream flow needs. It would also be possible to further determine these caps with respect to the water quality of the sea (endpoint).

The water quality (*immission cap*) and thus the total zonal load standards (emission cap) can be defined individually for each zone. The system is predestined to use spatially differentiated immission caps. The *emission caps* are derived from the immission cap and would thus also be spatially differentiated. Trading ratios reflect the dispersion characteristics between trading partners.

While this system appears suitable for integrating spatially differentiated immission caps, a differentiation over time seems to be more complicated.[230] The emission cap in the Trading Ratio System is derived from predetermined water quality standards. The relationship between water quality and discharges is thus taken into account. However, the relationship between immission and emission loads is assumed constant and is reflected by the initial allocation. Any modification of the immission settings would ask for the adjustment of the initial allocation and of all total zonal load standards. Changes over time are thus quite complex and costly. It would be reasonable to do so once or twice a year; however, frequent changes in the relationship, caused by water flow or temperature variation, for example, cannot be reflected in this system without going to too much effort.

[230] Hung and Shaw (2005, p. 88) mention in a footnote that changes in the water quality standard over time are possible and that the initial allocation of permits would be changing over time as a result. It is, however, assumed that permanent changes are not reasonable in this model.

Trading Ratio System (TRS)	
General Design Elements	
Cap type	- Ambient-based
Trading ratio	- Exogenous
Ecological Dimensions	
Geographical scope	- Entire river basin possible
Orientation	- Endpoint and/or instream possible
Differentiation of the immission cap in	
Space	- Predestined
Time	- Restricted
Adjustment of the emission cap in	
Space	- Predestined
Time	- Restricted
Water quantity aspects	- Only for the initial allocation, trading ratio exogenous

Table 4-13: TRS, Ecological Dimensions

The integration of changes in *water quantity* under the Trading Ratio System is not mentioned by Hung and Shaw. Water quality models apply to the transfer from immission caps to emission caps. Actual water quantity aspects would thus be integrated exogenously by simulations. However, the model implicitly assumes that the relationship between discharges and the ambient pollution level remains identical over time. Alteration of the water quantity over time would affect the remaining zonal load potentials over time. This is not reflected by the model. The value of discharge permits is defined once, and does not (often) change anymore, although variation in water quantity could permanently influence the impact of discharges on the ambient pollution level. Integrating these changing conditions into the system would require the adjustment of emission caps over time. The immission caps to be achieved do not necessarily change; but lower water quantities, for example, would require lower maximum emission loads in order to achieve the same immission cap. As we have seen above, a differentiation of emission caps over time would not be easy to implement using this model as the entire allocation of loads and permits must be adjusted. The model is thus more relevant for cases without permanent and significant changes in water quantity levels.

The step-by-step definition of the cap allows the setting of binding caps right from the beginning. Effects of discharges on other zones are included implicitly. Permits are allocated individually; an increase in emissions must be covered by additional permits. Impact trading ratios apply. The risk of hot spot formation is avoided. But again, this is only the case if the relationship between emissions and immissions is (nearly) constant over time. In cases with permanent changes in this relation, the Trading Ratio System cannot guarantee a world without hot spots.

Economic Criteria

The results of the economic analysis are presented in Table 4-14. The Trading Ratio System is *cost-effective* and achieves the specified environmental quality standard at the lowest total marginal abatement costs for any initial allocation. Benefiting from the uni-directional nature of water pollution, the cap setting strategy of the Trading Ratio System is able to set binding standards for each zone: the upstream permit fully accounts for its impact on downstream zones (Hung and Shaw, 2005, p. 88). This guarantees an efficient market solution. Furthermore, all discharge permits are allocated individually, an increase in emissions forces sources to purchase permits; no free rider behaviour occurs. Even if the selling of upstream zonal permits to downstream dischargers improve the situation upstream and in

Trading Ratio System (TRS)

Cost-effectiveness	- Yes - No free rider
Dynamic efficiency	- Yes
Transaction costs	- See discussion of Table 4-15 and Table 4-16 (p. 86-89).

Table 4-14: TRS, Economic Criteria

midstream zones, dischargers located there cannot free ride. One could think that they can increase their discharge without impairing the quality standard in their zone. But: this is not permissible since downstream zones are still binding after these trading activities; any additional discharges in the upstream or midstream zone would lead to a infringement of the quality standard in the downstream zone. Cost-effectiveness is thus not biased by free rider behaviour.

Trading ratios apply in order to allow trading between zones. The maximum number of participants is thus guaranteed. The typical problem of regional markets with a too small number of participants is not relevant.

As the Trading Ratio System is characterised by the typical elements of a permit trading with binding caps, the criterion of *dynamic efficiency* is fulfilled. Any increase in discharges requires the purchase of permits; additional emissions are not free. Sources have the incentive to lower discharges through technical innovation: each unit abated allows permits to be sold on the market.

Under the Trading Ratio System, a zonal approach, the location of sources influences the trading opportunities. Permits purchased from sources located upstream only influence the quality in a given zone. While the first upstream zone can only sell permits, the last downstream zone can only buy permits with an effect on discharge opportunities.[231] The first zone upstream can thus not increase the discharge opportunities; only purchase of upstream permits enables an increase of effluents. Even if the last source decreases effluents, it cannot sell permits because there is no source downstream that would benefit from this reduction. The trading situation is thus asymmetric. If more than one source is located within a single zone, these sources can trade with each other regardless of the zone position. The number of potential trading partners is, however, lower than for sources situated in the middle of the river. The importance of competition effects must be studied individually for any specific planned trading system.

The level of *transaction costs* is again discussed in comparison to other models. Let us begin with the transaction costs for the environmental authority (Table 4-15). Under the Trading Ratio System, *zones* must be defined in an adequate way. Sources within a zone should have similar impacts in discharges in order to allow the system to find adequate trading ratios for trading activities between zones and to avoid hot spots.[232] Also water use aspects or ecological aspects should be integrated in the zone formation process if ecological objectives require a spatial differentiation depending on some of these aspects. This setting is more complex than the definition of receptor points under the Ambient Permit

[231] Sources can buy and sell permits from any source if they want to participate in trading. However, permits purchased from downstream sources will not create additional discharge opportunities. Personal communication to the author by Daigee Shaw, Institute of Economics, Academia Sinica, Taipei, Taiwan, ROC, May, 05, 2005.

[232] See also Tietenberg (1995, pp. 104-105) for the case of air pollution. The flowing characteristics relativises central aspects of Tietenbergs argumentation; pollutants flow uni-directional within the river and are not dispersed to all directions as it is the case for the air.

Trading Ratio System (TRS)	
Transaction Costs (TAC), Environmental Authority	
Determination of receptor points, zones or blocks	
- Higher TAC - Lower TAC	X
Running Water Quality Model (WQM) before any *potential* change in discharges	
- Yes, higher TAC - No, lower TAC	 X
Real-time data	
- Yes, higher TAC - No, lower TAC	 X
Determination of the initial allocation of permits	
- Including Water Quality Model, higher TAC - Not including Water Quality Model, lower TAC	X
Definition of the dispersion coefficient (Water Quality Model)	
- Yes, higher TAC - No, lower TAC	X
Definition of the trading ratio	
- Endogenous (higher TAC) - Exogenous (lower TAC)	 X
Differentiation of the immission cap in space	
- Higher TAC - Lower TAC	 X
Differentiation of the immission cap over time	
- Higher TAC - Lower TAC	X
Determination of the emission cap	
- Environmental authority, higher TAC - Sources, no TAC environmental authority	X
Information requirements	
- Higher - Lower	 X

Table 4-15: TRS, Transaction Costs, Environmental Authority

System or the Pollution Offset System and thus causes higher one-off (trade-independent) transaction costs to the environmental authority.

The Trading Ratio System avoids, as do most of the other systems, running a *simulation model* before each potential change in discharge behaviour. This lowers the transaction costs for the environmental authority. No measurement of *real-time data* is necessary as it is the case for other models (Paragraph 5.3.3).

Under the Trading Ratio System, the determination of the *initial allocation* of permits already includes interdependencies between zones. The impacts of changes in discharges in one zone on the ambient pollution level in other zones are reflected by permits. In order to precisely determine the initial allocation the use of a water quality model is required. This initial permit setting is more complex than it is for most of the other models. It thus causes higher one-off (trade-independent) transaction costs for the environmental authority. At the same time, this step-by-step setting avoids high transaction costs for the sources; no portfolio of permits is necessary.

A water quality model is needed in order to define the *dispersion coefficients*. These are relevant for the initial setting of permits, but also for the definition of the exogenous (impact) trading ratios. This causes trade-independent, one-off, transaction costs for the environmental authority.

The *trading ratio* is defined exogenously. Trading ratios are not derived endogenously for each single trade and they are fixed for a certain period of time. Thus, the transaction costs paid by the environmental authority for the determination are relatively low and trade-independent, the definition is directly based on the dispersion coefficients.

While this system is predestined to have spatially differentiated *immission caps*, the spatial setting of water quality standards would cause additional one-off (trade-independent) transaction costs for the environmental authority. Depending on the water use and/or other aspects different immission caps must be defined (Paragraph 3.2.2). This causes higher costs, than the definition of one fixed cap for the entire river. Again, the additional costs could be justified by improved ecological performance. The value of permits (*emission caps*) would be defined according to the determined immission caps.

A differentiation of water quality goals over time would cause higher transaction costs, because the initial allocation (the permit value in emissions) has to be adjusted according to these new ecological settings. This would be reasonable if carried out once or twice a year, e.g. summer versus winter time. Constant changes in the immission cap would cause too high transaction costs, including monitoring costs, for the environmental authority.

The initial setting of the permits enables the environmental authority to define the actual emission value of the permits in order to fulfil the water quality standards (*determination of the emission cap*). These transaction costs are thus incurred by the environmental authority, one-off and trade-independent.

The necessity of informing sources is no higher than for other permit trading systems: sources need to be informed about the permit values (*information requirements*)[233] and allocation, trading processes and trading ratios. No additional information costs are incurred.

Turning to the transaction costs which arise for sources, their level is moderate (Table 4-16). No transaction costs arises when converting the immission values to emission caps at the source level (*determination of the emission cap*). Permits are defined as emission loads by the environmental authority. The Trading Ratio System avoids further transaction costs for the sources: they do not need to hold a *portfolio of permits* as they have to do under the Ambient Permit System. Permits are defined with respect to specific zones and already integrate the impacts on other zones. A modification of discharges does not require bargaining with the sources of all affected zones. The *value of permits* does not (often) change over time as doing this would be too costly; any modification of the immission cap over time would need to be followed by a new initial allocation.[234] Monitoring needs for sources are rather low.

[233] The determination process of the permit values is complex; but sources only need to be informed about the final value, not about the determination.

[234] We have seen that the differentiation of the immission cap over time is very complex. It would thus occur very rarely. Values of permits can thus be defined as constant over time.

Trading Ratio System (TRS)	
Transaction Costs (TAC), Sources	
Determination of the emission cap	
- Yes, higher TAC	
- No, lower TAC	X
Portfolio of permits	
- Yes, higher TAC	
- No, lower TAC	X
Constant adjustment of the permit value	
- Yes, higher TAC	
- No, lower TAC	X
Trading ratio	
- Endogenous, higher TAC	
- Exogenous, lower TAC	X

Table 4-16: TRS, Transaction Costs, Sources

Further on, the *trading ratio* is defined exogenously. This reduces the pre-eminent, more trade-independent, transaction costs in comparison with the Pollution Offset System which determines the trading ratio endogenously. Sources do not need to form expectations; trading ratios are fixed *ex ante*, no additional transaction costs arise.

The analysis of transaction costs shows that nearly all transaction costs under the Pollution Offset System are trade-independent. Trading activities themselves are thus not hampered by too high specific transaction costs. Again, general transaction costs that occur for any permit trading system to be implemented may be relevant. Very little additional trade-dependent transaction costs exist under the Trading Ratio System.

4.3.2 Integrated Water Quantity/Quality System

4.3.2.1 The Model

The link between the water quantity and the water quality aspects and the relevance for permit trading systems has been discussed in Section 2.4. Weber (2001) developed a theoretical model, called *Integrated Water Quantity/Quality System (IQQS)*, that incorporates water quantity aspects into a water quality trading approach. By combining a water (quantity) permit trading with a water quality permit trading system, the model includes endogenous changes in water levels, i.e. water extractions by sources. The direct impact of (endogenous) alteration of water flow on the resulting ambient pollution level is thus reflected in this model. Exogenous and stochastic changes in water flow, e.g. due to weather conditions, are explicitly excluded. This simplification has no impact on the general results of the model. Depending on the real conditions this aspect can still be added to the trading system.

The Integrated Water Quantity/Quality System is ambient-based. Permits are defined in terms of immissions rather than emissions. Participating sources can be water users and/or dischargers. They are located along a river ordered as $i = 1,...,n$ by increasing distance from the source of the river.

Water flows at the source are denoted as v_0. $v(i)$ indicates the amount of water available for diversion at the i th location and only depends on the consumption of water by upstream users. The total amount of water consumed by user i is equal to

$$c(i) = \left(1 - R^i\right)s(i),$$

where $s(i)$ indicates the amount of water diverted; $c(i)$ indicates the amount of water consumed. R^i is an exogenously defined return flow parameter and reflects the difference between diverted and consumed water, i.e. the water flowing back into the water system. The amount of water available for diversion at any point i along the river evolves according to the equation

$$v(i+1) - v(i) = -(1 - R^i)s(i).$$ [235]

In this model, it is assumed that water quality q at any point downstream from a specified point source of pollution is determined by a first-order difference equation. Thus the difference in water quality between users i (upstream) and $i+1$ (downstream) can be written as the site-specific function

$$q(i+1) - q(i) = f^i\left(c(i), e(i), v(i), q(i)\right).$$ [236]

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[235] This equation can be derived as follows: the amount of water available for diversion at the i th location minus the consumed amount of water at this location results in the amount of water available for diversion at the adjacent downstream location $i+1$: $\upsilon(i+1) = \upsilon(i) - c(i)$. This can be converted into the following equation: $\upsilon(i+1) - \upsilon(i) = -c(i) = -(1 - R^i)s(i)$.

[236] The conversion of immission loads (damages) to emission caps is not discussed in this approach. The function f, however, reflects the relation between emission at a specific location with specific conditions and the resulting immission load, a dispersion function.

Weber assumes that $f^i(c(i), e(i), \nu(i), q(i))$ is decreasing and strictly concave in c and e, and increasing and strictly concave in ν and q for all users, i.e. water quality is decreasing with increasing water consumption and discharges, whereas the assimilative capacity of the river at site i is increasing with increasing water quality or water flow.[237] The level of overall quality at any intake for user i can be written as

$$q(i) = \sum_{j=1}^{i-1} f^j(c(j), e(j), \nu(j), q(j)) + q_0 .$$

In this model two constraints for the allocation problem exist: the endpoint constraint and the instream flow need constraint.[238] The endpoint constraint defines a minimum quantity $\bar{\nu}$ and quality $\bar{q}$ of water which must be left after the nth user, e.g. at the estuary or mouth of the river. Instream flow need constraints specify minimum levels of flow $\tilde{\nu}$ and quality $\tilde{q}$ which must be maintained along the river. It is assumed that these constraints are set optimally by the environmental authority. The instream flow need constraints can be indicated as

$$\nu(i) - \frac{c(i)}{(1 - R^i)} \geq \tilde{\nu}$$ [239] and

$$f^i(c(i), e(i), \nu(i), q(i)) + q(i) \geq \tilde{q} .$$

$B^i = B^i(c(i), e(i), \nu(i), q(i))$ is the benefit function of each source/user which depends positively on the amount of water consumed, the emissions discharged, and the quantity and quality of water at the point of diversion.[240] If source i reduces water flow by means of additional diversion, this source benefits from additional water consumption. At the same time, costs of meeting the water quality constraint are increased due to lower water flow. This means that changes in the water diversion of a single source will have a significant impact on the water flow. The actual value of permits in terms of emissions varies with the water flow. This interdependency between water quality and endogenous changes in water quantity is directly taken into account by the model; even though the exact mechanism of emission adjustment is not discussed in further detail.

Upstream water diversion or pollution generates negative externalities for all downstream users through the state variables $\upsilon(i)$ and $q(i)$. The number of users affected by these externalities decreases when travelling downstream; the social cost of input use (diversion or pollution) declines from upstream to downstream, *ceteris paribus*. A shift in input use from upstream to downstream user i increases the

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237 This specification allows water quality to improve between users. That is, it is possible for $f^i(\cdot) > 0$, even if $c(i)$ and $e(i)$ are positive.

238 For the basic discussion of these constraints, see Paragraph 3.2.2.

239 The equation requires that the water flow at point i minus the amount of diverted water is equal to or greater than the minimum level of water flow. Return flows are thus not integrated; positive return flows would increase the water flow. Due to precautionary reasons, the constraint is formulated in a more stringent way. The uncertainty in the exact level of the return flows could otherwise provoque a violation of the minimum flow level.

240 The environmental authority maximises the total benefits of allocating water and pollution permits along the river by solving the control problem $\underset{s(i),e(i)}{Max} \sum_{i=1}^{n} B^i(c(i), e(i), \nu(i), q(i))$, subject to the quality and quantity constraints. Diversion s and discharges e are control variables; water flow υ and water quality q are state variables that are only indirectly influenced by sources' activities (Weber, 2001, pp. 57-59).

volume and quality of flows at site i, thus reducing i's instream flow need constraints and the shadow costs of allocating water or pollution. In consequence, the financial value of permits declines from upstream to downstream.

The market for tradeable permits covers water quantity and quality permits. An aggregate level of water (quantity) permits is allocated,

$$\overline{W} = \sum_{i=1}^{n} w_i^0$$

where w_i^0 is the initial allocation to each user i and

$$\overline{W} = v_0 - \overline{v}$$

is the total amount of water available for consumption. User i is in compliance if

$$w(i) \geq c(i).$$

The environmental authority also allocates water quality permits:

$\overline{D} = \sum_{i=1}^{n} d_i^0$ where

$$\overline{D} = q_0 - \overline{q}.$$

User i generates a permit that can also be traded, if $f^{i0}(\cdot) > 0$. In the model $d(i)$ equals the total number of pollution permits held by user i. User i is in compliance when

$$d(i) \geq -f^{i}(c(i), e(i), v(i), q(i)).$$

The market is assumed to be perfectly competitive. In the equilibrium there is no incentive for any pair of users to trade; no one holds excess permits in equilibrium.

Users control the flow and quality of water at their location by trading with other zones. However, $\upsilon(i)$ and $q(i)$ do not depend on downstream water use or pollution, therefore an asymmetry is introduced with respect to the willingness to pay for upstream versus downstream permits. One can show, that there is no incentive for non-adjacent trades for the users. Benefits from adjacent activities are higher.

4.3.2.2 Evaluation

The following evaluation of the Integrated Water Quantity/Quality System will focus on the relevant aspects of the water quality trading part, since this is the concern of this study. The water trading part is not discussed in details.[241]

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241 For the analysis of water (quantity) trading systems, see, for example, Bjørnlund (2005), Bjørnlund and O'Callaghan (2005) or Bauer (2004).

Integrated Water Quantity/Quality System (IQQS)	
General Design Elements	
Cap type	- Ambient-based
Trading ratio	- Not specified
Ecological Dimensions	
Geographical scope	- Entire river basin possible
Orientation	- Endpoint and instream integrated
Differentiation of the immission cap in	
Space	- Possible
Time	- Restricted
Adjustment of the emission cap in	
Space	- Possible
Time	- Possible
Water quantity aspects	- Integrated, endogenous

Table 4-17: IQQS, Ecological Dimensions

Ecological Dimensions

Table 4-17 shows the results of the ecological evaluation. The Integrated Water Quantity/Quality System is ambient-based (*cap type*); permits are defined in terms of immission caps. The model explicitly combines endpoint constraints with instream flow need constraints. This integration of both constraints is the most realistic. In the event that the environmental authority decides to include only one of these constraints, this could easily be reflected by the Integrated Water Quantity/Quality System through setting one of these constraints zero.

Weber (2001) does not explicitly mention the application of a *trading ratio*, even though the model bases on the Ambient Permit System where trading ratios are integrated. For a further discussion on that see the economic evaluation below.

Generally, the model is appropriate to cover the entire river basin, even if the original version does not integrate infeeding branches for simplification (*geographical scope*). An extension to the entire river basin would be possible without changing the results.

The Integrated Water Quantity/Quality System does not preview a differentiated maximum *immission cap* (nor in space neither over time) in this version. Only one value for the instream water flow $\tilde{\upsilon}$ and the instream water quality $\tilde{q}$ is assumed and integrated in the formal maximisation problem. This model could, however, be adjusted through the introduction of differentiated instream water flow and quality constraints (e.g. $\tilde{\upsilon}_{ij}, q_{ij}$ for different caps in space i and/or over time j). The definition of different caps in space would be integrated from the outset; the central aspects of the model and thus the results do not change. A differentiation over time is more complex as the total permit allocation (defined in immission loads) needs to be adjusted (see Section 6.1).

Emission caps for sources are adjusted flexibly according to the changing conditions (over time and in space) in order to reach a constant immission cap, even eventually spatially differentiated. Under a differentiated immission cap over time, additional adjustments of the emission cap due to changes in the total permit allocation would become necessary.

The Integrated Water Quantity/Quality System is the first model that combines water quality and *water quantity* issues in a single model. A water (quantity) market and a quality market are combined. The model thus explicitly reflects impacts on water quality (constraints) from water diversion. A modification of the permit allocation of the water (quantity) market changes the conditions at the water quality market. This is fully considered in this model and thus sets different incentives for sources than other models do.

The model assumes that a variation in the individual water diversion influences the water flow[242] and that changes in water quality constraints due to differences in the water diversion behaviour of a single source should be burdened to this same source by means of stricter individual discharge constraints. Alteration of water levels is assumed to be endogenous. This is a main difference to approaches which consider exogenous changes in the water level and thus burden all sources equally with higher or lower exogenous discharge constraints. We shall deal with such an approach in the case study on Australia (Paragraph 5.3.3).

The Integrated Water Quantity/Quality System explicitly excludes the aspect of stochastic, exogenous alteration of the magnitude of water flow, e.g. caused by weather conditions. However, particularly these exogenous aspects could be important for specific rivers. Generally, exogenous variation in water flow could be integrated in this model. But: stochastic (exogenous) alteration of the water flow would change the value of permits for all sources; quality constraints become more (less) stringent with lower (higher) water flow. Sources have no influence on this part of the changes in the permit value. The adjustment process of emission caps to achieve the same immission cap could be the same than for the endogenous variation in water flow.

An appropriate implementation of the system should avoid the problem of hot spots by allocating permits individually, setting binding caps and ensuring compliance. In this model, in particular the question of trading from downstream to upstream becomes relevant. This kind of trading is not explicitly forbidden. With regard to the ecological effectiveness this kind of trading is, however, dangerous. Modifications of the behaviour of downstream sources have no impact on the water flow or water quality situation for upstream sources. This is particularly important for water pollution control. If an upstream source purchases a downstream pollution permit and increases its pollution, the immission load at this location changes. If the cap has been fulfilled previously, this additional discharge would lead to an exceeding of the water quality standard. As may be reminded, under the Trading Ratio System the trading from downstream zones to upstream zones is therefore forbidden (Paragraph 4.3.1). It is unclear whether the lower willingness to pay for downstream permits mentioned under the Integrated Water Quantity/Quality System will alone be adequate to prevent trading of permits from downstream sources to upstream ones.

Economic Criteria

The economic evaluation is satisfactory (Table 4-18). The decentralised market for water quantity and quality permits in this model with the environmental constraints is *cost-effective*. The price will bring the market to the least-cost solution by reflecting different abatement costs. All third party effects are internalised: "When downstream users purchase upstream rights, they create positive benefits for all intermediate users. At the same time, intermediate users are willing to accept to sell the same license downstream since by selling the license downstream they will still enjoy the benefits of increased flows and quality at their own site. Thus the positive benefits generated from shifting consumption downstream are fully captured through reduced prices to downstream buyers. Similarly, the external

242 This has, of course, to be checked carefully for any river to be chosen for the introduction of a water quality trading system.

Integrated Water Quantity/Quality System (IQQS)	
Cost-effectiveness	- Depending on initial allocation (see Ambient Permit System) - No free rider
Dynamic efficiency	- Yes, if caps are binding
Transaction costs	- See discussion of Table 4-19 and Table 4-20 (p. 95-99).

Table 4-18: IQQS, Economic Criteria

costs associated with transfers of downstream rights to upstream users are captured by means of the increasing prices for upstream licenses" (Weber, 2001, p. 62).

When trading, sources consider that discharges from different locations (upstream versus downstream) have different impacts on the water flow and quality and thus cause different social costs; marginal abatement costs differ respectively. Trading activities guarantee that water use and effluent discharges are avoided where it can be done at least cost. The solution is thus cost-effective.

The Integrated Water Quantity/Quality System is based on a 'Montgomery-style market' (Weber, 2001, p. 54). If, however a pure Montgomery model with a non-degradation principle is chosen, cost-effectiveness would strongly depend on the initial allocation (Paragraph 4.2.1, APS). It is thus, from an economic point of view, advised to avoid the integration of a non-degradation principle.

The model avoids free rider behaviour if all water quality caps are binding. Only in this case does every increase in emissions effectively force the sources to purchase permits and to avoid free rider problems. Under the assumption that the constraints are set 'optimally' by the environmental authority, binding caps would be guaranteed.

If the Integrated Water Quantity/Quality System follows a pure Montgomery-style model, asymmetric trading conditions for sources as under the Ambient Permit System can not be avoided (Paragraph 4.2.1). The first source upstream cannot profit from reductions of sources upstream as no further source is located there. The last source downstream cannot sell surplus permits as no sources follow downstream. This asymmetry may be avoided if the model is supplemented by elements of other (practical) trading systems that will be discussed in Chapter 5 (Section 6.1).

For the Integrated Water Quantity/Quality System, the *dynamic efficiency* depends also on the cap setting. If caps are not set bindingly and free rider behaviour becomes relevant, the incentive to innovate is biased. An 'optimal setting' of the caps and a functioning price setting process would guarantee the incentive to innovate *ceteris paribus*.

Transaction costs caused by the Integrated Water Quantity/Quality System are again discussed in comparison to the other models and separated for the environmental authority and the sources (Table 4-19, p. 97 and Table 4-20, p. 98). Transaction costs for design elements that are not explicitly formulated under the Integrated Water Quantity/Quality System by Weber (2001), but mentioned in the Ambient Permit System, that forms the formal basis, are shown in brackets, as they cannot be determined.

If the Integrated Water Quantity/Quality System uses *receptor points* like the Ambient Permit System does, transaction costs for determination are on the low side (Paragraph 4.2.1).

As the Weber-model bases on the Ambient Permit System of Montgomery, it is assumed that no *simulation model* needs to run before any *potential* change in discharges like under the Pollution Offset System. This lowers the transaction costs for the environmental authority compared to the Pollution Offset System. The permanent collection of real-time data would not be necessary either. No additional transaction costs for the environmental authority are caused.

An open question is how the *initial allocation* of permits is organised. We assume that no special initial allocation applies as under the Trading Ratio System (Paragraph 4.3.1). The Weber-model instead allocates the permits without incorporating the relationship between different points (emissions versus immission) as is the case under the Ambient Permit System (Paragraph 4.2.1). The disadvantage of the Ambient Permit System is, however, that sources must hold a portfolio of permits in order to purchase or sell permits for all receptor points when changing discharges.[243]

As to trading ratios, sources know that changes in diversion and discharges in upstream zones are more valuable than the same changes downstream. Their willingness to pay, i.e. the price, thus reflects different impacts on water quantity and quality with respect to the location of the source. Theoretically, this replaces the trading ratio; the trading ratio weighs permit prices with respect to the impact of an abated unit on water quality. The Weber-model, however, assumes that sources are informed about the impacts of different discharges and thus able to differentiate their willingness to pay.

Weber does not specify the model in too much details. But we assume, that in practice, under the Integrated Water Quantity/Quality System, sources will also need to apply a kind of water quality models in order to be able to determine their willingness to pay with respect to the final impact on the water level and/or the water quality when purchasing permits from different sources.[244] In reality, it could thus be easier if the environmental authority determines the *dispersion coefficients* and trading ratios for all sources in a standardised way as under the Ambient Permit System in order to fill the information gap. These trading ratios would reflect the sources' willingness to pay as it reflects different impacts of discharges (upstream versus downstream).

The determination of the *trading ratio* under the Integrated Water Quantity/Quality System would, in contrast to the assumptions under the Ambient Permit System, be endogenous if it reflects the ratio of dispersion coefficients. As sources' decisions about water diversion results endogenously from the model, also the dispersion coefficients and thus the trading ratios are endogenous as they depend directly on the water quantity parameters. Transaction costs for the definition of the trading ratio would thus arise permanently.

[243] To find a solution, one could imagine to integrate a step-by-step initial allocation like the Trading Ratio System proposes. Sources could thus buy and sell permits from different zones and are not obliged to adjust the whole portfolio of permits. But: the allocation mode under the Trading Ratio System is only appropriate for a constant relation between emissions and immission. Endogenous changes in the water flow by means of diversion can thus not be reflected adequately. The allocation mode under the Trading Ratio System can thus not be adequate for the Integrated Water Quantity/Quality System.

[244] It does not make any difference whether exogenous and/or endogenous water quantity changes are integrated. The impact on the ambient pollution level needs to be defined anyway.

Integrated Water Quantity/Quality System (IQQS)

Transaction Costs (TAC), Environmental Authority

Determination of receptor points, zones or blocks	
- Higher TAC - Lower TAC	 (X)
Running Water Quality Model before any *potential* change in discharges	
- Yes, higher TAC - No, lower TAC	 X
Real-time data	
- Yes, higher TAC - No, lower TAC	 (X)
Determination of the initial allocation of permits	
- Including Water Quality Model, higher TAC - Not including Water Quality Model, lower TAC	 (X)
Definition of the dispersion coefficient (Water Quality Model)	
- Yes, higher TAC - No, lower TAC	(X)
Definition of the trading ratio	
- Endogenous, higher TAC - Exogenous, lower TAC	(X)
Differentiation of the immission cap in space	
- Higher TAC - Lower TAC	 (X)
Differentiation of the immission cap over time	
- Higher TAC - Lower TAC	(X)
Determination of the emission cap	
- Environmental authority, higher TAC - Sources, no TAC environmental authority	 (X)
Information requirements	
- Higher - Lower	X

Table 4-19: IQQS, Transaction Costs, Environmental Authority

A spatial differentiation of the *immission cap* would not cause high additional transaction costs for the environmental authority (Paragraph 4.2.1). From the beginning different immission caps would be defined for receptor points and relevant elements of the model would be adjusted. A differentiation over time would cause higher (trade-independent) transaction costs as in most cases the permit allocation would be influenced (see, for example, Paragraph 4.2.1 or Section 6.1).

Furthermore, the Integrated Water Quantity/Quality System does not specify the conversion from the immission cap defined by the permits to the *emission cap*; the model implicitly assumes that sources know the transfer conditions. Again, as the model is based on the Ambient Permit System of Montgomery and as sources could not be fully informed in reality, one could assume, that the environmental authority determines dispersion coefficients for any source and any receptor point. Sources convert their immission constraints into emission caps according to these dispersion coefficients (Paragraph 4.2.1). Again, dispersion coefficients show the impact of discharges on the ambient pollution level. These coefficients thus depend heavily on water quantity levels. As water quantity aspects are considered in this model and water levels and thus impacts of discharges are directly influenced by the sources, the definition of dispersion coefficients would be endogenous and changing over time. The environmental authority thus pays the (trade-independent) transaction costs for the installation of the water quality model and the determination of the dispersion coefficients. The endogenous determination causes higher permanent transaction costs than an exogenous determination would do.

Integrated Water Quantity/Quality System (IQQS)	
Transaction Costs (TAC), Sources	
Determination of the emission cap	
- Yes, higher TAC - No, lower TAC	(X)
Portfolio of permits	
- Yes, higher TAC - No, lower TAC	(X)
Constant adjustment of the permit value	
- Yes, higher TAC - No, lower TAC	(X)
Trading ratio	
- Endogenous, higher TAC - Exogenous, lower TAC	X

Table 4-20: IQQS, Transaction Costs, Sources

The need to inform sources is higher than under the Ambient Permit System (*information requirements*): sources should be informed about the water (quantity) and the water quality market. The general information requirements are the same: the sources need to be informed about trading conditions and if necessary about trading ratios and interdependencies between the water and the water quality market. As dispersion coefficients and trading ratios are defined endogenously, information is required more frequently thus causing higher permanent (trade-independent) transaction costs. Transaction costs arising for the environmental authority to inform sources and to monitor them are higher than under the Ambient Permit System. At the same time, the system reflects the reality much better than the Ambient Permit System does.

The evaluation of *transaction costs* for the sources comes to the following result (Table 4-20). It depends on the specification of the model whether sources *determine the emission cap* or not. Under an Ambient Permit System specification sources would define the emission caps (Paragraph 4.2.1).

If the assumptions of the Ambient Permit System are adopted, sources need to hold *portfolios of permits*. In this case, permanent transaction costs for sources would be high (Paragraph 4.2.1). Transaction costs for managing the portfolio arise also for potential as well as for actual trading activities (information and decision costs). Thus only part of these are trade-dependent. A permit trading between zones as is specified under the Trading Ratio System cannot solve this problem as the initial allocation is not able to integrate changing (endogenous) water levels; this is, however, the main element of the Integrated Water Quantity/Quality System.

The *value of permits* changes endogenously with alteration of allocation at the water (quantity) and water quality market. Sources can influence the permit values by their own activities, i.e. by means of water diversion. Adjustments in the value of (quality) permits might thus intentionally be influenced by sources. They spend permanent trade-independent transaction costs to adapt their decisions to changes in the permits' values. In most models, variation in the water level is exogenous. Under the Integrated Water Quantity/Quality System endogenous changes are reflected. By trading at the water (quantity) market, sources can influence their discharge constraints. Transaction costs do thus not only arise for determining the changing values of permits; as this term becomes endogenous, transaction costs also occur to find the optimal level of water diversion or permit portfolio. Monitoring and reporting costs increase enormously as the relationship between water quantity and quality aspects needs to be reflected by these data.

If a *trading ratio* applies, the definition is assumed to be endogenous. Independent of the determination (sources, by means of their willingness to pay or environmental authority, through dispersion coefficients) changes at the water quantity market influence the impact of sources' discharges. The trading ratio must be adjusted permanently to the changing conditions and is thus endogenous. This causes relatively high permanent transaction costs for sources; they must adapt their decisions to changing conditions.

Again, most of the transaction costs are trade-independent and do thus not additionally influence the trading activities, as discussed in Paragraph 3.3.4. Too high total transaction costs should be avoided in any case.

	Trading Ratio System (TRS)	Integrated Water Quantity/Quality System (IQQS)
General Design Elements		
Cap type	- Ambient-based	- Ambient-based
Trading ratio	- Exogenous	- Not specified
Ecological Dimensions		
Geographical scope	- Entire river basin possible	- Entire river basin possible
Orientation	- Endpoint and/or instream possible	- Endpoint and instream integrated
Differentiation of the immission cap in		
Space	- Predestined	- Possible
Time	- Restricted	- Restricted
Adjustment of the emission cap in		
Space	- Predestined	- Possible
Time	- Restricted	- Possible
Water quantity aspects	- Only for the initial allocation, trading ratio exogenous	- Integrated, endogenous

Table 4-21: TRS and IQQS, Ecological Dimensions

4.3.3 Summary

Both water specific models perform quite well from the ecological and economic point of view (Table 4-13 to Table 4-20). Some results for the Integrated Water Quantity/Quality System can not be answered in a final way, as some specific design elements of the system are not given in more detail in this model. The actual performance depends on the final design of certain elements.

Both approaches are ambient-based (*cap type*, Table 4-21). The Trading Ratio System uses an exogenous *trading ratio* defined as ratio of dispersion coefficients. Different impacts of discharges are thus reflected. Again, general problems of exogenous trading ratios are relevant (Section 2.5). The Integrated Water Quantity/Quality System does not explicitly incorporate a trading ratio. Any trading ratio to be implemented, e.g. with respect to the Ambient Permit System, needs to be endogenous to allow for the permanent integration of water flow changes.

Both approaches are provided to cover the entire river basin (*geographical scope*); both approaches can follow endpoint as well as instream flow constraints (*orientation*). Under the Trading Ratio System the definition of the constraint type would influence the setting of the emission caps derived by the water quality standard. The Integrated Water Quantity/Quality System explicitly includes endpoint *and* instream flow need constraints in the function to be maximised.

The zonal approach (Trading Ratio System) is predestined to set different water quality standards per zone (differentiated *immission caps* in space). Different water uses, e.g. swimming area or drinking water abstraction, can ask for different maximum immission loads. The water quality standard can be defined individually for each zone. The *emission cap* would then be derived in the form of a total zonal load standard. Differentiation of the immission cap over time is much more complex, because it directly influences the total allocation of permits.

The Integrated Water Quantity/Quality System does not preview a differentiated maximum immission load in this version. The model could, however, be adjusted by means of the introduction of differentiated instream water flow and quality constraints. Again, a differentiation in space would be

	Trading Ratio System (TRS)	Integrated Water Quantity/Quality System (IQQS)
Cost-effectiveness	- Yes - No free rider	- Depending on initial allocation (see Ambient Permit System) - No free rider
Dynamic efficiency	- Yes	- Yes, if caps are binding
Transaction costs	- See discussion of Table 4-23 and Table 4-24 (p. 102-104)	

Table 4-22: TRS and IQQS, Economic Criteria

more operable than a differentiation over time. Under both models a differentiation of emission caps over time is quite complex.

The main difference between these two models lies in the (non-)integration of alteration of water flows through input use (*water quantity aspects*). While the Trading Ratio System assumes that the relation between water quality and emissions is constant over a certain time period and thus does not consider short term changes in water flows, the Integrated Water Quantity/Quality System explicitly includes changing water flows. While the Trading Ratio System integrates (exogenous) water quantity aspects in the initial allocation of permits and in the exogenously fixed trading ratios, the Integrated Water Quantity/Quality System considers only endogenous changes in water quantities due to diversion by sources. This influences other aspects of the trading system.

The Trading Ratio System avoids the hot spot problem. Benefiting from the uni-directional nature of water flow in a river, the cap setting strategy of the Trading Ratio System is able to set binding standards for each zone: the upstream permit fully accounts for its impact on downstream zones. This avoids local concentrations.[245]

Not all design elements of the Integrated Water Quantity/Quality System are defined in much detail. An appropriate implementation of the system should avoid the problem of hot spot by setting binding caps and assuring compliance.

The economic analysis comes to the following results (Table 4-22). The Trading Ratio System is *cost-effective*, independent of the initial allocation. The consequent implementation of this approach guarantees thus the least-cost solution. The Integrated Water Quantity/Quality System is efficient. However, if the assumptions of the Ambient Permit System hold true, the cost-effectiveness can be highly dependent on the initial allocation (Paragraph 4.2.1.2).

The Trading Ratio System avoids free rider behaviour. All discharge permits are allocated individually, an increase in emissions obliges the sources to purchase permits; no free riding behaviour takes place. The Integrated Water Quantity/Quality System avoids free rider behaviour if all water quality caps are set binding and if permits are allocated individually. As the constraints are assumed to be set 'optimally', binding caps would be guaranteed. Implementing this system in reality would ask for a well designed definition of these constraints.

Sources under the Trading Ratio System have an incentive to innovate in order to decrease abatement costs and thus to be able to sell surplus permits (*dynamic efficiency*). The central mechanism of a permit trading system applies and guarantees for the incentive to innovate. For the Integrated Water

[245] It is assumed that permits are individually allocated and that additional emissions need to be covered by an additional purchase of permits.

	Trading Ratio System (TRS)	Integrated Water Quantity/Quality System (IQQS)
Transaction Costs (TAC), Environmental Authority		
Determination of receptor points, zones or blocks		
- Higher TAC	X	
- Lower TAC		(X)
Running Water Quality Model (WQM) before any *potential* change in discharges		
- Yes, higher TAC		
- No, lower TAC	X	X
Real-time data		
- Yes, higher TAC		
- No, lower TAC	X	(X)
Determination of the initial allocation of permits		
- Including Water Quality Model, higher TAC	X	
- Not including Water Quality Model, lower TAC		(X)
Definition of the dispersion coefficient (Water Quality Model)		
- Yes, higher TAC	X	(X)
- No, lower TAC		
Definition of the trading ratio		
- Endogenous, higher TAC		(X)
- Exogenous, lower TAC	X	
Differentiation of the immission cap in space		
- Higher TAC		
- Lower TAC	X	(X)
Differentiation of the immission cap over time		
- Higher TAC	X	(X)
- Lower TAC		
Determination emission cap		
- Environmental authority, higher TAC	X	
- Sources, no TAC environmental authority		(X)
Information requirements		
- Higher		X
- Lower	X	

Table 4-23: TRS and IQQS, Transaction Costs, Environmental Authority

Quantity/Quality System, the dynamic efficiency also depends on the specific design, especially on the cap setting (binding caps).

The analysis of the *transaction costs* shows that the costs for the environmental authority differ again (Table 4-23). The Trading Ratio System is a typical zonal approach. The environmental authority divides the river into *zones*, with similar dispersion characteristics. This is not the case for the Integrated Water Quantity/Quality System. When applying the assumptions of the Ambient Permit System, this would be a *receptor point* approach, causing lower transaction costs for the environmental authority.

The determination of the *initial allocation* requires the application of water quality models under the Trading Ratio System; the initial allocation already integrates the relation between emission loads of different zones. If the Integrated Water Quantity/Quality System follows a Montgomery-style design, no water quality model becomes relevant in order to set the initial allocation. Transaction costs are at the lower end.

The *trading ratio* under the Trading Ratio System is set exogenously and fixed for a certain period of time. Transaction costs for the environmental authority are thus lower. If one assumes that in the Integrated Water Quantity/Quality System the determination of the trading ratio also becomes necessary, additional transaction costs occur. The trading ratio would be defined endogenously as water

	Trading Ratio System (TRS)	Integrated Water Quantity/Quality System (IQQS)
Transaction Costs (TAC), Sources		
Determination of the emission cap		
- Yes, higher TAC		X
- No, lower TAC	X	
Portfolio of permits		
- Yes, higher TAC		(X)
- No, lower TAC	X	
Constant adjustment of the permit value		
- Yes, higher TAC		(X)
- No, lower TAC	X	
Trading ratio		
- Endogenous, higher TAC		X
- Exogenous, lower TAC	X	

Table 4-24: TRS and IQQS, Transaction Costs, Sources

quantity changes are endogenous and influence the trading ratio (in terms of dispersion coefficients). We assume that the trading ratio is defined by the environmental authority as supposed under the Ambient Permit System by Montgomery.

Under the Trading Ratio System, the conversion from the immission cap defined in the permits to the actual emission load (*determination of the emission cap*) is done by the environmental authority, once at the beginning (initial allocation). Under the Integrated Water Quantity/Quality System the sources would be responsible for defining the emission caps with respect to the dispersion coefficients as defined by the environmental authority, if the Ambient Permit System design holds true (Paragraph 4.2.1.2).

Information requirements under the Integrated Water Quantity/Quality System are assumed to be higher than under the Trading Ratio System as sources must be informed about endogenous changes on both the water and the water quality market. Information requirements under the Trading Ratio System are rather low as relevant trading parameters are defined exogenously and fixed for a certain period of time.

At the sources' level we come to the following conclusion: transaction costs are higher for sources under the Integrated Water Quantity/Quality System than under the Trading Ratio System. Under the Trading Ratio System sources are not obliged to hold a *portfolio of permits* (Table 4-24). This would be the case under the Integrated Water Quantity/Quality System if it follows a pure Ambient Permit System.[246]

Under the Trading Ratio System, the environmental authority defines the emission value of permits according to the immission caps (*determination of the emission cap*). No additional transaction costs arise for sources. If the Integrated Water Quantity/Quality System follows the Ambient Permit System design, sources determine the corresponding emission load according to the dispersion coefficients determined by the environmental authority. Sources thus pay these transaction costs.[247]

The initial allocation process under the Trading Ratio System means that sources do not need to manage *portfolios of permits*. This is not the case under the Integrated Water Quantity/Quality System, if

[246] For modifications of the Integrated Water Quantity/Quality System to avoid the disadvantages of the Ambient Permit System, see Section 6.1.

[247] The costs of determining the emission load under the Trading Ratio System could easily be transferred by the authority to the sources. The difference between the Trading Ratio System and the Integrated Water Quantity/Quality System concerning this aspect is thus small.

implemented in a Montgomery-style. Transaction costs differ. However, the disadvantage of portfolios under the Integrated Water Quantity/Quality System could be avoided by adequate design specifications (Section 6.1).

Under the Trading Ratio System the value of permits in terms of emissions in relation to the immission load is fixed with the initial allocation. Constant adjustments to the immission load would thus not be reasonable. Constant *values of the permits* reduce the uncertainty level for the sources and thus the transaction costs. Under the Integrated Water Quantity/Quality System, endogenously changing water levels are integrated in the water quality trading system. The absolute value of permits in terms of emissions changes over time with respect to changing water conditions. Sources are thus obliged to adopt their decisions to changing conditions. This causes higher transaction costs.

The *trading ratio* under the Trading Ratio System is exogenously defined. Uncertainties for sources are reduced and transaction costs decrease. Under the Integrated Water Quantity/Quality System the use of trading ratios is assumed according to the Ambient Permit System. These trading ratios are, however, endogenous as they are directly influenced by the water diversion activities of the sources that in turn develop within the trading system (endogenous). Trading ratios are thus changing and not fixed *ex ante*. This causes higher transaction costs, e.g. information costs and decision costs, for sources. One could state, that this reflects real conditions and thus helps the system to be more effective and efficient. The level of transaction costs alone cannot decide about the benefits of a model.

The analysis of the Trading Ratio System and the Integrated Water Quantity/Quality System demonstrated that transaction costs are mainly trade-independent. They are necessary to make the system function; as discussed in Paragraph 3.3.4, these transaction costs do not directly affect the allocation in a negative way.

4.4 Overall Result

This section discusses the results for the remaining approaches: the receptor point approach (Ambient Permit System) and both river-specific approaches, the zonal approach (Trading Ratio System) and the Integrated Water Quantity/Quality System. In contrast, the Pollution Offset System and the Exchange Rate System have been excluded from further discussion, because they are both confronted with significant problems of ineffectiveness and inefficiency (Paragraphs 4.2.3 and 4.2.4).

4.4.1 Ecological Dimensions

The analysis of the theoretical approaches has shown that certain design elements of a permit trading are crucially relevant for the efficient and effective functioning of a water quality trading system. Additionally, the discussion illustrated that neither one model or the other is more adequate *per se*. Whether the design of one or another model is more appropriate in order to guarantee homogeneity of traded entities, will depend on the ecological needs and on specific local conditions.

All approaches considered can integrate the entire river basin (*geographical scope*). All models are ambient-based (*cap type*) and can reflect instream flow need constraints as well as endpoint constraints (*orientation*). Depending on the constraint, the water quality model will come to different water quality goals and will thus ask for different emission caps. The Integrated Water Quantity/Quality System explicitly combines instream flow needs *and* endpoint constraints.

The differentiation of the *immission cap* in space can become relevant if, for example, different water quality levels are required depending on the water use. A spatial differentiation of the immission goal is only feasible, if the approach allows us to divide the river in zones or receptor points. The water quality goal would then be differentiated for zones or receptor points.

The differentiation of the immission cap over time is more complex. A differentiation of the immission caps over time means that the absolute ambient pollution level is changing over time. As permits are defined in immission loads, they have to be adjusted; the absolute amount allowed by permits changes (*emission cap*). Specific regulations could solve this problem, as will be shown in Section 6.1. Originally, none of these models provides a differentiation of the immission caps over time.

Section 2.4 has shown that water quality depends strongly on the current water level (quantity). While most of the models integrate *water quantity aspects* only very implicitly and as exogenous variable, the Integrated Water Quantity/Quality System explicitly considers (only) the endogenously caused part of changes in water quantity levels. In a more realistic version, both, exogenous and endogenous elements defining the water level (quantity) might be integrated by supplementation (Section 6.1).

Trading ratios can be defined exogenously or endogenously. General problems concerning the determination of a trading ratio were discussed in Section 2.5. The analysis of the theoretical models came to the result that only trading ratios determined as ratios of dispersion coefficients are ecologically effective.[248] Transaction costs under an endogenous trading ratio are higher than under an exogenous, for both, the environmental authority and the sources.

[248] The ratio of marginal abatement costs does not reflect relevant functions of a trading ratio (Paragraph 4.2.3).

The analysis of the models showed that hot spots can be caused by different reasons: the inadequate integration of the relationship between discharges and their impact on the ambient pollution level at specific points would cause hot spots. In some cases, free rider behaviour could cause hot spots. Caps need to be binding, permits should be allocated individually and increasing emissions should automatically lead to a purchase of permits to avoid hot spots.[249]

4.4.2 Economic Criteria

Cost-effectiveness

The discussion of the theoretical approaches has shown that several design elements can prevent the system from coming to a cost-effective solution by hampering the system from exhausting all trading potentials. Non-binding caps and free rider or the non-degradation principle affect the cost-effectiveness in a negative way. Cost-effectiveness in these models depends on the initial allocation of permits.

The existence of transaction costs influences the optimal allocation of abatement activities and thus the cost-effectiveness of the system. This is especially true for trade-dependent transaction costs paid by the sources themselves. The analysis of the theoretical models shows that most of the specific transaction costs analysed in this study are trade-independent and thus do not (additionally) affect the allocation process.[250]

The introduction of trading ratios supports the system to come to a cost-effective solution. Trading ratios indirectly reflect the individual marginal abatement costs which can be very different depending on the location of the source and/or time of discharge. One-to-one trading would not integrate these differences and thus not guarantee for a cost-effective solution exhausting all trading potentials (Section 2.5). Nearly all of the discussed models use a trading ratio to set well differentiated incentives.

Dynamic Efficiency

The criterion of dynamic efficiency is also influenced by the design. Cap setting, in particular, influences the dynamic efficiency. Only binding caps force sources to purchase additional permits when increasing discharges. If additional permits can be purchased without charge, there is no incentive to reduce discharges. Furthermore, if the design offers the possibility to free ride, sources would rather act as free riders than invest in new technologies to abate more. If, in turn, additional emissions force purchasing of additional permits and to paying for them, the incentives to invest in new technologies are given.

Dynamic efficiency also depends on the price of permits at the market. If prices are biased by the design, e.g. a lack of cost-effectiveness or incorrect trading ratios, this would set biased incentives and thus hamper the dynamic efficiency.

The incentive to abate pollution by reducing water diversion is only set by the model, if water quantity aspects are included endogenously, as under the Integrated Water Quantity/Quality System. Otherwise,

[249] The integration of effective sanction mechanisms is necessary in order to guarantee ecological effectiveness and thus to avoid hot spots.
[250] In this study only those transaction costs are discussed that are different for the analysed models and thus would additionally influence the allocation process (Paragraph 3.3.4)

the use of water does not influence the discharge possibilities; no incentive to reduce water use is created.

Transaction Costs

The analysis of specific water quality trading system transaction costs for different theoretical models has shown that it is important to evaluate transaction costs separately for the environmental authority and the sources.

The Trading Ratio System is the only model to use *zones*. The definition of zones, each of them characterised by similar dispersion processes, causes higher transaction costs for the environmental authority than the determination of receptor points.

The definition of the *initial allocation* of the permits causes higher transaction costs under the Trading Ratio System; the initial allocation already integrates interdependencies between discharges and immission standards. This is one peculiarity of the Trading Ratio System. No other model uses such a complex process of initial allocation. This causes higher transaction costs under the Trading Ratio System; at the same time, this lowers transaction costs for the sources that do not need to hold an entire portfolio of permits.

As we have seen (Section 2.5) exogenous *trading ratios* causes lower transaction costs for both the environmental authority (determination) and the sources (decision costs). Endogenously defined trading ratios cause higher transaction costs. They are, however, in some models, especially in the Integrated Water Quantity/Quality System, justified by a higher efficiency and effectiveness of the model. Higher transaction costs are thus not *per se* negative. They rather need to be analysed individually in detail for any trading system to be implemented. Most of the models define the trading ratio in terms of dispersion coefficients. These coefficients should be determined by the environmental authority thus guaranteeing a standardised determination process and lowering the total transaction costs as the water quality model needs to only be defined and specified once.[251]

The differentiation of the *immission cap* in space is possible without causing high additional transaction costs in most of these models. A differentiation over time, on the other hand, would only be possible under higher transaction costs as the total allocation needs to be adapted.[252]

In any ambient-based trading system, the *emission cap* needs to be determined according to the immission standards in order to make the system operable. Transaction costs for this transformation arise in any case; depending on who determines the emission caps, the environmental authority or the sources will pay the transaction costs.[253]

The requirements to inform sources differ for some models (*information requirements*). Generally, models with endogenous trading ratios would cause higher information costs than models with exogenously determined trading ratios. Higher transaction costs might thus be justified by a higher effectiveness and efficiency.

[251] For general problems of the determination of trading ratios, see Section 2.5.
[252] Chapter 5 will show that models exist that are capable of differentiating the immission caps over time without incurring too high additional transaction costs.
[253] Again, if the environmental authority determines the emission cap it could pass the transaction costs through to the sources.

At the sources' level the main difference lies in the need to manage a *portfolio of permits* for different receptor points. The Ambient Permit System and the Integrated Water Quantity/Quality System in the original version require a portfolio of permits. This causes high transaction costs for sources. Increasing discharges would be followed by trading activities with more than one other source to compensate changes at all affected receptor points.

Permanent variation of the *value of permits* (in term of emission loads) increases the transaction costs for sources. This would, however, often be the only way to reflect dispersion characteristics of substances in the water and would thus be necessary for an ecologically effective system.

As we have seen above, an endogenous definition of the *trading ratio* would cause higher transaction costs (information and decision costs) for sources than for exogenous trading ratios. At the same time, endogenous trading ratios could, if well defined, better reflect the conditions of the water medium and set differentiated incentives (time/space) for sources. Again, the decision about the type of the trading ratio must be taken case for case.

Most of the transaction costs (environmental authority and sources) are trade-independent. While they make the implementation of a water quality trading system costly, they do not additionally hamper the sources from trading activities. One may assume that only trade-dependent transaction costs which arise with each trading activity, hamper sources directly from trading (Paragraph 3.3.4). However, if transaction costs occur anyway, sources would have an incentive to trade in order to exploit all cost saving potentials.

4.4.3 Implications

The receptor point approach Ambient Permit System contains important design aspects. Nevertheless, the Ambient Permit System is characterised by some general problems. The main disadvantage may be the necessity of managing entire portfolios at the source level in order to cover any receptor point that is affected by discharge activities. One might think, that innovative computer programmes would be able to manage this complexity. But also well-established programmes would not be able to avoid the necessity of trading with more than one partner in case of modifications of the discharge behaviour. Too high transaction costs might hamper sources from trading; if not all trading activities are exhausted, cost-effectiveness cannot be realised. Again, also the initial allocation influences the cost-effectiveness. At the same time the Ambient Permit System is characterised by high ecological flexibility potentials.

In the following section the Ambient Permit System is mentioned in passing. The Ambient Permit System forms the main basis for the Integrated Water Quantity/Quality System, that combines water quantity and quality aspects. The application of the Ambient Permit System is thus relevant. At the same time, Section 6.1 will show, that instead of amendments of the Ambient Permit System rather the application of other approaches could be recommended.

The most relevant theoretical approaches are the zonal approach under the Trading Ratio System and the Integrated Water Quantity/Quality System, which incorporates water quantity aspects. Both approaches are discussed with respect to the implementation proposals made in Chapter 6.

5 Design Options: Practical Approaches

5.1 Basics

Especially in the United States different forms of permit trading systems have been tested and implemented for different rivers since the 1980/90s.[254] In Australia, water quality trading systems have been used as an instrument for the environmental protection of rivers since the 1990s, too (NSW EPA, 2003).

As a first step, one of the permit trading systems in the United States will be analysed: the *Tar Pamlico Nutrient Trading Program (TPNTP)* (Section 5.2). This example has been chosen because it is often mentioned in the literature, and therefore a good information basis is available.[255] But it is surprising that authors have rarely analysed this programme from an economic point of view.[256] This study seeks to fill this gap by analysing the Tar Pamlico Nutrient Trading Program in a comprehensive and standardised way in accordance with the criteria developed in Chapter 3.

As a second step, Section 5.3 describes the *Hunter River Salinity Trading Scheme (HRSTS)* in Australia. This example is selected for its outstanding structure: the system integrates the specific characteristics of the Hunter River as well as of the substances concerned (salt) in an impressive manner. Again, a comprehensive economic analysis of the model cannot be found in the literature. It is thus task of this study to evaluate this model with respect to the criteria developed in Chapter 3.

The analyses of the case studies start with a short introduction to the relevant statutory framework (Paragraphs 5.2.1 and 5.3.1). Legal regulations in place directly influence the design of a water quality trading system to be introduced. Therefore, the specific United States and Australian statutory framework is described in more detail in this chapter. The results of the analyses of both the practical approaches and the remaining theoretical approaches will be integrated in a comparative analysis (Section 6.1).

5.2 Experiences – United States

5.2.1 Statutory Framework

Environmental policy under the federal structure of the United States works as follows: while policy-making takes place at the supranational level, policy-implementation is done at a state level (Kelemen, 2004, p. 114). The States, however, can influence the environmental policy in the Congress.

Until the 1960s, environmental policy was mostly implemented individually by each single State. With increasing public interest in environmental policy, however, the Congress started to regulate the environmental policy on the federal level. In 1972, two years after the *Clean Air Act* (1970), the *Federal Water Pollution Control Act*, called *Clean Water Act (CWA)* was subscribed. The US Environmental

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[254] For an overview, see Environomics (1999), Kraemer *et al.* (2003).
[255] See Kraemer *et al.* (2004), Breetz *et al.* (2004) or Morgan and Wolverton (2005); see also website of the N.C. Division of Water Quality (`http://h2o.enr.state.nc.us/nps/tarpam.htm`, September 2006).
[256] For an exception, see Hoag and Hughes-Popp (1997) or Kerr *et al.* (2000).

Protection Agency (US EPA) was also founded at this time in order to implement regulative targets delegated by the Congress (Kallis and Nijkamp, 2000, p. 301). The environmental policy of the United States was initially Best Available Technology (BAT)-orientated (Wiener, 2004, p. 94). Relatively early on the United States started then to implement more and more marketable instruments, such as permit trading systems.[257]

Clean Water Act and its Implementation

The Clean Water Act (CWA) from 1972 sets the general framework for the US water policy.[258] The goal of the Clean Water Act is "to restore and maintain the chemical, physical, and biological integrity of the Nation's waters" (Section 101 (a) CWA). Even though the Clean Water Act does not explicitly stipulate the precautionary principle, it is assumed to be precautionary in intent (Tickner, 1999, p. 3).

Section 101 (a) of the Clean Water Act assigns certain well defined responsibilities like the abatement of specified pollutants and the development and implementation of programmes for nonpoint sources to each State. Section 101 (d) determines that the US Environmental Protection Agency administrates the Clean Water Act unless otherwise determined. Section 102 (a) CWA goes further and says that the US Environmental Protection Agency should in cooperation with the States and the organisations concerned prepare and develop programmes to avoid, reduce and abate the pollution (cooperation principle).[259]

The Clean Water Act further stipulates the development of (ambient) 'water quality standards'. These standards determine the type of water use, the water quality criteria which depends on the predetermined use of the water, and an anti-degradation condition.[260] The water quality standard is thus defined in accordance with the designated use of a body of water. This reflects the discussion from Paragraph 3.2.2.[261]

Section 402 of the Clean Water Act builds the basis for the National Pollutant Discharge Elimination System (NPDES). This system applies only to point sources. Within the National Pollutant Discharge Elimination System, the Clean Water Act stipulates that all discharges from point sources must be covered by a permit.[262] This permit defines – relatively low – source specific technology-based minimum standards, so called 'effluent limitations' (Section 301 CWA). These standards are defined for specific pollutants identified in the Clean Water Act (Section 304(a)(4)). Nutrients are not subject to such technology-based standards (Stephenson *et al.*, 1999, pp. 793-794). The application can, however, be extended to nutrients.

Such technology-based effluent standards are industry-specific and can, for example, require the installation of a second treatment process in a sewage treatment plant (Boyd, 2000, p. 83). This relatively low setting is only technology-oriented, not water quality-oriented and is often far from being sufficient to achieve the predetermined water quality. Whenever the environmental agency determines

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[257] See also Hansjürgens (2005).

[258] CWA as of November 27, 2002 (`http://www.epa.gov/watertrain/cwa/`, September 2006).

[259] The US Environmental Protection Agency is not directly responsible for the management of nonpoint sources; States must submit programmes to the Congress, which indicates clearly how they plan to reduce the pollution of the nonpoint sources (Section 319 CWA). The implementation of these programs can be supported by the federal government with up to 60 percent of the total sum (Section 319 (h,3) CWA).

[260] See also US EPA (1996a, Chapter 2). The 'anti-degradation' condition coincides with the 'non-degradation' principle discussed in Paragraph 3.3.2. The economic problems of this principle thus remain the same.

[261] For more details, see Novotny (2003, pp. 580-588).

[262] The term 'permit' as it is used here should not be confused with the term of the tradeable discharge permit.

that new cost-effective technologies are available, effluent limits can be set more stringently (Stephenson *et al.*, 1999, p. 95). Once a technology-based standard is fixed in a permit, the anti-backsliding principle applies. Antibacksliding requires a reissued permit to be as stringent in terms of technology requirements as the previous permit, with some exceptions (CWA 303(d)(4); CWA 402(c)). In addition, each permit requires the implementation of source specific monitoring and reporting systems as well as well-defined sanction mechanisms in case of non-compliance. The US Environmental Protection Agency controls the sources regularly. The polluter-pays-principle applies implicitly.

If the (low) technology-based effluent limits do not guarantee the achievement of the predetermined water quality standards, the State will put the body of water and the pollutants concerned on a list.[263] Bodies of water polluted with substances, e.g. nutrients, not being integrated in the National Pollutant Discharge Elimination System also follow this regulation. The State must then define a 'Total Maximum Daily Load (TMDL)' for this same water and its pollutants (Section 303 CWA). "The TMDL is the maximum amount of a specific pollutant that a body of water may contain, in 1 day, without adverse effects on its designated use" (Jarvie and Salomon 1998, p. 141).[264] The Total Maximum Daily Load is defined in emissions; the final goal is, however, an ambient pollution level. It considers seasonal changes as well as a security margin in order to reflect uncertainties concerning the relationship between discharges and water quality (Novotny, 2003, pp. 593-602). The Total Maximum Daily Load thus indirectly also introduces water quantity aspects. But again, nonpoint sources are not directly regulated (Boyd *et al.*, 2007).

The Clean Water Act (Title III) previews specific measures in the event of non-compliance. Violators are punished by a fine (per day or per year of violation) or even by imprisonment depending on the kind of violation. These punishments should ensure the enforcement of the Clean Water Act. It should be reminded that such a sanction mechanism can only be effective if it is automatically applied (see Paragraph 3.3.5).

The US Environmental Protection Agency plays an important role in implementing the Clean Water Act. The *US EPA Policy Statement 2003* (US EPA, 2003b) explicitly stipulates using innovative approaches such as a permit trading system in order to achieve the predetermined water quality goal. The US Environmental Protection Agency argues that the introduction of a permit trading would reach the predetermined environmental standards in a more flexible and more cost-effective way than traditional approaches do. This instrument would set economic incentives for innovations and for voluntary abatement measures. The US Environmental Protection Agency refers to experiences, gathered in already existing permit trading systems in the United States.

The Total Maximum Daily Load could be a good basis for a water quality trading system as it already defines the load standard for a specific body of water and a specified water quality standard. The determination of the Total Maximum Daily Load already requires some knowledge of the fate and transport processes of pollutants within the body of water. A water quality trading system that is being

[263] The reason for the non-achievement could lie in the too low technology-based standard; also the high part of nonpoint source pollution lead to violations of the water quality settings.

[264] Woodward and Kaiser (2002) note that the TMDL changes the perspective towards water quality management. Instead of asking, 'How much pollution should each source be allowed to emit?' the management questions under a TMDL become 'What is the total pollution load that should be permitted?' and 'How should the load be allocated among the various sources?' (p. 368). A more emission-based view is thus replaced by an ambient-based perspective.

introduced can access the water quality models already tested and used for the Total Maximum Daily Load.[265]

One should, however, take into account the fact that not all elements of the Clean Water Act allow for the introduction of a permit trading system. Technology-based effluent limits within the National Pollutant Discharge Elimination System might not hinder the implementation of a trading system if they are quite low. But even if this is the case under the Clean Water Act, effluents limits are industry-specific and non-negotiable. Trading between different types of industry is thus not possible for the pollutants concerned (Boyd, 2000).

On the other hand, National Pollutant Discharge Elimination System and thus technology-oriented effluent limits are, in general, not formulated for nutrients. Depending on the individual regulations, the restrictions for the introduction of a water quality trading system are thus not all relevant for nutrients. This is an important aspect for the evaluation of the Tar Pamlico Nutrient Trading Program (Paragraph 5.2.3).

The Clean Water Act and its elements mainly regulate point sources. As a result, significant proportions of point source pollution have already been avoided. The main problem is nevertheless the pollution caused by nonpoint sources.[266] These are not sufficiently regulated in the Clean Water Act. Water quality trading systems are often discussed and used to regulate the pollution of nonpoint sources (nutrients) although these sources, in particular, are extremely difficult to regulate by means of a permit trading system. Measurement problems make it extremely difficult to assign individual responsibility for pollution and thus to allocate permits; at the same time, it is difficult to guarantee homogeneity between traded units (Section 2.2).

Excursus: Pretreatment Trading

The US Environmental Protection Agency generally argues in favour of the implementation of a pretreatment trading programme, i.e. a permit trading for indirect discharges (Paragraph 2.2.1). The EPA discusses the potential cost savings, the incentive for technological innovations etc. At the same time, the US Environmental Protection Agency underlines that the coexistence with the existing legal framework needs to be analysed in detail; amendments might be necessary.

The Illinois Environmental Protection Agency (IEPA) evaluated the potential of pre-treatment tradings in more detail (Park, 1996): four sewage treatment plants with delegated pre-treatment programmes were studied for possible market trading opportunities. The Illinois Environmental Protection Agency found that the federal 'categorical pre-treatment standards'[267] were often more stringent than 'local limits'[268]. This would ask for a trading at the categorical level to be ecologically effective.[269] The

[265] The US Environmental Protection Agency established trading principles to facilitate the implementation of water quality trading systems (US EPA, 1996a, Jarvie and Salomon, 1998). These principles are, however, quite general and refer in most part to the integration in the legal framework.

[266] See also Boyd (2000, pp. 42-64).

[267] Categorical Pretreatment Standards are limitations on the discharge of pollutants to publicly owned treatment works (POTWs). They are promulgated by the EPA in accordance with Chapter 307 of the Clean Water Act and apply to specific process wastewaters of particular industrial categories. These are national, technology-based standards that apply regardless of whether or not the publicly owned treatment work (POTW) has an approved pretreatment program or the industrial user has been issued a permit. (http://cfpub.epa.gov/npdes/pretreatment/pstandards.cfm#categorical, November 2006).

[268] Local limits are developed in order to reflect specific needs and capabilities at individual publicly owned treatment works (POTWs) and are designed with the aim of protecting the publicly owned treatment work (POTW) receiving waters. POTW Pretreatment Programs must develop local limits or demonstrate that they are unnecessary; local limits are needed when

US Environmental Protection Agency, however, does not allow any infringement of categorical pre-treatment standards (technology-based); trading of these standards is thus not possible in the current legal situation (US EPA, 1996a, Chapter 6). The implementation of a pre-treatment trading system has not been pursued further (Breetz *et al.*, 2004, p. 106). The situation might be similar in other states.

5.2.2 Existing Programmes

The instrument of a permit trading system for water pollution control has been in use in the United States since the 1980s. Breetz *et al.* (2004) discuss existing trading programmes in the United States in a complex piece of research (see Table A-1, Appendix).[270] One can see, on the one hand, that roughly one third of the 40 plus programmes are trading with phosphorus and/or nitrogen (nutrients). Other integrated pollutants are selenium, heavy metals or even temperature.[271] On the other hand, this overview shows that in most of the programmes nonpoint sources are also included in the trading system.[272] Not all of these programmes are still working. Some of them are (were) pilot projects running only within a test phase.

In total, one has to conclude that, if at all, trading activities with these permits rarely actually takes place although high cost saving potentials had been expected (Boyd, 2000, p. 80). The descriptions of Breetz *et al.* (2004) show that trading activities took place for just 7 of these 40 plus programmes.[273] In most cases, only one or two trades have been observed (see Table A-1, Appendix). King and Kuch (2003) also state that, in total, very few trades have taken place in all water quality trading projects. These trades have often not been the typical standardised ones, but bilaterally negotiated agreements. The statement of Jarvie and Salomon (1998, p. 135) that "although only two of these four case study programs have involved actual effluent trades thus far, they all have resulted in more cost-effective reductions of water pollution" appears to be contradictory and must be analysed in more detail in the following discussion, at least for the Tar Pamlico Nutrient Trading Program.

One would expect that the reasons for missing trading activities often lie in the specific design of the trading programme. Indeed, as the example of the Tar Pamlico Nutrient Trading Program shows, both the strength of the effectiveness and the efficiency of the system are influenced by the design of the permit trading system.

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pollutants are received that could result in pass through or interference at the publicly owned treatment work. Essentially, local limits reflect site-specific needs. (`http://cfpub.epa.gov/npdes/pretreatment/ pstandards.cfm#categorical`, November 2006).

[269] The limits for indirect dischargers are set by publicly owned treatment works (POTWs) to ensure that the POTW is able to treat the waste water sufficiently to meet its own National Pollutant Discharge Elimination System permit limits. Although indirect dischargers are not required to have National Pollutant Discharge Elimination System permits, they must meet national categorical pre-treatment standards (Jarvie and Salomon, 1998, p. 140).

[270] See also Kraemer *et al.* (2004), Environomics (1999) or Morgan and Wolverton (2005).

[271] Kraemer *et al.* (2004) mentions additionally the case of BOD-trading. Sources can trade discharges that influence the biological oxygen demand (BOD) of the receiving water medium.

[272] In contrast, Paragraph 2.2.3 already clarified that the introduction of nonpoint sources into a 'pure' trading system is very difficult.

[273] Information on trading activities differ between different sources in the literature (King and Kuch, 2003; Kraemer *et al.*, 2004 etc).

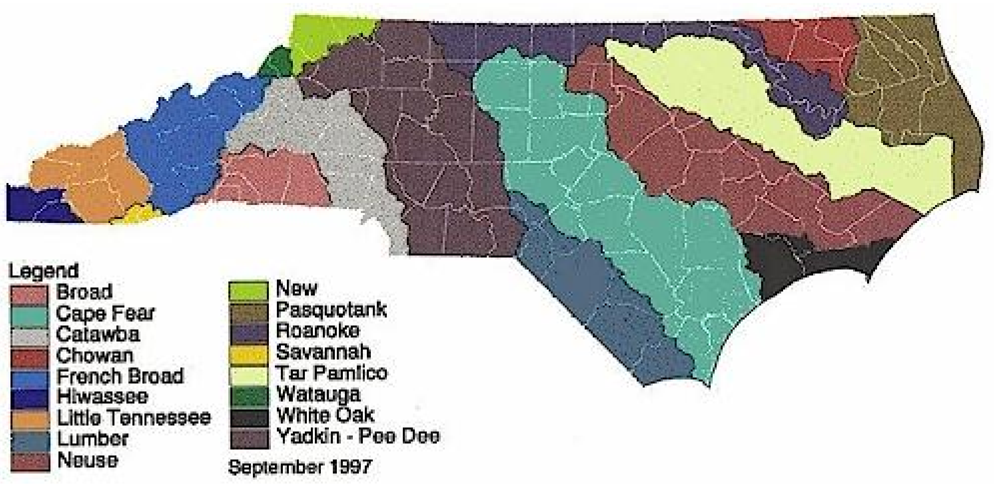

Figure 5-1: Tar Pamlico River Basin, North Carolina (http://www.ptrf.org, January 2007)

5.2.3 Tar Pamlico Nutrient Trading Program, NC, US

5.2.3.1 The Programme

The Tar Pamlico River is situated in North Carolina, US (see Figure 5-1). The Tar River and the Pamlico River are the upstream and estuarine portions of a single river system. Rising in the Piedmont Tar River flows southeasterly for 140 miles before flowing across the flat coastal plain. Near the city of Washington the name changes to the Pamlico River, and it travels another 37 miles before emptying into the Pamlico Sound. The agriculture along the river causes a large proportion of the river pollution;[274] but also point sources such as waste water treatment plants contribute to it (Faeth, 2000, p. 15). In 1989, the North Carolina Environmental Management Commission (NC EMC) declared the river 'nutrient sensitive' water (Kerr *et al.*, 2000, p. 13). In order to reduce the level of pollution, the Tar Pamlico Nutrient Trading Program was implemented in 1990. It spans the whole river basin.[275] The Tar Pamlico Nutrient Trading Program is already running through phase III (2005-2014) that follows phase I (1990-1994) and phase II (1995-2004).

This trading programme includes point sources as well as nonpoint sources. All point sources in the Tar Pamlico River are municipal waste water treatment plants; there are no industrial dischargers.[276] The traded pollutants are phosphorus and nitrogen, i.e. nutrients (Environomics, 1999, p. 26). The majority of the point sources, actually 15 waste water treatment plants, are members of the Tar Pamlico Basin Association (NC EMC, 2005, p. 24). The point sources of the Association are responsible for over 90 percent of point source discharges (Gannon, 2003).[277]

274 Up to 90 percent (!) of the nutrient load is caused by nonpoint sources (Kerr *et al.*, 2000, p. 12).

275 See the Agreements on the Tar Pamlico Nutrient Trading Program of the NC DEHNR (1991) and NC DEHNR (1994), NC EMC (2005) for Phase I to III of the Program.

276 Personal communication to the author by Richard Gannon, Nonpoint Source Program Supervisor, August, 16, 2005.

277 Pollution responsibility of sources that are not members of the Association is low (<10 percent). Specific regulations apply, see NC DEHNR (1991) or NC EMC (2005). Non-members of the Association are not further considered in this study.

In general, public owned waste water treatment plants are obliged to hold National Pollutant Discharge Elimination System permits and to respect technology-based effluent limits (Paragraph 5.2.1). However, nutrients are excluded from this regulation; a separate regulation strategy applies under the Tar Pamlico Nutrient Trading Program (Boyd *et al.*, 2007). Nevertheless, the Tar Pamlico Nutrient Trading Program is based on the nutrient Total Maximum Daily Load (TMDL) that defines ecological goals separately for point and nonpoint sources in order to achieve a specific water quality (Paragraph 5.2.1).[278] The polluter-pays-principle applies as far as groups of sources are held responsible for the pollution they cause. The trading programme is one element used to achieve this goal; the second element consists of a separate regulation of the nonpoint sources.[279]

The Tar Pamlico Nutrient Trading Program treats point sources in the Association as a single unit, with the aim of achieving the given cap within the group of the Association at a higher level of cost-effectiveness. Consequently, one collective nutrient cap has been defined and the individual Association members' nutrient limits were waived.[280] It would be the task of the Association to define individual caps. However, it seems that no final and precise allocation took place at the source level. As no data on that are available, it is unknown how bargaining within the Association takes place and how the reduction requirements are finally allocated to individual sources.[281]

When exceeding the cap, the Association has to pay an 'incentive fee' for each unit of pollution exceeding the cap.[282] This incentive fee is predetermined and paid into an agricultural fund set up by the state in advance, the Agriculture Cost-Share Program (ACSP)[283] which is then used to finance abatement measures according to the Best Management Practice (BMP) at the nonpoint source level.[284] Additionally, the Association can purchase permits from nonpoint sources. But point sources cannot sell permits to nonpoint sources.

For transactions between point sources and nonpoint sources a (n uncertainty) trading ratio applies, taking account of the fact that the impact of emission reductions at the nonpoint source level on the ambient pollution level cannot be accurately predicted and uncertainties remain.[285] Trading ratios are

[278] The nitrogen loading cap for the Association in Phase III (2005-2014) equals 404,274 kg/year; the phosphorus cap 73,060 kg/year (NC EMC, 2005, p. 14). Modelling results require a 30 percent reduction of nitrogen loading from a baseline of the year 1991 and no increase in the phosphorus loadings. Point sources are responsible for 8 percent of the total requirement of a 30 percent reduction of nutrient loads. Nonpoint sources are consequently confronted with 92 percent of the 30 percent reduction. Nitrogen and phosphorus loading caps for the Association are based on these values (NC EMC, 2005, p. 6).

[279] The nonpoint source regulation from 1995 has been insufficient to achieve the ecological settings. More stringent and obligatory regulations have since been installed (NC DWQ, 2001, p. 3; NC EMC, 2005, p. 19).

[280] US EPA (1996b). Rich Gannon confirmed this in a personal communication to the author (August, 24, 2004). At the same time, he states that the information basis for individual monitoring exists; however, neither the Association nor any other institution use this opportunity. Water quality trading in the Neuse River, NC, realised the opportunity to allocate permits individually.

[281] Consequences are drawn in the evaluation of the model (Paragraph 5.2.3.2).

[282] The exact amount per kilogram of pollutants is determined for a two-year period in advance by the Division of Environmental Management (DEM) (NC DEHNR, 1994, p. 16).

[283] North Carolina Division of Soil and Water Conservation (DSWC); Kerr *et al.* (2000, p. 13). More information can be found at the DSWC homepage: `http://www.enr.state.nc.us/DSWC/pages/ manual.html` (January 2007).

[284] The Best Management Practice for nonpoint sources corresponds to the Best Available Technology (BAT) for point sources (Paragraph 3.3.3). Generally, the same argumentation of inefficiencies holds for nonpoint sources. Sources are not free in choosing individual practices. At the same time, the measurement problems for nonpoint sources disallow the introduction of standard market instruments that starts directly at the final discharges. Nevertheless, it is assumed that marketable instruments can adopt to the specifics; higher effectiveness and efficiency is expected. Taxes, charges or even specific permits could base on estimated discharges. However, they need to be defined very carefully in order to be effective and efficient. For an overview, see, for example, Ribaudo *et al.* (1999) or Dosi and Zeitouni (2000).

[285] The trading ratios under the Tar Pamlico Nutrient Trading Program are 2 : 1 ('Animal Best Management Practices') or 3 : 1 ('Cropland Best Management Practices').

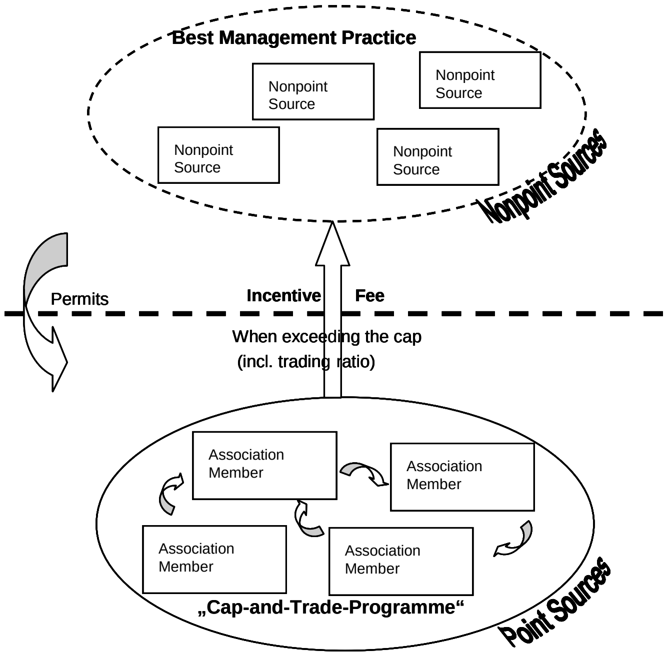

Figure 5-2: Tar Pamlico Nutrient Trading Program (TPNTP)

determined by the Division of Environmental Management (DEM) and are already integrated in the incentive fee currently set at 29$/kg.[286] Figure 5-2 gives an overview of the Tar Pamlico Nutrient Trading Program.

5.2.3.2 Evaluation

The Association did not surpass the cap in Phases I and II (1990-2004) of the Tar Pamlico Nutrient Trading Program. In the period from 1990 to 2003, the discharge load into the Tar Pamlico River decreased by approximately 45 percent for nitrogen and 60 percent for phosphorus despite increases in flow (NC EMC, 2005, p. 7).[287] The predetermined emission cap was over-fulfilled at all times. At a first glance, the Tar Pamlico Nutrient Trading Program appears successful. It is, however, surprising, that (almost) no trading activities took place, whether it be between point sources or between point sources and nonpoint sources.[288] The following analysis uses the criteria developed in Chapter 3 to analyse the Tar Pamlico Nutrient Trading Program in more detail in order to explain this phenomenon.

The evaluation mainly follows the point source perspective: although the participation of nonpoint sources is an important element in this trading system, this study focuses on the integration of point sources. Wherever necessary for a well-founded evaluation, nonpoint sources and their integration are included in the analysis.

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[286] This flat rate is fixed at least for the first two years of phase III (2005-2014) unless new information requires a different setting of the incentive fee (NC EMC, 2005, p. 17).
[287] For an overview of caps and loads for all years of the Agreement through 2003, see Table A-2 and Table A-3 (Appendix). For earlier periods, see NC DWQ (2001) or Hoag and Hughes-Popp (1997, p. 256).
[288] Kerr *et al.* (2000, p. 19). US EPA (1996b) states that "a few point-point trades have occurred". The following evaluation of the programme will show, that the realised trading activities are difficult to observe under the Tar Pamlico Nutrient Trading Program.

Ecological Dimensions

The Tar Pamlico Nutrient Trading Program follows a more emission-based approach (*cap type*). The cap for the Association is defined in nutrient loads (nitrogen and phosphorus) and does not (often) change once it has been fixed for a given period. Ambient quality goals are only indirectly integrated into the system; the emission load caps are based on the Total Maximum Daily Load (TMDL).[289] Total Maximum Daily Loads, in turn, are determined with the objective of achieving a predetermined water quality standard in the river. However, the direct link between emissions and immission has not been taken into account in the system's design; no dispersion coefficients or similar apply. Paragraph 3.2.2 has shown that for the water medium and especially for rivers an immission-based approach might be the most appropriate one in most cases.

A *trading ratio* applies in this model. But: the trading ratio applies for trades between point sources and nonpoint sources and is an uncertainty trading ratio. It reflects the fact that the emission reduction from nonpoint sources' Best Management Practice (BMP) measures cannot be determined without uncertainties. This kind of trading ratio must not be confused with the impact trading ratio discussed before. The impact trading ratio reflects the fact that discharges of sources have differing impacts on the water quality level, e.g. depending on their location (Section 2.5). No impact trading ratio applies under the Tar Pamlico Nutrient Trading Program. The system ignores the fact that discharges could have differing impacts on the water quality. The reason may be that homogeneity between discharges and their impacts is assumed. This is, however improbable (Chapter 2). As a result, trading rules should integrate the most stringent conditions in terms of possible impacts when defining permits in order to avoid hot spots. Again, costs of these different solutions should be compared to find the optimum.

The Tar Pamlico Nutrient Trading Program covers the entire river basin (*geographical scope*). But this river basin lies within national frontiers, therefore the organisational layer is the national level; a transnational cooperation is not necessary.[290] This thus simplifies harmonised water management for the entire river.

Under the construction of the Tar Pamlico Nutrient Trading Program, one assumed that nutrients are quite harmless (up to a certain amount) in rapidly flowing bodies of water (NC DWQ, 2001, p. 2). It is assumed that the difference between discharges and the concentration level does not play an important role. This example shows how important it is to analyse the specific river and the characteristics of the pollutants in this river. An analysis might come to the conclusion that a fine differentiation between emission and immission caps is not the adequate solution. The same goal might be achieved at lower costs by using a simple emission cap; even if the emission cap eventually needs to be set more stringently than would be usually the case under an ambient-based system. A more stringent emission cap would need a safety margin and thus guarantee for ecological effectiveness. The loss of efficiency caused by the integration of this safety margin (Paragraph 3.3.2) might be lower in cost than the introduction of a complex ambient-based system. This must be evaluated for each river and its pollutants specifically as the results will be very different.

The Tar Pamlico Nutrient Trading Program uses estuarine water quality models to define the emission load caps for sources and seeks to "achieve a nutrient reduction goal to address eutrophic conditions in the estuary" (NC EMC, 2005, p. 6). Nutrient caps for sources situated along the river are thus defined

[289] The regulations under the National Pollutant Discharge Elimination System (NPDES) and the technology-based effluent limits do not include nutrients.
[290] Cooperation between participants and organisational subjects remains, of course, indispensable.

in order to achieve a specific water quality standard in the estuary; the Tar Pamlico Nutrient Trading Program follows an endpoint *orientation*. It seems that instream flow needs are not taken into account separately. This is not surprising: as the Tar Pamlico Nutrient Trading Program is a more emission-based system, a differentiated constraint setting is not possible; the influence on the immission is not integrated (Paragraph 3.2.2). The most stringent constraint would define the cap for the whole river.

As the Tar Pamlico Nutrient Trading Program itself defines no explicit *immission cap* within the system, no differentiation of the quality standard in space or over time is possible.[291] This might not be required by the ecological settings. But the *emission cap* itself is fixed and does not change in accordance to varying conditions. If a differentiation of the caps in space is desired, some additional system elements need to be introduced. Without any spatial subdivision, spatial caps are not operable. The system would thus need to be supplemented by spatial elements like zones or receptor points.[292] This would not be the case for differentiated caps over time. A cap defined for the whole river could be adjusted for different time periods, e.g. summer and winter time. This would, however, demand for an adjustment of the permit allocation to sources. A differentiation of the emission cap is thus not possible without making some adjustments to the system. Again, a carefully conducted analysis would show whether the costs of adaptation outweigh the loss in efficiency by a simple emission-based approach. The Tar Pamlico Nutrient Trading Program decided in favour of the emission-based approach.

The constant emission cap does not pay regard to any *water quantity* aspects, neither endogenous nor exogenous variation. This obviously reflects the assumption, that changes in water quantity levels are not significant and/or do not influence the relationship between emissions and immission in a significant way.[293] Whether this reflects the real conditions and thus guarantee an effective trading system must be analysed in detail for every single system to be implemented, before introducing the model.

The emission cap of the Tar Pamlico Nutrient Trading Program itself does not guarantee that hot spots are avoided. It is assumed that the flow characteristics of the Tar Pamlico River generally prevent the occurrence of hot spots. If nevertheless, in single cases, hot spots problems become relevant, the North Carolina Division of Environmental Management (NC DEM) can define specific, more stringent, emission caps for single sources.[294] This has not been the case up to now (NC DWQ, 2001);[295] hot spots have not been a problem. Nevertheless, the question arises whether such a discrete adjustment of emission caps can adequately manage time lags between the measurement of hot spots and the adjustment of the emission cap. The system itself has no mechanism to avoid hot spots as it is an emission-based approach. Again, a safety margin in the emission cap setting could compensate for the lack of mechanism to avoid hot spots; the costs of this solution must be compared with costs of the implementation of a mechanism within the system.

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[291] Emission load caps are defined according to the Total Maximum Daily Load (TMDL) which seeks to reach a specific water quality level.

[292] The *Basinwide Assessment Report* of the NC DENR might be a basis for that (NC DENR, 2003). It analyses water quality as well as stream characteristics for different subbasins of the Tar Pamlico River.

[293] However, the cap setting process demonstrated that water quantity aspects strongly influence water quality at least at different times of the year. Emission load caps had been defined with respect to the year 1991, a very dry year with relatively water levels. They would have been less stringent for wetter years (NC EMC, 2005, pp. 12-13)

[294] NC DEHNR (1994, p. 19).

[295] If a single source needs to reduce its emissions to avoid hot spots, the Association receives credit to its permitted nutrient cap. But if the single source does not achieve the required reduction, the Association pays the penalty (Hoag and Hughes-Popp, 1997, p. 256). The system does thus not set any individual incentive; the Association is rather responsible for the group as a whole. This provokes free rider behaviour and impairs the effectiveness as well as the efficiency of the system.

The Tar Pamlico Nutrient Trading Program at least tries to integrate the nonpoint sources indirectly. The programme thus seeks to exploit the differences in marginal abatement costs between point and nonpoint sources to realise cost savings. While this idea is rational, in principle, the specific design of the Tar Pamlico Nutrient Trading Program risks of violating the ecological effectiveness. The following discussion shows this in more detail.

The Association is to achieve a specified cap. Only if the Association exceeds this cap, is it obliged to pay an incentive fee. Since the fee does not have the same function as a permit, the cap is still violated after the payment is made. The additional abatement to compensate this violation should be realised at the nonpoint sources level. It depends heavily on the funding mechanism of the Agriculture Cost-Share Program whether this additional reduction can be achieved. On the one hand, payments must be set at an adequate level to finance abatement measures; at the same time, abatement measures should be appropriate to achieve the reduction goal determined.

Actually, participation in projects that are funded by the Agriculture Cost-Share Program is voluntary; thus not all nonpoint sources participate. Even if the Agricultural Cost-Share Program projects would lead to reductions at the nonpoint source level,[296] it remains questionable whether the remaining nonpoint sources abate at an adequate level. As this has not been the case for a long time, an additional strategy for abatement measures at the nonpoint source level has been introduced. Even though incentives under this strategy are already higher,[297] measures are mostly oriented at the Best Management Practice (BMP), but not at the actual pollution the sources are responsible for. Emission reductions in return for the payment of the incentive fee are thus not guaranteed as it would be the case under a permit trading system in the original sense. It is thus unlikely that the group of nonpoint sources would be able to compensate for too high emission loads of the point sources; ecological effectiveness is put at risk. So far, the over-abatement of point sources allowed this problem to be ignored. Nevertheless, this design element needs to be reconsidered as it has the potential to erode the effectiveness of the trading system in the long-term.

The analysis of ecological dimensions shows, that the Tar Pamlico Nutrient Trading Program does not pay regard to many of the potential elements of a permit trading system as discussed in Chapter 3. Under the assumption that the difference between emissions and immission is not relevant in the specific case, the simplification of the model is desirable to achieve the ecological goal at least cost.[298] The underlying model cannot, however, be transferred one-to-one to other rivers with more complex dispersion processes. Table 5-1 (see below) summarises the derived results of the ecological analysis.

Even if the simplicity of this model might be appropriate from the ecological perspective in this very specific case, the Tar Pamlico Nutrient Trading Program needs to be analysed in detail from the economic point of view. An analysis regarding to the economic criteria follows.

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296 Another question is whether the capital of the fund is used in an efficient and effective way. A detailed analysis of the Agricultural Cost-Share Program cannot be done here; for further information, see DSWC websites: `http://www.enr.state.nc.us/DSWC/pages/manual.html` (January 2007).

297 Nonpoint sources can choose whether to develop an individual plan that guarantees the abatement requirements; otherwise they need to implement the Best Management Practice (BMP) (Kerr *et al.* 2000, p. 19). The application of sanction mechanisms remains unclear.

298 Intensive monitoring is necessary to determine specific emission/immission conditions for rivers; this is not the task of this study.

Tar Pamlico Nutrient Trading Program (TPNTP)	
General Design Elements	
Cap type	- Emission-based
(Impact) Trading ratio	- Not integrated

Ecological Dimensions	
Geographical scope	- Entire river basin
Orientation	- endpoint constraint
Differentiation of the immission cap in	
Space	- Not possible
Time	- Not possible
Adjustment of the emission cap in	
Space	- Not possible
Time	- Restricted
Water quantity aspects	- Not integrated

Table 5-1: TPNTP, Ecological Dimensions

Economic Criteria

The condition for *cost-effectiveness* has been defined as the equalisation of marginal abatement costs for the theoretical approaches. This condition is difficult to control for practical approaches due to a lack of data (Paragraph 3.3.2). But a careful analysis of the design already shows whether the programme is generally able to equalise marginal abatement costs and thus to come to the least-cost solution.

Trading under the Tar Pamlico Nutrient Trading Program is theoretically organised at the source level, which would be a good basis for the fulfilment of the cost-effectiveness criterion. However, the US Environmental Protection Agency does not allocate the total nutrient cap individually to the point sources within the Association. This would be the task of the Association; but it seems that no precise allocation took place at the source level.[299] Consequently, no 'real' trading within the group of point sources *can* take place. This would, however, be a crucial prerequisite for the system to come to the least-cost solution by balancing marginal abatement costs between sources.[300] Under the actual regulation, an equalisation of marginal abatement costs within the group of point sources would only take place by chance; the least-cost solution cannot be guaranteed.

For transactions between the two types of sources, equivalence of marginal abatement costs cannot be reached either as no real point-nonpoint source trading occurs. If the Association purchases permits from the nonpoint sources or pays the incentive fee, there is no direct equivalence of abatement measures at the nonpoint source level.[301] Due to the fixed incentive fee setting, marginal abatement costs lose their original function; they can no longer function as an indicator to find sources that can

[299] See Paragraph 5.2.3 and US EPA (1996b) and personal communication to the author by Rich Gannon, Nonpoint Source Program Supervisor, August, 24, 2004. It is not obvious, how and with which results sources bargain in order to determine individual emissions and reduction levels.

[300] Only sources within the Association are integrated in the 'permit trading'. Non-members are regulated separately. Reasons for the separation are not obvious. The criterion of cost-effectiveness would require integrating all point sources in order to realise as high cost savings as possible. On the other hand, this exclusion might seek to avoid too high transaction costs for smaller firms.

[301] "It's not individual-to-individual market based trading, but rather collective-to-collective offsetting [...]" (personal communication to the author by Richard Gannon, August, 12, 2004). Up to now, the Association did not pay any incentive fees into the fund; there was no need and therefore no demand for additional permits. Predetermined minimum payments have been made by the Association; but the permits earned by these payments have been banked by the Association. These payments have been used – as provided – for investments at the nonpoint source level, which were able to achieve a reduction of 3 percent from the required 30 percent (Kerr *et al.*, 2000, p. 19).

abate at least cost. Central elements of a permit trading system that are important to realise cost-effectiveness are thus lacking.

An equalising of the marginal abatement costs is not possible, neither within the group of point sources nor between the groups of point and nonpoint sources.[302] The criterion of cost-effectiveness is thus not fulfilled by the Tar Pamlico Nutrient Trading Program. Additionally, one would assume that the non-degradation principle stipulated under the Clean Water Act (Paragraph 5.2.1) applies by means of the implicit integration in the definition of the water quality standards behind the Total Maximum Daily Load (US EPA, 2003a).

Under the Tar Pamlico Nutrient Trading Program free rider behaviour cannot be avoided. Nutrient caps are not allocated to individual sources. The Association as a group needs to achieve the predetermined emission cap; the entire group is responsible in the event of non-compliance. For individual (and maybe small) sources no incentive exists to finance emission reductions as long as the group, as a whole, achieves the aim.[303] This also prevents the system from coming to a cost-effective solution. No sanction mechanism is in place in order to avoid free riding. This is surprising, because emissions are monitored at the individual source level. But this information basis is not used in order to allocate caps to the sources and thus to allow a sanction mechanism to become effective. No individual fining system has been put into action as yet; as a result "some smaller facilities have been 'free-riding' the entire time".[304]

In the case of the Tar Pamlico Nutrient Trading Program it is difficult to evaluate the (a-)symmetry in the position of the point sources as the allocation of permits is not obvious. It is thus questionable whether trading activities can be realised at all. But the analysis of general aspects show that even in the case of individual allocation, asymmetry in the location of sources is not integrated in the system. Trading between sources is not restricted. No trading ratios apply for trades between sources at different locations. No other mechanism is in place to ensure that sources are only able to increase their discharges if the corresponding reductions have been made upstream.[305] Downstream sources can sell their 'surplus' permits to any other source independent from their location; upstream sources can purchase permits from any other source. No restrictions for trading activities are in place. The starting position is symmetric for participating source; all trading potentials can be exhausted (Section 3.3.2). However, this is the result of the fact that the Tar Pamlico Nutrient Trading Program ignores river-specific conditions. Consequently, ecological effectiveness is not assured.

Under the nutrient trading programme in the Tar Pamlico River sources, i.e. sewage treatment plants, can reduce discharges by means of establishing new technologies which reduce the emission of nutrients within the discharged water. Studying the Tar Pamlico Program, one could assume that the Association (the point sources) has no incentive to reduce the emissions beyond the given cap (*dynamic efficiency*). Each reduction beyond the cap causes abatement costs, but no revenue can be generated by selling permits. No 'real' trading is possible, neither between point sources nor between point and nonpoint sources. The investment in new technologies seems to be logical as long as the given cap can be reached in a cheaper manner; any reduction beyond the cap would be surprising. Nevertheless: the members of the Tar Pamlico Association have achieved a reduction that even exceeds the requirements which compensates the missing ecological effectiveness of the nonpoint sources.

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302 For the nonpoint sources no permits exist, thus the question of equivalence is not relevant.
303 Personal communication to the author by Richard Gannon, August, 24, 2004.
304 Personal communication to the author by Richard Gannon, August, 24, 2004.
305 Only reductions upstream create a window for additional discharges downstream if caps are binding.

Tar Pamlico Nutrient Trading Program (TPNTP)

Cost-effectiveness	- No real trade - No equalisation marginal abatement costs, no cost minimisation - Free rider - Non-degradation principle (Clean Water Act)
Dynamic incentives	- Incentives biased - Binding caps?
Transaction costs	- See discussion of Table 5-3 and Table 5-4 (p. 123-126)
Competition, practicability, enforcement	- See discussion on p. 126

Table 5-2: TPNTP, Economic Criteria

Another aspect may be relevant. At the beginning of the Tar Pamlico Nutrient Trading Program each point source was obligated to analyse its management and operation practices for pollution abatement. As a result many low cost innovative methods of pollution control were detected and implemented without using the trading mechanism. "Many of the point sources were able to reduce discharge directly so trade has not been required for a while" (Bjørnlund, 2003, p. 31).[306]

However, this element contradicts the idea of permit trading; it circumvents the mechanism of a permit trading system and reduces trading potentials and thus cost saving potentials right at the beginning of the trading system. Potential efficiency gains by means of abatement measures at sources where it is cheapest thus get lost.

Another explanation for the lack of trading activities, even several years after the implementation, could be the ownership (state or private) of the point sources. In the Tar Pamlico Nutrient Trading Program, point sources are owned by municipalities.[307] If, at the same time, the incentives to compete are missing (Clausen and Scheele, 2002, p. 71), one could expect that the additional costs of abatement can be directly passed through on the next level of the value-added chain or be financed by cross subsidation rather than using the mechanisms of a trading system. From an ill-informed source perspective this would avoid unpopular transaction costs, in particular information and decision costs, within the trading system; potential benefits of trading are not considered due to a lack of information. Consequently, sources have no interest in participating in trading, leading to limited demand on the water quality market; this, in turn, also lowers the cost-effectiveness.

The analysis of theoretical approaches has shown that dynamic efficiency requires binding caps. The cap setting under the Tar Pamlico Nutrient Trading Program is often criticised as 'too weak'; the fact that sources over fulfil their objectives seems to confirm this (see Hoag and Hughes-Popp, 1997). If the determination of the emission cap under the Tar Pamlico Nutrient Trading Program does not need additional reductions, no incentive for further investment in abatement technologies exist. Dynamic efficiency is hindered. Cap setting is thus very important and should be done carefully and with respect to the special conditions.[308] Table 5-2 shows the results of the economic analysis.

The Tar Pamlico Nutrient Trading Program integrates nonpoint sources into the system in order to realise cost savings resulting from differences in the marginal abatement costs. In principle, this idea is

[306] See also NC EMC (2005, p. 6).
[307] NC EMC (2005, p. 24). It has been 12 of 13 according to Hoag and Hughes-Popp (1997, p. 255).
[308] Nonpoint sources that participate at the State's Agriculture Cost-Share Program (ACSP) must apply the Best Management Practice (BMP), which is not dynamically efficient.

rational from an economic point of view, as long as all sources are incorporated equally in the system; however, implementation is complicated. In the case of the Tar Pamlico Nutrient Trading Program, nonpoint sources do not directly participate in trading activities (Paragraph 5.2.3). As a result, already-regulated point sources, that had reduced high parts of discharges in the past, are now burdened with additional abatement costs for reductions at the nonpoint source level. The proportion of pollution from nonpoint sources is still high. It is widely recognised that "a more effective, efficient and fair approach would be to regulate nonpoint sources directly" (Boyd *et al.*, 2007). This would rather correspond to the polluter-pays-principle and avoid high additional financial transfers from point to nonpoint sources.[309]

The majority of *transaction costs* probably arose during the implementation of the system (Hoag and Hughes-Popp, 1997, p. 257). The current transaction costs for the point sources are low – compared to a 'real' trading system. The following analysis shows this in more detail. Again, it should be underlined, that a low level of transaction costs *per se* cannot say anything about the efficiency and the effectiveness of the system.

The simple structure of the Tar Pamlico Nutrient Trading Program does not require any *zone or receptor* point formation. No additional transaction costs need to be paid by the authority.

Also the Tar Pamlico Nutrient Trading Program needs no *simulation model* before any potential change in discharge in order to check the impacts on the ambient pollution level. A water quality model would only be used under the Program in order to define the total emission load for the Association. The emission load bases on the predetermined Total Maximum Daily Load (TMDL). Modelling costs for the environmental authority are thus relatively low.

The Tar Pamlico Nutrient Trading Program prescribes one fixed emission load, no *real-time data* need to be acquired to make the system work. No additional transaction costs is incurred by the environmental authority.

The environmental authority defines the emission load cap for the Association. The allocation to individual sources (initial allocation) is the job of the Association. As the environmental authority is not responsible for the *initial allocation*, no additional transaction costs occur (low monitoring frequency and density). Monitoring costs for the environmental authority are low; only the discharges for the Association as a whole must be monitored.

The Tar Pamlico Nutrient Trading Program follows an emission-based approach. The relationship between discharges and the ambient pollution level is not taken into account or assumed not to be relevant. No *dispersion coefficient* applies. Again, no additional transaction costs need to be paid by the environmental authority.

Trading within the group of point sources is not defined in further detail. The Association should allocate the cap to individual sources; *trading ratios* could be used in order to reflect different impacts of discharges. As, however, the nutrient cap is not allocated to individual point sources no trade in the original sense can take place. The application of trading ratios is thus not relevant. Generally, this

[309] King and Kuch (2003, pp. 10358-10359) also state that additional reductions at the point source level would be cost-intensive. More stringent regulations at the nonpoint source level are politically difficult to implement. The financial burden lies at the point source level, even though the reduction at the nonpoint source level would be less cost-intensive.

trading system assumes that the fact that the discharge of different sources could cause different immission loads can be ignored. From this perspective, no trading ratio becomes necessary. No additional transaction costs arise.

The Tar Pamlico Nutrient Trading Program does not integrate any differentiation of the *emission cap*.[310] Modifications of the cap in space or over time would, in any case, cause additional transaction costs as the programme needs to be adjusted. In order to allow for spatially differentiated emission caps, receptor points or zones need to be defined to make the system operable. This would cause higher transaction costs. A differentiated emission cap over time would be easier to be implemented; different caps in summer and winter time, for example, would not cause too high additional transaction costs for the environmental authority; adjustments in the permit allocation are the job of the Association. These extensions of the model would only be reasonable if the additional benefit of the differentiation compensates the additional costs.

Under the emission-based Tar Pamlico Nutrient Trading Program no conversion from the immission cap to operable emission caps is necessary; no additional transaction costs arise (*determination of the emission cap*).

The environmental authority needs to inform the Association about the emission cap and the incentive fee settings. There are no additional *information requirements*. (Trade-independent) transaction costs in form of information requirements are thus very low for the environmental authority.

Table 5-3 (see below) summarises the transaction costs occurring for the environmental authority. Transaction costs for the environmental authority are relatively low and mainly trade-independent. This fact must be handled with care; the level of transaction costs alone cannot say anything about the efficiency of the system. Low transaction costs can even be the result of abandonment of efficiency.

The transaction costs for the point sources (Association) are relatively low, too (Table 5-4, p. 126). “The only remaining transaction costs for firms is the effort and paperwork to buy the credits” (Hoag and Hughes-Popp, 1997, p. 257).

Permits are defined in terms of emissions from the start; no conversion from immission caps to emission caps becomes necessary (*determination of the emission cap*).

Sources do not need to hold an entire *portfolio of permits* for different receptor points as no spatial differentiation takes place. As permits are not allocated individually, no portfolios can be hold by sources anyway. A permit allows for a certain load of emissions; the impact on other areas of the river (third-party effects) are not integrated nor are they assumed to be relevant. Theoretically, only one single market for permits develops. No additional transaction costs arise.

[310] The differentiation of the immission cap is not treated as the Tar Pamlico Nutrient Trading Program follows an emission-based approach. No direct influence on the immission load is possible.

Tar Pamlico Nutrient Trading Program (TPNTP)

Transaction Costs (TAC), Environmental Authority	
Determination of receptor points, zones or blocks	
- Higher TAC - Lower TAC	 X
Running Water Quality Model (WQM) before any *potential* change in discharges	
- Yes, higher TAC - No, lower TAC	 X
Real-time data	
- Yes, higher TAC - No, lower TAC	 X
Determination of the initial allocation of permits	
- Including Water Quality Model, higher TAC - Not including Water Quality Model, lower TAC - No initial allocation (environmental authority)	 X
Determination of the dispersion coefficient (Water Quality Model)	
- Yes, higher TAC - No, lower TAC	 X
Definition of the trading ratio	
- Endogenous, higher TAC - Exogenous, lower TAC - No trading ratio, no TAC	 X
Differentiation of the emission cap (time/space)	
- Higher TAC - Lower TAC	X
Determination of the emission cap	
- Environmental authority, higher TAC - Sources, lower TAC environmental authority	X
Information requirements	
- Higher - Lower	 X

Table 5-3: TPNTP, Transaction Costs, Environmental Authority

The *value of permits* (in total) is constant and does not change with changing conditions, i.e. relationships between discharges and immission loads. The sources do not incur transaction costs in order to adjust their decisions to the varying values of the permits. Monitoring and reporting costs are on the low side.

No (impact) *trading ratio* applies; sources are not burdened with additional transaction costs (information costs). The uncertainty trading ratio for 'trading activities' between point and nonpoint sources is fixed *ex ante* and included into the incentive fee. (Trade-independent) Information costs for sources are thus relatively low.

When purchasing permits from the nonpoint sources typical transaction costs such as information, bargaining and decision costs in order to find a trading partner do not occur. Point sources pay the incentive fee that has been fixed, no concrete trading partner exists. The environmental authority administrates the funds.[311] Furthermore, the State is responsible for monitoring and verification of Best Management Practice (BMP); once point sources have purchased credits, they are no longer liable for implementation of corresponding reductions at the nonpoint source level (Bjørnlund, 2003, pp. 30-31).[312] No *buyer liability* and thus no additional transaction costs for sources arise (Paragraph 3.3.4).

311 Woodward and Kaiser (2002, p. 376) denote this as a (government) clearinghouse. See also Morgan and Wolverton (2005).

312 Morgan and Wolverton (2005) show that seller liability does not apply in all US trading systems. Buyer liability is more common; a buyer of permits, often point sources, is responsible for compliance of the seller, often nonpoint sources. The discussion in Paragraph 3.3.4 showed that buyer liability can not be reasonable in the case of water quality trading systems with individual trading partners.

Tar Pamlico Nutrient Trading Program (TPNTP)

Transaction Costs (TAC), Sources

Determination of the emission cap	
- Yes, higher TAC	
- No, lower TAC	X
Portfolio of permits	
- Yes, higher TAC	
- No, lower TAC	X
Constant adjustment of the permit value	
- Yes, higher TAC	
- No, lower TAC	X
Trading ratio	
- Endogenous, higher TAC	
- Exogenous, lower TAC	
- No trading ratio, no TAC	X
Buyer-liability	
- Yes, higher TAC	
- No, lower TAC	X

Table 5-4: TPNTP, Transaction Costs, Sources

The fact that the government is responsible for the Agriculture Cost-Share Program (ACSP) and thus for the implementation of reduction measures at the nonpoint source level, reduces information and decision costs for the point sources enormously. It is, however, this same element which prevents the system from coming to an effective and efficient solution.

No specific trade-dependent transaction costs arise for the sources. According to Stavins (1995) this would not additionally provoke the general problem, that the trading system cannot achieve the optimum in the presence of too high transaction costs per trade. However, the analysis shows that other (missing) elements hamper the system from being cost-effective.

It is assumed that no previous instruments regulated nutrient discharges into the Tar Pamlico River before. All transaction costs caused by the establishment and the maintenance of the Tar Pamlico Nutrient Trading Program have thus been relevant for this permit trading system.

Competition, Practicability and Enforcement

For the practical approaches, additional criteria developed in Paragraph 3.3.5 become relevant. The question of distortion of competition, practicability and enforceability plays an important role in existing systems. It lies in the nature of theoretical approaches that these criteria cannot be analysed in detail.

Under the Tar Pamlico Nutrient Trading Program it is difficult to analyse the risk of distortion of competition: permits cannot be traded between sources in the original sense; permits are not allocated to the sources. Typical competitive risks like permit concentrations or price fixing are thus not possible. As no initial allocation takes place under the actual regulation, we can not analyse whether sources are discriminated against by the allocation of permits. As the Tar Pamlico Nutrient Trading Program is contained within North Carolina no competitive bias between participating States or countries is possible (Paragraph 3.3.5); the same regulation applies for all participating sources.

The Tar Pamlico Nutrient Trading Program appears to have been implemented with no resistance. This may be due to the fact that ambitious caps have not been defined either for point sources or for nonpoint sources. One could derive, based on the reductions which exceed the cap at the point source level, that the point sources themselves support the given system; but the caps should be determined in a more challenging manner in order to ensure that the system can achieve full impact and that trading occurs. Then, however, a stronger resistance can be expected. That the point sources also are to enforce the abatement on the part of the nonpoint sources, is – considering the pollution abatement already exceeding the cap on the part of the point sources – certainly justified. An expansion of the system regarding the inclusion of incentives for the nonpoint sources participating at the Agriculture Cost-Share Program (ACSP) as well as for the nonpoint sources not participating, however, would also lead to a stronger resistance.

The general institutional framework needed to introduce a permit trading system exists in North Carolina. However, nutrient discharges introduced into the Tar Pamlico River Basin have not been regulated prior to the implementation of the Tar Pamlico Nutrient Trading Program (interaction with former regulations). As Paragraph 5.2.1 already indicated, nutrients are in general not regulated by the National Pollutant Discharge Elimination System (NPDES) and its technology-based effluent limits. The nutrient trading program can thus be designed beyond this regulation and would thus avoid the general conflicts with the National Pollutant Discharge Elimination System as discussed in Paragraph 5.2.1. The system can profit from higher flexibility in the dischargers' decision making (Stephenson *et al.*, 1999, pp. 804-805).

It is difficult to evaluate the level of planning reliability under the Tar Pamlico Nutrient Trading Program. The lack of individual permits within the group of point sources makes it difficult to guarantee planning reliability. For activities between point sources and nonpoint sources, the planning reliability granted is higher. The cap as well as the price (incentive fee) are fixed in advance and do not change within a time frame of two years. But one has to assume that the incentive fee could be set either too high or too low compared with the fixed objective (cap) and should then be adjusted. Accordingly, the incentive fee set in the Tar Pamlico Program was reduced from 56$/kg nitrogen in Phase I to 29$/kg in Phase II.[313] Planning reliability is thus only granted for two years and the subsequent modification can be extreme.

The ecological settings have been reached within the Tar Pamlico Nutrient Trading Program until now, although the enforcement mechanism in this programme is quite weak. Point sources are acting as a group within the Association. Individual sources do not bear any risk. The group is always sanctioned as a whole; free rider behaviour is provoked. Individual discharges are monitored, a necessary basis for an effective sanction mechanism; nevertheless no individual fines are defined.[314] This is not surprising as the emission cap is not allocated to individual sources. Sources thus have no individual discharge limit; it is thus difficult to define whether a source is in compliance or not. Sanctions, in form of a fine, however, should base on a clearly defined term of non-compliance to be able to be applied automatically. An effective sanction mechanism would be necessary in order to avoid free rider behaviour and to ensure the efficiency of any trading system.

313 Kerr *et al.* (2000, p. 20) and NC DEHNR (1994, p. 17). This fee is continued at least for the first two years of Phase III (NC EMC, 2005, 17).

314 Also in the case of discrete hot spot avoidance at the individual source level, the Association pays the penalty in case of non-compliance of this same source (Hoag and Hughes-Popp, 1997, p. 256).

As mentioned above, in individual instances the environmental authority can set more stringent caps for individual sources in order to avoid hot spots. But even if the source is not in compliance with this specific limit, and hot spots consequently continue to exist, it is the Association who pays the penalty, not the source. The individual source has thus no incentive to reduce emissions, also not in the specific case of hot spots. It is surprising that the ecological objective has nevertheless been reached.

The analysis comes to the result that the Tar Pamlico Nutrient Trading Program is neither ecologically flexible nor economically efficient. The Tar Pamlico Nutrient Trading Program represents an emission-based programme excluding water quantity aspects. It can thus only be used for those rivers and pollutants for which the relationship between emissions and immission can generally be ignored. Differentiations in space or over time are not possible.

The economic analysis shows the main flaccidities of the Tar Pamlico Nutrient Trading Program: no trading activities in the original sense are possible. Consequently, neither cost-effectiveness nor dynamic efficiency is guaranteed by the Tar Pamlico Nutrient Trading Program. Transaction costs are relatively low for both the environmental authority and the sources. A main reason lies, however, in the failure of central elements of a trading system, in the first place the lack of individually allocated permits. At least source-specific allocation of permits is required; this would be the main prerequisite for the efficient functioning of the system.

Additional Remarks

The Tar Pamlico Nutrient Trading Program is only one of the 40 plus trading systems and projects in the United States (Paragraph 5.2.2). But: low trading activities for nearly all trading systems and further analyses in the literature lead to the assumption that the design of other systems has also not set the right incentives.[315] Indeed, the analysis of the Tar Pamlico Nutrient Trading Program and of other permit trading projects in the United States shows that these approaches do not fully coincide with a permit trading in its original sense.[316] Boyd *et al.* (2007) even states for the United States that "no existing water quality management trading program conforms to this image of a market, though some programs do include significant discharger decision making discretion".

Gawel (1993) discusses the term of 'permit trading' in more detail. He finds that it is interpreted differently in the United States and in Europe. While the European term 'permit trading' refers to a permit trading system in its original sense (Chapter 1), the United States 'permit trading' corresponds to an offset system. Offset systems allow regulated sources, generally after having gone to the limits of technology, e.g. by means of a Best Available Technology (BAT) regulation, to invest in abatement measures at the level of unregulated sources in order to offset additional discharges at the own facility.[317] Gawel argues that the correct differentiation between permit trading and offsets forms one

315 See van Mark *et al.* (1992) or Woodward (2003).

316 Huckestein (1996, p. 62, fn. 1) finds similar results for US permit trading systems in air pollution control.

317 Literature often underlines that the offset solution avoids a break in the environmental policy structure; necessary political changes could be hold at a minimum (van Mark *et al.*,1992, pp. 20-21). This argumentation supposes that the CAC is a quantity based allocation instrument, providing permits although without fungibility. By making the advance source-specific CAC permits transferable the creation of a permit-like procedure is assumed. Gawel (1993, pp. 43-45) criticises the generous abstraction behind this argumentation. The authorisation to emit within the CAC does not equal a permit in the original sense of a permit trading system. Permits to be traded within a permit trading system need to be clearly defined in quantity, time and quality to guarantee homogeneity of traded entities (see Paragraph 3.2.2). It is assumed that CAC originally set for pollution control rarely fulfil this requirement. Additionally, legal restrictions might hamper the transmission from CAC to tradeable permits (Gawel, 1993). For further criticism, see Stengel and Wüstner (1997, pp. 29-30).

part of the explanation, why nearly no trading activities took place in the United States permit trading systems.[318]

The structure of the Tar Pamlico Nutrient Trading Program confirms these considerations in part. No permit trading in the original sense takes place. Activities between point and nonpoint sources could rather be interpreted as 'offsetting'. Only in the event that the Association (point sources) exceeds the predetermined cap, does it purchase additional 'permits' by means of payments to the funds in order to offset the exceedance. Nonpoint sources are not allowed to buy permits from point sources. The cap for the Association corresponds to the (low) technical-based command-and-control setting based on the Total Maximum Daily Load (TMDL). One specification of the Tar Pamlico Nutrient Trading Program is the fact that point sources are treated as a single unit; this could at least be the basis for permit trading in the original sense between point sources. The analysis of the Tar Pamlico Nutrient Trading Program demonstrated that even within the group of point sources no 'pure' trading occurs due to specific (lacking) design elements (Paragraph 5.2.3.2).

The analysis of the Tar Pamlico Nutrient Trading Program shows, on the one hand, that the specific design can not be recommended for the implementation of water quality trading systems due to the lack of ecological effectiveness and economic efficiency. Although transaction costs are rather low under the Tar Pamlico Nutrient Trading Program, the lack of elements of a permit trading system in the original sense creates inefficient results. The example of the Tar Pamlico Nutrient Trading Program shows how important it is to design a water quality trading system very carefully in order to integrate the specific river conditions and to achieve an effective and efficient result. As the design of the Tar Pamlico Nutrient Trading Program is not advisable, this programme is not consequently considered in the further discussion. Nevertheless, it is recommended to re-examine the structure of the Tar Pamlico Nutrient Trading Program. At least the precise allocation of permits is necessary. This would create one of the most important design elements and allow for trading activities that in turn guarantee for the effectiveness and efficiency of the system. In addition, only the precise allocation of permits allows for individual sanction mechanisms which are, in turn, important for the enforcement of the system.

[318] Woodward *et al.* (2002, p. 976) identify another difference between a pure permit trading and an offset system. While prices are known *ex ante* under a permit trading system, prices are a result of negotiations under an offset system (*ex post*).

5.3 Experiences – Australia

5.3.1 Statutory Framework

Australia is, similar to the United States, organised in a federal system: the Commonwealth with its States and Territories. The constitution does not give the federal level direct legislative power over the environment (DEH, 2006). The Federal Government's possibility to pursue an environmental policy arises from its capacity to legislate on trade, external affairs, corporations and taxation. Originally, all other policy fields have been assigned to the States and Territories. In the late 1990s, the Federal Government and all State and Territory governments signed an agreement that seeks to establish a clear legislative basis to enable the federal level to actively participate in environmental policy. The power of the federal level under the new legislation is mainly derived from the Federal Government's constitutional responsibility for implementing international treaties and agreements.

The amendment in the policy strategy is based on the fact that most of the environmental issues in Australia are inland-transboundary and concern more than one single State or Territory. Even though most of the decisions on environmental policy are taken at the State level they are part of a stringent environmental policy at the federal level. Bilateral and multilateral agreements for transboundary issues supplement this structure.

The following reflections concern the special case of water policy and in particular salinity policy as a part of the Australian environmental policy. Guidelines, determined at the federal level, exist. They are, however, not all binding. The final implementation lies still under the responsibility of the States and Territories.

Water Policy

Since 1993, the Australian Government has developed the *National Water Quality Management Strategy* in cooperation with the States and Territories. This strategy mainly provides information and tools in order to help communities to manage water resources and to achieve predetermined goals. The National Water Quality Management Strategy forms the framework for the water quality management and is based on policies and principles that apply nationwide (ANZECC and ARMCANZ, 2000, pp. 4-8). The strategy follows the goal, "to protect and enhance the quality of water while maintaining economic and social development" (DAFF, 2006). A three-tiered approach is required: national, State or Territory, and regional or catchment (cooperation principle).

Moreover, the Strategy contains *21 Guidelines* (DAFF, 2006). The application of these Guidelines is not obligatory; they are very broadly defined. They stipulate the determination of the designated water use; water quality standards are based on the water use. States and Territories are responsible for the determination of water quality guidelines and objectives;[319] they are, however, well advised to follow the Guidelines in order to realise an environmental management that is both effective and efficient. The Guidelines show current water use specific issues (salinity, toxic substances,…) and potential indicators,

[319] A 'water quality guideline' is a recommended numerical concentration level or a descriptive statement with respect to the designated use of a body of water. They form the basis for determining 'water quality objectives'. Water quality objectives go a step further; they are targets agreed amongst stakeholders or set by the local authorities (ANZECC and ARMCANZ, 2000, p. 8). They are the relevant indicators, corresponding to water quality standards.

but they do not provide any specific water quality standards as it is not possible given the very wide range of ecosystem types.

The development of the national strategy provides the opportunity to share a national objective and to, at the same time, allow for flexibility at the State and Territory level. The adjustment to local conditions is required. Furthermore, the strategy proposes to use an integrated catchment approach for regulation (ARMCANZ, 1994, p. 6). Finally, it suggests to mix market and command-and-control instruments in order to reach predetermined goals (ANZECC and ARMCANZ, 1994, p. vi). The strategy is supplemented by bilateral and multilateral agreements.[320]

Australian water policy is moving from control to prevention; the emphasis is put on outcomes rather than on prescriptive regulation (ANZECC and ARMCANZ, 2000, p. 5). Pollution should be prevented instead of combating damages that has already been done (precautionary principle). Furthermore, the National Water Quality Management Strategy explicitly postulates the polluter-pays-principle. Sources that cause the pollution should be held responsible for the damage they cause.

Within the national policy framework, water quality protection in the State of New South Wales is handled under the *Protection of the Environment Operations Act 2005 (POEO Act,* NSW EPA, 1997*)*.[321] The Protection of the Environment Operations Act explicitly stipulates the cost-effectiveness of measures and offers the opportunity to introduce a tradeable permit system (Part 9.3A POEO). This Act mainly regulates point sources.[322]

The Protection of the Environment Operations Act requires the prevention of pollution rather than the abatement of damages (NSW, 1997, Chapters 1 and 3) and thus implicitly provides the precautionary principle (Section 3.1). Reduction measures need, according to the polluter-pays-principle, to be implemented at source; under a recommended permit trading system the polluter-pays-principle would apply anyway.

Salinity Policy

Salinity is an issue of great importance in the Australian water policy. The reason is that natural salinity is high in most of the rivers in this country. Therefore, a specific policy developed for salinity problems. In August 2000, the Federal Government of Australia and all States and Territories agreed on the *National Action Plan for Salinity and Water Quality*.[323] This plan forms the basis for detailed agreements between the contracting parties. It contains some key elements like standards and targets, i.e. 'caps', particularly for salinity and water quality or the requirement of integrated catchment management. Standards and targets are developed by the parties with respect to the national standards. The integration of regional natural resource conditions is required explicitly. The National Action Plan for Salinity and Water Quality has no voluntary part; the parties agreed that all elements of this plan must be acted on.

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320 The *National Water Commission Act 2004* established the National Water Commission which has the mandate to implement the *National Water Initiatives* with the aim of improving water management.

321 This Act is the amendment of the *Protection of the Environment Operations Act 1997*.

322 The *Catchment Management Authorities Act 2003* establishes authorities to ensure the appropriate management of the catchments.

323 Commonwealth of Australia (2005).

The cooperation between States and Territories is called for explicitly (cooperation principle).[324] The National Action Plan for Salinity and Water Quality underlines that consistent strategies in the form of integrated approaches for catchments or regions crossing jurisdictional borders are important. Bilateral and multilateral agreements must be established where necessary. The National Action Plan for Salinity and Water Quality applies to 21 predetermined priority regions. The Hunter River, that will be discussed in Paragraph 5.3.3, is not regulated within this plan.

Additionally, New South Wales (NSW) developed its own salinity policy: the *NSW Salinity Strategy*. The Salinity Strategy requires the development of 'end of valley salinity targets' as well as 'within-valley targets' (NSW DLWC, 2000a, p. 4-8). While the former target defines the cap at the lower reach of the river, the latter cap is water or land-based and defines the salinity level to be aimed for at that location, e.g. for the protection of sensitive locations.[325] Translated into terms used in Paragraph 3.2.2, this coincides with an endpoint constraint combined with instream flow needs. The combination of both types is thus explicitly demanded.[326]

The Strategies themselves do not recommend any specific instrument to reach the ecological objectives. However, a working group developed recommendations for the NSW Government (NSW DLWC, 2000a, p. 35). They propose to use market-based instruments and to build upon previous trading experiences in order to install salinity trading systems. They require the recognition of specific catchment salinity issues and the selection of the most suitable system. Furthermore, it is important to define the most appropriate trading unit. The Hunter River Salinity Trading Scheme has been in place since 1995 and is given as an example.

5.3.2 Existing Programmes

In Australia, permit trading as an instrument for environmental protection of rivers has been used since the 1990s (NSW EPA, 2003). The Interstate Salinity Trading for the Murray-Darling Basin came into force in 1992 as part of the *Murray-Darling Basin Salinity and Drainage Strategy* (Kraemer *et al.*, 2004, p. 13). This trading system is administrated by three Australian States: New South Wales, Victoria, and South Australia. Discharge permits (salt) are thought to be exchanged between States, not between individual sources. 'Salt credits' can be earned by investing into the reduction of salt introduced into the water or the increase of water flow.[327] Nevertheless, trading activities between States did not come off; credits have rather been used within States to offset debits from drainage entering the river system (Kraemer *et al.*, 2002, p. 248).

While sources were able to reduce salinity at first, during the 1990s increasing salinity was observed. A new strategy (2001-2015) was developed to make the system more effective (MDBC, 2006). A basin-wide end-of-valley target, with Queensland also participating, had been introduced. Any reduction work within a State that reduces salinity in the shared river will attract additional credits for that State. A State

[324] Neither the polluter-pays-principle nor the precautionary principle is explicitly mentioned under the National Action Plan for Salinity and Water Quality.
[325] Water quality models are used to define these targets.
[326] In addition, management targets can be determined. Management targets define the impact of actions on salinity or the quantity of actions required by the DLWC (NSW DLWC, 2000a, pp. 27-28). An important function of the management targets is to detect where best to invest in actions to reduce salinity.
[327] This system thus integrates the interdependencies between water quantity levels and pollution concentrations in the water (water quality). The importance of this integrated view has been discussed in detail in Section 2.4. It also becomes relevant for the further analysis.

incurs debits for measures that increase the salinity in the shared river. Trading potentials are thus higher than before. The system will be reviewed after 2015.

Australia also has a nutrient trading programme: the South Creek Bubble Licence in the Hawkesbury-Nepean River started in 1996 (Kraemer and Banholzer, 1999, pp. 94-95). Nitrogen and phosphorus loads in the river have been quite high. Sewage treatment plants and other point and nonpoint sources have been responsible for additional discharges. The NSW Environment Protection Authority firstly introduced a 'bubble licence' incorporating a number of sewage treatment plants.[328] The regulator thus only controls the load generated by the group of point sources; no individual discharges are measured and controlled. Sources within the bubble can adjust their discharges by trading parts of their allocations; the total limit is not exceeded. In this way, the bubble solution seeks to achieve a predetermined goal at least cost. Indeed, cost savings are assumed to be AUD 45.6 million or 37 percent (Kraemer *et al.*, 2002, pp. 250-251).

The presentation and analysis of the Hunter River Salinity Trading Scheme follows. This example is selected due to its outstanding structure: in an impressive manner the system integrates the special characteristics of the Hunter River as well as of the substances concerned. Nevertheless, the system has rarely been analysed from an economic point of view. This is the starting point for the following analysis that is based, again, on the results of Chapter 3.

5.3.3 Hunter River Salinity Trading Scheme, NSW, Australia

5.3.3.1 The Scheme

The Hunter River is a major river in New South Wales. It rises in the Liverpool Range and reaches the Pacific Ocean at Newcastle (Figure 5-3). In the region of the Hunter River, agriculture is very important. Also, it is home to more than 20 coal mines and three power stations (NSW EPA, 2003, p. 3). The critical substance in the Hunter River is salt.[329] Salt exists naturally in many of the rocks and soils of the Hunter Valley and thus in the river. In addition, sources such as coal mines and power stations contribute to an increasing salinity in the river by introducing saline water.[330] During coal mining, salty water collects in mine pits and shafts and must be pumped out in order to allow mining operations to continue. Furthermore, power stations use large volumes of river water for cooling. As a proportion of the water evaporates during cooling, the concentration of salt in the remaining water is relatively high. This water is then subsequently reintroduced into the river.[331]

[328] There are only three sewage treatment plants within the bubble. Nonpoint sources are not integrated (Kraemer *et al.*, 2002, p. 250).
[329] The salinity can (quite) precisely be measured using the electrical conductivity (EC) of the water.
[330] It is estimated, that each one EC (electrical conductivity units) increase in river salinity causes a $10,000 per annum loss due to reduced agricultural yields and increased cost of supply and treatment (James, 1997).
[331] Salt does not degrade within the water.

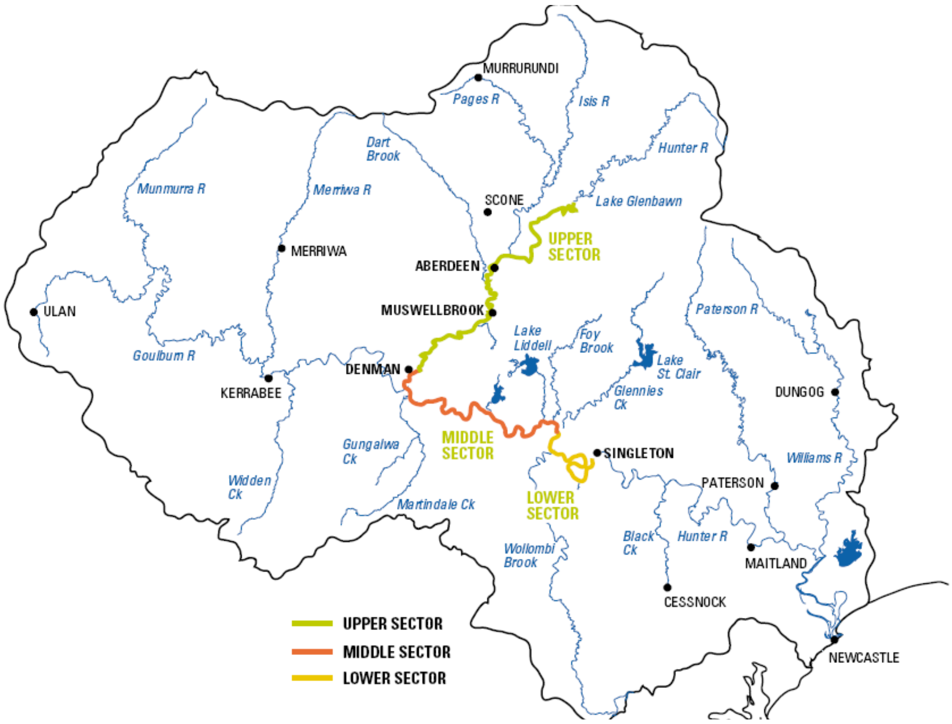

Figure 5-3: Hunter River Salinity Trading Scheme (HRSTS), Geographical Scope (NSW EPA, 2003, p. 3)

Consequently, ecosystems were at risk and the water could no longer be used for irrigation in agriculture. This led the NSW Department of Land and Water Conservation (NSW DLWC) and the NSW Environment Protection Authority (NSW EPA) introducing the Hunter River Salinity Trading Scheme, a system with dynamic and tradeable discharge permits.[332] Figure 5-3 shows the relevant geographical area. The relevant catchment is divided into three sectors: upper, middle and lower sector.[333]

River monitoring has shown that at the beginning of a high flow period ('event', see broken line, Figure 5-4) the salinity of the water increases heavily for a short time, followed by a severe decrease (see continuous line, Figure 5-4).[334] The idea behind the Hunter River Salinity Trading Scheme is the following: point sources can introduce saline water at that moment at which the impact on the water quality is – due to the high volume of fresh water – the lowest possible. While in low flow periods no emissions are allowed, they are allowed – according to the permits – during periods of high flow using the 'window of opportunity'.[335] The permitted amount of emission loads (cap) depends directly on the current salinity and water flow and can thus change permanently (*dynamic* discharge permits).

[332] After a pilot phase the Hunter River Salinity Trading Scheme was finally implemented in September 2002. The basic document is the *Regulation of the Environmental Operations (Hunter River Salinity Scheme) Regulations 2002* (NSW DEC, 2004a).
[333] The geographical scope of the Hunter River Salinity Trading Scheme and the creation of subsectors will be discussed in more details in the following analysis.
[334] The additional volume of water washes salt from the ground and surface into the river. The subsequent volume of fresh water diminishes the salinity of the water (HITS, 2004).
[335] For the exact definition of the terms 'high flow' and 'low flow', see NSW EPA (2002, Part 2, Division 11-14 and Part 3, Division 1, 17). The water quality standard is 600EC for the upper sector of the catchment and 900EC for the middle and lower sector (NSW EPA, 2001, p. 15).

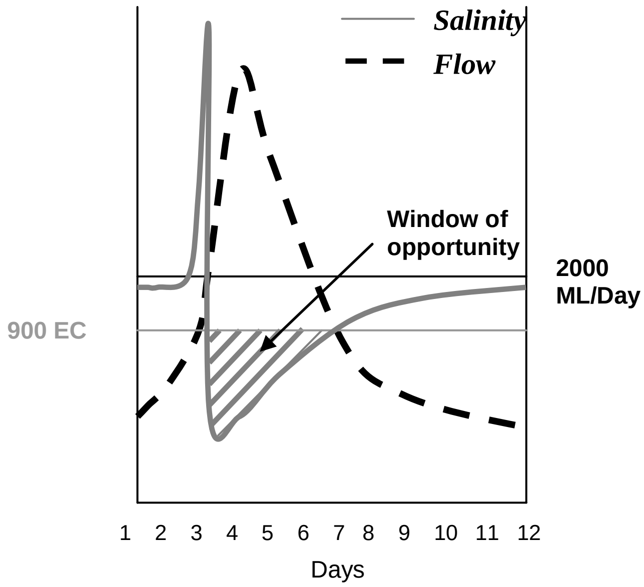

Figure 5-4: Hunter River Salinity Trading Scheme (HRSTS), Salinity and Water Flow According to (NSW EPA, 2003, p. 6)

In a period of flood, saline water can be emitted without permits.[336] This combination of flood periods with tradeable permits should be an incentive, especially to mines, to invest in onsite storage. They can thus wait until flood periods before discharging and would thereby be able to sell their permits during high flow periods to other sources in need or sources incapable of onsite storage.[337]

The total amount of permitted emission loads is defined for 'blocks'. A block is "a body of water that flows down the Hunter River and that is predicted to pass the [...] reference point in a 24-hour period" (NSW EPA, 2002, Division 1, 9, 2).[338] These blocks represent a volume unit of water that is flowing through the river bed from the origin to the estuary. Figure 5-5 illustrates that: block j passes source A within a 24-hour period; some days later block j, i.e. the same volume of water, passes source B. Both sources can discharge (according to the regulations) in block j when it passes through. Although source B introduces its discharges two days later than source A, both sources use the same block j, i.e. the same volume of water.

The water flow as well as the salinity are permanently measured for each individual block (HITS, 2004). Based on these real-time data, the amount of salt that can be introduced in addition (the cap) is defined.[339] In consequence, the application of this system makes it imperative that an intensive

336 Brady (2004, p. 11); NSW EPA (2003, p. 4). While discharges are unlimited during periods of flood flow, NSW EPA (2002, Clause 27) provided that the cumulative effect of discharges under these conditions exceeds the agreed conductivity target, then the same rules that apply to high flows may be used to regulate discharges during subsequent flood flow periods. This provides an incentive to the participants in the Scheme to ensure that the targets are met, even when discharging into flood flows. Personal communication to the author by Mitchell Bennett, Head Regional Operations Unit, NE Branch, Environment Protection and Regulation, Department of Environment and Conservation, July, 14, 2006.

337 Personal communication to the author by Henning Bjørnlund, Associate Research Professor, University of South Australia, July, 11, 2006.

338 In total there are 365 blocks per year which are numerated per day and year. A block is classified, in relation to each sector that it passes through as low flow or high flow or flood flow (NSW EPA, 2001, p. 16).

339 (NSW EPA, 2003, p. 4; NSW EPA, 2002, Division 1, 9).

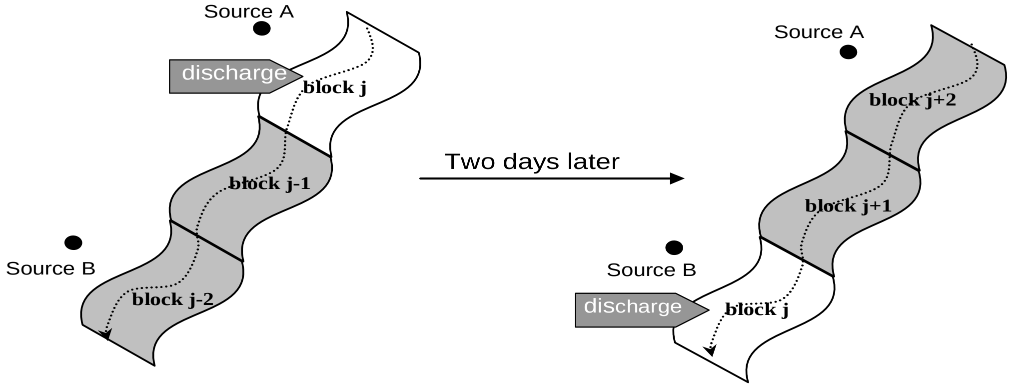

Figure 5-5: Hunter River Salinity Trading Scheme (HRSTS), A Block Approach

monitoring system is available.[340] Such a precise monitoring is also required on behalf of the sources (NSW EPA, 2002, Part 5, Division 3).

In total, 1000 permits were allocated (grandfathering).[341] Each permit allows the licensee – this is another particularity of the system – to introduce 0.1 percent of the total amount of emissions allowed for a defined block into this very block.[342] Due to the determination in percentages, the absolute value of permits (in terms of emission loads) changes according to the actual cap. Sources can trade these permits (*tradeable* discharge permits) for a single block or for sequential blocks.[343] Nonpoint sources are explicitly excluded under the Hunter River Salinity Trading Scheme.[344]

On the one hand, the Hunter River Salinity Trading Scheme allows for a regulated distribution of discharges over time. Only in cases where the water flow is high enough, are discharges allowed. On the other hand, the Hunter River Salinity Trading Scheme can require that the total of discharges decreases by means of more stringent salinity caps.

[340] Monitoring points select information for the whole length of the river. Every 10 minutes, data on the water flow and the salinity of the water are collected and transmitted via radio or telephone to the central data warehouse. River modelling experts use these data to calculate the total emissions allowed (NSW EPA, 2003, p. 7).

[341] Since 2002, the initial permits have different life spans; every two years 200 new permits will be created to replace those that have expired. The total number of active permits is limited to 1000. New permits are sold by auction; newcomer can enter the system by buying permits at auction or from other system participants (NSW EPA, 2003, p. 9). Initially credits had been tradeable only between persons who are allocated those credits by the EPA. Since the first auction of credits, any person is able to buy, hold and trade in credits. A credit does not confer any rights on the holder to discharge saline water unless the person also holds a licence that authorises such discharges NSW EPA (2001, p. 8).

[342] HITS (2004); NSW EPA (2003, p. 5). A 'sector credit discount factor' may be applied, if necessary, to additionally protect water quality (Murtough *et al.*, 2002, p. 30).

[343] Each trade must be approved by the EPA (HITS, 2004; NSW EPA, 2002, Part 5, Division 2, 56). Permits do not expire upon use (NSW EPA, 2001, p. 37). For further information about the trade of permits, see NSW EPA (2002, Part 5, Division 2).

[344] Nonpoint sources have not been integrated into the Australian trading system for the following reason: many of the nonpoint sources discharge on a more or less continuous basis, while the industrial facilities emit on a discrete basis. It is in question whether an emission permit earned by the industries through abatement measures on the nonpoint source level can also allow the point sources to discharge continuously regardless of the river conditions. Personal communication to the author by Qinghong Pu, Centre for Resource and Environmental Studies, Australia National University, October, 27, 2004.

5.3.3.2 Evaluation

The Hunter River Salinity Trading Scheme started as a pilot project in 1995 (Murtough *et al.*, 2002, p. 31). The previous regulation allowed sources to release small amounts of saline water at any time, irrespective of the river flow. Thus, on the one hand, salinity could be very high in low flow periods (risk of hot spots); on the other hand, sources could not discharge more in periods of high flow even though this would not have caused environmental impairments. Consequently, under the previous regulation (by 1992), salinity levels in the Hunter River exceeded the benchmark of 900 EC for 49 percent of days (Bjørnlund, 2003, p. 27).

Under the new trading system this is now down to 3 percent of days. Due to drought conditions in the Hunter region during the early years of implementation participants did not have a great need to discharge saline water.[345] Most of the saline water that participants need to discharge originates from rainwater that has dissolved salt in disturbed soil around active mining areas. Consequently, only few trading activities could be observed prior to 1998. Since 1998 trading activities increased.

Additionally, operators became more familiar with the system and the introduction of an online trading facility in 2000 simplified trading activities. Participants can trade at short notice whenever opportunities arise due to changes in river flow. More than 2000 permits per year were traded between sources in 2003 and 2004 in around 60 transactions per year.[346] Given the relative small number of participants this is an impressive number. The Hunter River Salinity Trading Scheme is one of the first water quality trading systems, where trading activities really occur.

Ecological Dimensions

The Hunter River Salinity Trading Scheme is clearly ambient-based (*cap type*). Permits are defined in salt discharge limits (expressed in tonnes); they are, however, very precisely derived from ambient pollution goals. Salinity concentration in the river should not exceed 900 EC, respectively 600 EC in some cases.[347]

The Hunter River Salinity Trading Scheme does not rely on a *trading ratio*. Permits are traded on a one-to-one basis. Different impacts of the emissions depending on the local position of the sources are reflected by a combined time and space component within the block system. In consequence, the impact of the sources' discharges into a specific block is identical for all sources: all sources directly discharge their pollutants into the same block. Different locations of sources are thus not relevant for the impact.

The Hunter River Salinity Trading Scheme does not span the entire river basin (*geographical scope*). The NSW Environment Protection Authority explicitly defines the relevant catchment for the scheme as the Hunter River and its tributaries upstream of Singleton as shown in Figure 5-3 (p. 134); excluding

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345 Before the Hunter River Salinity Scheme commenced it was calculated that in a median year low flow periods would occur on 89 percent of days, high flow on 7 percent of days and flood flow on 4 percent of days. However, since the implementation in 1995, the Hunter River Valley has experience persistent drought and low flow conditions. For example, during the 2005/2006 financial year only 3 blocks (days) (corresponding to less than 1 percent) were declared high flows and there were no flood flows (Personal communication to the author by Mitchell Bennett, July, 14, 2006). On the one hand, dry conditions lower the discharge possibilities of sources; at the same time, discharge need is lower in dry periods, see above. For flow categorisations for all days since mid 2000 see the River Registers, `http://www.hits.nsw.gov.au/rr/rrindex.html` (January 2007).

346 Own calculation based on EPA data: `http://hrs1.epa.nsw.gov.au/` (January 2007).

347 NSW EPA (2002, Part 3, Division 1, Clause 17).

the Goulburn River upstream of Kerrabee and any tributary that drains into Lake St Clair (NSW EPA, 2001, p. 8). This generally contradicts the criteria developed in Paragraph 3.2.2. A water quality trading system should cover the entire river in order to be effective and to avoid free rider behaviour. In the case of the Hunter River Salinity Trading Scheme the trading system mainly covers coal mines. Most of them are situated upstream from Singleton. The few mines downstream of Singleton are regulated separately with specific licence conditions.[348] Under these conditions a restricted geographical coverage seems to be reasonable. At the same time, one could imagine that this would set incentives for sources of pollution to relocate in order to benefit from less regulated regions. However, in the specific case of coal mines, many other aspects such as coal deposits would mainly influence the choice of location. Nevertheless, should it ever become necessary, the trading system can be extended to further regions.

The cap is defined for the entire catchment; the determination of this cap seeks to reach a predetermined quality at the end-valley (Singleton). At the same time, the Hunter River Salinity Trading Scheme integrates instream flow needs by setting differentiated immission caps. The scheme thus follows a combined instream flow need and endpoint strategy (*orientation*).

As mentioned above, the river is divided into three sub-sectors (upper, middle and lower sector, see Figure 5-3, p. 134) and the *immission cap* is defined separately for these sectors.[349] The immission cap is thus differentiated in space, even though the definition of the sectors is quite broad. The system assumes that instream flow needs do not ask for a finer differentiation in space.

The complexity of the system increases with different immission caps in space. Blocks have different immission caps at different points in time, i.e. for the different locations they pass through. The definition process of the permitted amount of additional saline water should be adapted to that. The model behind the Hunter River Salinity Trading Scheme which defines the final caps needs to consider different immission caps at different time periods, i.e. for different locations. Permits are defined in percentages of the current total emission cap; they are thus prepared to reflect changes in the absolute value of permits, in this case due to modifications of immission caps in space.

A differentiation of the immission cap over time is also possible. The immission cap varies for some sectors with changes in the type of flow period, i.e. over time.[350] Again, the model that permanently defines the actual cap for each block needs to integrate these time specifications. Under a block approach in general, a more accurate differentiation than the one currently integrated under the Hunter River Salinity Trading Scheme would also be possible, should it ever become necessary.

The Hunter River Salinity Trading Scheme is predestined to incorporate changing relations between discharges and resulting water quality (salinity). The cap is permanently redefined integrating actual values of water level and salinity. The *emission cap* is thus adjusted very flexibly in space and over time, even if the immission cap is fixed.

Water quantity aspects are reflected in considerable detail under the Hunter River Salinity Trading Scheme. Real-time data permanently show the conditions in water flow and salinity for every block. The actual cap is defined depending on these parameters. The fact that changing water levels *ceteris*

348 Personal communication to the author by Mitchell Bennett, October, 27, 2006. High in the catchment there are practical limitations restricting the ability of any mine to participate in the Hunter River Salinity Trading Scheme.

349 At the same time, the determination of low, high or flood flow periods, and thus the value of permits, integrates the connection between these sectors.

350 NSW EPA (2002, Part 3, Division 1, Clause 17).

Hunter River Salinity Trading Scheme (HRSTS)		
General Design Elements		
Cap type	-	Ambient-based
Trading ratio	-	Not integrated, not necessary
Ecological Dimensions		
Geographical scope	-	Main parts of the river basin
Orientation	-	Endpoint and instream integrated
Differentiation of the immission cap in		
Space	-	Possible
Time	-	Possible
Adjustment of the emission cap in		
Space	-	Possible
Time	-	Possible
Water quantity aspects	-	Integrated, exogenous

Table 5-5: HRSTS, Ecological Dimensions

paribus influence the water quality, i.e. the salinity level, is thus adequately incorporated in the system. Water quantity aspects are, however, assumed to be exogenous. Endogenous variation in the water level due to water diversion or introduction by the sources themselves is not reflected as such.[351] The incentive to reduce water abstraction in order to increase water levels in the river resulting in a higher value of the permit are not explicitly integrated in the system; changes in the water level are assumed to be exogenous. Reduction requirements that result from changes in the water level are always burdened proportionally to all sources.[352]

The Hunter River Salinity Trading Scheme avoids the risk of hot spots, even for the case of complex ecological conditions. The emission cap for each block is updated permanently with respect to the actual flow and salinity characteristics and is well defined in space and time. Also the immission cap is differentiated in space and over time. The total amount of permits cannot exceed 100 percent. The defined maximum level of salinity cannot be surpassed in any part of the river. Even changing relations between emissions and immission are reflected in time. The results of the ecological analysis can be found in Table 5-5.

Economic Criteria

Bjørnlund states that “the Salinity Trading Schemes emerged as the most cost-effective way of meeting the environmental objectives of the catchment” (Bjørnlund, 2003, p. 27). Again, *cost-effectiveness* would, in the ideal case, lead to equivalent marginal abatement costs. Under the Hunter River Salinity Trading Scheme cost-effectiveness is principally given: sources compare their abatement or storage costs for different blocks with the current price at the market. Given their cost curves, they decide to sell or to buy permits, in other words to abate or not to abate. However, as the permit allows for a discharge defined as an amount in percentages, the absolute amount of discharge is changing over time (dynamic) which influences the development of prices. In consequence, marginal abatement costs – in theory – equalise only for the single block at a particular point in time. The question arises whether this is represented by reality.[353] Nevertheless, costs savings are achieved. The permanent comparison of costs and prices moves abatement measures to sources where it can be realised at least cost.

351 This is the case for the Integrated Water Quantity/Quality System of Weber (2001) (Paragraph 4.3.2).
352 Under the Integrated Water Quantity/Quality System, changes in the water level are assumed to be caused by single sources; resulting reduction requirements are thus also burdened to single sources. Other sources are not influenced by this activities (Paragraph 4.3.2).
353 Confirmed in a personal communication to the author by Qinghong Pu, April, 20, 2005.

It is difficult to define the level of marginal abatement costs. If trades take place mainly among the coalmines of same companies, often very little real money changes hands. Therefore it could be difficult to obtain any information regarding the permit values, and, in turn, use this information to estimate the sources' marginal abatement costs. Additionally, it might be complicated to separate the costs of storing saline water from other waste water management activities.[354]

One criterion for high cost saving potentials is a high diversity of the sources' marginal abatement costs (Paragraph 3.3.2). The final average price at the auction may be an indicator of the actual marginal abatement costs.[355] At the 2006 auction, prices ranged between AUD 532-658 per permit with an average price of AUD 564 per permit.[356] One could assume that marginal abatement costs are similar for all sources.[357] The participants of the Hunter River Salinity Trading Scheme are almost all coal mines and power stations. They all use the water for cooling and the marginal abatement costs are probably not so different in general. But they are for different points in times and therefore for different blocks. In consequence, the condition of diverse marginal abatement costs is assumed to be fulfilled.[358]

Trading activities are brisk under the Hunter River Salinity Trading Scheme. More than 2000 permits per year were traded between about 30 sources in 2003 and 2004.[359] One should take into account that in a median year only 7 percent of days are high flow periods and 4 percent flood flow periods. During the 89 percent of days in low flow no discharges are possible.[360] Taking into account the relatively small number of participants and the few days with discharge opportunities, these trading activities seem high. This may be an indicator of cost saving potentials and differences between the marginal abatement costs for different blocks, i.e. for different points in time.[361]

There can be no free rider behaviour: permits are allocated individually, discharge processes are well defined, and the standards are binding, at every point in time and in any place. No free rider thus prevent the system from coming to the cost-effective solution.

Under the Hunter River Salinity Trading Scheme, the problem of asymmetric starting positions of sources is avoided. The determination of blocks creates a symmetric situation: independent of the location, a source can buy or sell permits for any block. Consequently, also the first source upstream can buy permits for a specific block from downstream sources in order to create an additional discharge window. Similarly, a source located at the end of the river (downstream) can sell surplus permits for specific blocks, if it can reduce discharges for this same block. The block approach is thus able to solve the asymmetric situation on the market. All trading opportunities can be exploited, competition is not biased.

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354 Personal communication to the author by Qinghong Pu, April 20, 2005.

355 In total 200 credits with a life span of 10 years (1 July 2004 to 30 June 2014) were auctioned. These credits replaced the 200 credits which expired on 30 June 2004 (Paragraph 5.3.3.1).

356 NSW DEC (2006). Results for the auction in 2004 are similar: prices ranged from AUD 487 to AUD 551 with an average price of AUD 507 (NSW DEC, 2004b, p.1). The auction raised AUD 112,770 in 2006, respectively AUD 101,467 in 2004. These proceeds are used to fund the cost of running the system (NSW DEC, 2004b, p. 1).

357 Actual prices at the market would give more information on marginal abatement costs. However, data on actual market prices are not available.

358 Personal communication to the author by Qinghong Pu, April, 20, 2005.

359 See `http://hrs1.epa.nsw.gov.au/` (January 2007).

360 Personal communication to the author by Mitchell Bennett, July, 14, 2006.

361 It is assumed that no speculative trading activities take place.

Hunter River Salinity Trading Scheme (HRSTS)

Cost-effectiveness	- Trading activities - Equalisation marginal abatement costs (blocks), cost minimisation assumed - No free rider
Dynamic efficiency	- Incentive to innovate assumed
Transaction costs	- See discussion of Table 5-7 and Table 5-8 (p. 141-145)
Competition, practicability, enforcement	- See discussion on p. 145

Table 5-6: HRSTS, Economic Criteria

The Hunter River Salinity Trading Scheme seeks to reallocate the (reduced amount of) discharges over time depending on water flow conditions. At the same time, a reduction of the total amount of salinity is required. In the case of salinity, reduction possibilities and thus the dynamic incentives set by the system differ from the case of nutrients. It is assumed that changes in production processes are not the main source for reducing emissions of salinity. Sources would rather invest in storage capacities in order to retain discharges in low flow periods and to introduce them in high flow periods according to the permits.[362] The Hunter River Salinity Trading Scheme thus sets the right incentive to abate discharges of saline water, e.g. through storage. Thereby, a surplus of permits is created at a specific point in time and can be sold at the market. As long as the revenues exceed the storage costs, any additional 'abatement' makes sense – even beyond a fixed objective. Binding caps are a crucial condition for the dynamic efficiency; the Hunter River Salinity Trading Scheme guarantees binding caps and thus creates the right incentives.

Again, it is difficult to identify real data; it is even assumed that the *dynamic efficiency* of the Hunter River Salinity Trading Scheme appears insignificant at this stage.[363] The fact that coal mines are moving toward to the upper Hunter River, "where the discharge window might be reduced due to lower river flows, the coal mines might still have incentives to improve their saline water management technology".[364] Table 5-6 summarises the results of the economic analysis.

The discussion of *transaction costs* under the Hunter River Salinity Trading Scheme can be found below, again separated for the environmental authority and the sources. Again, transaction costs cannot be defined in absolute terms. They are instead determined in comparison to the other models discussed.

Under the Hunter River Salinity Trading Scheme (rather permanent) transaction costs arise in order to define the *blocks* and the corresponding monitoring system. Sources need exact information regarding when the block passes through and whether it is a low, high or flood flow block. This information is sent to the sources and is updated permanently.[365] One might think that this causes higher transaction costs than the definition of zones. On the other hand, dispersion characteristics need not to be integrated as under a zonal approach.

Under the Hunter River Salinity Trading Scheme a water quality model is required to define the endpoint and the instream constraint. No prior *simulation* for any potential change in discharges as

[362] The desalination of the water to be discharged into the river would lower the salinity of this water. These processes are, however very cost-intensive; at the same time, the salt remains and cannot be recycled.
[363] Personal communication to the author by Qinghong Pu, April, 20, 2005.
[364] Personal communication to the author by Qinghong Pu, April, 20, 2005.
[365] See `http://www.hits.nsw.gov.au/rr/rrindex.html` (January 2007).

under the Pollution Offset System (Krupnick *et al.*, 1983) is used; no transaction costs are caused for the environmental authority. The discussion of the Hunter River Salinity Trading Scheme, however, showed that water quality models are necessary in order to be able to inform sources in time and permanently about potential discharge opportunities. These models can use the current real-time databases; forecasts are developed. Although permanent running of water quality models is indispensable, additional (trade-independent and permanent) transaction costs might be relatively low, as these models can be directly fed with the current real-time data that are gathered anyway.

The Hunter River Salinity Trading Scheme is the only permit trading system worldwide that uses *real-time data* rather than forecasts (Bjørnlund, 2003, p. 27).[366] Even though this is a crucial element for the impressive performance of this scheme, this causes high (trade-independent) transaction costs (e.g. monitoring costs) for the environmental authority. It is, however, assumed that these transaction costs occur mainly once-off in order to install the monitoring system.[367] The salinity and the water flow must be monitored at numerous monitoring points as well as in short intervals and must then be promptly communicated to the sources (high monitoring frequency and density). Although these costs are indirectly financed by the sources – the auction revenue as well as contributions from the licensees are used for financing – they are indicated at the environmental authority as original payer.

The *initial allocation* under the Hunter River Salinity Trading Scheme does not require any water quality modelling. The relation between emissions and immission is reflected by the block approach and thus does not need to be taken into account by means of additional modelling for the initial allocation.

As the Hunter River Salinity Trading Scheme uses real-time data, other elements that are crucial in the previously discussed models do not matter under this scheme. No water quality models are thus necessary to estimate the impact of discharges on the water quality. Interdependencies between subsectors are also directly determined by the acquisition of real-time data under the block system. The estimation of dispersion coefficients is not relevant for the functioning of the Hunter River Salinity Trading Scheme. This lowers the transaction costs for the environmental authority enormously.

Additionally, no *trading ratio* applies: sources introduce their effluents directly into the block. No differences in impacts between different zones or receptor points are of relevance; they are already integrated in the block system. No additional transaction costs are thus caused for the environmental authority.

Additional differentiation of the *immission cap* would not cause too high transaction costs. The new value should be inserted into the cap definition process. Transaction costs arising from a differentiation in space and/or over time would be very modest. The immission cap would be different for some block and thus at some locations.

The current *emission cap* ('salt discharge limit') in absolute terms (in tonnes) is changing permanently depending on water flow and salinity. Permits are defined in percentages. It is the task of the environmental authority to permanently define the changing emission load cap, derived from the

[366] Forecasts are only used to inform sources in time about potential discharge windows. The trading system itself bases on real-time data.
[367] The maintaining costs of the automatically initialised monitoring processes are lower.

Hunter River Salinity Trading Scheme (HRSTS)

Transaction Costs (TAC), Environmental Authority

Determination of receptor points, zones or blocks	
- Higher TAC	X
- Lower TAC	
Running Water Quality Model (WQM) before any *potential* change in discharges	
- Yes, higher TAC	
- No, lower TAC	X
Real-time data	
- Yes, higher TAC	X
- No, lower TAC	
Determination of the initial allocation of permits	
- Including Water Quality Model, higher TAC	
- Not including Water Quality Model, lower TAC	X
- No initial allocation (environmental authority.)	
Determination of the dispersion coefficient (Water Quality Model)	
- Yes, higher TAC	
- No, lower TAC	X
Definition of the trading ratio	
- Endogenous, higher TAC	
- Exogenous, lower TAC	
- No trading ratio, no TAC	X
Differentiation of the immission cap in space	
- Higher TAC	
- Lower TAC	X
Differentiation of the immission cap over time	
- Higher TAC	
- Lower TAC	X
Determination of the emission cap	
- Environmental authority, higher TAC	X
- Sources, no TAC environmental authority	
Information requirements	
- Higher	X
- Lower	

Table 5-7: HRSTS, Transaction Costs, Environmental Authority

immission standard. Transaction costs for this definition is thus incurred by the environmental authority.[368]

The *requirement to inform* sources is quite high under the Hunter River Salinity Trading Scheme. The changing absolute values of the permits in terms of emissions need to be communicated in time and permanently to the sources in order to ensure planning reliability. To guarantee a sufficient information flow, computer programmes have been installed. On the one hand, sources thus receive detailed information about potential discharges.[369] No active information activities by sources are necessary, they are automatically served with updated data which lowers transaction costs. On the other hand, these systems enable online trading. Thus, a trading partner can quickly be identified. The installation of these information systems finally avoids an extreme increase in transaction costs for point sources.

In order to inform sources in time about the potential discharge windows in the near future, estimations regarding salinity and water flow development are necessary. Water quality models become relevant. The real-time database is available and allows forecasting without causing too high additional

368 In the following, sources are responsible to convert the determined discharge limit into volume discharge limits that depend, in turn, amongst others on the concentration of the discharged water. NSW EPA (2002, Part 3, Division 5, Clause 29) describes the conversion in detail.

369 DLWC monitoring of "weather reports, rainfall in the catchment, streamflows, instream salinity levels and surface conditions (wet or dry) allows the timing and extent of high flow events to be predicted" (HITS, 2004).

Hunter River Salinity Trading Scheme (HRSTS)

Transaction Costs (TAC), Sources	
Determination of the emission cap	
- yes, higher TAC	
- no, lower TAC	**X**
Portfolio of permits	
- yes, higher TAC	
- no, lower TAC	**X**
Permanent adjustment of the permit value	
- yes, higher TAC	**X**
- no, lower TAC	
Trading ratio	
- endogenous, higher TAC	
- exogenous, lower TAC	
- no trading ratio, no TAC	**X**
Buyer liability	
- yes, higher TAC	
- no, lower TAC	**X**

Table 5-8: HRSTS, Transaction Costs, Sources

(permanent) transaction costs. The water quality model 'only' has the task of informing about the discharge window. No further information like dispersion coefficients or trading ratios is necessary. Table 5-7 (see above) summarises the results for the transaction costs to be paid by the environmental authority.

Transaction costs for the sources can be described as follows (Table 5-8). The environmental authority is responsible for the transformation of the immission standard into an emission cap (*determination of the emission cap*). This does not impose additional transaction costs on sources. Sources need then only to convert this limit into volume discharge limits, as is the case for all trading systems.

Sources must hold permits for each block they want to discharge into. A single market evolves for a specific block at a specific point in time. But when trading with other sources, only the permits for the block concerned are affected. Sources are not obliged to change the allocation for other blocks if they buy or sell a permit for a single block as it is the case under the Ambient Permit System, discussed in Paragraph 4.2.1. The environmental authority installed a computer programme which constantly informs sources about the current discharge opportunities. The permanent and precise information load makes the system manageable for sources. These trade-independent transaction costs are not significantly higher than for other trading approaches.

The absolute emission *value of the permit* changes permanently in line with variation in salinity and water flow. But this information is also provided continuously and precisely by the computer programmes. The system seeks to inform sources as early as possible about the future values of their permits and thus try both to lower permanent (trade-independent) transaction costs and to guarantee planning reliability. Transaction costs as well as monitoring and reporting requirements are nevertheless assumed to be more cost-intensive than for the other models presented above.

No *trading ratio* applies in this permit trading system. Sources will not be burdened with additional transaction costs.

The environmental authority is responsible for the monitoring of sources and has to meet the corresponding monitoring costs. No *buyer liability* applies. A source that purchases permits at the market is not obliged to control the seller's compliance. This lowers transaction costs.

Most of the transaction costs for the environmental authority and the sources are trade-independent. The efficiency of trading activities is thus not additionally hampered by specific transaction costs in accordance with the model of Stavins (1995).

The Hunter River has been subjected to intensive monitoring processes even before the implementation of the Hunter River Salinity Trading Scheme. The water quality trading system can thus directly profit from existing monitoring processes. Even though transaction costs appear rather high under the Hunter River Salinity Trading Scheme, the majority of the cost causers, e.g. monitoring systems etc., were already implemented. Transaction costs that arise additionally with the final introduction of the trading system are thus attenuated.

Competition, Practicability and Enforcement

The online portal of the Hunter River Salinity Trading Scheme gives information about the sources' permit holdings.[370] A closer look at the data shows that permits are allocated quite uniformly between sources. Few of them have zero permits;[371] on the other hand, few sources have over hundred permits for most of the blocks. These differences appear to be more influenced by storage possibilities and the size of the source than by market power. As a matter of fact, the final report of the auction in 2004 concludes that no collusion or anti-competitive behaviour was has been detected (NSW DEC, 2004b, p.3). Murtough *et al.* (2002, p. 31) even see more of an advantage in the relative small number of participants than a risk of market power. They assume that the limited number of participating sources in the catchment makes it feasible to regulate the quantity and timing of discharges from each site in such detail to effectively protect the quality of the water. Further research is necessary in order to find the final answer on that.

The region has been looking for a solution for conflicts between the agricultural nonpoint sources and the point sources in the Hunter River Region for sometime already. The Hunter River Salinity Trading Scheme offers a solution which is profitable for all participants. While point sources can continue to discharge saline water – even though only at specific points in time – an appropriate level of water quality is guaranteed for irrigation. Additionally, the possibility of trading offers high flexibility to the point sources, which they appear to exploit. Since the introduction of the trading system (September 2002), around 150 trading transactions took place and nearly 5000 permits have been traded.[372] This data appears rather impressive when taking account of the fact that trading periods (high flow) have been rare. Both point sources as well as the agricultural sector benefit from the permit trading system. This, of course, supports the enforceability of its implementation.

The regulations of the National Action Plan for Salinity and Water Quality do not apply to the Hunter River Basin as it is not under the 21 priority regions (Paragraph 5.3.1). Before the installation of the Hunter River Salinity Trading Scheme, other regulation instruments existed which did not achieve their objectives. Thus, with the introduction of the Hunter River Salinity Trading Scheme all former regulations have been displaced (NSW EPA, 2001). In this way, the problem of the parallel existence of non-compatible instruments is avoided.

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[370] See `https://hrs1.epa.nsw.gov.au/` (January 2007).

[371] This is, for example, the case for the NSW Environment Protection Authority. The EPA can participate in trading activities. However, the EPA does not rely on discharges and is thus not obliged to hold permits. Other sources can only use the flood flow periods for discharges that are permit free.

[372] See NSW Environment Protection Authority website `http://hrs1.epa.nsw.gov.au/`(January 2007).

The Hunter River Salinity Trading Scheme tries to guarantee a high level of planning reliability by installing information systems for point sources which very accurately predict the amount and the point in time for discharges of saline water.

Under the Hunter River Salinity Trading Scheme, the starting position for sources is symmetric. Any source, regardless of the position at the river (downstream versus upstream), can buy or sell permits for any block. Sources with different locations are not discriminated against. Even the first and the last source can buy or sell permits for any block and benefit from transactions. The problems typically associated with the asymmetric situation of sources, like under the zonal or receptor point approaches, are avoided. Competition between sources with respect to the trading potentials is not biased.

The NSW Environment Protection Authority considers financial penalties for violating conditions defined in the system. If necessary, permits can be suspended and transfers of further permits prohibited. Depending on the type of violation (false and misleading information, contravention of trading conditions etc.) the NSW Environment Protection Authority stipulated financial penalties.[373] Only if these penalties are set at an appropriate level and applied automatically in the instances of non-compliance, the sanction mechanism is capable of enforcing the permit trading conditions (Paragraph 3.3.5). The consistent implementation of sanctions also relies on precise monitoring at the source level. Only if the discharge behaviour of sources can be identified for any point in time, i.e. for any block passing through, can non-compliance be detected. Therefore, the precise monitoring by the sources becomes quite important.

[373] NSW EPA (2001, pp. 42-45).

5.4 Conclusion

The analysis of the theoretical approaches has already determined the central elements of water quality trading systems that influence the ecological design and effectiveness of the system. The discussion of selected practical approaches supplements these results by further design elements. The results of the two preceding Sections (5.2 and 5.3) are summarised in a more generalised version.

5.4.1 Ecological Dimensions

As the Tar Pamlico Nutrient Trading Program is emission-based and does not integrate any receptor points or zones, no (impact) trading ratio between point sources can be used.[374] The Hunter River Salinity Trading Scheme also works without a trading ratio; however, the reasons for this are quite different. Under the block approach, sources always introduce their discharges directly into the predetermined block, regardless of their location. The impact on the immission within this block is thus the same, regardless of the location of the source and the time of discharge. This is one crucial advantage of the block approach.

The Tar Pamlico Nutrient Trading Program follows an emission-based approach with endpoint constraints. Crucial amendments to the system would be necessary in order to enable an ambient-based approach reflecting instream flow needs in a differentiated way. The Hunter River Salinity Trading Scheme is ambient-based and also follows an endpoint constraint. The characteristics of the block approach would, however, allow for instream flow need constraints under certain conditions. Whether this additional specification is necessary or not will, again, depend on the ecological settings.

The integration of the entire river into the management system is important in order to guarantee the effectiveness and efficiency of a water quality trading system (Paragraph 3.2.2). However, in the real world, political conditions as well as other aspects could hinder the integration of entire rivers despite the fact that all parties would benefit from this. The Tar Pamlico River Basin lies within North Carolina and is managed as a whole by the environmental authority. This is not the case for the Hunter River Salinity Trading Scheme. Only the central part of the river, where the coal mines are located, is subject to the scheme. It is assumed that the remaining parts of the river could be kept unregulated as no relevant sources of pollution are situated there. While this appears intuitive, this limited coverage might set unwanted incentives. Sources of pollution might react by changing location in order to profit from less regulated areas. In the case of coal mines, as under the Hunter River Salinity Trading Scheme, this migratory process might not be possible due to the local existence of natural resources. For industry, in particular, the non-integration of some parts of the river could set the ‘wrong’ incentives indeed.

The Tar Pamlico Nutrient Trading Program has no possibility of differentiating the immission cap in space or over time as the approach is emission-based. The programme is thus not able to guarantee differentiated ecological results as they may be appropriate. The block approach under the Hunter River Salinity Trading Scheme is predestined to change the quality standards over time; a differentiation in space is possible as well, it requires, however, a higher complexity of real-time data modelling.

[374] An uncertainty trading ratio for ‘trades’ between point and nonpoint sources applies; it is defined *ex ante* and integrated in the incentive fee. No impact trading ratio applies.

	Tar Pamlico Nutrient Trading Program (TPNTP)	Hunter River Salinity Trading Scheme (HRSTS)
General Design Elements		
Cap type	- Emission-based	- Ambient-based
Trading ratio	- Not integrated	- Not integrated, not necessary
Ecological Dimensions		
Geographical scope	- Entire river basin	- Main parts of the river basin
Orientation	- Endpoint constraint	- Endpoint and instream integrated
Differentiation of the immission cap in		
Space	- Not possible	- Possible
Time	- Not possible	- Possible
Adjustment of the emission cap in		
Space	- Not possible	- Possible
Time	- Restricted	- Possible
Water quantity aspects	- Not integrated	- Integrated, exogenous

Table 5-9: TPNTP and HRSTS, Ecological Dimensions

An approach without any spatial subdivision of the river cannot integrate spatially differentiated emission caps. Thus, the difference between emissions and immission is not reflected under the Tar Pamlico Nutrient Trading Program. Any differentiation of the emission cap over time in order to guarantee for the same immission goal is thus not intended. The Hunter River Salinity Trading Scheme reflects any alteration in the relationship between emissions and immission by means of the real-time data basis. The emission cap is thus automatically adjusted to reach the same water quality standard for different flow and salinity conditions.

The simple structure of the Tar Pamlico Nutrient Trading Program does not allow for the inclusion of water quantity aspects, neither exogenous nor endogenous influences. This might be reasonable if it can be proved that water quantity levels very rarely change and thus do not significantly influence the water quality aspects. The Hunter River Salinity Trading Scheme explicitly considers water quantity aspects: salinity levels alter depending on the water flow conditions. Real-time data are used to integrate the current water flow conditions; the actual value of permits and thus trading conditions are based on these real-time data. Water quantity aspects are thus adequately dealt with in the Hunter River Salinity Trading Scheme. They are assumed to be exogenous parameters. The fact that the water diversion activities of sources could also influence the water flow (endogenously) is not considered in the model.[375]

Under the Tar Pamlico Nutrient Trading Program no local concentrations (hot spots) have appeared as yet, although the programme has no mechanism to prevent these concentrations. Nevertheless, it would be suitable to add such a mechanism in order to guarantee an effective trading system, also in the future. Under the Hunter River Salinity Trading Scheme, the individual allocation of permits and the block construction avoids the hot spot problem. The predetermined water quality standard cannot be infringed at any point in time or space if all participants act according to their permits. Our results of the ecological analysis are summarised in Table 5-9 (see above).

The Tar Pamlico Nutrient Trading Program incorporates – as most of the US programmes do – point sources as well as nonpoint sources. Organisers of the Tar Pamlico Nutrient Trading Program have

[375] Paragraph 4.3.2 has shown, that the theoretical model Integrated Water Quantity/Quality System only integrates endogenous water quantity aspects.

realised that the pollution caused by nonpoint sources is still high; therefore, they decided to extend the application of marketable instruments to nonpoint sources. The concept of profiting from lower marginal abatement costs at the nonpoint source level is reasonable from an economic point of view. The specific design elements of the trading system under the Tar Pamlico Nutrient Trading Program show how difficult it is to regulate nonpoint sources in a combined permit trading system. Therefore, it is advisable to regulate nonpoint sources separately from point sources with specific and well adopted programmes.[376]

Nonpoint sources have not been included in the Hunter River Salinity Trading Scheme, because many of the nonpoint sources discharge on a more or less continuous basis, while the industrial facilities emit on a discrete basis.[377] It is in question whether an emission permit earned by the industries through abatement measures on the nonpoint source level can also allow the point sources to discharge continuously regardless of the river conditions (homogeneity of traded entities).

5.4.2 Economic Criteria

For the economic analysis, in particular, the discussion of the practical approaches show that the implemented trading systems do not reflect the textbook version of a permit trading system one-to-one. It is thus crucial to consider in advance which parameters, e.g. conditions of implementation and/or design elements, could prevent the system from coming to an efficient solution (Table 5-10, see below).

5.4.2.1 Cost-effectiveness

The equalisation of marginal abatement costs, can only be achieved if trading incentives and thus demand and supply exist. The process of trading is necessary in order to realise cost saving potentials (Paragraph 3.3.2). This chapter attempts to generalise the experiences from the Hunter River Salinity Trading Scheme and the Tar Pamlico Nutrient Trading Program.

One crucial element of a permit trading system is the allocation of the cap. If the cap is not individually allocated, no trade is possible. Free rider behaviour within the group is probable. No individual sanction mechanism can be applied. In the presence of free riders not all trading potentials are exhausted and further cost saving potential exists. Cost-effectiveness is impaired.

Under the Tar Pamlico Nutrient Trading Program, no individual permit allocation in its original sense takes place. It is thus not surprising that no trading activities have been observed since the implementation. The Hunter River Salinity Trading Scheme, however, does allocate permits to individual sources. The main basis for a water quality trading system is thus realised.

The price of permits is a crucial element of a permit trading system in order to guarantee for efficiency and cost-effectiveness. Participants compare individual marginal abatement costs with the actual price for permits. Depending on their cost structures, sources will abate and sell permits or not reduce pollution and eventually purchase additional permits. If the price is not set freely by the market, the decision making of sources will be distorted.

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[376] The regulation of nonpoint sources cannot be discussed in more detail in this study. For a general discussion of the integration of nonpoint sources, see Segerson and Wu (2006), Camacho and Requate (2004) or Cochard *et al.* (2005).
[377] Personal communication to the author by Qinghong Pu, October, 10, 2004.

	Tar Pamlico Nutrient Trading Program (TPNTP)	Hunter River Salinity Trading Scheme (HRSTS)
Cost-effectiveness	- No real trade - No equalisation marginal abatement costs, no cost minimisation - Free rider - non-degradation principle (Clean Water Act)	- Trading activities - Equalisation marginal abatement costs (blocks), cost minimisation assumed - No free rider
Dynamic incentives	- Incentives biased - Binding caps?	- Incentive to innovate assumed
Transaction costs	- See discussion of Table 5-11 and Table 5-12 (Paragraph 5.4.2.3)	
Competition, practicability, enforcement	- See discussion in Paragraph 5.4.3	

Table 5-10: TPNTP and HRSTS, Economic Criteria

Under the Tar Pamlico Nutrient Trading Program, the price for trading activities within the group of point sources cannot develop as permits can not be traded in the original sense. The price for trades between point and nonpoint sources is fixed by the incentive fee. Prices do not develop at the market. They do not adapt adequately to changing conditions on the market. If the fee is not defined appropriately, market activities would be biased. The result is not efficient.

Under the Hunter River Salinity Trading Scheme the price develops on the market. No intervention distorts development. A crucial condition for the criterion of the cost-effectiveness is fulfilled.

If the environmental cap is set too generously, no demand for additional permits arises and trading activities will be curbed. Caps are not binding, economic efficiency is hampered. On the other hand, sources should be able to comply with the cap; therefore, making the setting too stringent would not be reasonable either.

The cap under the Tar Pamlico Nutrient Trading Program appears to be rather 'weak' compared to the abatement possibilities of sources. Even before the trading programme started, sources had reduced their emission and reached a level which was higher than the standard required. Although no further trading activities occurred, sources did not exceed the cap. Caps are not binding. As long as there is no necessity to reduce discharges, no demand and supply for permits exist; no trading activities take place.

The cap under the Hunter River Salinity Trading Scheme changes with varying water flow conditions. In total, the salinity cap obliges the sources to reduce discharges at least at specific point in times and at specific locations. Binding caps are set; thus there is no room for free rider behaviour.

Only if property rights are clearly defined and guaranteed for a long-term period, sources have an incentive to trade and to invest in pollution reductions. An unclear definition or an uncertainty in the validity of permits can make trading and thus abatement activities too risky.

A changing element under the Tar Pamlico Nutrient Trading Program is the incentive fee; the fee is adjusted every two years. These modifications can be high (Paragraph 5.2.3). Uncertainty regarding the fee could influence the decision processes of point sources. Modifications of the fee change the financial trading conditions. Uncertainty regarding the future level could hamper trading activities.

Under the Hunter River Salinity Trading Scheme the actual value of the permit is changing continuously. To avoid a higher uncertainty at the source level, information programmes have been installed in order to use the flexibility of the model without reducing the planning reliability of sources and thus the trading activities.

If the legal framework stipulates the application of the non-degradation principle, as it is the case under the Clean Water Act in the United States, cost-effectiveness is also hindered (Paragraph 3.3.2).

Both approaches avoid asymmetry between sources: each of the sources can trade with any other source independently of their location. The reasons for this are very different. The Hunter River Salinity Trading Scheme allows for symmetric trading potentials by means of the construction of blocks and the integration of impact differences; immission goals are not at risk. The Tar Pamlico Nutrient Trading Program avoids the asymmetry through the emission-based approach. Different impacts are ignored; sources are not discriminated against by their location.

5.4.2.2 Dynamic Efficiency

Whether dynamic incentives can be realised in practice depends heavily on the specific design and the institutional embeddedness. For theoretical models, Section 4.4 has shown that binding caps are a critical prerequisite for the existence of the dynamic incentive to innovate. Also, in practice, the setting of the cap is of great importance. If the cap is set too generously, as this is assumed for the Tar Pamlico Nutrient Trading Program, sources are not obliged to reduce discharges; caps are not binding. They consequently have no incentive to innovate and to install new abatement technologies. Free rider behaviour might emerge. Sources might rather attempt to benefit from other sources' activities. This behaviour is also encouraged by the missing individual allocation of permits. The price setting also plays an important role. Prices set by the regulator (see Tar Pamlico Nutrient Trading Program) risk of reflecting incorrect conditions. Incorrect price signals could hinder the willingness of sources to innovate. The Hunter River Salinity Trading Scheme, on the other hand, guarantees binding caps and market prices and realises dynamic incentives.

The dynamic incentive is also influenced by the co-existence of other instruments. The Tar Pamlico Nutrient Trading Program obligated sources to approve their production and operation processes before the trading system as such started. Sources thus reduced discharges by changing their production conditions without using the trading mechanism; market results are thus hampered. The incentive to sell surplus permits at the market is biased; all sources reduced their discharges, no (significant) demand emerges.

The incentive to invest in new technologies in order to reduce water use by means of diversion is not integrated, neither under the Tar Pamlico Nutrient Trading Program nor under the Hunter River Salinity Trading Scheme. The Tar Pamlico Nutrient Trading Program does not reflect water quantity aspects at all. The Hunter River Salinity Trading Scheme only considers exogenous changes in water quantities. The direct link between water diversion and water quality is thus not reflected. Sources have in turn no incentive to invest in new technologies in order to reduce water diversion.

	Tar Pamlico Nutrient Trading Program (TPNTP)	Hunter River Salinity Trading Scheme (HRSTS)
Transaction Costs (TAC), Environmental Authority		
Determination of receptor points, zones or blocks		
- Higher TAC		X
- Lower TAC	X	
Running Water Quality Model (WQM) before any *potential* change in discharges		
- Yes, higher TAC		
- No, lower TAC	X	X
Real-time data		
- Yes, higher TAC		X
- No, lower TAC	X	
Determination of the initial allocation of permits		
- Including Water Quality Model, higher TAC		
- Not including Water Quality Model, lower TAC		X
- No initial allocation (environmental authority)	X	
Determination of the dispersion coefficient (Water Quality Model)		
- Yes, higher TAC		
- No, lower TAC	X	X
Definition of the trading ratio		
- Endogenous (higher TAC)		
- Exogenous (lower TAC)		
- No trading ratio, no TAC	X	X
Differentiation of the immission cap in space		
- Higher TAC	-	
- Lower TAC	-	X
Differentiation of the immission cap over time		
- Higher TAC	-	
- Lower TAC	-	X
Determination of the emission cap		
- Environmental authority, higher TAC	X	X
- Sources, no TAC environmental authority		
Information requirements		
- Higher		X
- Lower	X	

Table 5-11: TPNTP and HRSTS, Transaction Costs, Environmental Authority

5.4.2.3 Transaction Costs

The most relevant transaction costs for both practical approaches, Tar Pamlico Nutrient Trading Program and Hunter River Salinity Trading Scheme, will be compared below. For the environmental authority, the analysis shows that transaction costs are mostly higher under the block approach of the Hunter River Salinity Trading Scheme than under the rather emission-based Tar Pamlico Nutrient Trading Program (Table 5-11). The determination of blocks, the use of water quality models and real-time data, the initial allocation of permits, and the information requirements cause higher transaction costs for the environmental authority under the Hunter River Salinity Trading Scheme. These design elements are necessary in order to reflect river conditions and to allow for effectiveness and efficiency (Paragraph 5.3.3).

In addition, the Hunter River Salinity Trading Scheme causes transaction costs that do not arise under the Tar Pamlico Nutrient Trading Program: the differentiation of the caps. The Tar Pamlico Nutrient Trading Program is emission-based and thus not capable of integrating differentiated caps. The Hunter River Salinity Trading Scheme reflects constantly changing impact conditions and in time. Again, the higher transaction costs are justified by better performance.

Both approaches, Tar Pamlico Nutrient Trading Program and Hunter River Salinity Trading Scheme, do not use dispersion coefficients and impact trading ratios; in consequence, there are no additional transaction costs.[378] It is, however, important to analyse the specific reasons for not integrating these design elements. While the Hunter River Salinity Trading Scheme integrates different impact conditions by defining the blocks and thus does not rely on the definition of trading ratios, the Tar Pamlico Nutrient Trading Program simply ignores the potential differences in the impacts of discharges as is typical for emission-based approaches. Both approaches avoid additional transaction costs. But while this does not affect effectiveness and efficiency in a negative way under the Hunter River Salinity Trading Scheme, non-integration under the Tar Pamlico Nutrient Trading Program hinders effectiveness and efficiency.

At first glance, it seems that the transaction costs for the sources are rather similar (Table 5-12, see below). It is not the task of sources to determine the emission cap, resulting from the immission cap. The Tar Pamlico Nutrient Trading Program is emission-based and permits are already determined in terms of emissions by the environmental authority. Also under the Hunter River Salinity Trading Scheme, the environmental authority is responsible for the conversion from the immission goal to emission-based values for any block.

Both approaches avoid the need to hold portfolios for numerous receptor points. Under the Tar Pamlico Nutrient Trading Program, the river is not divided spatially, e.g. in form of receptor points. This lack of subdivision of the river enables trading activities to be carried out independently of the location; due to the emission-based approach, no holding of portfolios is necessary. The Hunter River Salinity Trading Scheme avoids the needs for portfolio by defining blocks. Relationships between discharges of different locations are neutralised by these blocks; impacts on other parts of the river are integrated. No portfolio is necessary.

While absolute permit values are fixed under the Tar Pamlico Nutrient Trading Program, they are constantly changing under the Hunter River Salinity Trading Scheme. This causes higher transaction costs for sources. The constant adjustment is justified by changing river conditions which need to be reflected by the trading system.

No trading ratio applies under both the Tar Pamlico Nutrient Trading Program and the Hunter River Salinity Trading Scheme. While the reason for that differs significantly for both models, the resulting transaction costs for sources are low in both systems.

In both systems buyer liability applies and avoids high transaction costs for sources. After the purchase of permits the buyer is not obliged to control compliance of the seller.

The analysis of the Tar Pamlico Nutrient Trading Program and the Hunter River Salinity Trading Scheme leads to two important conclusions. Firstly, the level of transaction costs alone gives no information about the efficiency and the advantage of a trading system. Transaction costs under the Tar Pamlico Nutrient Trading Program are relatively low, for both the environmental authority and the sources. At the same time, the analysis shows that the low level of transaction costs is caused

[378] The Tar Pamlico Nutrient Trading Program uses uncertainty trading ratios. These are only relevant for transactions between point and nonpoint sources. These trading ratios are exogenous, fixed for a certain period of time and already included in the incentive fee. Additional transaction costs for sources are low.

	Tar Pamlico Nutrient Trading Program (TPNTP)	Hunter River Salinity Trading Scheme (HRSTS)
Transaction Costs (TAC), Sources		
Determination of the emission cap		
- yes, higher TAC		
- no, lower TAC	X	X
Portfolio of permits		
- yes, higher TAC		
- no, lower TAC	X	X
Permanent adjustments of the permit value		
- yes, higher TAC		X
- no, lower TAC	X	
Trading ratio		
- endogenous, higher TAC		
- exogenous, lower TAC		
- no trading ratio, no TAC	X	X
Buyer liability		
- yes, higher TAC		
- no, lower TAC	X	X

Table 5-12: TPNTP and HRSTS, Transaction Costs, Sources

significantly by the lack of central elements of a permit trading system. The low level of transaction costs is thus overcompensated by high losses in efficiency.

Secondly, the discussion has shown that the Hunter River Salinity Trading Scheme causes rather high transaction costs for specific design elements. The measurement of real-time data creates high transaction costs; the constantly changing value of permits (in percentages) requires flexibility at the source level. On the other hand, this design allows to pass on the development of trading ratios and the implementation of water quality trading models. Real-time data avoid the uncertainty of water quality forecasts. However, higher transaction costs under the Hunter River Salinity Trading Scheme allows a better ecological and economic performance.

5.4.3 Competition, Practicability and Enforcement

Aspects of competition cannot be evaluated under the Tar Pamlico Nutrient Trading Program due to the missing clear-cut individual allocation of permits. Under the Hunter River Salinity Trading Scheme no market power can be observed. Different aspects might be responsible. Sources are active in different economic sectors and thus not in direct competition. The buying-off of permits does not appear to be reasonable for individual firms as they do not act in the same relevant market. Furthermore, the financial capacity of individual sources might be to small to buy the necessary number of permits. Asymmetric trading positions and discrimination in competition are avoided under both the Hunter River Salinity Trading Scheme and the Tar Pamlico Nutrient Trading Program.[379]

The practicability and feasibility of a permit trading system depends on the attitude of those concerned. No significant resistance of the potential participants could be observed, neither under the Tar Pamlico Nutrient Trading Program nor under the Hunter River Salinity Trading Scheme. This might have different reasons. Caps are not set bindingly under the Tar Pamlico Nutrient Trading Program, no

[379] In the case of the Tar Pamlico Nutrient Trading Program, this lies in the emission-based approach and thus leads to other inefficiencies.

stringent reduction requirements exist. In the case of the Hunter River Salinity Trading Scheme, different previous regulations did not come to satisfactory results. But all parties can benefit from the implementation of the permit trading system: sources can continue to introduce saline water; agriculture can use the water for irrigation. Resistance was thus also low (NSW EPA, 2003).

The coexistence of a trading system and previously installed incentives like subsidies or other state programs can hinder the system from coming to a sufficient solution by compensating the incentive to participate in trading activities. It is assumed that this problem does not exist under both the Tar Pamlico Nutrient Trading Program and the Hunter River Salinity Trading Scheme. No other instruments exist in parallel.

The enforcement depends heavily on the design of (individual) sanction mechanisms. If sanctions are lacking or not automatic in their application, enforcement of the system would be hampered. The Tar Pamlico Nutrient Trading Program does not define individual fines for non-compliance.[380] The Hunter River Salinity Trading Scheme previews financial penalties for different cases of non-compliance. Only if these sanctions are automatically applied in the event of non-compliance, can they enforce the trading system.

5.4.4 Implications

The Tar Pamlico Nutrient Trading Program is excluded from further discussion within this study. The previous analysis comes to the conclusion that the current design of this programme cannot be recommended. The lack of central design elements of a permit trading system cause inefficiencies and may risk the effectiveness of the system. The criteria-based analysis has shown where the design of this trading system must be improved in order to be effective and efficient.

The trading mechanisms of the Hunter River Salinity Trading Scheme (HRSTS) lead to an effective and efficient solution. The design of this trading system is innovative and enables to consider specific river conditions and to fulfil ecological requirements. Thus, this approach is integrated in the following chapter, which develops design proposals and links the results form the analysis of theoretical and practical approaches.

[380] As permit are not allocate individually, non-compliance on an individual level is very difficult to determine!

6 Linking Theoretical and Practical Approaches: Proposals

Chapter 6 provides an integration of the experiences of the theoretical and practical approaches by linking the discussed models. In the current literature, only the specific problems of one trading system or another were examined in greater detail. An integrated analysis of the theoretical and practical approaches does not yet exist. The standardised and comparing analysis shows that some design elements of the different trading systems are similar. On the criteria-based fundament developed in Chapter 3, proposals concerning the adequate design of trading systems can be developed.

This Chapter has two main objectives. Firstly, the results from Chapters 4 and 5 are integrated in Section 6.1. The discussion of theoretical and practical approaches identified four main relevant types of water quality trading system: receptor point approaches such as the Ambient Permit System; zonal approaches such as the Trading Ratio System; block approaches such as the Hunter River Salinity Trading Scheme in New South Wales, Australia, and the combined water quantity and quality approaches such as the Integrated Water Quantity/Quality System.[381] Each of these types is characterised by a different ecological flexibility, i.e. differentiation potential. At the same time, the economic performance is slightly different for these approaches. In some cases it might be reasonable to combine design elements within a further, modified model. This study develops, for the first time, proposals that are derived from the standardised and comparable analysis of theoretical and practical water quality trading approaches. We shall evaluate and develop design-specific proposals by following an approach-oriented analysis. The final (criteria-based) results of the ecological and economic analysis for the remaining approaches can be found in Table A-4 to A-7 (Appendix).

Secondly, Section 6.2 makes general proposals concerning the area of implementation. It seeks to identify conditions that may hinder or encourage the implementation of a water quality trading scheme. Two countries provide examples which are examined in more detail in order to give an initial idea where the application of water quality trading schemes may be reasonable.

6.1 Design-specific Proposals

6.1.1 General Design Elements

The performance of an approach depends in any case on the specific river conditions and on the determination of ecological objectives. Proposals can thus not be developed for a single approach as long as the implementation area and the ecological settings are not defined.

One main difference between the four remaining approaches lies in the definition of the reference unit of the permits. While permits under the receptor point approach are related to a fixed point within the river, permits under a zonal approach are defined with respect to zones based on predefined sections of the river bank. Permits under a block approach are defined for a volume of water at a specific location, i.e. for a specific point in time. It stands to reason that these approaches are able to realise different forms of 'depth' of homogeneity, i.e. different ecological dimensions (Section 3.1).

381 The Pollution Offset System has been excluded from the discussion due to a lack of efficiency (Paragraph 4.4.3). Also, the approach of the Tar Pamlico Nutrient Trading Program is not discussed in the following as this design is weak due to several inefficiencies (Paragraph 5.4.4)

Before design-specific proposals follow, one aspect that is relevant for all remaining approaches is discussed here: the differentiation of the immission cap. The differentiation of the immission cap in space is generally possible for any approach which divides the river into sectors or similar. Water quality standards and thus immission caps can be spatially differentiated from the beginning. All remaining approaches use design elements which result in a higher flexibility in the cap setting process.

At the same time, it is difficult for most approaches to integrate an immission cap that is differentiated over time. This may be the reason for the fact that this kind of differentiation is rarely mentioned in the literature. Changes in the immission cap over time, e.g. motivated by different water uses, would automatically require a re-adjustment of the allocation as the total number of permits cannot be kept constant. The discussion in Chapters 4 and 5 occasionally has referred to the possibility of a flexible adaptation to immission caps over time by means of the introduction of permits defined in relative values. A permit would thus not contain the right to emit a constant amount, e.g. in terms of absolute emission loads; the permit would rather be defined in terms of percentages. Instead of a definition in absolute terms, relative terms are used. Any permit would thus allow the emission of a certain proportion of the changing total emission cap resulting from the changing total immission cap for a specified spatial area. The allocation would thus change proportionally for all sources with any modification of the immission cap.

While in the case described above, relative permit values are used in order to reflect the changing immission caps over time, this design element can also be used under constant immission caps if a changing relationship between emissions and immission needs to be reflected. Discharge windows in terms of total emission loads may change with changing conditions in water flow etc. even if the immission cap is fixed. Relative permit values are able to adjust the emission cap with respect to the immission cap.

This element can be found under the Hunter River Salinity Trading Scheme.[382] The definition of permits in relative terms is assumed to cause additional transaction costs. Sources must adapt their decisions to the changing conditions. The environmental authority is responsible for determining the changing immission caps over time; permanent information about the current conditions within the river must be re-evaluated in order to find the effective cap.

Note again, that not all cases ask for a constant adjustments of the immission cap. In these cases, additional transaction costs would not be justified as neither ecological effectiveness nor economic efficiency would be higher afterwards. However, in cases that require further and constant differentiation over time, a relative definition of the permit value is an effective design element. Transaction costs caused by the integration of this element could be kept within a limit by means of the integration of computer-based information programmes for sources as well as for the regulating authority.

6.1.2 Receptor Point Approaches

Paragraph 4.2.1 defined the receptor point approach (Ambient Permit System) as an immission-based approach; each source must hold a portfolio of permits for any of the affected receptor points since

[382] Under the Hunter River Salinity Trading Scheme the determination in percentages is mainly used in order to realise a constant immission cap with changing emission caps due to changing relations between emissions and immission.

each receptor point forms a single market. This is an important disadvantage compared to all other approaches.[383]

Furthermore, the receptor point approach has been criticised, in particular, for the lack of cost-effectiveness. Only for specific initial allocations, that are very difficult to define, would the least-cost solution be achieved. The optimum would be reached rather by chance. Only binding caps would guarantee cost-effectiveness.

The receptor point approach is nevertheless still considered as it forms the basis for the Integrated Water Quantity/Quality System which creates an innovative model with combined water quantity and quality markets. The next paragraph will make proposals in order to maintain the advantages of the receptor point approach without having to tolerate the disadvantages. A main advantage of the Ambient Permit System is the high degree of homogeneity of traded entities.

Both of these problems, namely the management of portfolios and non-binding caps, could be solved by setting caps zone-by-zone and integrating interdependencies in terms of emissions and immission levels as under the zonal approach (Trading Ratio System). However, the main differentiation potential of the receptor point approach would thus be lost; a constant adjustment of the emission cap would no longer be possible. The zonal approach is only appropriate for cases with a constant relationship between emissions and immission.

If higher flexibility regarding the nexus between discharges and water quality impacts is required, by natural or political conditions, it would be more reasonable to choose a block approach. A receptor point approach in its 'pure' version as presented in Paragraph 4.2.1 cannot be suggested for implementation. The risk of a lack of efficiency is too great.

6.1.3 Zonal Approaches

The zonal approach (Trading Ratio System) is characterised by high economic performance. The step-by-step initial allocation of permits which integrates interdependencies between zones allows for the setting of binding caps and can thereby guarantee cost-effectiveness. This specific initial allocation, however, influences the ecological flexibility of this model (Paragraph 4.3.1). Ecological suitability is only attained when the relationship between emissions and immission can be assumed to be (almost) constant. At the same time, the zonal approach is predestined to incorporate spatially differentiated immission caps.

One could consider mechanisms for integrating dynamic aspects. Relative emission caps, for example, would allow for changing immission caps over time. The zonal approach could than also be implemented in cases where the relationship between emissions and immission changes over time. In this case, the remaining discharge window for a given water quality standard in a specific zone would need to be constantly redefined with respect to all other zones. The system would be much more complex, water quality models would need to be run permanently in order to redefine the emission load window and the trading ratios. Specific advantages of the model in its original design such as

[383] One could argue that receptor point approaches are a specification of a zonal approach. In this study we defined these approaches according to the design proposals from different researchers. The zonal approach (Trading Ratio System) avoids the need to hold portfolios for sources by integrating the emission-immission relation into the permits by means of the initial allocation (Paragraph 4.3.1). the pure receptor point approach (Ambient Permit System, Paragraph 4.2.1) does not integrate such specific initial allocation.

exogenous trading ratios and constant permit values would be lost. At the same time, the asymmetric trading position between sources would persist.

If flexibility of this nature is required, it would be advisable to introduce a block approach that is extremely resilient with respect to changing conditions and the adjustment of caps. At the same time, the turning our back on the zonal concept allows the application of trading ratios to be abandoned. The asymmetric trading position of sources would also be avoided. The block approach would thus be able to reflect more ecological dimensions and is highly efficient; transaction costs are assumed to be lower.

The zonal approach is thus well suited for rivers with a constant relationship between emissions and immission at least for certain periods of time, e.g. summer or winter months. In the case of rivers that ask for greater flexibility, i.e. an increasing 'depth' of homogeneity, a block approach is more advisable.

One major disadvantage of the zonal approach is the asymmetry in the trading positions of sources. The relevance of the effects on competition and cost-effectiveness depends on the number of sources per zone, their cost structure, and discharge conditions. If specific conditions make it possible to ignore competitive and cost-effectiveness effects, the application of a zonal approach remains reasonable. In the case of strong competitive distortions, however, it would be more reasonable to choose the block approach. The block approach may often be too complex with respect to the environmental conditions, i.e. too complex in comparison with the actual capacities required. The potential to flexibly reflect constant changes in the relationship between discharges and impacts may not be necessary in some cases. However, if the asymmetry in the starting positions prevents the system from coming to an efficient solution, a simplified block approach would be recommendable. A constant relationship between discharges and impacts would lower the complexity of the block approach and thus reduce transaction costs. Permit values would be almost stable if the relationship between emissions and immission is constant over time; sources do not need to adapt to permanently changing permit values. The central cost causers of the original approach become less important.

To conclude: a zonal approach is preferable in cases with constant relationships between discharges and their impacts on water quality. Only for this constellation, can the zonal approach guarantee hot spot prevention. But: a zonal approach can only be taken into account without reservations, if the asymmetric trading position of sources is negligible in the specific case.

6.1.4 Block Approaches

If high flexibility is necessary for accomplishing the river-specific and/or ecological objectives, it is advisable to implement the block approach. The block approach, as under the Hunter River Salinity Trading Scheme in New South Wales, Australia, is able to permanently reflect changing conditions in space and time. The fact that the block approach is based on real-time data avoids the permanent application of water quality simulations. Dependencies between blocks are integrated by the definition of blocks. It is not necessary to hold portfolios.

One important advantage of the block approach is the precise integration of the changing relationship between emissions and immission due to the changes in water flow and salinity conditions. The relative definition of permit values enables a flexible adaptation. Only the block approach is capable of integrating pollutants which have very different impacts on the immission level, and also depending on water quantity aspects.

The block approach of the Hunter River Salinity Trading Scheme is the only approach in which the application of trading ratios can be eluded. Sources discharge directly into the block as it passes through; the impact on the water quality is thus the same for all sources even if they trade with each other. At the same time, the definition of the blocks, in the form of volumes of water provides the relationship between discharges of different locations by means of time and space components. Ecological effectiveness is guaranteed; transaction costs are avoided.

At the same time, the block approach is the only model that avoids, from the very beginning on, asymmetric trading positions of sources. Sources hold permits for discharges into a specific volume of waters, i.e. into blocks. Regardless of their location, sources are able to sell or to buy permits for any block. The maximum trading potential is realised as sources are not restricted in their choice of trading partners, as it is the case for an asymmetric position of sources in (almost) all other models (Chapter 4). There is no asymmetry in the trading position and competition is not biased. These properties are vitally important. They could lead to the exclusive proposition to make use of the block approach. Depending on the river conditions, measurement requirements may be lower. If, for example, the relationship between emissions and immission does not change often, the frequency of real-time data acquisition could be lower. At the same time, the value of permits would not change constantly; adaptation costs for sources would be lower. This means that the complexity of the block approach depends on the river and its pollutants.

The use of a block approach is thus not only suitable in the event of a rapidly changing relationship between emissions and immission. Also in the case of a constant relationship, we would suggest applying a block approach. Transaction costs are lower as the complexity decreases. At the same time, the advantages of the block approach can be realised. Should ever become necessary, it is still possible to make a spontaneous adaptation to changing conditions. Ecological effectiveness and economic efficiency are guaranteed.

Depending on the pollutant traded under the block approach, the dynamic incentive would vary. In the case of salinity, the trading system would set an incentive for choosing the optimal storage capacity. This allows discharges to be distributed over time with respect to the different water flow conditions. If the block approach applies to nutrients, it would set incentives for production process innovations in order to reduce nutrient discharges (Chapter 5).

The block approach, as presented in its original version, is only limited with respect to the integration of water quantity aspects: the block approach only provides exogenous variation in water quantities. This is reasonable if sources' activities, such as water diversion, have no significant impact on water levels. For cases with high importance of water diversion, water quantities could be endogenously included in the system. As under the Integrated Water Quantity/Quality System, sources would then have an incentive to integrate their diversion decisions. Lower water use helps to ensure a certain water level and thus to hold the discharge opportunities. In simple terms, sources can thus decide whether they want to reduce pollution or water diversion. In special cases an extension of the block approach might be reasonable.[384]

[384] Diversion activities need to be high enough to significantly influence the water level and thus the water quality. See next paragraph.

6.1.5 Integrated Approaches

In specific cases, endogenous changes in water levels may be relevant (Section 2.4). Water diversion at a single source may affect the discharge opportunities in a negative or positive way through significant alteration of the water level.[385] If only exogenous variation is regarded in the trading system, free rider behaviour might develop. A source could use more water in its own interest while the required reduction in discharges caused by that activity is burdened to all sources. If endogenous changes in water levels, by means of water diversion are integrated in the trading system, incentives to use water efficiently are also created on the water (quantity) market at the individual source level.

The application of the polluter-pays-principle would imply charging those sources that are actually responsible for the pollution; the Integrated Water Quantity/Quality System with the endogenous integration of water changes would then be the adequate approach. Sources are individually burdened with changing discharge constraints, if they use water for abstraction and thus lower the water quantity and quality *ceteris paribus* (Paragraph 4.3.2.2). According to this system, the diversion behaviour of individual sources significantly influences the water level and thus the quality constraints. This connection should be checked for each individual case. If the impact of endogenous modifications of the diversion behaviour is negligible with respect to the total water level and to changes in the constraints, one could justify that only the exogenous variation is considered. This would lower the complexity of the trading system.

As we stated in Paragraph 4.3.2, Weber (2001) does not specify all of the design elements of the Integrated Water System in a final way. In general, it follows a Montgomery-style, i.e. a receptor point approach. Previous analyses underline the fact that the receptor point approaches have two main disadvantages: one is that sources must manage entire portfolios of permits for any of the receptor point they affect; the other disadvantage is that cost-effectiveness can be biased if caps are not all binding. One could thus imagine selecting another basis for the Integrated Water Quantity/Quality System.

A zonal approach, as characterised in this study, cannot be used to supplement the Weber-model: the Integrated Water Quantity/Quality System explicitly requires the integration of changing impact conditions caused by endogenous water changes. The zonal approach cannot reflect the resulting constantly changing relationships between discharges and their impacts; the zonal approach is not able to regularly redefine the pollution constraints.

It is thus advisable to combine the Integrated Water Quantity/Quality System with the block approach, if the endogenous variation in the water level is significant. In a simple version, endogenous alteration of water levels could be regarded as if they were exogenous. All of the concerned sources would thus proportionally bear the effects of changes in the diversion behaviour of an individual source, i.e. of changing water levels; the value of permits is proportionally adapted to changing caps. But: the idea behind the Integrated Water Quantity/Quality System is, in particular, to create individual incentives to ensure an adequate water level and a specific water quality by means of the connection of water use and water pollution.[386]

[385] The Integrated Water Quantity/Quality System assumes that diversion activities of single sources significantly influence the water flow within the river.
[386] The Integrated Water Quantity/Quality System requires the complete integration of an integrated water trading system additionally to the water quality trading system. This aspect must also be integrated, independently of the model selected.

This differentiated integration of the Integrated Water Quantity/Quality System into a block approach would extremely complicate the mechanism behind the trading system. Real-time data would not only be needed in order to define the 'window of opportunity' and thus the cap, these data should also identify the proportion of endogenous changes in water level as caused by individual sources and then adjust permit values individually. Under an Integrated Water Quantity/Quality System-style block approach, individual sources must be directly assigned responsibility by means of changing discharge constraints if they modify their diversion activities.

If stochastic (weather) and sources' impact on water levels are required to be separated, the model also needs to distinguish exogenous from endogenous changes. While endogenous changes would only influence the permit values of the individual source, namely the acting one, exogenous variation in water quantity levels would have an impact on the permit values of all permits held by the sources. Even if this can be realised by the application of complex monitoring programmes, the increase in transaction costs should be considered and must be compared with the additional advantage in the form of a higher differentiated ecological dimension.[387]

Finally, the Integrated Water Quantity/Quality System can only be proposed for cases in which the endogenous alteration in the water level is relevant. Additional transaction costs would be justified by the higher degree of accuracy of the system. However, if the exogenous integration of changes in water levels is deemed sufficient, the additional transaction costs caused by the endogenous integration would no longer be justified. No additional effectiveness or efficiency would be achieved. Furthermore, it is assumed that a block approach might be overloaded by the separated integration of endogenous *and* exogenous changes in water levels. The final design of an integrated water quantity/quality approach should thus be selected very carefully.

6.1.6 Conclusion

Figure 6-1 shows in a stylised form, how the river conditions and ecological settings would influence the choice of the specific water quality trading approach. If the relationship between discharges and the resulting water quality is more constant in space and over time and if water quantity aspects are not relevant, i.e. permanent adjustments of the emission cap are not necessary, then a zonal approach would be more advisable. The zonal approach would also be favourable if spatially differentiated immission caps are required by the regulating authority. A zonal approach, such as the Trading Ratio System, is able to reflect these needs in an economically efficient way. However, a crucial condition for the implementation of a zonal approach is that the asymmetric position of sources does not bias competition and/or cost-effectiveness. In this case, the Trading Ratio System is effective and efficient and does not cause any hot spots. It would not be reasonable to introduce a permit trading system with greater complexity if the current conditions do not require that. Under the conditions given, the trading system would be effective and efficient; and too high transaction costs could be avoided.

In other cases, the relationship between emissions and immission may change frequently; water quantity levels change permanently (and exogenously), thus influencing the relationship between emissions and immission, for example. A zonal approach cannot adequately reflect these conditions. In this case, a block approach which is economically efficient, would be more appropriate. The block approach explicitly considers the changing impacts of discharges on the water quality in space and over

[387] The block approach, in its original design, does not integrate a complete water quantity trading system as it is the case under the Integrated Water Quantity/Quality System.

Decision Parameters	Approach
- Relationship between emissions and immission is constant. - Water quantity aspects are not relevant. - Spatial differentiation is particularly important. - Asymmetric trading positions can be ignored.	Zonal approach
- Relationship between emissions and immission changes constantly. - Exogenous or endogenous water quantity aspects are relevant. - Differentiation (space/time) is possibly required. - Symmetric trading positions are important.	Block approach
- Relevance of endogenous changes in water quantity levels.	Block-based Integrated Water Quantity/Quality System (IQQS)

Figure 6-1: Design Proposals

time. The block approach could even, if necessary, integrate further differentiation elements to a certain degree (differentiated immission cap in space and over time). Another advantage of the block approach lies in the avoidance of asymmetric trading positions of sources. All trading potentials can be exhausted; competition is not distorted. If the asymmetric trading position cannot be ignored under the zonal approach, we suggest that a block approach is implemented.

If endogenous variation in water quantity levels is significant and must be applied instead of exogenous changes, for ecological or political reasons, it is advisable to adopt a block-based Integrated Water Quantity/Quality System. Although the monitoring would be more complex, the block approach has the capacity to include endogenously changing water levels and to reflect their impact on individual discharge constraints. If specific conditions ask for a separate integration of exogenous and endogenous changes of water levels, the complexity of the adapted block approach may no longer be manageable. The question then arises whether another water quality trading approach would be capable of integrating this complexity. In order to identify the relevance of endogenous and exogenous changes in water levels, the precise evaluation of the specific river conditions is necessary.

6.2 Country-specific Considerations

6.2.1 General Aspects

So far we have suggested to establish one or another trading system depending on the specific river and political conditions. It is reasonable to also characterise more general aspects which influence implementation potentials in specific countries. Five aspects, in particular, will determine whether the introduction of water quality trading systems would be reasonable in a specific country:

- the institutional capacity,
- the density of water-specific regulations,
- the current water quality,
- the structure of pollution, including sources of pollution, etc.,
- the political integration of riparian countries.

A well developed *institutional capacity* is a critical condition for the successful implementation of a water quality trading system. Functioning administrative processes are the prerequisite for the integration of this instrument. The environmental administration should be able to fulfil its tasks, such as monitoring, information transfer and enforcement. It is self-evident, that the legal framework must also be appropriate for the integration of water quality trading systems. Existing environmental instruments must be analysed with respect to the compatibility with a water quality trading system (Paragraph 3.3.5).

Proceeding from the assumption that most industrialised countries have already been pursuing a rigorous environmental policy for several decades, one can take for granted that institutional capacity is already in place. Of course, the institutional structure may need to be supplemented or amended in some cases; the general basis, however, exists. Member States of the European Union, for example, have been obliged to implement stringent environmental policy systems for decades. As a result, the institutional framework is already well developed in the advanced countries. The situation differs for the acceding countries from Central and Eastern Europe: these Member States have still to invest a lot in the installation of a proper environmental institutional framework, since the emphasis placed on ecological objectives has been rather low in the past.

Other countries, outside the European Union, have only just started implementing an adequate environmental policy. In emerging markets, in particular, the discussion of a deliberate environmental policy is increasing in importance.[388] But as environmental policy has been disregarded for a long time, specific regulations and thus the specific institutional capacity to well-manage ecological tasks such as the preservation of a specific water quality is still insufficient to deal effectively with increasing pollution problems within these countries.

At the same time, the previous *regulation density* in a country influences the potential for the integration of a water quality trading system. It is questionable whether the introduction of a completely new instrument would be suitable if regulation in a given country is already very dense. Germany, for

[388] For emerging countries, the relationship between environmental policy effects and the economic growth has been discussed in the literature, see Gradus and Smulders (1993), Grossmann and Krueger (1995) or Smulders (1995).

example, already displays high regulation of water pollution[389] and the regulated sources have adapted to this. It would not necessarily be efficient to replace the existing regulations.

With respect to the improved *quality of waters*, the question arises, for these countries in particular, whether it would be efficient to close the remaining, usually small gap in the water quality level by implementing a new instrument. Adaptation costs for sources and administration would be high; the additional benefit in terms of ecological effectiveness would be rather small.

On the other hand, countries that have not focused on water policy in the past and that need to establish a new instrument for water pollution control, may consider setting up water quality trading systems. Sources and the environmental authority need, in any case, to adapt to a new regulation system, and transaction costs for the installation of a new instrument will be incurred anyway. Thus, the leeway to introduce a new instrument exists. The implementation of an efficient and effective permit trading system would thus not necessarily cause higher transaction costs, e.g. information or monitoring costs, than the introduction of any other instrument.

Structural conditions may also be different between countries. The following example can be used to illustrate this: in European countries, the proportion of pollution caused by agriculture is still of great importance; their discharges influence water quality significantly (EEA, 2003). In emerging countries, such as China, rapid industrial growth, i.e. mainly point sources, causes a large proportion of river pollution.[390] Paragraph 2.2.3 clarified that nonpoint sources can only be integrated into a 'pure' permit trading system when subject to severe restrictions.

Furthermore, it is important to examine the type and number of pollutants concerned. Not all types of pollutants are adequate for the integration in a water quality trading system (Section 2.3). Moreover, discharges into a river may consist of a conglomeration of different types of pollutants which must be regulated. The integration of divers pollutants into a single water quality trading scheme is only possible, if a standardised trading unit (equivalent) can be defined. Otherwise, one could think of implementing several water quality trading systems; one for each pollutant. The feasibility and reasonability of the introduction of parallel trading systems must be examined carefully for any specific case.

As to riparian countries, the cooperation principle requires them to work together by means of a basin-wide coordinated water policy. In the case of transboundary rivers, the aspects mentioned above may differ significantly between riparian countries. Transboundary rivers require close cooperation between riparian countries. Cooperation is more or less challenging according to the countries concerned: Durth (1996a) states that cooperation between states that are already integrated in another form, such as Member States of the European Union, would be easier to be accomplished than cooperation between states with different previous regulations (*political integration*). A higher homogeneity in political and institutional structures facilitates cooperation processes.[391]

To sum up, one can conclude that the implementation of water quality trading systems would be more suitable in countries where previous regulation is low and new instruments need to be implemented.

[389] Kraemer *et al.* (1997), Keudel and Oelmann (2005, pp. 219-220), Oelmann (2005), and the literature mentioned there.
[390] See de Graaf (2004, p. 19) for a Chinese example.
[391] For an in depth discussion, see Bressers and Kuks (2004). Durth (1996b) illustrates the successful cooperation within the Rhine River Basin between Switzerland, Austria, Germany, France and the Netherlands. Barrett (1992) analyses international cooperation under a theory of games perspective.

Water pollution in these countries is still assumed to be relevant. At the same time, the institutional framework must be sufficiently stable in order to carry out the administration of the new instrument.

In the literature, two particular regions are discussed as potential areas of application for water quality trading systems. One region is the eastern part of the European Union. Most of the eastern EU countries are riparian states of the Danube River. Environmental policy mainly started developing within the last decade. As Member States of the European Union, most of the riparian countries must adapt to the European environmental water policy. Instruments to improve water quality in rivers must be defined. This could be a prerequisite for the introduction of a water quality trading system.

The other region under consideration for the integration of water quality trading systems is China, standing for the group of emerging economies. The economy of China is developing rapidly. At the same time, the water pollution is increasing.[392] China has only just started to define a stringent environmental policy. Therefore, the possibility exists to introduce a market-based instrument practically from the very beginning.

Both examples are outlined in the following paragraph. The final decision regarding the trading system must be based on the specific conditions in the region; intensive monitoring must be used in order to identify these conditions in advance. Political opinions will also influence the selection of the final design. It is not task of this study to analyse the conditions in the country in great detail. This section instead tries to give an initial idea about where it may be reasonable to implement water quality trading systems. The following paragraphs only discuss the most relevant aspects, i.e. the five factors mentioned above, in order to display their potential for final implementation of a water quality trading system.

6.2.2 Eastern European Countries: The Case of the Danube River

Statutory Framework

Currently the European Union does not have a water quality trading system for rivers.[393] The most important river which passes through the European Union, the Danube River, included. The basin of the Danube River covers 19 countries, approximately two thirds of these are Member States of the European Union. Nutrient pollution is particularly high for some parts of the river (UNDP/GEF, 2005a). The Danube River is the main contributor of nutrients to the Black Sea (UNDP/GEF, 2005b, p. 8).

Numerous supranational directives regulate the environmental water policy in the European Union.[394] Since 2000, the Water Framework Directive (WFD) (European Union, 2000) sets the general

[392] De Graaf (2004), Wang and Wheeler (2000).

[393] As in the United States, the EU environmental policy of the 1970s/80s was predominated by approaches based on the Best Available Technology (BAT). While the United States started to introduce more pro-market instruments for the protection of the environment relatively early, it is only recently that the European Union has also started to discuss and introduce marketable instruments like the permit trading.

[394] Carius *et al.* (2000). Within the European Union, the national as well as the European legal framework defines the conditions for the introduction of new instruments (see Michaelis, 1996, pp. 142-144 or Hansjürgens, 1992, p. 224); for Germany, see Betz (2003, pp. 301-305) or Huckestein (1996, p. 149). For the case of the Danube River, additionally, the Danube River Protection Convention is of importance (ICPDR, 1994).

framework for the European Union water policy.[395] The Water Framework Directive stipulates that any river management addresses entire river basins (Article 3 and 11 WFD) which is the only way to guarantee the ecological effectiveness of implemented measures (Paragraph 3.2.2).[396] In the event of transnational river basins the Water Framework Directive requires an international coordination of the river management (cooperation principle) (Article 3, 4 WFD).[397]

The main objective of the Water Framework Directive is to guarantee a 'good water status'[398] of all European waters. The quality goal is aimed at both surface water as well as groundwater.[399] The Directive explicitly requires the combined regulation of point and nonpoint sources (Article 10 WFD).

Moreover, the European Water Framework Directive (WFD) is the only directive that explicitly recommends a combined approach in order to adequately reflect the relation between emissions and immission.[400] The Directive stipulates the achievement of water quality levels by setting water quality standards and effluent limits.[401] The Water Framework Directive thus requires, on the one hand, the determination of a specific ambient pollution level and on the other hand, the Directive recognises that immission standards need to be transferred into emission caps to make them operable in the real world. Therefore, the Directive additionally requires the definition of effluent limits (Article 2.4 WFD).

All types of waters are integrated in the Water Framework Directive regulations. As a result, not only instream flow needs constraints of the river, but also endpoint constraints of the receiving body of water that must be achieved. This equals a combined instream flow need/endpoint constraint.

Furthermore, the Directive includes water quantity aspects: Article 8 WFD, for example, explicitly requires monitoring the water quantity, the water level and the flow rate, if relevant, as well as the monitoring of the water quality.[402]

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395 For current information about the Water Framework Directive, see `http://ec.europa.eu/environment/water/water-framework/index_en.html` (January 2007); for questions about the implementation of the Water Framework Directive, see Interwies and Kraemer (2001). For a general presentation of European environmental policy and its development, see Aubine and Varone (2004).

396 For an illustration of the European river basins, see Figure A-7 (Appendix).

397 The Water Framework Directive came into force on 22. December 2000. On a broader basis, also the UNECE Water Convention (UNECE, 1992) and the Bucharest Convention (Black Sea Commission, 1992) are relevant. Another important document is the Directive 96/61/EC (European Union, 1996) concerning integrated pollution prevention and control. This Directive requires the integrated management of different environmental concerns/areas, like water, air and soil pollution. However, these international legal documents are very broad in their definition. The European Water Framework Directive integrates the main principles of these Conventions.

398 According to Article 2, 18 WFD, the good water status for surface water is achieved wh0en both its ecological and its chemical status are at least 'good'. For the definitions of these terms, see Article 2, 21 and 24 WFD as well as Annex V WFD. The Water Framework Directive additionally incorporates hydromorphological requirements (Article 4,3 WFD).

399 Article 17 WFD requires a complementary Groundwater Directive; the proposal of this Directive already exists (Commission of the European Communities, 2003). In this proposal the European Commission requires measures to protect and to monitor the quality of groundwater and to guarantee good chemical water status.

400 Further on, the Water Framework Directive explicitly requires a combined approach for point and nonpoint sources (Article 10 WFD). When implementing a permit trading system, however, one needs to decide on the degree to which the nonpoint sources can/should be involved from the technical and the economic point of view (indirectly or not at all). It is, however, indispensable – not only for reasons of the Water Framework Directive requirements – to set strict individual incentives for the nonpoint sources to abate.

401 Emission limits are defined on the basis of Best Available Technology (BAT) and Best Environmental Practices (BEP). The Best Available Technology or Best Environmental Practice must be introduced in any case, even if the water quality is already achieved without the application of these technological requirements (precautionary principle) (Hecht and Werbeck, 2006, Chapter 7). From an economic point of view, this requirement is too strict and causes inefficiencies. Additional abatement costs are generated although the ecological objective has already been achieved.

402 Furthermore, the Water Framework Directive proposes monitoring frequencies (Annex V WFD).

The Water Framework Directive also incorporates economic aspects: Annex III WFD requires cost-effectiveness and dynamic efficiency of instruments.[403] In addition, the application of the polluter-pays-principle is made obligatory. At the same time, Article 4, a WFD stipulates the non-degradation for all bodies of water.[404] While this appears to be intuitive from an ecological point of view, Paragraph 3.3.2 demonstrated that the application of the non-degradation principle hinders cost-effectiveness.

The introduction of a water quality trading system is not explicitly required by the Water Framework Directive in order to achieve water quality standards.[405] In principle, one can conclude that the Directive sets the correct framework for the introduction of a permit trading system by determining crucial elements as the combined approach.[406] At the same time, the Water Framework Directive sets the necessary leeway for Member States in the event that individual and project orientated decisions are necessary. The sole problem for the implementation of a water quality trading system under the Water Framework Directive would be the predetermined integration of the non-degradation principle. The analysis in Paragraph 3.3.2 showed that the application of the non-degradation principle would prevent a permit trading system from coming to the least-cost solution. It is questionable whether this element of the Water Framework Directive could be changed from a political point of view. We expect that even in presence of this principle cost savings will be realised, but not all potentials could be exhausted.

For the specific case of the Danube River the *Danube River Protection Convention* (*DRPC*, ICPDR, 1994) is also of importance. The main focus of this Convention lies in the cooperation between riparian states (cooperation principle). At the same time, the Danube River Protection Convention emphasises two economic principles: the polluter-pays principle, on the one hand, and the precautionary principle, on the other (Article 2 DRPC). Article 5 and 8 of the Danube River Protection Convention explicitly postulate that the taken measures must be efficient.

The Danube River[407]

Riparian States of the Danube River are heterogeneous. While the German and the Austrian sections of the river have implemented a stable institutional capacity for stringent water policy during the last few decades, the eastern European countries, that have only recently acceded to the European Union, are in the process of implementing an adequate institutional framework in order to be able to comply with the

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[403] When talking about environmental 'instruments' and the Water Framework Directive, a precise understanding of the terms and definitions in the Water Framework Directive is indispensable. The term 'instrument' is not explicitly defined in the Water Framework Directive; the term used in the Water Framework Directive is 'measure'. However, according to Article 11, the Water Framework Directive differentiates between 'basic' and 'supplementary' measures. The term 'instrument' as it is used in this paper is a subgroup of the supplementary measures as defined in Annex VI, Part B WFD. In other words, in this paper the differentiation between instruments and measures is – according to the definitions in the Water Framework Directive – the following: While the term 'measure' describes a more concrete technical (structural or architectural) action, 'instruments' are more of an administrative or economic nature. While measures become effective in the short term and directly address their aim, the effect of instruments evolves over the long term. Instruments set the correct incentives to realise certain measures. Measures are relevant on a local level, for example, at a plant or installation. Instruments are on a broader level; correspondingly, they have to be regulated at a higher administrative level. This differentiation follows Interwies *et al.* (2004, p. 4).

[404] Requirements for groundwater are even stricter, see Article a, b) of the Water Framework Directive.

[405] Keudel (2005) analyses in more detail to what extent the Water Framework Directive has already made decisions on the concrete design of water pollution control in the European Union and in which areas the Member States can individually influence the design.

[406] As a result, transaction costs which arise when implementing such a system will occur anyway when implementing the Water Framework Directive, independently of the environmental instrument chosen; these transaction costs would not arise in addition when implementing a permit trading system.

[407] For an illustration of the Danube River Basin, see Figure A-8 (Appendix). For more information on the Danube River Basin, see website of the *International Commission for the Protection of the Danube River (ICPDR)* (`www.icpdr.org`, October 2006).

EU requirements. The institutional system is thus assumed to achieve, in the mid-term, the capacity that is necessary in order to administrate a water quality trading system.

Regulation history varies also a lot when passing from the western to the eastern part of the Danube River. Germany and Austria have developed a high regulation density (Schönbäck *et al.*, 2003). As a result, the water quality of the Danube River is rather high in these countries. The eastern European countries have not followed a strict river management. Environmental concerns have been neglected for a long time. Thus, water pollution remains a problem in the eastern part. Agriculture plays an important role; waste water treatment plants must be installed; the potential for reduction is high (Section 2.2). The European environmental and water policy requires consistent measures to be taken. This could be the opportunity to decide in favour of a water quality trading system; any instrument to be introduced would cause additional transaction costs for sources as well as for the environmental authority. Furthermore, the implementation of the Water Framework Directive requirements would already form a stable prerequisite, e.g. in terms of monitoring; the transaction costs of the final introduction of a permit trading system would thus be partly absorbed.

The Danube River basin is an extreme case with respect to the cooperation conditions. The river flows through a large number of countries with very different historical and political backgrounds. Not all of these countries are Member States of the European Union. Even within the group of EU countries, differences between countries might complicate effective and efficient cooperation. Nevertheless, the identical political basis within the European Union may facilitate, sooner or later, international cooperation.[408]

The *Danube Regional Project*, conducted by the United Nations Development Programme and the Global Environmental Facility, analyses in more detail, whether and in which form permit trading system could be an adequate instrument to achieve the water quality aimed at in the Danube River.[409] In general, this study supports the introduction of such a trading system. At the same time, it is aware of the complexity, created by the heterogeneity of the countries with respect to the (environmental) policy, institutional and even economic perspective. Actually, particularly the eastern European countries are concerned with the implementation of the Water Framework Directive. At the moment, resources that are necessary in order to discuss the introduction of a permit trading are assumed to be missing. It may thus be too early for implementation of such a system. On the other hand, the Water Framework Directive already determines some technical instruments and mechanisms, particularly monitoring. Even though this level of monitoring might not be appropriate for any of the water quality trading approaches presented above, the Directive builds an important basis and would facilitate the introduction of a water quality trading system.

One can conclude that permit trading systems in the European Union are far from implementation. Even the discussion regarding the potential introduction of such systems is at the very early stages. In the case of the Danube River further examination is necessary. It should be underlined that the institutional basis that is currently established under the Water Framework Directive is an adequate one for the introduction of a permit trading system in a not too distant future. This the more so as the Directive already sets the framework for the final definition of water quality standards with respect to different sectors of the river.

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[408] The International Commission for the Protection of the Danube River (ICPDR) developed in order to manage the Danube River Basin as a whole. Also non EU members are integrated in this commission.

[409] For more information, see `http://www.icpdr.org/undp-drp/` (January 2007).

Propositions

According to the results of this study, generally a zonal approach or a block approach could be regarded as recommendable. In the case of high significance of endogenous variation in water flow, an integrated water quantity/quality system might be sensible. If the relationships between emissions and immission do not often change, a zonal approach might be chosen. For the Danube River, in particular, one could consider setting zones with respect to the national borders. In this case, different immission caps, if in accordance with the Water Framework Directive, might be set for different zones and thus for different countries (spatial differentiation of the immission cap). This might facilitate international cooperation as the individual leeway remains. However, the application of a zonal approach would only be reasonable, i.e. effective and efficient in the result, if the river conditions in the Danube permit the integration of a (n almost) constant relationship between discharges and resulting water quality. In addition, the asymmetric trading position of sources under the zonal approach must be analysed in more detail. It depends very much on the number and type of the participating sources, and their locations, whether the starting position for trading will lead to competitive bias or not (Section 6.1). If the zonal approach is identified as not being suitable for the Danube River, a block approach would be a worthwhile alternative.

A block approach would be able to reflect the permanently changing relationship between emissions and immission and avoids the problem of asymmetric trading positions. Sources are able to trade with each other independently of their location. Monitoring processes are more intensive under this approach. At the same time, the block approach guarantees ecological effectiveness and economic efficiency. It goes without saying that such achievements would be quite welcome.

6.2.3 Emerging Markets: The Case of the Yangtze River in China

Emerging markets may be a potential area for the implementation of water quality trading systems. Current water pollution control policy is rather weak. In countries such as China, the high growth rates of the economy have been accompanied by high growth rates of waste water discharges and water abstraction. Pollution in the majority of rivers is very high which has a negative effect on human health but also on economic activities. In the following section, policy options for the example of China are analysed more accurately.

Statutory Framework

The Chinese environmental policy seeks to adapt to the recent situation by means of a strict water policy which regulates discharges from point sources in particular (Pu, 2003, p. 2). The general framework for the Chinese water policy is set by the *Water Law (WL,* NPC, 2002*)* and the *Law on Prevention and Control of Water Pollution.* Article 12 WL requires unified management for entire river basins; at the same time, administrative regions need to be recognised. In order to enforce the Water Law, water standards have been defined at the national, provincial and local level. Water standards are defined by the *Environmental Quality Standards for Surface Water* (ambient-based) and the *Integrated Wastewater Discharge Standard* (discharge standard, emission-based) (NPC, 1996; Pu, 2003, p. 3).

China currently follows a command-and-control approach in form of a hybrid charge-subsidy system. Two components are relevant: on the one hand, a ‘discharge fee’ is imposed for every unit of waste water; on the other hand, a ‘pollution levy fee’ must be paid on waste water in excess of the official effluent concentration standards. The revenue from this two-component system is used to subsidise pollution abatement measures. This means that, in principle, market-based policy instruments come into play which turns China into an interesting example among emerging countries (Sterner, 2003, p. 321). However, Wang and Wheeler (2000) states that even after the payment of the pollution levy fee discharges by the sources concerned might exceed the standard. This, of course, hampers ecological effectiveness.

Yangtze River

Currently, no water quality trading systems exist in China. However, a few authors analyse the potential of water quality trading systems for specific rivers in China. Pu (2003) looks at the case of the Yangtze which is the longest river in China. Due to strong economic growth, both water consumption and waste water discharge have increased, the level of water pollution is high.[410] The overall situation in the main stream is acceptable. Due to the swift flow the assimilative capacity of the water is higher. However, the quality varies significantly between urban and non-urban sectors of the river. At the same time, the water quality of tributaries is generally lower.

In general, the institutional capacity in China is assumed to be adequate for the integration of a water quality trading system (Tao *et al.*, 2003, p. 13; Pu, 2003). However, some aspects need to be improved. De Graaf (2004), for example, finds inefficiencies in the collaboration between the different administrative levels.[411] Monitoring processes exist, even though they would need to be ameliorated with respect to a permit trading system. Actually, the enforcement capacity appears rather low. Not all sources are covered by the water policy instruments and not all potential charges are collected.

The existing density of regulations is assumed to be low in China. Environmental policy has gained in importance with the rapidly growing economy in China which has been accompanied by high rates of pollution. There is, thus, room for the implementation of new instruments.

Due to the strength of economic development in China, point sources, in particular, e.g. industrial facilities, cause river pollution (over 60 percent of the total discharge received) (Pu, 2003, p. 1). Pollution responsibility of nonpoint sources is lower, but not negligible. In addition, there is the potential to convert nonpoint source pollution into point source pollution by establishing area-wide waste water treatment plants.[412]

Cooperation challenges would be lower as the majority of Chinese rivers are not transboundary. Cooperation between different administrative levels is necessary; however, no international cooperation is required.

Studies show that the control strategies have produced a positive outcome. At the same time, they identify room for improvement. As a result, the ecological effectiveness of the current policy is lower than it could be (Pu, 2003, p. 8).[413] It is advisable to improve effectiveness by means of the introduction of market based instruments such as a permit trading.[414] Pu (2003) proposes the implementation of a pilot water quality trading project as the general conditions are assumed to be appropriate. This requires the establishment of comparable water quality models and an improved monitoring system in order to

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[410] Tao *et al.* (2003) analyse the case of the Nanpan River in China. Conditions for this river are similar to those in the Yangtze and are thus not discussed here in detail. Results of the study are, however incorporated.
[411] See also Dasgupta *et al.* (1996), or Wang and Wheeler (2000).
[412] As long as municipal waste water is not treated in waste water treatment plants, it may instead be part of the group of nonpoint sources.
[413] Tao *et al.* (2003) find similar results for the Nanpan River.
[414] De Graaf (2004), Wang and Wheeler (2000) or Dasgupta *et al.* (1996).

determine adequate and specific caps for all parts of the river. Marginal abatement costs curves are assumed to vary considerably between sources; as a result, high cost saving potentials are expected. At the same time, enforcement mechanisms and the collection of charges need to be more stringent.[415]

Propositions

Again, an intensive observation of the river and its sources and pollutants is indispensable in order to define the best water quality trading approach for the specific river. It is advisable to chose a zonal approach or a block approach, possibly combined with the integration of endogenous water quantity aspects. For rivers in China, the monsoon may also be relevant as these rainfalls change the water flow significantly during specific periods of time. It is probable that these changes would influence the water quality conditions.[416] If the water flow is only influenced once a year and if this variation is rather predictable it may be integrated in a zonal approach. Hung and Shaw (2005) also propose to readjust the zonal approach once a year in order to reflect important changes in conditions. Again, the relevance of asymmetric trading positions must be dealt with in an appropriate manner.

If river conditions require constant adjustments of the emission load cap and/or if symmetric trading positions are indispensable, a block approach would be the better solution even though an improvement in the monitoring processes would be a necessary prerequisite. Depending on the relevance of endogenous and/or exogenous changes in water levels, the integration of water quantity aspects through a combined water quantity/quality approach must be analysed.

6.2.4 Conclusion

The analysis in this section showed that only some countries are suitable for the introduction of a water quality trading system. Aspects of institutional capacity, regulation density, current water quality, structural conditions, as well as political integration will influence the suitability. Countries of eastern Europe and the emerging markets, in particular, have been identified as potential areas of application. Most of these countries are in the process of introducing proper institutional frameworks. The existing level of regulation density and the quality of the water are low in the majority of cases. The introduction of a water quality trading system may thus be the most effective and efficient answer to environmental needs.

[415] For the Nanpan River, Tao *et al.* (2003) deliver first results on cost saving potentials which are remarkable. It seems that regulatory authorities and sources are not opposed to a permit trading system.

[416] The other extreme would be that rivers run dry once a year. De Graaf (2004, p. 13) mentions this for the Yellow River in China. In the periods of extreme drought, the suitability of water quality trading systems is restricted for the most part.

7 Conclusions

This is a new attempt to develop standardised and comparable (river-specific) criteria and to identify the design possibilities for water quality trading systems in order to evaluate the existing theoretical and practical approaches for rivers and to be able to formulate well-founded propositions for the introduction of new trading systems.

An analysis of the specific structures and characteristics of rivers, sources of pollution and pollutants identified the general requirements made of a water quality trading system (Chapter 2). The results of Chapter 2 lead to the conclusion that common criteria of ecological effectiveness and economic efficiency cannot be used for the evaluations in their original definition. They instead must be fine-tuned specifically for rivers (Chapter 3). The ecological objective, in particular, is formulated much more precisely in this study than has been done elsewhere in order to reflect river-specific conditions. Several ecological dimensions have been defined that are incorporated in the following analysis of specific approaches. Economic criteria must also be adapted. Transaction costs, in particular, are identified with respect to the specific design elements of different water quality trading systems. As a result, Chapter 3, developed a catalogue of standardised and comparable (ecological and economic) criteria for the comprehensive evaluation of water quality trading systems for rivers.

Chapters 4 and 5 discussed the existing theoretical and practical approaches to water quality trading systems. The application of the previously developed criteria allows us to identify and evaluate different design elements. As a first step, the analysis of theoretical models showed how water quality trading systems can be adapted to the river-specific conditions (Chapter 4). However, it becomes obvious, that some of the models presented loose some of their economic efficiency. Only a few approaches are capable of achieving different ecological dimensions in an economically efficient way. This chapter clarified that any water quality model must reflect the river-specific conditions. This is absolutely essential. The approaches that already exist for air pollution control can be used as foundation to build river-specific permit trading schemes on. But design elements of approaches developed for the medium air cannot be adapted one-to-one to water pollution control.

As a second step, the standardised examination of practical approaches, implemented in the United States and Australia, identified additional design elements (Chapter 5). It becomes obvious that the implemented trading systems do not always revert to theoretical examinations. Some practical approaches ignore river-specific conditions and are neither ecologically effective nor economically efficient. But this does not hold true for all of the implemented water quality trading systems. The analysis of one practical approach in Australia, in particular, detected additional design elements and mechanisms which enable a water quality trading system to manage rivers in an ecologically effective and economically efficient way. The design enables fine-tuning of the cap setting and the trading mechanism which permits high differentiation potentials on the ecological side. At the same time, these revealed design elements create incentives for sources which bring the system to an efficient solution.

Chapter 6 is directly based on the results of the preceding Chapters 4 and 5. The criteria-based analysis of theoretical and practical approaches enabled the identification of the suitability of different design elements of water quality trading systems. It enables us to find out, in the case of successful trading systems, those potential design options that may sensibly be incorporated in existing or new water quality trading schemes. In the case of less effective and/or less efficient water quality trading schemes, a re-examination which integrates the developed criteria, would help to identify specific problems and to solve them. The analysis may detect missing or ill-designed elements that hinder the system from being effective and efficient. Adjustments can be suggested in order to ameliorate the effectiveness and efficiency of the system.

In the event of a completely new introduction of a water quality trading system, we are now able to disclose the best, i.e. effective and efficient, design for the given river conditions and ecological requirements. By linking the results of the criteria-based analysis of theoretical and practical approaches, Section 6.1 traces design-specific proposals. On the one hand, this section identifies in which cases the one or the other approach will be appropriate. On the other hand, new combinations of designs are developed for specific cases.

The identification of the adequate design of a potential water quality trading system is of great importance and forms the main prerequisite for a successful implementation. At the same time, it is essential to analyse the potential area of implementation accurately and to do this, before establishing a permit trading system. Section 6.2 provides country-specific proposals. This section identifies major characteristics that will hinder or enhance the introduction of water quality trading systems in different countries. Firstly, the institutional capacity of a country must be sufficient in order to administrate and manage a water quality trading scheme. Secondly, the density of existing river-specific regulations must be taken into account. The introduction of a water quality trading system may not be reasonable for countries that already pursue a stringent water policy that has led to high water quality levels. The introduction of a completely new instrument would, in this case, cause high additional transaction costs; and at the same time, the additional environmental effect would be negligible. It is advisable to instead consider the introduction of water quality trading systems for countries that have a stable institutional framework, but that have not had an intensive water policy in the past. Previous regulation density in these countries is assumed to be low; this is often accompanied by low water quality levels. Thirdly, the structure of pollution may influence the choice of design: in some countries, nonpoint sources are mainly responsible for the river pollution; in other countries, point sources cause high proportions of pollution. While a water quality trading system is, in general, suitable for the regulation of point sources, the integration of nonpoint sources into this regulation is very difficult. In this case, other instruments will be more appropriate. Fourthly, the previous integration of riparian states, in the case of transboundary rivers, hinders or enhances the effective and efficient introduction of a water quality trading system. The examples using eastern European countries and China have been chosen in order to give an initial impression of what such an analysis could look like. Further research will hopefully generate further insights in this area.

APPENDIX

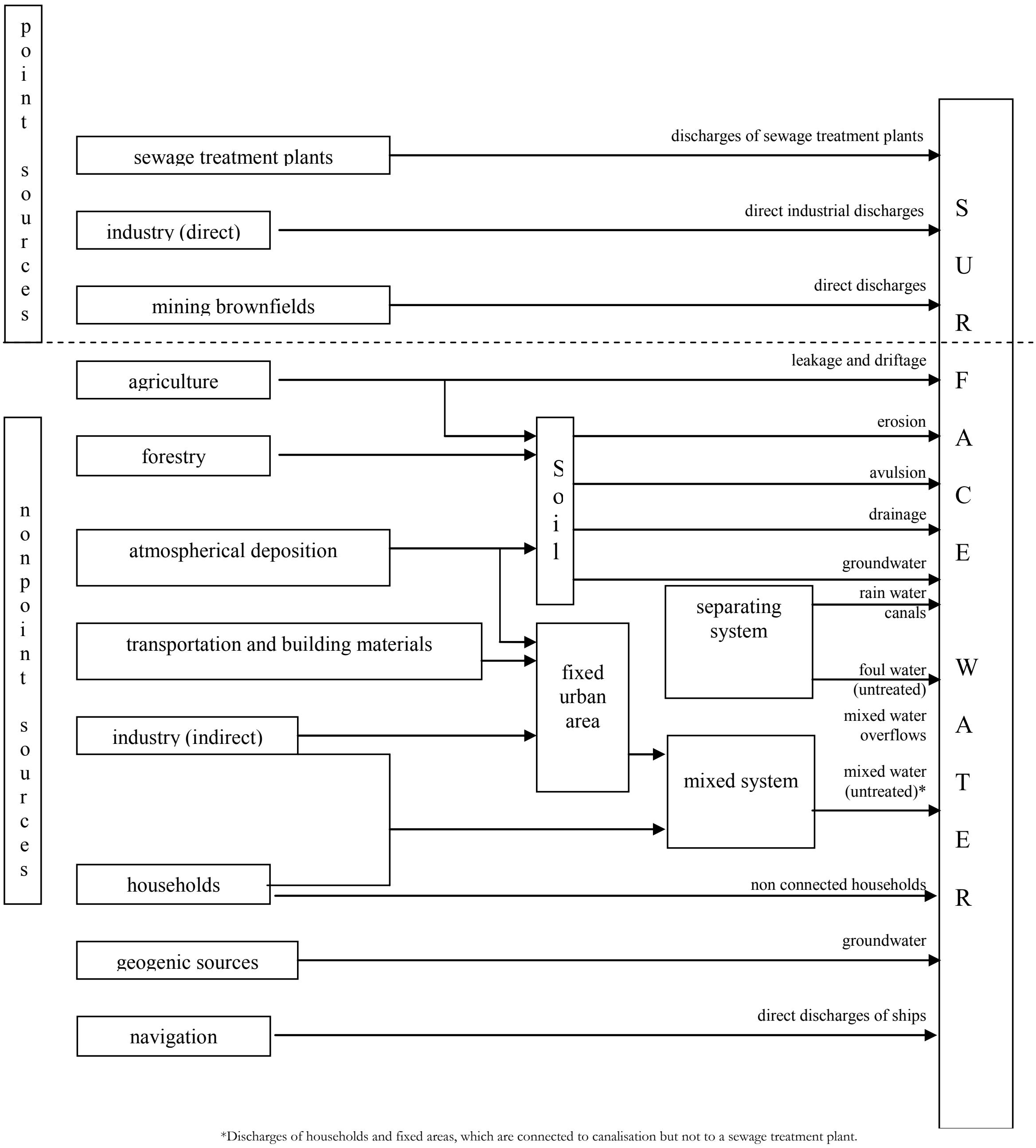

Figure A-1: Point and Nonpoint Sources
Rumm *et al.* (2006, p. 139, own translation)

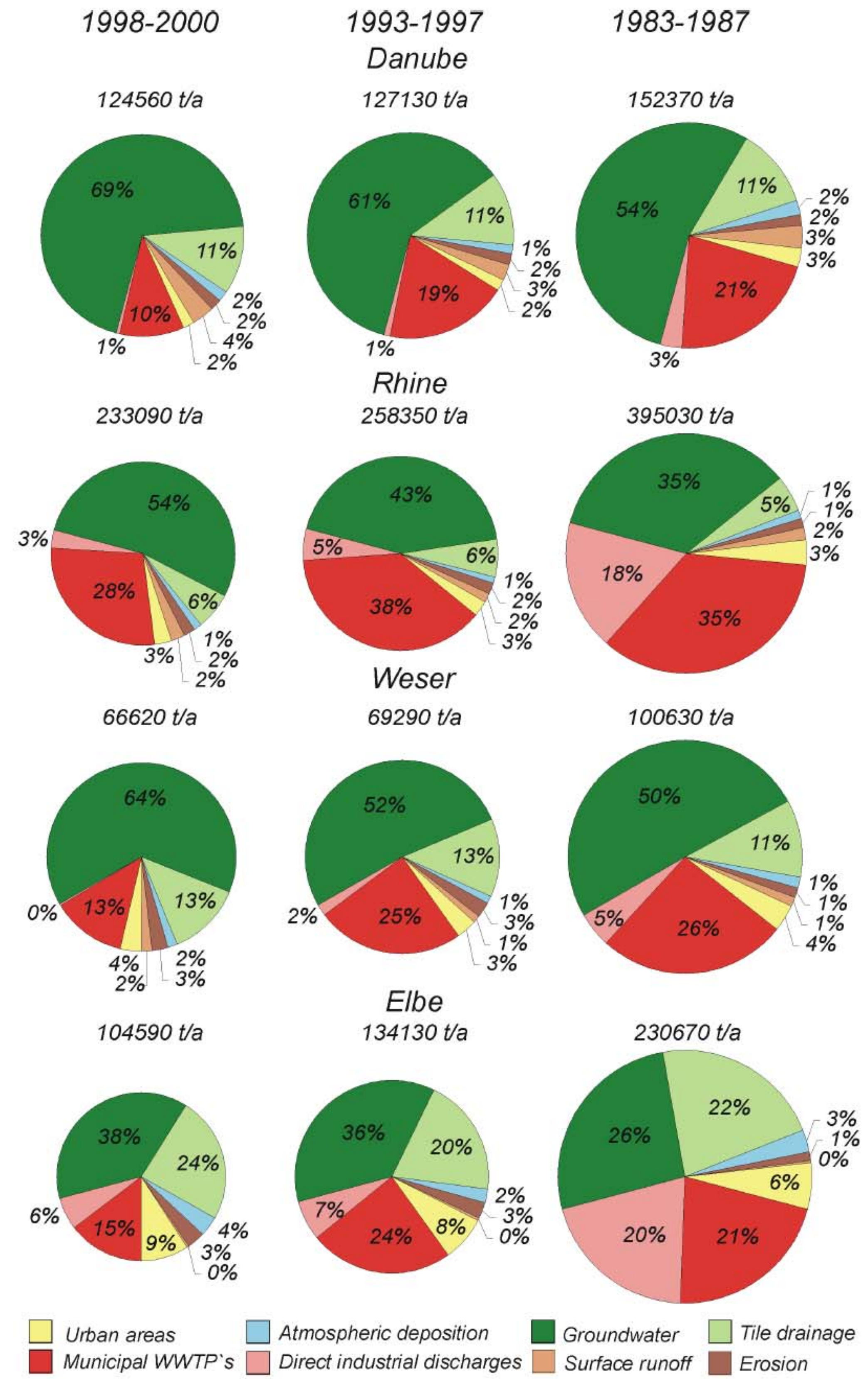

Figure A-2: Nitrogen emissions via the various pathways into German rivers (1983-1987, 1993-1997, 1998-2000)
Behrendt *et al.* (2003b, p. xvii)

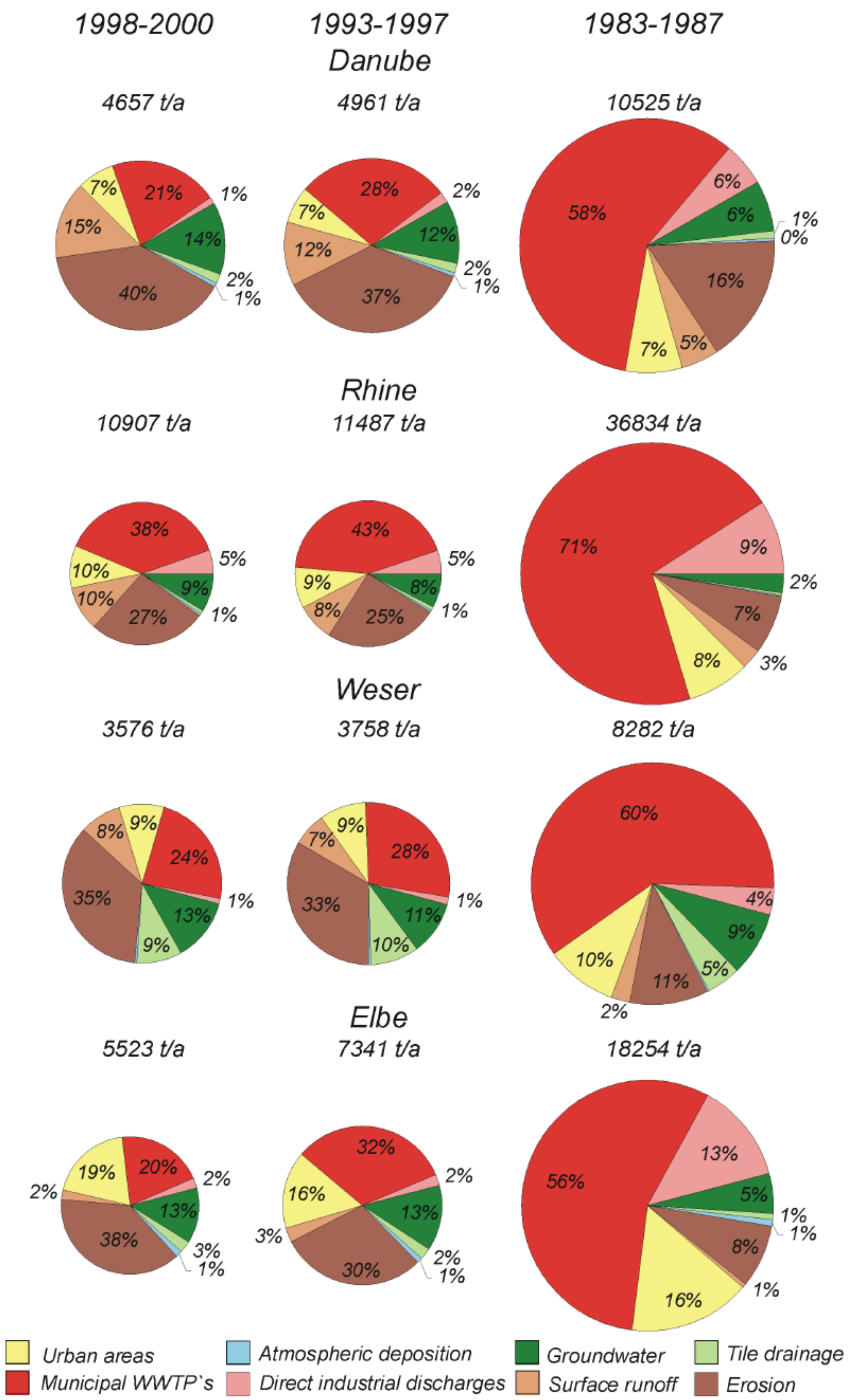

Figure A-3: Phosphorus emissions via the various pathways into German rivers (1983-1987, 1993-1997, 1998-2000)
Behrendt *et al.* (2003b, p. xix)

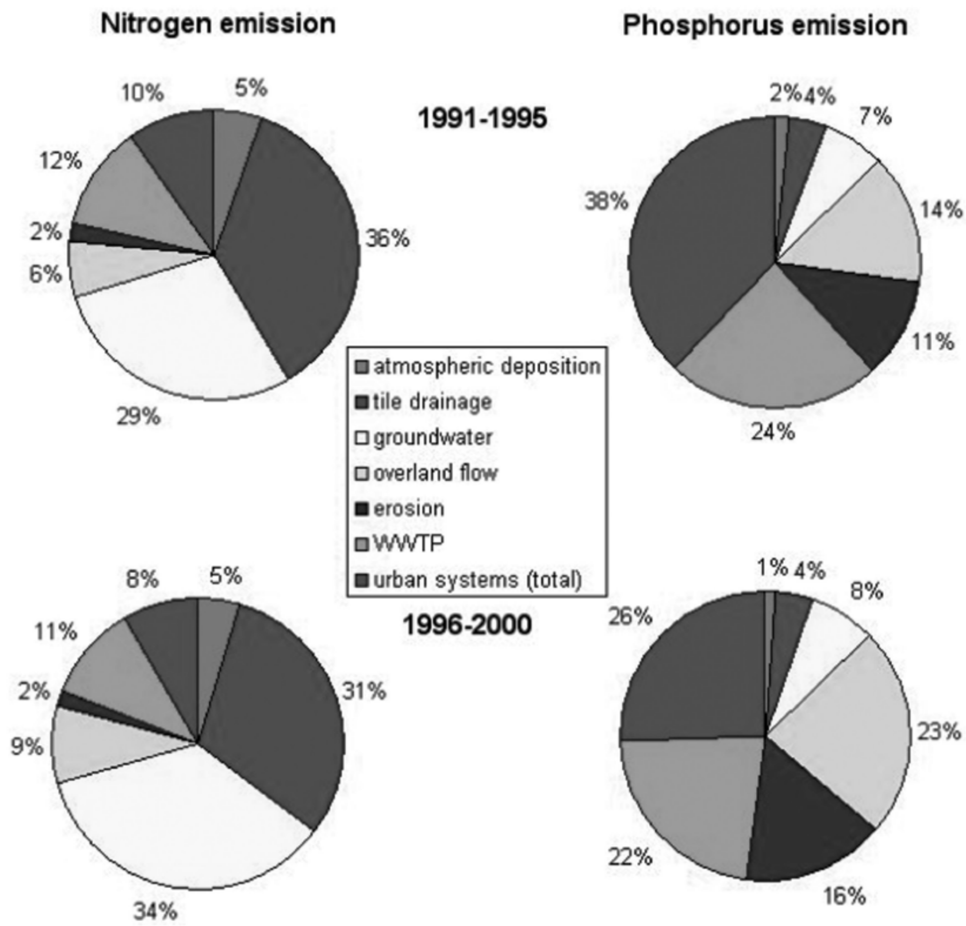

Figure A-4: Nutrient emissions via the various pathways into the Vistula River, Poland (1991-1995, 1996-2000)
Kowalski and Buszewski (2006, p. 619)

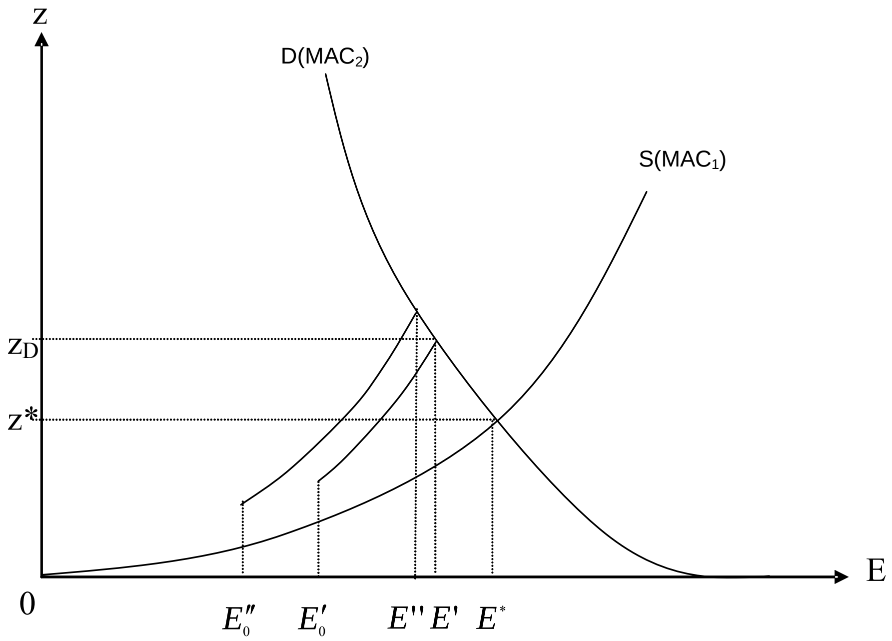

Figure A-5: Increasing Transaction Costs
According to Stavins (1995, p. 142)

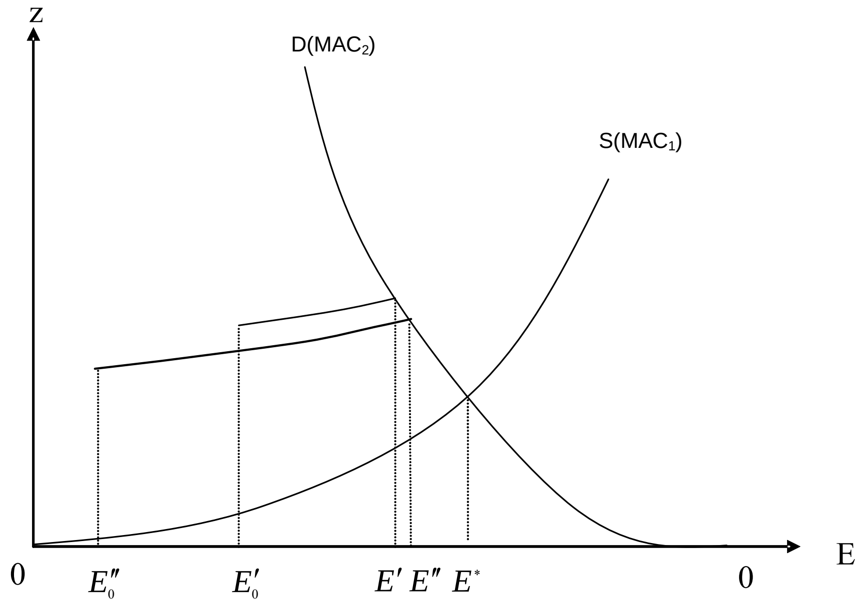

Figure A-6: Decreasing Transaction Costs
According to Stavins (1995, p. 143)

State	Name	Pollutant	Potential Types of Trading (Point or Nonpoint Sources)	Trades
		Trading Initiatives		
CA	Grassland Area Farmers	Selenium	NPS-NPS	39
CA	San Francisco Bay	Mercury	Not determined	
CO	Bear Creek	Phosphorus	PS-PS	1
CO	Boulder Creek	Nitrogen	PS-NPS	
CO	Chatfield Reservoir	Phosphorus	PS-PS and PS-NPS	1
CO	Cherry Creek	Phosphorus	PS-NPS	
CO	Clear Creek	Heavy metals (e.g. Arsenic, Copper)	PS-NPS	1
CO	Lake Dillon	Phosphorus	PS-NPS and NPS-NPS	(3)
CO	Lower Colorado River	Selenium, possibly habitat	PS-PS, PS-NPS, and NPS-NPS	
CT	Long Island Sound	Nitrogen	PS/PS	(63)
FL	Tampa Bay	Nitrogen	No trading actually occurs	
ID	Lower Boise River	Phosphorus	PS-NPS	
IL	Illinois Pretreatment Trading Program	Multiple (indirect discharges)	PS-PS	
IL	Piasa Creek Watershed Project	Sediment	PS-NPS	
MA	Acton WWTP	Phosphorus	PS-NPS	
MA	Charles River	Water flow	PS-NPS	
MA	Edgarton WWTP	Nitrogen	PS-NPS	
MA	Falmouth WWTP	Nitrogen	PS-NPS	
MA	Massachusetts Estuaries Project	Nitrogen	PS-NPS	
MA	Specialty Minerals, Inc.	Temperature	PS-NPS	
MA	Wayland Business Center	Phosphorus	PS-NPS	
MI	Kalamazoo River	Phosphorus	PS-NPS	
MN	Minnesota River	Phosphorus	PS-PS	
MN	Rahr Malting Co.	Phosphorus, nitrogen, 5-day carbonaceous biochemical oxygen demand (CBOD-5), and sediment	PS-NPS	2
MN	Southern Minnesota Beet Sugar Cooperative	Phosphorus	PS-NPS	
NV	Truckee River	Nitrogen, Phosphorus, or Total Dissolved Solids (TDS)	PS-PS and PS-NPS	(33)
NJ	Passaic Valley Sewerage Commission Pretreatment Trading	Heavy metals (Cadmium, Copper, Lead, Mercury, Nickel, and Zinc)	PS-PS	2
NY	New York City Watershed	Phosphorus	PS-PS and PS-NPS	
NC	Neuse River Basin	Nitrogen	PS-NPS	1
NC	Tar-Pamlico Basin	Nitrogen and Phosphorus	PS-NPS	
OH	Clermont County	Nitrogen, Phosphorus, or Total Dissolved Solids (TDS)	PS-NPS	
OH	Great Miami River Watershed Trading Pilot Program	Nitrogen and Phosphorus	PS-NPS	
PA	Conestoga River	Nitrogen and Phosphorus	PS-NPS	
PA	Pennsylvania Water-based Trading Simulations	Simulations include: CBOD, phosphorus, nitrogen, suspended solids, ammonia, acid and metals	PS-PS, PS-NPS, and NPS-NPS	
VA	Blue Plains	Nitrogen	PS-PS	
VA	Henry County	Total Dissolved Solids (TDS)	PS-PS	
WI	Fox-Wolf Basin	Phosphorus	PS-PS and PS-NPS	
WI	Red Cedar River	Phosphorus	PS-NPS	
WI	Rock River	Phosphorus	PS-PS and PS-NPS	
Regional	Chesapeake Bay	Nitrogen and Phosphorus	PS-PS and PS-NPS	
		STATE POLICIES AND PROGRAMS		
MD	Maryland Nutrient Trading Policy	Nitrogen and Phosphorus	PS-PS, PS-NPS, and NPS-NPS	
MI	Michigan Water-Quality Trading Rules	Nitrogen, Phosphorus, potentially sediments	PS-PS, PS-NPS, and NPS-NPS	
PA	Pennsylvania Multi-media Trading Registry	Multiple (potentially nutrients, habitat, carbon, etc.)	PS-PS, PS-NPS, and NPS-NPS	
VA	Virginia Nutrient Trading Program	Nitrogen and Phosphorus	PS-PS and PS-NPS	
WV	West Virginia Trading Framework	Multiple (could potentially include nutrient, metals, or cross-pollutant trading for dissolved oxygen)	PS-NPS and NPS-NPS	
WI	Wisconsin Nutrient Trading Rules	Phosphorus	PS-PS, PS-NPS, and NPS-NPS	

Table A-1: Water Quality Trading Systems, US
According to Breetz *et al.* (2004, pp. 8-9) (numbers in brackets: Morgan and Wolverton (2005, p. 22))

Annual Nutrient Loads and Caps, Tar-Pamlico Basin Association
Phase I

Combined N+P	1991[1]	1992[1]	1993[1]	1994[1]
Loading Cap[a] (kg/yr)	525,000	500,000	475,000	425,000
Actual Load (kg/yr)	461,394	436,128	417,217	371,200
% of Cap	88	87	88	87
Average Daily Flow (MGD)	24,88	26,86	28,46	26,65

Loads were estimated by NC Division of Water Quality as the sum of calendar-year monthly load values for each facility, which are based on minimum biweekly nutrient concentrations and daily mass flows.

[a] Cap values and changes result from the following:

1. Phase I – Original 12-member Association.
2. Phase II through 2000 – 14-member Association.
3. Robersonville added in 2001, making a 15–member Association.
4. Scotland Neck added in 2002, making a 16-member Association.

Table A-2: TPNTP, Annual Nutrient Load and Caps, Phase I (NC, EMC, 2005, p. 23)

Annual Nutrient Loads and Caps, Tar-Pamlico Basin Association
Phase II

Separate N, P	1995[2]	1996[2]	1997[2]	1998[2]	1999[2]	2000[2]	2001[4]	2002[4]	2003[4]	2004[4]
Loading Cap[a] (kg/yr)	N 405,256 P 69,744	N 405,256 P 69,744	N 405,256 P 69,744	N 405,256 P 69,744	N 405,256 P 69,744	N 405,256 P 69,744	N 421,972 P 73,060	N 426,782 P 73,694	N426,782 P 73,694	N 426,782 P 73,694
Actual Load (kg/yr)	N 372,582 P 37,360	N 354,219 P 43,266	N 320,670 P 36,532	N 344,781 P 36,804	N 309,476 P 32,052	N 297,988 P 30,277	N 279,958 P 32,730	N 279,330 P 34,076	N 309,724 P 30,856	N 256,791* P 33,566*
% of Cap	N 92 P 54	N87 P 62	N 79 P 52	N 85 P 53	N 76 P 46	N 74 P 43	N 66 P 45	N 65 P 46	N 72 P 42	N 60* P 45*
Average Daily Flow (MGD)	31.03	33.57	29.84	33.31	33.39	32.74	30.21	30.54	36.54	29.56

Loads were estimated by NC Division of Water Quality as the sum of calendar-year monthly load values for each facility, which are based on minimum biweekly nutrient concentrations and daily mass flows.

[a] Cap values and changes result from the following:

1. Phase I – Original 12-member Association.
2. Phase II through 2000 – 14-member Association.
3. Robersonville added in 2001, making a 15–member Association.
4. Scotland Neck added in 2002, making a 16-member Association.

* These values provided by Association, not yet reviewed by DWQ.

Table A-3: TPNTP, Annual Nutrient Load and Caps, Phase II (NC EMC, 2005, p. 23)

	Ambient Permit System (APS)	Trading Ratio System (TRS)	Integrated Water Quantity/ Quality System (IQQS)	Hunter River Salinity Trading Scheme (HRSTS)
		General Design Elements		
Cap type	- Ambient-based	- Ambient-based	- Ambient-based	- Ambient-based
Trading ratio	- Exogenous	- Exogenous	- Not specified	- Not necessary
		Ecological Dimensions		
Geographical scope	- Entireriver basin	- Entire river basin	- Entire river basin	- Parts of the river basin
Orientation	- Endpoint/instream	- Endpoint/instream	- Instream and endpoint	- Endpoint and instream
Differentiation of the immission cap in				
Space **Time**	- Possible - Possible	- Predestined - Restricted	- Possible - Restricted	- Possible - Possible
Adjustment of the emission cap in				
Space **Time**	- Possible - Possible	- Predestined - Restricted	- Possible - Possible	- Possible - Possible
Water quantity aspects	- If at all: implicitly in the dispersion coefficient	- Only for the initial allocation, trading ratio exogenous	- Integrated, endogenous	- Integrated, exogenous

Table A-4: Ecological Dimensions (APS, TRS, IQQS, HRSTS)

	Ambient Permit System (APS)	Trading Ratio System (TRS)	Integrated Water Quantity/ Quality System (IQQS)	Hunter River Salinity Trading Scheme (HRSTS)
Economic Criteria				
Cost-effectiveness	- Depending on initial allocation - No free rider	- Yes - No free rider	- Depending on initial allocation (see Ambient Permit System) - No free rider	- Trading activities - Equalisation marginal abatement costs (blocks), cost minimisation assumed - No free rider
Dynamic efficiency	- Yes, but possibly biased	- Yes	- Yes, if caps are binding	- Incentive to innovate assumed
Transaction costs	- see Table A-6 and Table A-7			

Table A-5: Economic Criteria (APS, TRS, IQQS, HRSTS)

	Ambient Permit System (APS)	Trading Ratio System (TRS)	Integrated Water Quantity/ Quality System (IQQS)	Hunter River Salinity Trading Scheme (HRSTS)
Transaction Costs (TAC), Environmental Authority				
Determination of receptor points, zones or blocks				
- Higher TAC		X		X
- Lower TAC	X		(X)	
Running WQM before any *potential* change in discharges				
- Yes, higher TAC				
- No, lower TAC	X	X	X	X
Real-time data				
- Yes, higher TAC				X
- No, lower TAC	X	X	(X)	
Determination of the initial allocation of permits				
- Including WQM, higher TAC		X		
- Not including WQM, lower TAC	X		(X)	X
Definition of the dispersion coefficient (Water Quality Model)				
- Yes, higher TAC	X	X	(X)	
- No, lower TAC				X
Definition of the trading ratio				
- Endogenous (higher TAC)			(X)	
- Exogenous (lower TAC)	X	X		
- No trading ratio, no TAC				X
Differentiation of the immission cap in space				
- Higher TAC				
- Lower TAC	X	X	(X)	X
Differentiation of the immission cap over time				
- Higher TAC	X	X	(X)	
- Lower TAC				X
Determination of the emission cap				
- Environmental authority, higher TAC		X		X
- Sources, no TAC environmental authority	X		(X)	
Information requirements				
- Higher			X	X
- Lower	X	X		

Table A-6: Transaction Costs, Environmental Authority (APS, TRS, IQQS, HRSTS)

	Ambient Permit System (APS)	Trading Ratio System (TRS)	Integrated Water Quantity/ Quality System (IQQS)	Hunter River Salinity Trading Scheme (HRSTS)
Transaction Costs (TAC), Sources				
Determination of the emission cap				
- Yes, higher TAC	**X**		**(X)**	
- No, lower TAC		**X**		**X**
Portfolio of permits				
- Yes, higher TAC	**X**		**(X)**	
- No, lower TAC		**X**		**X**
Constant adjustment of the permit value				
- Yes, higher TAC	Depending on the final design		**(X)**	**X**
- No, lower TAC		**X**		
Trading ratio				
- Endogenous, higher TAC			**X**	
- Exogenous, lower TAC	**X**	**X**		
- No trading ratio, no TAC				**X**
Buyer-liability				
- Yes, higher TAC	-	-	-	
- No, lower TAC	-	-	-	**X**

Table A-7: Transaction Costs, Sources (APS, TRS, IQQS, HRSTS)

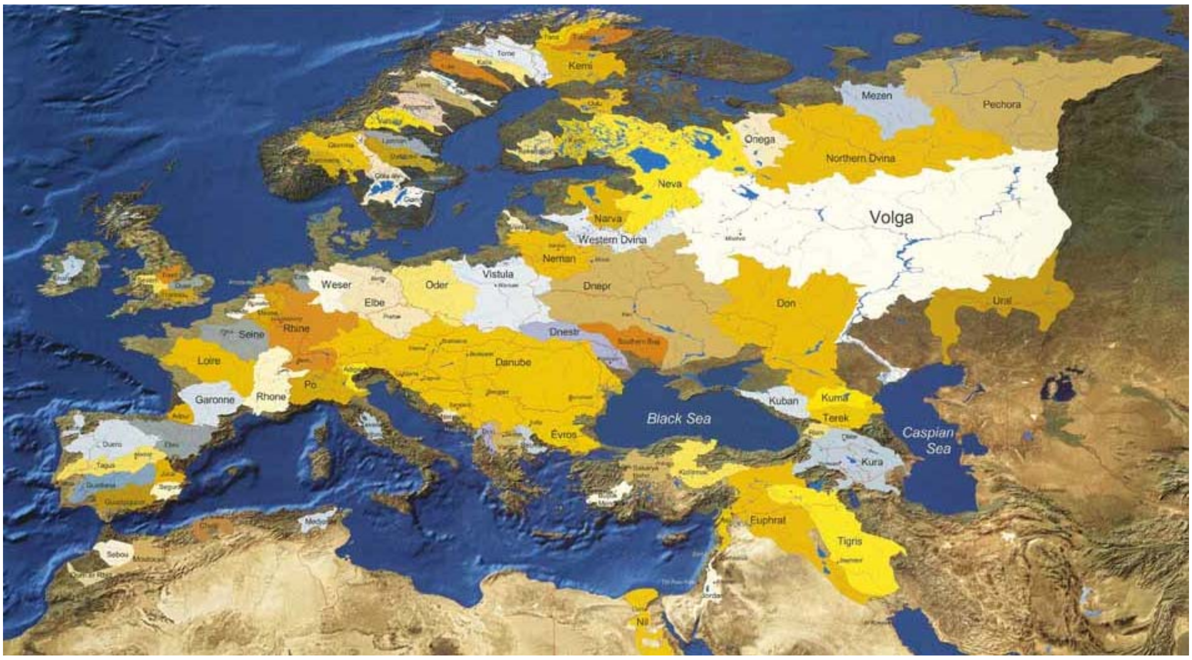

Figure A-7: European River Basins
European Rivers Network (`http://www.rivernet.org/`, December 2006)

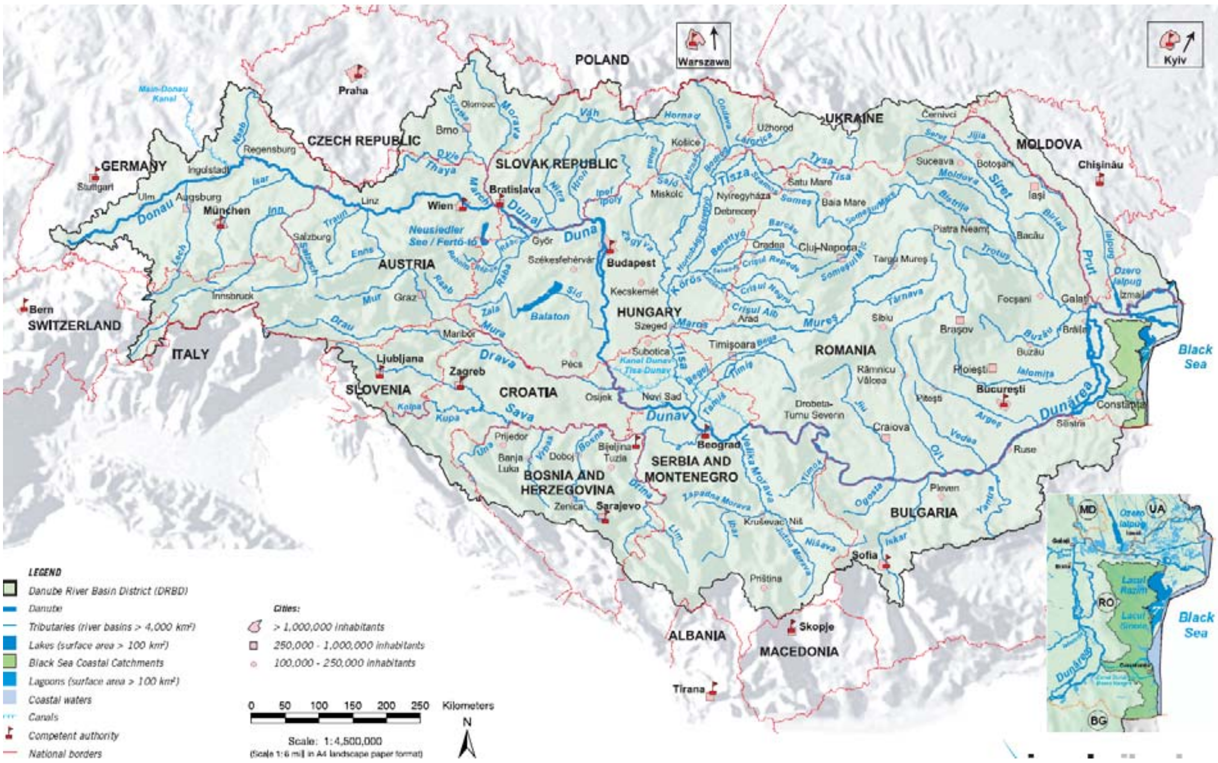

Figure A-8: Danube River Basin
ICPDR (`http://www.icpdr.org/icpdr-pages/river_basin.htm`, December 2006)

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