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Water Quality Trading:
Theoretical and Practical Approaches

by

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Abstract

Permit trading as an instrument to control air pollution has already been implemented in several countries, so in Europe since 2005. Could this instrument, however, also be adequately used for water pollution control of river basins in form of a water quality trading? Specific characteristics of rivers, pollutants and pollution sources strongly influence the design of such an instrument. This paper reviews theoretical and practical approaches on water quality trading. It is surprising that these approaches have never been linked by the literature. To fill this gap, this paper gives a first idea, how different water quality trading approaches (in theory and practice) can be made comparable.

Keywords: water quality trading, water pollution control, river basin management

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1 Introduction[1]

Permit trading as an instrument to control air pollution has already been implemented in several countries, so in Europe since 2005. Could this instrument, however, also be adequately used for water pollution control of river basins in form of a water quality trading?[2]

This paper starts with a short presentation of the relevant characteristics of rivers, sources and pollutants (chapter 2), because these characteristics crucially influence the design of a permit trading system. The paper then reviews theoretical and practical approaches to water quality trading (chapter 3 and 4). The theoretical approaches are mainly based on models for air pollution control. The analysis of implemented permit trading systems (practical approaches) focuses on the often cited „Tar Pamlico Nutrient Trading Program, North Carolina, U.S." and on the innovative „Hunter River Salinity Trading Scheme, New South Wales, Australia". For a better understanding, a short overview of the institutional framework in these countries will be given.

It is surprising that theoretical and practical approaches on water quality trading have never been linked by the literature. To fill this gap, this paper gives a first idea how these approaches can be made comparable. This will be the basis for the further research work, by which the author will develop a guideline that, depending on the defined quality objectives and the characteristics of the relevant river basin and its sources and pollutants, will be able to criticise existing theoretical and practical approaches on a consistent and standardised basis. Moreover, this guideline will offer the opportunity to examine whether the introduction of a new water quality trading system for a certain river basin is reasonable. If this holds, this guideline will be able to formulate precise requirements for an adequate design of such a trading system.

[1] This paper has been presented at the *Kieler Nachwuchsworkshop „Umwelt- und Ressourcenökonomik"*, IfW Kiel, 20.-21. February 2006. The author is indebted to Andreas J. Schmidt for very useful comments. Furthermore I would like to thank Nadine Kalwey for her support.

[2] For the general discussion about the advantages of a permit trading over other environmental instruments, also for water pollution control, see Tietenberg (2003, pp. 343).

2 Water Pollution Control: Relevant Factors

2.1 Sources

In managing river basins two potential sources of pollution have to be taken into account: on the one hand, point sources, for example industrial facilities or sewage treatment plants. They discharge their pollutants at a fixed and well identifiable point into the river (Shortle and Horan, 2001, p. 256). The discharges of these point sources can therefore be precisely measured; the assignment of individual accountability for the pollution is possible.

On the other hand, nonpoint sources, for example agriculture, do not discharge the pollutants at a precise point into the water. The emissions are thus not exactly measurable (Shortle and Horan, 2001, p. 256). It is not possible to assign individual accountability for the resulting pollution. That is the reason for not integrating nonpoint sources in the discussion of this paper; the assignment of permits to individual sources is very complex and makes a “pure” water quality trading nearly impossible.[3]

2.2 Pollutants

Depending on their specific characteristics, discharged pollutants influence water quality in a different way, which in turn influences the adequate design for an instrument. Therefore, one has, on the one hand, to distinguish between assimilative and accumulative pollutants (Tietenberg, 1985, p. 15). In the case of assimilative pollutants, the medium has a certain capacity to absorb these pollutants; they do not accumulate.[4] Accumulative substances can not be absorbed and thus do accumulate over time.[5]

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[3] Even if modern technologies can precisely predict the impact of different activities (for example in agriculture) on the ambient pollution level of a river concerned, the precise individual assignment of a defined share of the resulting ambient pollution at a certain point in time is impossible. A permit trading can not be introduced on a clear individual level for nonpoint sources.
But: this does not mean that nonpoint sources should be excluded from any environmental regulation. On the contrary: considering the high share of pollution and the relatively low activities of abatement, strong economic incentives for nonpoint sources to abate are required.

[4] The absorbing capacity for assimilative substances may be different in space and time, i.e. for a different intensity of use, regional or local specifications etc. (Kemper, 1993, p. 71). Exceeding the capacity, the substances take over the characteristics of accumulative substances and should be treated as such (Kemper, 1993, p. 71).

[5] Kemper (1993, p. 77) states that in the case of assimilative substances, which biodegrades, current emissions are responsible for damages, i.e. immission loads, at present. All future damages depend only on future emissions, not on the present ones. Thus, different time periods can be examined independently.

On the other hand, pollutants can be classified by their degree of mixing in the medium: uniformly mixed pollutants spread quickly and uniformly throughout the medium, non-uniformly mixed pollutants do not disperse quickly.[6] While the former are of global relevance, the latter are rather of local relevance.

The following table shows classification examples for different substances. Relevant for this paper are non-uniformly mixed assimilative substances in the water medium.

Substances	Uniformly mixed	Non-uniformly mixed
accumulative	(CO_2, (air))[7]	heavy metals (water)
assimilative	GHG (air)	Sulfur dioxide (air), nutrients, salt (water)

Table 1 – Substances (own compilation based on Tietenberg, 1985, pp. 14).

While the extent and spatial pattern of damage caused by uniformly mixed pollutants (assimilative or accumulative) depends only on the level of emissions, this does not hold for non-uniformly mixed pollutions. The extent and the spatial pattern of damage are affected by the level of emissions, but also by the location and the dispersion characteristics, i.e. the distribution in the medium, of the emissions. This has to be taken into account when designing a water quality trading system.

2.3 The River itself

This paper focuses on the environmental management of rivers.[8] In contrast to lakes, rivers are flowing water bodies. This has to be taken into account when thinking about the introduction of a water quality permit trading. In contrast to CO_2 control, the point in time and the location of discharges are of crucial relevance; the problem of hot spots has to be addressed. In addition, rivers are characterised by different water levels and flow rates over time. This, as well as different temperature levels at different times, results in varying assimilative capacities for pollutants.

[6] For the definition see Tietenberg (1985, pp. 14). Uniformly mixed pollutants can be controlled by an emission-based system; time and location of the emission do not matter. Non-uniformly mixed pollutants call for an ambient-based control system; depending on time and location of the emission, the impact on the ambient pollution level is very different.

[7] CO_2 is originally characterised as an assimilative pollutant. The actual amount of CO_2 in the atmosphere exceeds the assimilative capacity of the air medium and thus becomes accumulative.

[8] Out of the scope of this paper are coastal waters or other inland waters such as lakes; due to their different characteristics, an individual discussion of environmental instruments is required. Also groundwater is not explicitly treated in this paper. But an increasing quality of surface water, here rivers, automatically leads to a higher quality of the groundwater (and to a limited extent of coastal waters as well). Nevertheless, additional, individual instruments need to be introduced for these water bodies.

If ecological effectiveness should be reached in all parts of the river and at every point in time, the cap, i.e. the maximum emission or immission load, should be formulated in detail for each river basin. Any instrument to be introduced to achieve this ecological effectiveness[9] has to be flexible in the sense that it can deal with – eventually changing – restrictions (and thus changing caps) in time, quantity and location of discharge.[10]

3 Theoretical Approaches

In the following, different permit trading models for non-uniformly mixed assimilative pollutants will be presented. In the literature, models for these pollutants do mostly apply to air pollution. They are nevertheless shortly presented, because the following water specific model bases on these air pollution models.

All these models base on the central mechanism of a permit trading: each source has to hold permits for the pollutants it discharges into the environmental medium (air or water). When reducing its discharges, a source can sell the surplus of permits; when increasing its discharges, a source has to purchase permits from other sources to cover these additional emissions. The total amount of emissions allowed is determined by the total number of permits distributed in the system.

Depending on the location of the sources, the discharge of different sources can have different impacts on the immission load, i.e. the environmental quality of the medium at a specific point. If this holds, a one-to-one trade of permits between these sources could affect the ecological effectiveness of the system in a negative way. To avoid this, one could introduce so-called trading ratios for all transactions between sources. A trading ratio determines by how much source 1 has to decrease its discharges if source 2 wants to increase its discharges by one unit, purchasing permits from source 1. A 2:1 trading ratio for example means that source 1 has to abate two units if source 2 increases its emissions by one unit. This would be the case if one unit of source's 1 discharge has a weaker impact on the environmental quality at a specific point than one unit discharged by source 2. The system can thus guarantee for a constant environmental quality level, i.e. immission load.[11]

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[9] For reasons of cost-effectiveness this strong interpretation of ecological effectiveness may not hold for all cases.

[10] Additionally, any instrument to be introduced should be able to deal with the requirements in the river basin concerned. This becomes especially relevant if the river basin comprises many countries of different characteristics as this is the case for the Danube River Basin. For more details see NIRAS (2004).

[11] The application of a trading ratio of course influences the price of the concerned permit.

The models can roughly be subdivided into zonal and non-zonal approaches applying trading ratios set endogenously or exogenously. Zonal approaches divide the environmental medium into zones; quality standards can then be set for each individual zone separately. Other models use receptor points: a mesh of receptor points is installed; the quality standard (immission) has to be achieved at these points.[12]

If a trading ratio applies, it can be fixed ex ante, i.e. exogenously, or it can be derived by the model, i.e. endogenously. Most of the models are immission-based, i.e. the environmental quality level should be influenced by the permit trading. Other, however, are emission-based, i.e. the rate of emissions should be affected by the model; and thus, the immission load is not directly affected by the model. Trading ratios are only relevant in the case of immission-based models. An emission-based model does not account for the resulting immission load; different impacts of emissions on the water quality between trading sources are not assumed to be a problem.

3.1 General Approaches

Montgomery (1972): Ambient Permit System (APS)

Montgomery (1972) was the first to develop a theoretical approach for a permit trading with non-uniformly mixed assimilative pollutants reflecting spatial considerations of damages.[13] In his *Ambient Permit System (APS*) Montgomery mentions a model with n industrial sources of pollution and m receptor points in the river.[14] The ecological objective is immission-oriented.[15] Each source needs to cover its discharges with permits, which can be traded between sources. The total amount of permits is defined by the required water quality standard (immission load).

An environmental authority defines water quality standards (immission loads) of one pollutant concerned at various receptor points, denoted as a vector $Q^* = (q_1^*,...,q_m^*)$,

[12] A broader definition could even include zones as one type of receptor points.

[13] Montgomery (1972) formulated his theoretical approach for air as well as for water. The specific case chosen by Montgomery for analysis is air-related: he 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 the case of a river, where the water transports the pollutants in a constant direction with constant speed.

[14] These sources are fixed in location and owned by independent, profit-maximising firms.

[15] Montgomery also proposes a so called *Emission Permit System* (EPS) with emission-based permits. Because EPS has both theoretical a practical problems, it will not be considered in the analysis that follows (for further discussion see Krupnick et al. (1983) or Hung and Shaw (2005, p. 84)).

where q_j^* ($j = 1,...,m$) is the determined water quality standard (immission load) of the pollutant at receptor point j.[16] Each of the sources ($i = 1,...,n$) emits a single pollutant at the rate e_i, which can be shown by an emission vector $E = (e_1,...,e_n)$. As the emissions of the sources have, e.g. depending on their location, different impacts on the water quality, i.e. the immission load, this emission vector will be mapped into water quality levels (immission loads) by a matrix H, so that $E \cdot H = Q$. The matrix H thus shows how water quality is affected by the emissions. 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 water quality (immission load) at receptor point j.[17] The elements of the matrix h_{ij} are thus called dispersion coefficients.

The Q^*, set by the environmental agency, defines the water quality standard (immission) constraint which should not be exceeded, i.e. $E \cdot H \leq Q^*$. In Montgomery's model, the dispersion coefficients of the matrix H are taken as exogenously defined trading ratios for transactions between sources. The trading ratio reflects that emission discharges of different sources could affect the water quality in a different way. Additionally, a non-degradation principle applies. This disallows any lowering of water quality in any part of the river, even if this quality level is higher than the water quality standard would require.

Montgomery arrives at the conclusion that a competitive equilibrium exists and corresponds to the cost minimum attainment of a set of predetermined water quality standards. Krupnick et al. (1983, p. 240), however, demonstrate that the market equilibrium coincides only with the cost-minimum solution if the initial allocation of

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[16] 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.

[17] Montgomery mentions a model with only one relevant pollutant. If the desired air quality in terms of one pollutant concerned is independent of the desired air quality of any other pollutant, nothing is lost. This problem could be generalised by adding constraints representing emission vectors which 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 (Montgomery, 1972, p. 398). Furthermore, Montgomery assumes 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 (Montgomery, 1972, p. 398).

permits makes the water quality standard, i.e. a pollution constraint, binding at all receptor points. The required water quality standard at any receptor point j is not binding if the initial allocation of permits set by the environmental authority results in a higher water quality, i.e. a lower immission load, than the allowable level set by the water quality standard. Under the non-degradation principle the water quality at the receptor point can not be less than under the initial allocation; the water quality standard will not become binding. The least-cost solution, however, can only be reached if the initial allocation of permits is set such, that the water quality standard is binding at all receptor points; only in this case all trading potentials can be exhausted (for a more detailed discussion see Krupnick et al. (1983)). Hung and Shaw (2005, p. 84) argue that this is usually unattainable, which would mean that the cost-minimum solution can not be achieved. The non-degradation principle is thus quite restrictive in an economic sense and causing inefficiencies.

Furthermore, the Ambient Permit System (APS) causes high transaction costs: when increasing or decreasing discharges, a source should obtain or sell permits for each receptor point affected. Thus, for each receptor point a single market establishes; sources need to hold and to manage a portfolio of permits.

The exogenous, i.e. beforehand, and fixed determination of the trading ratios will cause additional transaction costs.[18] Furthermore, the mesh of receptor points needs to be fine enough to avoid too high local pollution (hot spots); but the number of receptor points also influences the level of transaction costs.

The free riding problem, occurring in other models (see below), seems to be avoided: all discharge permits are allocated to individual sources; a source can increase its discharges only if it purchases an adequate amount of additional permits. No potential for free rider behaviour exists.

Krupnick et al. (1983): Pollution Offset System (POS)

Based on their own criticism against Montgomery's Ambient Permit System (APS), Krupnick et al. (1983) developed a so-called *Pollution Offset System* (POS) for air pollution control.[19] The permits held by the sources confer the right to discharge pollutants; but the permits are not associated with a particular receptor point or

[18] ...which will be lower than in other models where trading ratios have to be determined endogenously.

[19] In this approach, new sources have to purchase permits from other sources to "offset" the effects of these additional emissions in such a way that the pollution constraint is everywhere satisfied. Therefore Krupnick et al. (1983) call their system *Pollution Offset System* (POS).

market as under the Ambient Permit System (APS). Sources can obtain permits from the environmental authority as long as a beforehand environmental model simulation shows that the proposed transaction will not cause a violation of the environmental quality standard at any receptor point. If this transaction will cause a violation of the predefined standard at any receptor point (i.e. the receptor becomes binding), the sources have to trade applying a trading ratio equal to the ratio of the two sources' dispersion coefficients, which indicate how the environmental quality (immission load) at the receptor point is influenced by the discharge of pollutants of the concerned sources (see above, Ambient Permit System). These trading ratios are endogenously given by the simulation model (Hung and Shaw, 2005, p. 85).
To avoid the inefficiencies of Montgomery's model, no non-degradation principle applies under the Pollution Offset System (POS); the model thus allows increasing discharges at any point as long as the simulation model shows that the quality standard will not be exceeded at any receptor point, and this, even if it lowers the beforehand quality at any receptor point. Consequently, the quality standards are not all and not always binding.

Krupnick et al. (1983) demonstrate that this model comes to a least-cost solution independently of the initial allocation of permits.[20] Furthermore, the sources are only concerned with those receptor points whose quality would violate the quality standard due to an increase in emissions. This lowers the transaction costs compared to the Ambient Permit System (APS), under which sources need to hold a portfolio of permits, because each receptor point establishes a single market. On the other hand, a high number of receptor points might be required to avoid hot spots, causing high transaction costs. Furthermore, the beforehand simulation causes high transaction costs; the impact on all other receptor points and thus sources has to be analysed before every transaction. The trading ratio is not known until the sources are not obliged to apply one, which creates uncertainty about the required abatement activity and the price of the permits. Førsund and Nævdal (1998, p. 404) underline that the endogenous definition of the trading ratio is very complex and causes, if ever feasible, too high transaction costs.

[20] McGartland (1988), however, shows that the cost minimum solution under the POS will be achieved only if all potential gains from trade are exhausted (McGartland, 1988, p. 36). Certain conditions, e.g. perceived uncertainty among sources, too high transactions costs, may prevent the system to reach the least-cost solution.

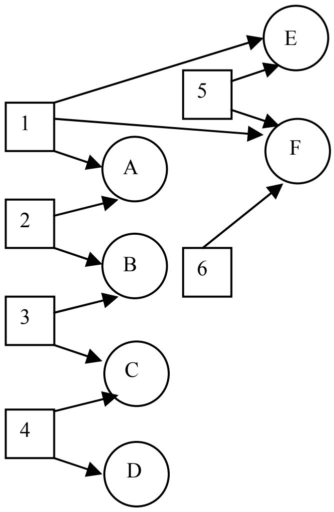

Figure 1 – Dispersion Characteristics, Air (McGartland, 1988, p. 39).

The main problem of the Pollution Offset System (POS) is the occurrence of a free rider effect due to the fact that sources can always obtain additional discharge permits if they do not violate the quality standard. Assume that pollutants discharged by source 1 affect the quality at the receptor point A, E, and F (Figure 1). Suppose that source 1 purchases permits from source 2 (affecting common receptor point A) and source 5 (affecting common receptor points E and F); consequently, source 2 and 5 reduce their discharges. The mentioned trade will improve the environmental quality at receptor point B, due to the abatement of source 2, even though source 1 is not directly concerned. Other sources affecting receptor point B, e.g. source 3, will benefit: they can increase their discharges contributing to the quality at receptor point B at no cost, if the quality standard is not violated; they are free riding[21] (McGartland, 1988, pp. 39; Klaassen, 1996, p. 55).

Førsund and Nævdal (1994): Exchange Rate System (ERS)

A third model, developed by Førsund and Nævdal (1994), has originally been developed for the abatement of SO_2, i.e. air pollution control: the *Exchange Rate System* (ERS), a non-zonal and rather immission-oriented approach. A fixed and exogenously set "exchange rate" that applies at any transaction between sources reflects regional differences in the impacts of discharges and thus corresponds to a trading ratio as defined above. Under the ERS, no testing of violation at the receptor

[21] Krupnick et al. (1983) argue that the efficient solution would be to integrate the concerned sources (affecting B) in the bargaining process. This could, however, become a very complex target if more sources are concerned.

points before transactions is necessary: trades are allowed as long as the exchange rate (trading ratio) applies (Førsund and Nævdal, 1994, p. 309). The exchange rate equals the ratio of the sources' marginal abatement costs in the optimum, i.e. the least-cost solution (Førsund and Nævdal, 1998, p. 309). Førsund and Nævdal assume that the marginal abatement costs differ for the sources depending on their dispersion coefficient related to the binding receptors: depending on the impact that discharges of a certain source have on the environmental quality (immission load), the abatement activity required to achieve a better environmental quality will vary, thus causing different marginal abatement costs.

The exogenous definition of the exchange rate lowers the transaction costs. No simulation has to be done before each single transaction as under the Pollution Offset System (POS). But: the cost-minimum solution and the individual marginal abatement costs should be known in advance to calculate the exchange rates (trading ratios). Førsund and Nævdal (1998, p. 310) remark, that for the case of SO_2 this information is available. This might not be the case for other substances and other environmental media. This might thus, if ever feasible, cause high transaction costs in form of high information costs.[22] Additionally, Førsund and Nævdal note themselves that only for specific initial allocations the exchange rate (trading ratio) can be defined such that it guarantees for an efficient result (Førsund and Nævdal, 1998, p. 409).

Finally, the definition and application of exchange rates (trading ratios) for transactions between sources integrates an immission-based view; it can not guarantee for the avoidance of hot spots; exchange rates reflecting the ratio of the marginal abatement costs can not guarantee for ecological effectiveness, i.e. compliance with the quality goal (immission load). Therefore this approach can not be unambiguously classified as immission-based.

3.2 A Water-specific Approach

Hung and Shaw (2005): Trading Ratio System (TRS)

Hung and Shaw (2005) argue that the main aspects of the theoretical approaches for the medium air hold for water pollution control. The medium water, however, has at least one characteristic that makes the design of the trading scheme vary: while pollutants emitted into the air disperse in many directions, the pollutants discharged

[22] See also Hung and Shaw (2005, p. 98).

into rivers always flow to the lowest level uni-directionally. This property influences the design of a water quality trading system enormously.

The so called *Trading Ratio System* (TRS) of Hung and Shaw (2005) represents a zonal approach: the river basin is divided into n zones ($n \in N^{+}$).[23] Hung and Shaw (2005, p. 86) assume that the dispersion characteristics of effluents and the environmental impact within a zone are very close. For transactions between these zones, trading ratios apply, which are exogenously determined and equal the dispersion coefficient and thus reflect the impact of individual discharges on the water quality level (Hung and Shaw, 2005, p. 89). These trading ratios are promulgated ex ante.

The environmental authority sets a Zonal Total Load Standard E_j for each zone.[24] This determined emission load guarantees for a certain water quality standard (immission load). The environmental authority defines zonal effluent 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 Tradable Discharge Permits ($\overline{T}_j$), which are defined in terms of their original zonal locations. The first zone obtains discharge permits equal to the Zonal Total Load Standard, $\overline{T}_1 = E_1$. The zones following downstream obtain permits depending on the Zonal Total Load Standard E_j and the weighted upstream pollution (emission load) transferred: $\overline{T}_j = E_j - \sum_{k=1}^{j-1} t_{kj}\overline{T}_k$.[25] t_{kj} is the dispersion coefficient, indicating the contribution that one

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[23] For simplification the zones are ordered by location and only one representative discharger is located in each zone.

[24] Hung and Shaw (2005, p. 86, fn 6): "Note that 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."

[25] Assume that $k < j$; i.e. zone k is an upstream zone to zone j. In some cases the downstream zone may be a critical one; 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 needs to be stronger. Analytically, this means that if $t_{(j-1)j}E_{j-1} > E_j$, the zonal tradable 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 $j-1$, the

unit of discharge from zone k makes to the total load in zone j, and equals the trading ratio, indicating by how much a source in zone j can increase its discharges if it purchases one unit of $\overline{T}_k$ from any other discharger.

Hung and Shaw (2005) argue that the Trading Ratio System (TRS) avoids the problems of the trading approaches mentioned above; they demonstrate that the TRS is cost-effective and achieves the given environmental quality standard with the least total marginal abatement costs for any initial allocation.[26] Benefiting from the uni-directional nature of water pollution, the cap setting strategy of the TRS 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 for an efficient market solution, avoids local concentrations (hot spots) and avoids that sources need to hold a portfolio of permits as under the Ambient Permit System (Montgomery), which in turn lowers their transaction costs (Hung and Shaw, 2005, p. 88). Further on, the trading ratio is defined exogenously, which lowers the transaction costs in comparison to the Pollution Offset System (POS) determining the trading ratio endogenously. Finally, all discharge permits are allocated individually, an increase in emissions obliges the sources to purchase permits; no free riding behaviour occurs.

3.3 Results

Hung and Shaw (2005, p. 100) developed a scheme for a classification of (theoretical) trading schemes, which could give a first idea about a valuation of different trading schemes. Table 2 shows how the theoretical approaches would be integrated in the scheme. At first glance, one could assume that the water specific approach of Hung and Shaw surpasses the other theoretical approaches: the transaction costs are, also due to the exogenous trading ratio, lower, no hot spot or free riding problems do occur and the quality standard is easy to set as binding.

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zonal tradable 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 stronger to guarantee for the water quality standard in zone j.

[26] The compliance will be controllable by monitoring the emissions e_j of each source. The emissions discharged into the river should be equal to or lower than the zonal tradable discharge permit plus the tradable discharge permits sold to downstream zones: $e_i \le \overline{T}_i + \sum_{k=1}^{i-1} t_{ki}T_{kj} - \sum_{k=i+1}^{n} T_{ik}$ (Hung and Shaw, 2005, p. 89).

Trading Systems	Effects				
	Trading Ratio (Exchange Rate)	Transaction Costs	Hot Spots	Free Riding	Environmental Constraint
APS	Exogenous	High	Yes	No	Difficult to set as binding
POS	Endogenous	High	Yes	Yes	Easy to set as binding
ERS	Exogenous	High	Yes	No	Difficult to set as binding
TRS	Exogenous	Lower	No	No	Easy to set as binding

Table 2 –Theoretical Approaches (Hung and Shaw, 2005, p. 100).

This classification gives only quite general information about the trading schemes. To give one example: the level of transaction costs alone can not tell us anything about the efficiency of the system. The absence of central elements of a permit trading can lower the transaction costs, but at the same time they might reduce the efficiency of the system. Additional criteria have to be taken into account when intending to give a well-founded and comparable analysis of a theoretical or practical permit trading approach (see below).

4 Practical Approaches: Case Studies

Theoretical approaches presented in the literature do rarely refer to already implemented water quality trading schemes (practical approaches) and the other way around. It thus would be interesting to have a look on the practical approaches and to link them to the theoretical ones.

For a better understanding of the case studies in the USA and in Australia, an overview of the institutional framework of each country will be given before the trading schemes themselves will be presented.

4.2 Tar Pamlico Nutrient Trading Program, USA

4.2.1 Institutional Framework – USA

The general framework for the US water policy is set by the *Clean Water Act* (CWA) from 1972.[27] The goal of the CWA is “to restore and maintain the chemical, physical, and biological integrity of the Nation’s waters“ (Sec 101 (a) CWA). Sec 101 (a) assigns well defined targets, like the abatement of specified pollutants and the

[27] Here: CWA as amended on 2002.

development and implementation of programs for nonpoint sources, to each State. Sec 101 (d) determines that the US Environment Protection Agency (US EPA) administrates the CWA, unless otherwise noted. Sec 102 (a) precises that the US EPA should in cooperation with the States and the organisations concerned prepare and develop programs to avoid, reduce and abate the pollution.

The US EPA plays an important role in implementing the CWA. The US EPA *Policy Statement* from 2003 recommends explicitly using innovative approaches like a permit trading to achieve the predetermined water quality standard.[28] The US EPA argues that the introduction of a permit trading would reach given environmental standards in a more flexible and less cost-intensive way than traditional approaches. This instrument would set economic incentives for innovations and for voluntary abatement measures.[29] The US EPA refers to experiences gathered from already existing permit trading systems in the USA.[30]

It is thus not surprising that permit trading is often used for water pollution control in the USA.[31] A critical analysis of some of these programmes, however, shows, that a permit trading in the original definition does not exist. Only rarely permits are really traded. This could create the impression that this is, at least in some cases, due to the given design of the system. The example of the Tar Pamlico Nutrient Trading Program shows how important the specific design of the permit trading scheme is for the effectiveness and the efficiency of the system. This example has been chosen, because it is often mentioned as a success in the literature, and therefore a good information basis is available.[32] But: it is surprising that rarely an author analysed this programme from an economic point of view.[33]

[28] US EPA (2003, p. 1).
[29] US EPA (2003, p. 2).
[30] For more information on the US institutional framework see Keudel and Oelmann (2005).
[31] For an overview of the trading systems see Breetz et al. (2004), Kraemer et al. (2004) or Environomics (1999).
[32] See for example the website of the N.C. Division of Water Quality (http://h2o.enr.state.nc.us/nps/tarpam.htm, January 2006).
[33] For an exception see Hoag and Hughes-Popp (1997), Kerr et al. (2000), Keudel (2005) or Keudel and Oelmann (2005).

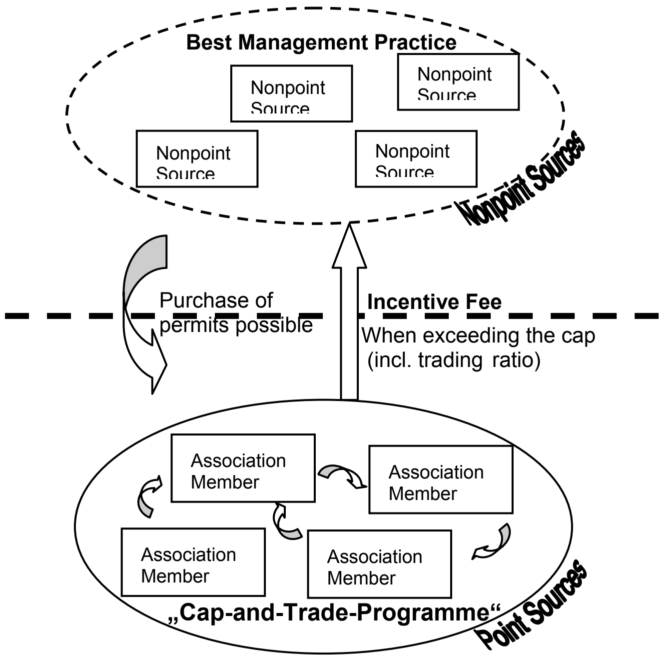

Figure 2 – Tar Pamlico Trading Program

4.2.2 The Trading Scheme

The Tar Pamlico River is situated in North Carolina (USA). The agriculture along the river is responsible for a high share of the river pollution; but also point sources such as sewage treatment plants contribute to it (Faeth, 2000, p. 15). In 1989, the North Carolina Environmental Management Commission (EMC) declared the river as „nutrient sensitive“ water. To decrease the level immission load, the Nutrient Trading Program was implemented in 1990, which spans the whole river basin. This trading programme includes point sources as well as nonpoint sources; traded pollutants are phosphorus and nitrogen (nutrients). According to the theoretical approaches, the Tar Pamlico Nutrient Trading Program could be classified as a non-zonal, rather immission-based trading scheme, applying an exogenous trading ratio for trades between point and nonpoint sources. The major part of the point sources is integrated in the so called Tar Pamlico Basin Association. The system treats point sources in this Association as one single unit, with the goal to reach the given water quality standard, i.e. immission load (cap), within the Association at a higher level of cost-efficiency. When exceeding the cap (the predetermined immission load), the Association has to pay a so called „incentive fee“ for each unit of pollution exceeding the cap. This incentive fee is predetermined, payments flow into an – established beforehand – agricultural fund set up by the state (Agriculture Cost-Share Program (ACSP)) to finance abatement measures according to the Best Management Practice (BMP) at the nonpoint source level. Additionally, the Association can purchase permits from nonpoint sources. But point sources can not sell permits to nonpoint sources. For transactions between point sources and nonpoint sources, a trading ratio applies, taking into account the fact that the impact of emission reductions at the nonpoint source level on the water quality (immission load) can not be accurately predicted. Figure 2 gives an overview of the system.

4.2 Hunter River Salinity Trading Scheme, Australia

4.2.1 Institutional Framework – Australia

The political structure of Australia is a federal one, consisting of the Commonwealth with its States and Territories. While the national level has competences in foreign trade relations, defence and migration affairs, States and Territories are responsible for all other political fields, so the environmental policy. As especially environmental issues are often of cross-border relevance, the Australian environmental policy is characterised by a certain number of bi- and multilateral agreements. Also, the water policy consists of numerous guidelines on the national level and bi- and multilateral agreements between States and Territories.

As the HRSTS controls the salinity of the Hunter River, the Australian agreements on salinity are of relevance. On the national level, the *National Action Plan for Salinity and Water Quality* is of importance; bi- and multilateral agreements between States and Territories implement this plan. In New South Wales the *NSW Salinity Strategy* provides the basis for salinity control. This strategy requires the determination of targets (water quality, immission loads) and efficiency of the measures; the environmental authority does not, however, recommend a specific environmental instrument, like the US EPA does. Water quality trading schemes are rare in Australia; one of them is implemented in the Hunter River basin.

4.2.2 The Trading Scheme

Following, the Hunter River Salinity Trading Scheme (HRSTS) will be presented. This example is selected due to its outstanding structure: in an impressive manner the system integrates special characteristics of the Hunter River as well as of the substances concerned.

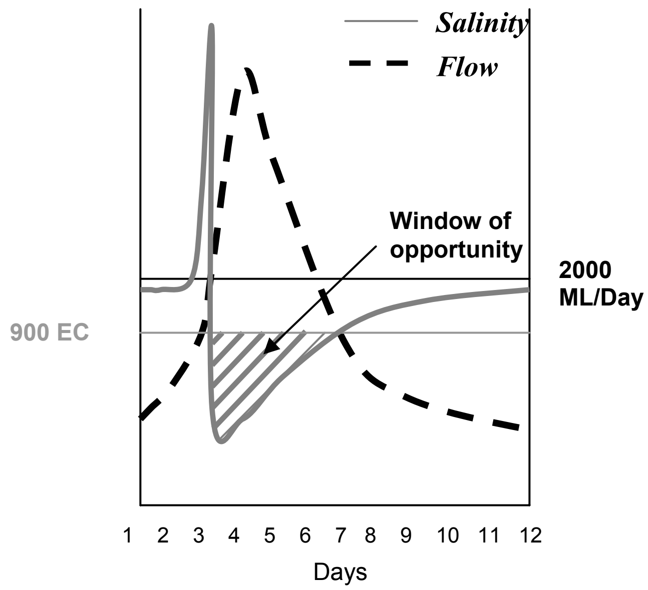

Figure 3 – Salinity and water flow, HRSTS (NSW EPA, 2003, p. 6)

In the region of the Hunter River, the agricultural sector is of high importance. Also, it is a home to more than 20 coal mines and to three power stations (NSW EPA, 2003, p. 3). The critical substance in this river is salt. Salt occurs naturally in many of the rocks and soils of the Hunter Valley and thus in the river. But by introducing saline water, sources as coal mines and power stations contributed to an increasing salinity in the river (HITS, 2004; NSW EPA, 2003, p. 3). Consequently, the water could not be used any more for irrigation in the agriculture (NSW EPA, 2003). This led the NSW Department of Land and Water Conservation (DLWC) and the NSW Environment Protection Authority (NSW EPA) to introduce the HRSTS, a system with dynamic and tradable discharge permits.[34] According to the theoretical approaches, the HRSTS is a non-zonal immission-based approach; no trading ratio applies.

River monitoring has shown that at the beginning of a high flow period („event", see broken line, Figure 3) salinity of the water increases strongly for a short while, followed by a strong decrease (see continuous line, Figure 3).[35] The idea behind the system is the following: licensees can introduce saline water in the moment in which the impact on the water quality is – because of the high volume of fresh water – the lowest possible (HITS, 2004). While in low flow periods no emissions are allowed, they are allowed (according to the permits) in periods of high flow using the „window

[34] After a pilot phase the HRSTS was finally implemented in December 2002. The basic document is the *Regulation of the Environmental Operations (Hunter River Salinity Scheme) Regulations 2002* (NSW DEC, 2004).

[35] The additional amount of water washes salt from the ground and surface out in the river. The following volume of fresh water diminishes the salinity of the water (HITS, 2004).

of opportunity" (NSW EPA, 2003, p. 4).[36] The allowed amount of emissions (cap) depends on the actual salinity and can be changing every day (dynamic discharge permits).[37]

The total amount of allowed emissions is defined for „blocks" (see Figure 4). 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).[38] These blocks represent a volume unit of water that is flowing through the river bed from the origin up to the estuary. The water flow as well as the salinity is measured for each individual block (HITS, 2004). Based on these data, the amount of salinity which can be introduced additionally (the cap) is defined (NSW EPA, 2003, p. 4; NSW EPA, 2002, Division 1, 9). In consequence, the application of this system requires an intensive monitoring system.[39] Also on part of the licensees, a precise monitoring is required (NSW EPA, 2002, Part 5, Division 3).

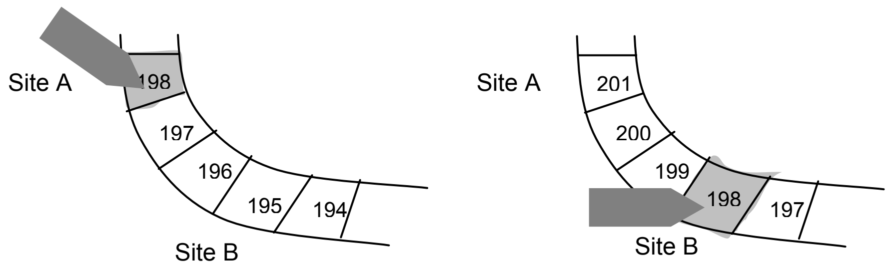

Figure 4 – Trading in blocks, HRSTS (NSW EPA 2003, p. 5)

In total 1000 permits have been allocated. Each permit allows the licensee – and 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 (HITS, 2004; NSW EPA, 2003, p. 5). Licensees can trade these permits (tradable discharge permits) for a single block or for sequential blocks.[40]

[36] 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).
[37] In a period of flood, saline water can be emitted without permits (Brady, 2004, p. 11; NSW EPA, 2003, p. 4)).
[38] In total there are 365 blocks per year which are numerated per day and year.
[39] 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).
[40] Each trade has to 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 trading of permits see NSW EPA (2002, Part 5, Division 2).

Trading Systems	Effects				
	Trading Ratio (Exchange Rate)	Transaction Costs	Hot Spots	Free Riding	Environmental Constraint
APS	Exogenous	High	Yes	No	Difficult to set as binding
POS	Endogenous	High	Yes	Yes	Easy to set as binding
ERS	Exogenous	High	Yes	No	Difficult to set as binding
TRS	Exogenous	Lower	No	No	Easy to set as binding
Tar Pamlico	**PS: 1:1 PS-NPS: Exogenous**	**Lower**	**Yes**	**Yes**	**No binding cap**
HRSTS	**No trading ratio**	**Higher**	**No**	**No**	**Binding cap**

Table 3 – Tar Pamlico Trading Program (based on Hung and Shaw, 2005, p. 100)

4.3 Results

The scheme of Hung and Shaw (2005, p. 100) mentioned above implicitly provides first information about the evaluation of a trading scheme. But it is not detailed enough to give a well-founded recommendation. To give a first idea, the practical approaches are, however, integrated in this scheme (Table 3).

According to the scheme, the Tar Pamlico Nutrient Trading Program would be integrated as follows: for transactions between point sources and nonpoint sources, an exogenous trading ratio, fixed beforehand, applies. For transactions within the group of point sources, no trading ratio applies. The transaction costs for the determination of the trading ratio are thus relatively low. The major part 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 when trading with nonpoint sources are – compared to a „real“ trading – relatively low. Transaction costs for example occur for monitoring, but do not occur for searching a trading partner and the determination of the permit price (information costs), as the incentive fee is paid directly into the agricultural funds and no direct trading takes place. The lower level of transaction costs is due to the absence of central trading elements, which overcompensates the effect of lower transaction costs (see below).

The system sets one cap for all sources; neither for the cap nor for the trading rules, a differentiation in time or place applies. The system thus risks the occurrence of hot spots.

The main criticism against the Tar Pamlico Nutrient Trading Program is that the cap for the point sources is not allocated individually; point sources thus are not able to trade with each other (see Keudel, 2005, p. 298). This absence of an individually allocated cap causes inefficiencies by offering the possibility of free riding behaviour within the group of point sources.[41] Within the group of nonpoint sources no mechanism exists to make the abatement of pollution obligatory for all sources; this can also lead to free rider behaviour. Finally, the determined cap is not a binding one. This can prevent the system from coming automatically to a cost-effective solution.

As the discussion about the Tar Pamlico Nutrient Trading Program shows, the scheme of Hung and Shaw (2005) does not give enough information for a complete (and comparable) analysis of trading systems. An economic analysis for the Tar Pamlico Nutrient Trading Program, for example, shows that the common criteria (ecological effectiveness, cost-efficiency, level of transaction costs) are not fulfilled.[42] Furthermore, other criteria, such as ecological ones, and the interrelation between these, need to be defined to guarantee for a well-founded and standardised analysis.

According to the scheme of Hung and Shaw (2005, p. 100), the HRSTS can be classified as follows. No trading ratio applies; the permits are traded at a one-to-one basis. The different impact of the emissions depending on the local position of the sources is indirectly presented in the permits by a time component: the permits apply to blocks, which are defined in a combined time and space unit.

Probably, the major part of the transaction costs arises from monitoring, including the computer systems installed and the auction processes. The salinity and the water flow have to be monitored at many monitoring points as well as in short intervals and must then be promptly communicated to the sources.[43] To guarantee for a sufficient information flow, computer programmes have been installed. On the one hand, the

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[41] Richard Gannon, Nonpoint Source Planner NC DWQ, email 12.08.2004. The emissions are monitored at the individual source level and the introduction of an individual fine has been considered. It has never been put into action. In consequence, „some smaller facilities have been ‚free-riding' the entire time" (Richard Gannon, Nonpoint Source Planner NC DWQ, email 24.08.2004).

[42] For a detailed analysis see Keudel (2005) or Keudel and Oelmann (2005).

[43] These costs are indirectly financed by the sources: the auction revenue as well as contributions from the licensees are used for financing.

sources thus receive detailed information about potential actual discharges.[44] On the other hand, these systems allow for online trading. Thus, a trading partner can be quickly identified. The installation of these information systems finally prevents an extreme increase in transaction costs for point sources.

The HRSTS avoids the risk of hot spots: the cap for each block is updated permanently depending on the actual flow and salinity characteristics and is well defined in space and time. The defined maximum level of salinity can not be exceeded in any part of the river. No free rider behaviour will occur: the permits are allocated individually, the discharge processes are well defined, and the standards are binding in every point in time and at any place.

This classification only gives a first idea about the effectiveness and the efficiency of the HRSTS. An economic analysis assigns high efficiency and effectiveness to the HRSTS.[45] But: as for all approaches represented here, other aspects might be relevant to give a well-founded, complete, and comparable judgement of the HRSTS.

5. Conclusion

Different theoretical and practical approaches on water quality trading exist; but they have rarely been linked in the literature. Without a standardised and complete criteria catalogue it is not possible to compare these different systems and to formulate recommendations for specific cases. More information on the specific characteristics of the river concerned, the pollutants, the ecological goals etc. need to be integrated in the design of a water quality trading. By her research, the author tries to fill the gap in the literature. Keudel (2005) as well as Keudel and Oelmann (2005) already give an economic and ecological analysis on practical approaches (Tar Pamlico and HRSTS), which extends the criteria catalogue mentioned by Hung and Shaw (2005, p. 100). Without a well-founded criteria analysis, no reasonable comparison of different approaches or even the selection of the adequate instrument design is possible. In her further research, the author will develop a consistent and standardised guideline to allow criticising existing theoretical and practical approaches. This guideline will offer the opportunity to examine whether the introduction of a new water quality trading system for a certain river basin is

[44] “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).

[45] For a detailed economic analysis of the HRSTS see Keudel (2005).

reasonable. If this holds, it will be able to formulate precise requirements for an adequate design of such a trading system.

References

Brady, Katy (2004): Salinity and Nutrient Trading in Australia, Department of Environment Conservation, presented on “International Conference on Compliance and Enforcement of Trading Schemes in Environmental Protection”, 16-18 March 2004, Oxford.

Breetz, Hanna L., Karen Fisher-Vanden, Laura Garzon, Hannah Jacobs, Kailin Kroetz and Rebecca Terry (2004): Water Quality Trading and Offset Initiatives in the U.S.: A Comprehensive Survey, Darthmouth.

Environomics (1999): A Summary of U.S. Effluent Trading and Offset Projects, Bethesda, M.D.

Faeth, Paul (2000): Fertile Ground: Nutrient Trading’s Potential to Cost-effectively Improve Water Quality, World Resources Institute, Washington, D.C.

Førsund, Finn R. and Eric Nævdal (1994): Trading Sulfur Emissions in Europe, in: Ger A.J. Klaassen (1994): Economic Instruments for Air Pollution Control, chapter 11, Dordrecht.

Førsund, Finn R. and Eric Nævdal (1998): Efficiency Gains under Exchange-Rate Emission Trading, Environmental and Resources Economics, 12, pp. 403-423.

HITS (Hunter Integrated Telemetry System) (2004): Hunter River Salinity Trading Scheme, http://www.hits.nsw.gov.au/salinity_trading.html (January 2006).

Hoag, Dana L. and Jennie S. Hughes-Popp (1997): Theory and Practice of Pollution Credit Trading, Water Quality Management, Review of Agricultural Economics, 1, 2, pp. 252-262.

Hung, Ming-Feng and Daigee Shaw (2005): A Trading-Ratio System for Trading Water Pollution Discharge Permits, Journal of Environmental Economics and Management, 49, 1, pp. 83-102.

Kemper, Manfred (1993): Das Umweltproblem in der Marktwirtschaft – Wirtschaftstheoretische Grundlagen und vergleichende Analyse umweltpolitischer Instrumente in der Luftreinhalte- und Gewässerpolitik, Volkswirtschaftliche Schriften, Heft 390, Duncker Humblot: Berlin.

Kerr, Robert L., Steven J. Anderson and John Jaksch (2000): Crosscutting Analysis of Trading Programs – Case Studies in Air, Water, and Wetland Mitigation Trading Systems, Research Paper Number 6, June 2000, National Academy of Public Administration, Washington, D.C.

Keudel, Marianne (2005): Permit Trading as an Instrument for Water Pollution Control – A practically oriented analysis considering the European Water Framework Directive, Zeitschrift für Wirtschaftspoltitik, 54, 3, pp. 280-304.

Keudel, Marianne and Mark Oelmann (2005): Zertifikatehandel im Gewässerschutz: Ansätze in den USA und ihre Übertragbarkeit auf die EU, Zeitschrift für Angewandte Umweltforschung (ZAU), 17, 2, (in press).

Klaassen, Ger (1996): Acid Rain and Environmental Degradation – The Economics of Emission Trading, Edward Elgar: Cheltenham, UK and Brookfield, US.

Kraemer, R. Andreas, Eleftheria Kampa and Eduard Interwies (2004): The Role of Tradable Permits in Water Pollution Control, Working Paper, Inter-American Development Bank, Sustainable Development Department, Washington, D.C.

Krupnick, Alan K., Wallace E. Oates and Eric van de Verg (1983): On Marketable Air-Pollution Permits: The Case for a System of Pollution Offsets, Journal of Environmental Economics and Management, 10, pp. 233-247.

McGartland, Albert (1988): A Comparison of Two Marketable Discharge Permit Systems, Journal of Environmental Economics and Management, 15, pp. 35-44.

Montgomery, W. David (1972): Markets in Licenses and Efficient Pollution Control Programs, Journal of Economic Theory, 5, pp. 395-418.

Montgomery, W. David (1974): Artificial Markets and the Theory of Games, Public Choice, 14, pp. 25-40.

NIRAS (NIRAS Consulting Engineers and Planners A/S) (2004): Danube Study on Pollution Trading and Corresponding Economic Instruments for Nutrient Reduction – Review of existing Economic Instruments, May 2004, Allerød.

NSW DEC (New South Wales Department of Environment and Conservation) (2004): Hunter River Salinity Trading Scheme Credit Auctions – General Information for the 2004 Auction, Sydney.

NSW EPA (New South Wales Environment Protection Authority) (2001): Proposed Protection of the Environment Operations (Hunter Salinity Trading Scheme) Regulation 2001, Sydney.

NSW EPA (New South Wales Environment Protection Authority) (2002): Protection of the Environment Operations (Hunter River salinity Trading Scheme) Regulation 2002, Sydney.

NSW EPA (New South Wales Environment Protection Authority) (2003): Hunter River Salinity Trading Scheme – Working Together to Protect River Quality and Sustain Economic Development, Sydney.

Shortle, James S. and Richard D. Horan (2001): The Economics of Nonpoint Source Pollution Control, Journal of Economic Surveys, 15, 3, pp. 255-289.

Tietenberg, Tom (1985): Emissions Trading an Exercise in Reforming Pollution Policy, Resources for the Future, Washington, D.C.

Tietenberg, Tom (2003): Environmental and Natural Resource Economics, Boston.

US EPA (U.S Environmental Protection Agency) (2003): Water Quality Trading Assessment Handbook: EPA Region 10's Guide to Analyzing Your Watershed, Washington, D.C.