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Can tradable permits be applied for biodiversity conservation? A conceptual analysis

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Silvia Wissel* and Frank Wätzold*

Abstract

There is an increasing demand for flexible and cost-effective biodiversity conservation instruments, which are able to successfully conserve biodiversity while allowing for economic development. One promising option could be the application of tradable permits. We analyze on a conceptual level the challenges arising from applying the instrument of tradable permits to biodiversity conservation. Our analysis is centred around six key elements which are relevant for the functioning of a permit market: initial allocation, market size & activity, market power, habitat assessment, exchange rules and monitoring & enforcement.

* Helmholtz Centre for Environmental Research - UFZ, Department of Economics

Contact author: silvia.wissel@ufz.de

1. Introduction

In biodiversity conservation, there is an increasing demand for flexible and cost-effective conservation instruments which are able to successfully conserve biodiversity while allowing for economic development at the same time. Ecological research indicates that some species are resilient to a moderate level of land use changes and can adapt to small scale disturbances (e.g. Opdam et al. 2006). This implies that in such cases habitat patches may be destroyed for reasons of economic development as long as there is compensation elsewhere. One suitable way to organize such a habitat exchange is to use the instrument of tradable permits.[1] Tradable permits have been successfully applied in many fields of environmental policy (e.g. Tietenberg 2006, Freeman and Kolstad 2007) with the most prominent example being probably the EU CO_2 emissions trading scheme (e.g. Hansjürgens 2005). Potential cost savings to achieve the targets of the Kyoto Protocol through CO_2 trading in the European Union have been estimated to amount up to 56% as compared to a situation without trade (Gusbin et al. 1999).

Applying the instrument of tradable permits to biodiversity conservation is, however, not straightforward. Analyses of existing conservation policies which exhibit some similarities with tradable permit systems (e.g. Wetland Mitigation and Conservation Banking in the USA, Eco-accounts and the Impact Mitigation Principle in Germany, Brazil Forest Trade)[2] indicate challenges as well as opportunities. While each of the above mentioned schemes has been evaluated under selected aspects, an overarching analysis about the challenges that arise when tradable permits are applied to biodiversity conservation is missing. The purpose of this paper is to carry out such an analysis.

We start by explaining the basic idea of applying tradable permits to biodiversity conservation including the reasoning why permit markets offer opportunities for improved conservation. We continue by identifying six conceptual issues along which the challenges of creating a tradable permit market for conservation will be discussed: initial allocation, market size & activity, market power, habitat assessment, exchange rules and monitoring & enforcement.

[1] The terms tradable permits and tradable development rights (TDRs) will be used equivalently throughout the paper.
[2] For references on these policies, see e.g. US: Shabman and Scodari 2004, Fox and Nino-Murcia 2005, Germany: Wittrock et al. 2003, Brazil: Chomitz 2004.

2. The idea of applying tradable permits to biodiversity conservation

2.1. Basic design features

The idea of applying tradable permits to biodiversity conservation is to fix the overall ecological value of habitats in a particular region (cap), while allowing for small scale changes of the landscape and destruction of habitat if tradable permits can be submitted. Conversion of land into new habitat or upgrading of existing habitat will create such a right which is tradable on the market. Exchange rules have to ensure that the value of the destroyed habitat and the value of the newly created habitat is equivalent.

The supply of development rights is provided by private landowners such as farmers and forest owners but also from state authorities or conservation groups that own land. The demand for development rights may result from both private developers who want to build new houses or industrial areas and from government agencies that need development rights for infrastructure projects such as, e.g. new roads. However, demand may also come from private individuals or conservation groups who aim to enhance the ecological value within a region. In contrast to a private or governmental developer they do not submit the development rights but instead keep it (see Thum and Wätzold 2007).

To ensure the ecological and economic effectiveness of tradable permit markets a regulatory framework needs to be designed. This includes the rules according to which the value of destroyed and created habitats is assessed and rules that determine which habitats can be considered exchangeable. Furthermore a regulatory authority is required to ensure compliance with the rules and their enforcement.

2.2. Opportunities through tradable permits

Traditional conservation policies have concentrated on conserving land by designating certain areas as reserves and leaving other areas for economic development. Although the allocation of land for purposes of conservation and economic development may have once been carried out in a cost-effective manner (e.g. Ando et al. 1998), land prices (and hence opportunity costs[3] of conserving biodiversity on this land) can change over time, implying that a previously cost-effective allocation may not be cost-effective anymore. Cost-effectiveness is here understood in a way that land allocation is such that a given conservation

[3] Opportunity costs are the foregone benefits from the best alternative uses.

aim (e.g. protecting a certain number of endangered species) is achieved at least costs. In order to re-establish a cost-effective allocation, a conservation agency would be confronted with the problem of designating new areas for conservational purposes and allowing economic development on land formerly used for conservation. Such a planning approach requires high levels of information on the side of the conservation agency which can be costly to gather. Furthermore, changing the designation of land is also highly sensitive to lobbying activities regarding the choice of areas to be released for economic development and to be designated reserves.

In contrast, a market of tradable permits allows for cost-effectiveness under dynamic conditions without direct governmental interference, facilitating cost-effectiveness advantages when opportunity costs for conservation differ. Owners of economically developed land with decreasing prices would have an incentive to designate their area for conservation and sell the generated development rights to developers who aim to destroy habitats on land with high opportunity costs for conservation. Cost-differentials may arise at different levels:

- Regionally: E.g., in urban areas we expect higher land prices and consequently higher opportunity costs for each provider.
- Between individual providers: Costs also differ between individual providers due to e.g. different technologies or interests in conservation.
- Temporarily: Over time, costs can change, e.g., due to economic development.

The extent of cost-effectiveness advantages and consequently the market potential depends on the degree of cost-heterogeneity on each level.

Alternative market-based instruments are charges or taxes on developing land for economic purposes. The revenue can then be spent on developing new or improving existing habitats.[4] In a first-best world (where marginal costs and damages are known), taxes can be regarded as equivalent to permits in terms of achieving a cost-effective land allocation (Ellerman 2005). However, if assuming that marginal costs and damages are not known with certainty, tradable permits are preferable when a certain ecological target shall be reached (Norregaard and Reppelin-Hill 2000). The problem with charges in a dynamic environment is that their levels need to be constantly adapted and due to limited information it may be difficult to obtain the level that secures a certain conservation goal.

[4] e.g. landscape charge (Landschaftsabgabe) in Austria, since 1994. Charges are required for mining of gravel, sand and stone where the collected funds are allocated to projects of particular purposes.

A system of tradable permits can also contribute to *enhancing* biodiversity conservation in a region in a cost-effective way. In principle there are two possibilities: as mentioned earlier conservation groups or the government may buy development rights and choose not to destroy habitats.[5] Another possibility to increase the ecological value within a market framework lies in altering the rule of exchange, e.g.: for one destroyed habitat patch not only one patch of equal ecological value but two patches have to be converted into habitats.[6]

3. Conceptual Issues

The analysis is structured along six main design issues which we identified as relevant for the market's functioning:

- Initial allocation
- Market size and activity
- Market power
- Habitat assessment
- Exchange rules
- Monitoring and enforcement

3.1. Initial allocation

The initial allocation of permits mainly affects distributional questions of the instrument and efficiency considerations. For initial allocations, the discussion for tradable emission permits mainly focused on trade-offs between "grandfathering", where the initial allocation is based on historical data about emissions, and a form of auctioning permits (Tietenberg 2006). Auctions promise higher efficiency while they often imply property rights changes. In practice, this makes auctions politically difficult to implement.

Decisions regarding the allocation of permits among land owners also have to be made for TDR markets. Grandfathering for TDR markets implies that the existing allocation of land between land for conservation and land for economic development is accepted as given and no development rights are needed for the land already used for economic purposes.[7] In

[5] More than a thousand non-profit land trusts in the US are supporting habitat conservation through incentive programs (Shogren 2005).
[6] Average in Wetland Mitigation Banking: for 1 hectare destroyed, 1.8 restored (Salzman and Ruhl 2007).
[7] Such a system is comparable to, e.g., the German Impact Mitigation Principle or the American Incidental Take Permits.

contrast, auctioning permits for conservation would mean that a regulatory authority auctions off the rights to develop and such rights are also needed for land which is already developed. A landowner who does not obtain such a right needs to stop economic development on his or her land and has to use the land for conservation. An auction might only take place for the first round; subsequently the allocation may be organized via trading permits on the market. A third alternative for tradable permits in biodiversity conservation is to set an ecological target, obliging each landowner to submit TDRs according to the size of his land.[8] Within a region, the overall amount of TDRs would be determined depending on the overall ecological aim. Then each unit of land entitles the right for a certain amount of TDRs (depending on the land's ecological value) but also implies the obligation to submit a certain amount of TDRs.[9] Such a system cannot be applied for emissions trading between firms, since there is no fixed benchmark like the available area to apply a percentage to (neither output, nor input remain constant) and consequently the emissions cap could not be held fixed.

These initial allocations follow different ethical logics (compare Tietenberg 2006 for emissions trading). Grandfathering requires permits only for impacts above current entitlement while auctioning requires paying for any impact. Percentage rules imply a form of charge on property. Any landowner is obliged to care for conservation. The difference to ordinary charges is that the price is determined by the market and that landowners have a choice to either conserve or buy permits.

With respect to cost-effectiveness, the initial allocation should have no influence according to theory. Through auctioning, permits should directly be allocated to the individual who values them most; other allocations would achieve the same result via trading. If, however, dealing with preexisting tax distortions, general equilibrium analysis has shown auctioning to be less costly than free distribution (Goulder et al. 1999). Revenues generated from auctioning can be recycled to compensate for tax distortions in factor markets. So-called "double dividends" refer to revenues being employed to reduce income or other distortionary taxes (Parry et al. 1999).

Auctioning would lead to higher transaction costs than grandfathering since each

[8] Compare the Brazil Forest Trade, where forest owners are obliged to maintain a reserve covering at least 20 percent of their property.

[9] Note that this implies that the overall area designated for conservation may vary, due to the quality of land. High quality implies high eco-points and less area for conservation.

landowner would directly be involved and would have to care for obtaining sufficient permits for his land. Similarly, this holds for an initial allocation based on the size of land. At the same time, the obligations for each landowner arising through the latter allocation automatically initiate a certain market activity.

3.2. Market size and activity

Market size and activity are closely interrelated and are essential for market efficiency. Large markets, with large numbers of participants and high trading activity facilitate finding the adequate market partner implying reduced search costs, leading to lower costs for each transaction. The frequency of interaction depends on several factors:

1. The *regional size* of the market, where larger opportunity-cost differences with increasing market size are expected (compare Newell and Stavins 2003). Thereby incentives for interactions to exploit gains from trade are created.
2. *Economic development* is another external factor driving interaction. In regions with little growth demand for TDRs will be low and irregular, reducing the frequency of interactions.
3. Whether any interactions take place on the market also depends on *existing regulations*. For instance, an existing regulation might require landowners to conserve a given percentage of their land, which is not tradable. Only permits created beyond this level can be traded. Such regulations may be so restrictive that they lead to less trade on the market.[10]
4. High *transaction costs* are another source of market activity impediment, which can be caused by complicated, time-consuming and costly administrative procedures. Both buyers and sellers may be reluctant to participate in markets with high transaction costs. Linked to this is also the aspect of market transparency which is, besides constant price developments and a reliable political environment also relevant for planning certainty (Gagelmann and Hansjürgens 2002). A market platform for exchange provides information for the individuals and eases information exchange between buyers and sellers. Without such an institution, party-to-party transactions might be the norm, including bilateral bargaining. In such cases, the efficiency may be reduced due to lacking information when wrong price signals are sent and in consequence, wrong “abatement” or trading decisions accrue (Baron 1999).

[10] Such a situation occurred in Basel with emissions trade. Existing regulations for emissions were too strict such there was no volume anyone could offer on the market (Staehelin-Witt and Spillmann 1994).

Small markets with low trading activity also increase the uncertainty of price development which is relevant for individual planning decisions (Tietenberg 2006). When expectations about the risk for investors who restore habitat are high due to lack of demand, investors are deterred from participating in the market (e.g. Gagelmann and Hansjürgens 2002, Panayotou 1994). In such a situation the market would break down. An instrument which can help to cope with variations in demand and prices over time to reduce risk is banking (Newell et al. 2005, for further discussion see below).

In order to facilitate market activity and enhance efficiency, it is reasonable to choose large markets for trading permits. This view is supported by a study from Chomitz (2004) on forest trade in Brazil. He analyzes the cost-effects when altering the geographical size of the market. Therefore a static equilibrium landuse model is used, in which landowners are required to hold TDRs for twenty percent of their total area as forest and this forms the base for simulations. Including data on current land use and qualities, he finds substantial gains from trading among larger areas.

But there are also drawbacks to having large markets, since the economic analysis must take into account important ecological aspects such as which species requires habitat to be within reach and the preferences of people to maintain conservation within their region. The instrument of zoning can provide an option to split a market into zones which act as independent submarkets (e.g. Thornes and Simons, 1999). Within each zone an individual ecological target can be determined and trade between zones can or cannot be allowed for. When separating such markets, the question arises what are the optimal sizes for the zones. From the literature on non-uniformly distributing emissions, it is known that the regulators face a trade-off between higher control on the environmental targets and lost potential for cost-effectiveness for smaller markets. Krysiak and Schweitzer (2006) find that depending on firm heterogeneity and pollutant mobility, the zone size can be a partial substitute for abatement.

3.3. Market power

Economic analyses for tradable permit markets, particularly for emissions trading, have pointed out the relevance of market power. Two different ways have been identified of how market power can be exploited: either a firm can exert power on the permit market itself or on the product market. Product markets include input and output markets, where a dominant firm in the permit market might try to use its power to harm competitors in other markets, e.g. for industrial or agricultural products (Burneaux 1999). Here, the interest lies only on the permit market itself, since the impacts from permit markets on product markets are regarded as negligible. This is because in the areas where TDR may be potentially implemented, we are often dealing with world markets (e.g. as with many markets for forestry or agricultural goods) and also competitors might not act within the same permit market. Market power is present when a firm can exert influence on prices, usually when it holds a large market share. This may be a dominant firm on the demand side (monopsony), beating down prices below the competitive level, or supply side (monopoly), raising prices above the competitive level.

A broad range of analyses have been carried out for emissions trading in the presence of market power. The debate started with Hahn (1984) who shows that a monopsony as well as a monopoly may lead to higher total abatement costs, implying reduced efficiency compared to a competitive permit market. He presents the case where one firm has market power and acts as price setter, other firms being price takers. Price takers minimize costs (sum of abatement and permit costs) with respect to the quantity they emit and yield: marginal abatement cost equals price of permit (mc=p). The firm holding market power minimizes with respect to the price. Following Hahn's analysis this leads to efficiency losses unless the quantity of permits the market power firm was being allocated initially equals the amount it chooses to emit in equilibrium. This refers to the case where the market power firm does not enter the market, and in consequence, does not manipulate the price.

For a market of tradable permits in conservation this analysis can best be transferred to a case where landowners are obliged to maintain a certain amount of permits depending on their land. The initial allocation could be based on the overall size of land or on current ecological qualities. Then each landowner can decide whether to provide or buy permits and the decision process, concerning the cost minimization as introduced by Hahn, is equivalent to emissions trading. The assumption about increasing marginal costs can be expected to hold for conservation, where conservation costs increase with lower suitability. As a result, landowners holding market power might influence the price and in consequence impact cost-effectiveness equivalently to firms involved in emissions trading.

The question for TDR markets is how market power could occur and to what extent it may be exhibited. Entry to the market, and therewith potential competition, for TDRs is constrained by the amount of land which is available. However, given large enough areas overall TDR markets can be regarded rather open.

From the demand side, market power can arise when there is a large scale investor, e.g. for a business park, being the only developer in the region, or in a situation of very infrequent demand. An investor who can exhibit market power will try to beat down the price below the competitive level. This can imply a risk of investment for suppliers. When the risk is too high, participation by suppliers might be prevented, such that they would not be willing to make investments, which they base on their expectations of demand, and a market might not develop.

On the supply side, market power arises, e.g., if there are only few large landowners or a large conservation organization. Again, the question is how much influence suppliers can exercise on the market when they have market power. This is dependent upon demand elasticity. Suppose permit prices differ between regions:

- Considering an investor for a business park. High prices for permits may lead to a situation where an investor moves to another market and suppliers would not be able to extract extra profits.
- Considering a public investor for infrastructure projects, his demand might be more inelastic. Thus, he cannot choose about switching to other markets for TDRs. In such a situation, the supplier might raise the price.

3.4. Habitat Assessment

In order to achieve the environmental target of maintaining a certain habitat quality in a region, a method for assessing habitats and creating tradable permits following some form of measurements must be developed. An assessment is required to ensure the ecological value destroyed will be equivalent[11] to the value created. In that respect an assessment is crucial for the ecological effectiveness of the instrument.

Habitats differ in many ways, e.g. in terms of type (wetland, forest, etc.), temporal aspects (e.g. succession stages, resulting in suitabilities for different species) and spatial distances to other habitats (relevant for dispersal and metapopulation dynamics).[12] Moreover, habitats also perform ecosystem functions not directly related to biodiversity conservation to different

[11] Equivalency depends on environmental targets, supposed we want to maintain the status quo an approximation to identity is required.
[12] Nonfungibility of space, type and time (Salzman and Ruhl 2007).

extents, e.g. as places for human recreation and watersheds. Given the target of biodiversity conservation, the focus will be on heterogeneities in type, time and space.

If an exchange via the market is to be operationalized, units of exchange for habitat patches are necessary, which will in the following discussion be called “eco-points”. This includes the need for a way of measurement and, depending on the chosen index, relevant information. A range of indexes have been proposed and applied for similar economic instruments so far.[13] Before deciding about the index itself, the requirements it should fulfill have to be laid out. Some requirements include:

- The information required for the index needs to be accessible: this means the costs for the information should be rather low, information should be available and transparent for the market participants.
- The index should be relevant and robust for the ecosystem under analysis, combining considerations of quality and quantity, still remaining objective and its application should be repeatable at different times and by different agents. This contributes to the reliability and overall trust agents have in the market.
- Generally a simple index is preferable to a complex and detailed index.[14]

It has to be recognized, that any habitat assessment involves normative decisions.

3.5. Exchange Rules

Since habitats are heterogeneous, rules must determine what can actually be traded against each other. Based on an index, incorporating various information from assessments on type-specific, temporal and spatial aspects, a decision must be made by a regulatory authority about the criteria that would allow the corresponding eco-points to be traded against each other and according to which rules. For example, it may not be feasible to exchange wetlands for forest. As discussed above, how these rules are decided upon determines the market size and activity. If limits are decided for what can be traded against each other, these will be separate markets.

3.5.1. Exchange Ratios

[13] These include, e.g., the environmental benefits index (USA), the habitat hectares approach (Australia) and Eco-accounts (Ökokonten, Germany), one which has only been proposed but is not yet applied is the habitat equivalency analysis (HEA), compare Bruggeman et al. 2005

[14] For discussions about the requirements of such an index compare, e.g.: Wilcove 2004, DPI and University of Melbourne 2003, Parkes et al. 2003.

Tradable permit markets which have made an attempt to deal with the heterogeneity of the resource under consideration are markets for sulfur dioxides and nitrogen oxides. A reason for combining these markets was seen in the comparability of environmental impacts caused by these two emissions. A ratio of 3 to 1 was proposed following the environmental damage caused. This reflects a pragmatic approach which was discussed as a trade-off between the potential for a larger market and the ecological externalities which might not have been fully identified by research. Since the costs for compliance differ between SO_2 and NO_x, there lies a potential for reducing total costs. When prices for permits diverge disproportionately to their relative damages trade between the pollutants can enhance cost-effectiveness given the exchange ratio reflects the relative difference in environmental damage.[15]

Ratios can also be used for habitat exchange rules. Suppose trade among different types (e.g. grassland and forest) is allowed for. The regulator can then determine a ratio depending on the scarcity and importance of the types for the region under consideration, e.g. forest eco-points can be traded for grassland eco-points 2:1. Thereby a higher ecological value is imposed on grasslands and incentives are created for landowners to rather restore grassland than forest as well as for developers to choose forest instead of grassland. Similarly, ratios can be applied for trade between different succession stages of habitats. Old forest might then receive more eco-points then young forest, which is of particular relevance for habitats of long regeneration time.

One problem which occurs with ratios is decreasing marginal utility when more habitat of one type is created. To accommodate for this, exchange rates may permanently have to be adjusted. Disadvantages from flexible adjustments are uncertainties created for market participants about future values of their land as well as the level of obligations.

3.5.2. Temporal dimension

For species persistence, the continuous existence of habitat is relevant. High turnover rates and long time lags between habitat destruction and restoration may negatively impact survival probabilities of species. Habitats can take long time to regenerate, e.g. old forests require decades of growth before providing adequate habitats for certain species. Also species need time to recolonize their new habitats (Keymer et al. 2000).

This raises the question whether there should be a minimum duration for permits on how

[15] An analysis by Lutter and Burtraw (2003) estimates the compliance cost savings to amount $1.1 billion annually for the US.

long a land must be maintained for conservation purposes once it has been converted. This decision involves a trade-off between the ecological requirements for species and economic requirements for efficiency. While frequent changes in habitat structure endanger the ability of species to adapt, permanent spatial structure of habitat leaves no room for improving cost-effectiveness.

Another aspect which is related to this question refers to who bears the economic risk of future costs. Permit prices might fluctuate over time. With long-term contracts the risk can be transferred to the providers of eco-points, otherwise it is left with the developer.

An uncertainty related to temporal dimensions arises when dealing with habitats that take a long time for restoration. If it is required that such habitats have to be fully restored before a landowner gets a permit several problems arise. For a developer it becomes difficult to find an exchangeable permit at the time of destruction. Likewise, there is uncertainty involved for landowners when restoring habitat without knowing whether they will find a buyer for the permit in the future.[16] An instrument to facilitate trade when there are temporal discontinuities is banking.[17]

In emissions trading the introduction of banking for an intertemporally optimized allocation has been discussed. Kling and Rubin (1997) analyze the efficiency gains through banking from the perspectives of the individual firms who seek to optimize the net present value and from the perspective of a social optimal allocation. Keeping the overall level of emissions fixed, they find that due to discounting of firms[18], more is emitted in early periods, causing higher damage to the society. Therefore they propose a trading rule, including a penalty at the level of the firms' discount rate counterbalancing the effect.

Similarly, such effects from discounting occur in conservation markets. Incentives can be created for suppliers to restore habitat before explicit demand exists when they receive some form of interest on their permits. For developers, borrowing permits obliges them to pay interest. In order to achieve a social optimal allocation the rates should reflect the ecological benefit or damage. This damage has to incorporate the risk that habitat restoration might not be successful or when species might not be able to colonize land. Incorporating these rules

[16] Existing policies, e.g. 'Ökokonten' (eco-accounts) in Germany, allow for banking to facilitate finding an appropriate exchange partner on time.
[17] Here: banking stands for emitting less in the current period to save permits for later emissions; borrowing means higher emissions in the current period and using up permits which have to be repaid in the future.
[18] When firms discount they value the present higher than the future.

into a TDR market would cover the problems raised by Kling and Rubin about the deviation of private and social optimization.

An effect which can result from allowing for banking and paying interest is that the environmental cap cannot be held fixed anymore. Suppose a supplier restores habitat and does not sell the permit immediately on the market. He banks his permit and receives interest for this – which will amount to the additional ecological benefit he provides during this time. Effectively the ecological target is higher than the fixed level in this period. Later, when the permit is sold, the difference in eco-points which results from the interest on the permit will also be sold. Then, the overall ecological benefit lies below the fixed environmental target. Thus, allowing for banking implies a fluctuating environmental target. A limitation to variation could be introduced via corridors within which fluctuation of the ecological benefit is accepted.

3.5.3. Spatial Considerations

Another aspect which needs to be taken into account to ensure the survival of species is the spatial size and configuration of habitats. When concerned with species, their habitat needs to be of sufficient size. Furthermore, metapopulation dynamics is important which implies that connected habitats are more valuable than isolated ones because species can migrate between habitats allowing a recolonialisation of habitats where the population has gone extinct (Hanski 1998). This means that the ecological value of habitats is space dependent, i.e. it depends on the presence and location of other habitats. When the spatial dimension is neglected in permit trading, resulting configurations are likely to be less cost-effective than with a scheme which includes a suitable spatial approximation (Drechsler et al. 2007).

If neighboring or close patches have an influence on the ecological value of land, the spatial externalities cause interdependencies among contiguous lands and this should be included in the decision making. However, this is not trivial. For purpose of illustration suppose a landowner A restores habitat, which is close to landowner B’s already existing habitat, and both habitats’ ecological values increase.[19] A very simple example is:

[19] When a certain size for a network of habitat is reached the increase of the ecological value may only be small.

A's restored land on its own would receive	2 eco-points
B's land on its own was valued	2 eco-points
Both in conjunction value an additional	2 eco-points

But who will receive the added value? And vice versa: if one area is developed who is responsible to pay for the losses in value?

Case 1: Landowner A who restored the new habitat. At the same time this must imply that for a situation when a landowner decides to develop his habitat, he would have to account for the overall - global - ecological loss he causes. Each action will incorporate global benefits and costs for habitat.

Case 2: Landowners receive eco-points corresponding to local ecological benefits for their land. This only incorporates a share of the added value for their land. In a situation where one of them decides to develop his land, he will also impact the ecological value of the other habitat.

Case 3: Landowner A can only restore or destroy habitat with approval of landowner B. This implies that the additional value generated or abolished is split up between the two depending on the local costs.

In all cases, landowners are dependent on the actions of their neighbors. Incentives for cooperation and interests differ though. For case 1 landowners would generally be reluctant when their neighbors restore habitat because their costs of leaving a habitat increase when having neighbors (as long as there are positive benefits from adding new habitat to a plot). Then landowners might even want to deter their neighbors from providing eco-points. Mutual interests for landowners are created in cases 2 and 3, where landowners can benefit from their neighbors offering eco-points.

In case 3 a landowner will only receive the share which is left after potential costs arising for his neighbors are covered and benefits are allocated to neighbors. This situation is most efficient, since a landowner would join or leave a network only when considering costs and benefits for all his added value is at least zero. In case 1 and 2, a new habitat might not be created, even though it would generate a global surplus, or a habitat might be maintained too long while it causes higher costs than benefit.

In reality, there is a lack of information and uncertainty involved with changes in the ecological value when neighborhood changes. As has been proposed for emissions trading there are different market-based regulation concepts for dealing with spatial distribution:

- ambient permit system (where emissions are measured at the particular receptor)
- trading zones with/ without interzone trading
- use trading rules imposing restrictions on individual permit transactions (e.g. Tietenberg 2006, Krysiak and Schweitzer 2006)

Only the two latter systems have been applied for emissions trading and similar systems are feasible for an application to conservation.

Zoning is already used for transferable development rights in land markets without particular focus on biodiversity conservation. Each zone is assigned a level of development intensity. Usually there is a receiving area, where demand for development is strong, and a sending area, where development is halted to compensate for the loss of open and farm lands in the receiving area (compare e.g. Mills 1980). Zoning is yet another way how to care for denser habitat formation but only on a regional level, incentives for direct neighborhood cannot be created through this instrument.

A possible trading rule for spatial incentives dealing with direct neighborhood is the agglomeration bonus, which has originally been proposed for conservation payments (Parkhurst et al. 2002). It can also be applied to biodiversity conservation (Drechsler and Wätzold 2007). Such a bonus would be accredited when suppliers coordinate their action and convert habitat on contiguous lands. This raises questions about the assumptions made on the behavior of the market participants – here particularly of the farmers. The agglomeration bonus creates an incentive for the coordination of their action and it is a crucial question whether the individuals achieve cooperation enhancing the overall market efficiency.

Zoning and trading rules might reduce the economic cost saving potential in the market, while restricting trade by zoning or making it profitable for high-cost landowners to produce habitat when they are surrounded by other producers. Given that these market rules care for better ecological performance, there might still end up being overall market efficiency gain.

3.6. Monitoring and Enforcement

For a functioning market and to ensure the ecological effectiveness of trading development rights, effective monitoring and enforcement mechanisms are required. These ensure the implementation of trading rules and measures. If compliance cannot be ensured, the overall effectiveness and maintenance of habitat quality are at risk.

A monitoring agency must acquire information in order to ensure compliance, which is costly. However, gathering this information is essential for other conservation instruments as well. To make site selection, e.g., the authority must also have a high level of information on species and habitat suitabilities. Also for enforcement of conservation measures the requirements are alike to those for other instruments.

When designing rules for exchange an authority can influence the costs of monitoring, which also depend on the type of habitat and the species to be preserved.

4. Discussion and Conclusion

Permit trading schemes are a promising instrument for biodiversity conservation when searching for a flexible and target-oriented policy. Tradable permits, in contrast to other conservation instruments, open opportunities for dynamic adaptation on different levels, e.g. price or cost changes and changes in the neighborhood conditions.

Many results which have been obtained from studies of other permit markets can as well be applied to TDR markets and similar problems are identified in relation to market power and the choice of an initial allocation. One important difference to the permits traded in other markets (like emissions trading) is the necessity of some assessment for biodiversity to facilitate measurement and evaluation of biodiversity.

A major challenge for an application to biodiversity is the crucial dependence on the spatial and temporal context. This adds additional complexity to the design of permits and makes a careful analysis of ecological and economic implications necessary before implementing any trading system.

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