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Measuring the Effectiveness of International Environmental Regimes

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Although past research has emphasized the importance of international regimes for international governance, systematic assessments of regime effects are missing. This article derives a standardized measurement concept for the effectiveness of international environmental regimes. It is based on a simultaneous evaluation of actual policy against a no-regime counterfactual and a collective optimum. Subsequently, the empirical feasibility of the measurement concept is demonstrated by way of two international treaties regulating transboundary air pollution in Europe. The results demonstrate that the regimes indeed show positive effects—but fall substantially short of the collective optima.

In a major review of research on international environmental policy, Zürn (1998, 649) concludes that regime effectiveness has become a “driving force in the analysis of international relations” (see also Martin and Simmons 1998). Much of this research has been undertaken in the environmental field. The first phase was characterized by a focus on the conditions that account for the rise of international regimes as instruments for managing or resolving conflicts over environmental problems (e.g., Keohane 1984; Keohane and Nye 1989; Young 1989a, 1989b; Young and Osherenko 1993; Gehring 1994; Rittberger 1995; Hasenclever, Mayer, and Rittberger 1996). However,

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although international institutions may be successfully initiated, this does not guarantee that they will have effects.

In the second phase of research, attention shifted toward regime implementation and compliance (e.g., Chayes and Chayes 1993; Victor, Ravstiala, and Skolnikoff 1998; Brown Weiss and Jacobson 1998; Underdal and Hanf 2000). In the present third phase of research on international regimes, we return to the core question of whether the international regimes formed actually matter (Haas 1989).

In a broader sense, the analysis of regime effectiveness is related to the literature on public policy evaluation (see Mohr 1988). Project evaluation routinely forms part of the standard public policy cycle; it is applied to domestic and comparative political domains such as the evaluation of public health care systems, pension plans, and military expenditures. Given the rise of international regimes to combat environmental and other problems on the regional and global scale, it is important for governments to find out which of the international regulatory regimes they have joined actually yield returns on their investments and where progress has been minute. This necessitates both aggregate (regimewide) assessments and disaggregate results on the level of countries. Such a comparison of the relative effectiveness of different regimes serves also as a prerequisite for an inquiry into the causal impacts of various regime design factors.

If public policy evaluation is more broadly understood as evaluating government performance against standardized yardsticks, it is quite conceivable that the concept of regime effectiveness is more broadly applied across the various issue domains of international relations. Generalizations of the concept of regime effectiveness, as introduced in this article, can be applied to problems of international political economy (e.g., negotiations on the reduction of tariffs on trade) or international security. The literature on environmental security (e.g., Homer-Dixon 1994) actually lacks a systematic inclusion of the response strategies at the hands of national governments or international security regimes. By focusing on this crucial intervening variable in between environmental degradation and the onset of violent war, one of the central puzzles of the environmental security literature might be solved: why do we find so few cases in which a strong link between environmental degradation and the onset of violent conflict can be found (e.g., Hauge and Ellingsen 1998; Sprinz 1999)? Conversely, once we witness the onset of violent conflict, it is very plausible that effective international regimes will be able to limit the duration and severity of such conflicts as opposed to ineffective regimes. In both cases, international regime effectiveness would be a key variable to enhance our understanding of international conflict.

In this article, we develop a general measurement concept for assessing the degree to which international environmental regimes contribute to environmental problem solving. This concept will subsequently be formalized for the case of transboundary environmental problems, and its feasibility is illustrated with data from the regulation of "acid rain" in Europe. Furthermore, the article highlights the benefits of an assessment tool for the effectiveness of international environmental institutions by comparing the results with those derived from different methodological approaches.

THE GENERAL MEASUREMENT CONCEPT FOR REGIME EFFECTIVENESS

The present literature does not offer a unified approach to assess a regime's effectiveness. Nevertheless, there exists considerable agreement about the conceptual problems. These have been succinctly summarized by Underdal (1992, 228-29):

(i) What precisely constitutes the *object* to be evaluated? (ii) Against which *standard* is the object to be evaluated? (iii) *How* do we operationally go about comparing the object to our standard; in other words, what kind of measurement operations do we perform in order to attribute a certain *score* of effectiveness to a certain object (regime)?¹

The method outlined below systematically builds on each of these questions.

THE OBJECT OF EVALUATION

In his literature review on environmental regime effectiveness, Jacobeit (1998) concludes that much research has focused on variables of political behavior in the economic-political domain (e.g., Keohane and Levy 1996), the legal-political domain (e.g., Victor, Ravstala, and Skolnikoff 1998), the comparative-political dimension (enhanced by multilevel explanations relating domestic and international environmental policy) (e.g., Schreurs and Economy 1997), or the process dimension of international regimes (e.g., Oberthür 1997). Probably the most inclusive concept of regime effectiveness has been advanced by Young (1999), who combines several of the above aspects. However, the challenges in devising operational measures of regime effectiveness increase with the comprehensiveness of the underlying concept.

Most authors have used relatively simple indicators as the object of evaluation. An obvious candidate is the degree of problem solving: the actual impacts of a regime. In *Institutions for the Earth*, Keohane, Haas, and Levy (1993, 7) ask the crucial question: "Is the quality of the environment or resource better because of the institution?" However, reliable data are often lacking. Furthermore, especially for environmental problems, there is sometimes a long time lag between the action triggered by a regime and the impacts that follow from this action. This is particularly severe for pollution stock problems, in which the recovery process of the environment may last long (as for stratospheric ozone depletion) or the impacts of pollutive activities are felt only after a long time lag (as for climate change).

Such problems are also acknowledged by Keohane, Haas, and Levy (1993, 7), who therefore suggest to "focus on observable political effects of institutions rather than directly on environmental impact." This evaluation of a regime along its output may take place either on the level of the regime itself, analyzing its norms, principles, and rules (see Underdal 1992, 230), or on the national level in terms of the regulations and

1. Similarly, Bernauer (1995, 335) suggests, "The concept of institutional effect raises three questions: Which outcomes do institutions affect and which of these outcomes should analysts focus on? How can these outcomes be evaluated in terms of institutional success or failure? Which measurement operations are required to assess the effect of an institution?"

other decisions that have been agreed on by the members of the regime. However, a high political output does not necessarily lead to the desired impacts because rules may prove ineffective or simply be neglected.

We therefore believe that a policy instrument that lies in between those two extremes and covers aspects of both of them will be the most appropriate object of evaluation. This policy instrument should be closely related to the primary goals of an institution, and sufficient reliable data must be available. In many of the most prominent environmental regimes, emission reductions (of greenhouse gases, CFCs, SO₂, or NO_x) will be an obvious candidate because they follow more or less directly from the political output of the regime and are deterministically or at least probabilistically related to environmental impacts. This is in line with the conclusions by Zürn (1998) and Jacobeit (1998, 348), both of whom regard emission-based approaches to the measurement of international regime effectiveness as particularly promising.

THE STANDARD OF EVALUATION

Having decided on the object of evaluation, the next question is against which standard this object should be evaluated. The first candidate is the no-regime counterfactual or “the hypothetical state of affairs that would have come about had the regime not existed” (Underdal 1992, 231). Despite its widespread use in the literature, there is a common feeling of uneasiness in doing so (Tetlock and Belkin 1996). For example, Bernauer (1995, 360) criticizes that the counterfactual component “introduces an element of more or less informed speculation”—hence, one is very much tempted to ask whether one can do without it.

However, Fearon (1991) has convincingly argued that counterfactuals cannot be avoided in nonexperimental hypothesis testing, and all one can do is to be explicit and careful in their use. Similarly, Tetlock and Belkin (1996, 3) summarize a recent volume on *Counterfactual Thought Experiments in World Politics* by concluding that “we can avoid counterfactuals only if we eschew all causal inference.” It is identification of effects that have been *caused* by a regime that constitutes the very essence of research on regime effectiveness.

In particular, for transboundary air pollution, it would be inappropriate to use emission levels prior to regime formation as a standard against which to evaluate the regime effects. This would neglect that countries might have undertaken substantial emission reductions even without international cooperation—for a multitude of reasons, such as increased environmental awareness, improved abatement technologies, or the collapse of Eastern European economies. Only after we have systematically explored the counterfactual of what would have happened without the regime can we ascribe the remaining effects to the international regime.

Having accepted the indispensability of counterfactual reasoning in any analysis of regime effectiveness, the main challenge is to find methods by which its “speculative element” can be minimized. Many studies of regime effectiveness in the field of international environmental policy employ process tracing to establish the causal effect of international regimes (e.g., Underdal 1997; Young 1999). By familiarizing themselves with the subject matter, expert authors try to understand the role that international

regimes play across their life cycle. However, the subjective component of the particular researcher figures strongly in this approach, and as Zürn (1998, 640) concludes, "The reader . . . wonders whether the method could not be made more systematic."

An alternative approach is to explicitly model regime and nonregime factors—and thereby construct a tool to simulate different states of the world. This exercise is still in its infancy and has probably not yet reached a stage where it can be implemented reliably for complex policy issues.² On the other hand, the simulation of baseline scenarios of emission trajectories has become a standard exercise in many environmental policy areas. Systematic assessment of those scenarios and their ex-post correction using the actual development of critical parameters (such as population and GDP growth) may offer some guidance in constructing no-regime counterfactuals.

Useful information may also be obtained from econometric studies. For example, Murdoch and Sandler (1997) build an impure public subscription model of emission reductions that accounts for emission transports across borders (see also Murdoch, Sandler, and Sargent 1997). Based on this theoretical model, they derive an econometric specification for the demand for emission reductions that is tested empirically with data for sulfur emissions in Europe during the period from 1980 to 1985. However, the estimated parameter values for the period before the signature of the Helsinki Sulfur Protocol provide only a very rough indication of what would have happened after 1985 without the European regime on transboundary air pollution. This is particularly true because the data used for the estimation do not reflect the structural break caused by the dramatic political changes in Eastern Europe (Murdoch, Sandler, and Sargent 1997, 288).

Based on these considerations, in this article we have opted to seek advice from a number of long-standing policy experts in the particular domain under investigation and elicit their best assessment of the no-regime counterfactual via standardized interviews. The presumption behind this approach is that the assessment of the counterfactual should be undertaken on the basis of the best knowledge available in a particular field. Furthermore, by interviewing different groups of actors, this method makes it possible to incorporate different perspectives and, by averaging their assessments, derive estimates that are less biased toward the subjective assessment of any particular individual. However, it is important to note that the quality of the data derived from interviews affects the substantive findings on regime effectiveness. In conclusion, we believe that progress in the construction of counterfactuals is probably the most pressing area of improvement not only for the viability of the approach followed in this article but for any study on regime effectiveness.

The no-regime counterfactual does not suffice as the only evaluative criteria because it gives only a very vague indication of how well a regime serves the purpose it has been designed for. For example, some environmental problems might require higher aggregate reductions of pollutive emissions than others, an important aspect

2. This is not to deny recent advances in this area (e.g., in the literature on two-level games or domestic policy models). Examples are Pahre and Papayoanou (1997) as well as Mo (1994).

that would be neglected if the effectiveness of a regime were only judged according to changes relative to the no-regime counterfactual. Evaluating a regime “against some concept of collective optimum” (Underdal 1992, 231) circumvents such problems; however, its specification poses a research challenge by itself.

One avenue followed by the compliance literature is to use the targets specified in environmental treaties. However, these treaties need not prescribe an optimal abatement strategy. The Helsinki Sulfur Protocol, with its 30% emission reductions applied to all participants regardless of their abatement cost functions, provides a good example. Using the regime’s rules (e.g., a specific emission reduction) creates an endogeneity problem because the attainment of inappropriate treaty targets would be mistaken as an indicator of high regime effectiveness. As Downs, Roke, and Barsboom (1996, 383) observe, severe selection effects may have led researchers to find high degrees of compliance when, in fact, the “treaty’s depth of cooperation” (i.e., the incentives for countries to defect from an international environmental agreement) are low.

Bernauer (1995, 369) proposes the use of broader institutional goals instead. However, not only the specification of explicit treaty targets but also the setting of broader institutional goals are part of the regime process and, therefore, are susceptible to the endogeneity problem. Furthermore, broader institutional goals are often formulated very vaguely—such as the objective “to prevent a dangerous anthropogenic interference with the climate” in the United Nations Framework Convention on Climate Change—to ensure that the goals will be widely acceptable. This vagueness makes it extremely difficult to derive clear-cut evaluative criteria.

In those cases in which ecosystems are characterized by large discontinuities in their response to pollutants, thresholds such as the critical loads concept for acid rain could be used as the collective optimum. If environmental vulnerability has been eliminated, then a collective optimum is reached. In the context of transboundary air pollution, policies aim to avoid exceeding critical loads, which are defined as

a quantitative estimate of the exposure to one or more pollutants below which significant harmful effects on specified sensitive elements of the environment do not occur according to present knowledge. (United Nations Economic Commission for Europe [UNECE] 1988, Article 1(7))

However, not only do many environmental problems lack such discontinuities, but from a welfarist perspective, attainment of the environmental optimum is also not necessarily desirable. To take a somewhat extreme example, developing a worldwide protection system against the potentially devastating impact of asteroids seems prohibitively expensive, even though most of us would agree that it would be a good thing in principle.

Therefore, we will derive the collective optimum by way of another counterfactual—namely, the hypothetical state of affairs that would have come about with a perfect regime. Although constructing this second counterfactual may appear demanding at first sight, we later present a method for deriving it by game-theoretical reasoning from knowledge of the no-regime counterfactual.

DEFINING AND OPERATIONALIZING REGIME EFFECTIVENESS

To assess a regime's effectiveness, we have previously suggested incorporating a no-regime counterfactual as well as a collective optimum as standards of evaluation. In combination with our comments on the object of evaluation, this can be synthesized into the following measurement concept for regime effectiveness.

Regime effects are improvements in the object of evaluation (dependent variable) that can be attributed to the regime. Usually, this will be evaluated along the degree of instrument use such as percentage emission reductions. A lower bound is determined by the no-regime counterfactual (NR) (see Figure 1): the degree of instrument use that would have occurred in the absence of the international regime under investigation. An upper bound is established by the collective optimum (CO): the degree of instrument use that would have been obtained by a perfect regime. Accordingly, the *regime potential* is the distance between the no-regime counterfactual (NR) and the collective optimum (CO), expressed in units of instrument use. Usually, countries (or a group of countries) will execute actual policies (AP) that fall into this interval. The effectiveness of a regime (E) can then be measured as the relative distance that the actual performance has moved from the no-regime counterfactual toward the collective optimum or as the percentage of the regime potential that has been achieved (see Figure 1). This score falls into the interval $[0, 1]$.³

By construction of the effectiveness score, a small regime potential (CO – NR) would imply that even small deviations of either the no-regime counterfactual, the collective optimum, or the actual performance can lead to relatively large changes in the results. To assess this effect, we define a sensitivity of effectiveness score (S) as the absolute change of the effectiveness score resulting from a change in the actual performance by one percentage point. This is, of course, just the derivative of the effectiveness score with respect to the actual performance.

This definition and measurement concept of regime effectiveness show a range of advantages: by merging the two evaluative criteria of relative improvement from the no-regime counterfactual and distance from the collective optimum into one dimension, we overcome the bias toward either of the two, which characterizes large parts of the literature on regime effectiveness (see Underdal 1992, 230-34). Furthermore, the measurement concept is expressed in very general terms, and it is not limited to a particular policy instrument or a specific method to derive the upper and lower bounds. The appropriate method to be chosen depends on a variety of factors, including the type of international regime, data availability, and the methodological orientation of researchers. By providing a common standard of evaluation that can be used by

3. Note that our "effectiveness score" differs from the "index of easy riding" and the "index of tragedy" as introduced by Cornes and Sandler (1983, 1984). The latter describes the relative distance of the Pareto optimum and of the Nash solution from a common reference point. Applied to our article, this reference point would be the level of emissions without any abatement measures. If the Nash solution coincides with the Pareto optimum, the index of tragedy is 0, and if the Nash solution coincides with the reference point, the index of tragedy is 1. Therefore, the index of tragedy roughly describes the "malignity" of a problem; as such, it has some similarity to our "regime potential": the more malign a problem is, the greater the potential benefits of an effective regime.

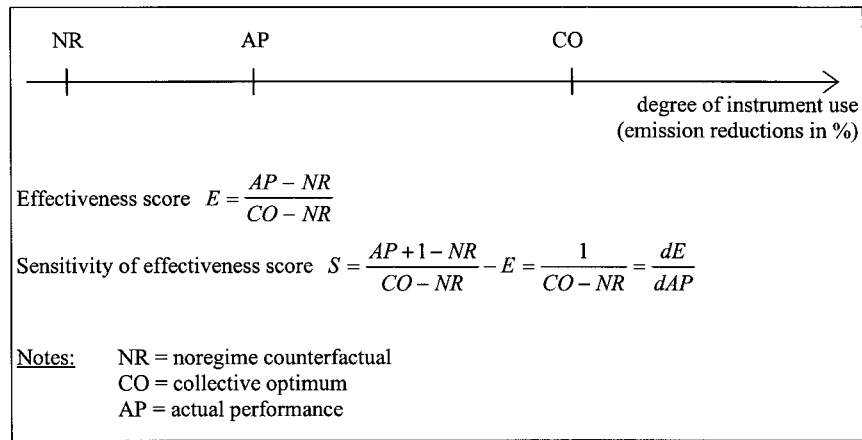


Figure 1: Measuring Regime Effectiveness (general concept)

researchers from different schools of international relations, the communication and comparison of results are facilitated. Finally, the effectiveness scores are easy to interpret in the applied context by policy makers.

A RATIONAL-CHOICE APPROACH TO MEASURING REGIME EFFECTIVENESS

Based on the general measurement concept for regime effectiveness, we will introduce a formal modeling approach to demonstrate how the no-regime counterfactual and the collective optimum can be determined. In line with the empirical example presented later, we develop the solution for transboundary environmental regimes; however, the extension to many other problems of international cooperation is fairly straightforward.

We perceive of states as self-interested actors, which choose their strategies in accordance with the principal goal of maximizing their individual pay-offs. Strategies are defined in terms of instrument use such as pollutive emissions or emission reductions, respectively. Pay-offs are measured as the difference between the (political) benefits and costs of emissions reductions. However, the goal-seeking behavior is effected by the strategic interdependency of the international system. This arises from the fact that national depositions (i.e., the total pollution leading to environmental damages in a country) do not only originate from one's own emissions but also from emission exports by other countries. As a consequence, national strategies have to simultaneously take national and foreign sources of environmental damages into account and optimize their national emission (reduction) policy accordingly.

This simple game-theoretic description of transboundary environmental problems leads to a straightforward interpretation of the no-regime counterfactual and the collective optimum. In particular, the no-regime counterfactual can be interpreted as the noncooperative solution of the transboundary pollution game that would follow from the uncoordinated choice of one's best reply to the strategies of the other countries (Nash equilibrium). In choosing their emissions levels, states would take only those emissions into account that are deposited in their own country and neglect the damaging effect of their exported emissions to other countries.

To state this more formally, we elaborate on the partial equilibrium approach as used by Mäler (1989) and others. The implicit assumption behind this is that environmental expenditures for the particular problem under consideration constitute only a small fraction of total expenditures so that we can abstract from income effects.⁴ In contrast to Mäler, damage cost functions are allowed to be nonlinear, and the idea of critical loads below which no damages occur is explicitly incorporated into the problem formulation.

Each country, indexed alternatively by i and j , follows the objective of minimizing its own "total (political) costs" of pollutive emissions:⁵

$$\min_{E_i} C_i(E_i) + p_i D_i(L_i). \quad (1)$$

$C_i(E_i)$ are the abatement costs of reducing emissions to the level E_i ; following standard assumptions, marginal costs of emission reductions increase with the level of abatement. Environmental damages $D_i(L_i)$ are assumed to increase exponentially—with b being the exponent—in the exceedance of critical loads L_i^* . This is calculated as the difference of depositions and the level of critical loads L_i^* , where depositions depend on emissions and the transboundary transport coefficients t_{ji} . The latter specify the share of emissions from country j that is deposited in i . Assuming that no damages occur if the exceedance of critical loads is reduced to zero, this can be expressed as

$$L_i = \sum_{j \in N} E_j t_{ji} - L_i^* \quad \text{and} \quad D_i(L_i) = \begin{cases} L_i^b & \text{if } L_i > 0 \\ 0 & \text{if } L_i \leq 0 \end{cases}, \quad (2)$$

with derivatives $\partial L_i / \partial E_i = t_{ii}$ and $\partial D_i / \partial L_i = bL_i^{b-1}$ for $L_i > 0$.

It remains to explain the term p_i in equation (1). Although governments are assumed to pursue policies in accordance with optimality criteria such as the equalization of marginal abatement and damage costs, they are dependent on domestic political pressure in choosing their policies. In particular, they are endogenous to proenvironmental political actors favoring strong emission reductions and pressure from industries worrying about abatement costs. Introducing this domestic political component (see Pastor and Wise 1994) into the measurement concept of regime effectiveness provides both a heuristic in determining the empirical values for the no-regime counterfactual

4. For a general equilibrium formulation that conceptualizes the degree of environmental protection as a public good, see Murdoch and Sandler (1997).

5. It is more convenient to use emissions rather than emission reductions as the choice variable because the latter would require specification of a baseline relative to which emissions are reduced.

and a bridge to the literature on multiple-level analysis (see Evans, Jacobson, and Putnam 1993; Iida 1993).⁶ In substantive terms, the influence of political pressure groups, which in turn depends on their political capabilities and issue salience, is represented by the weighing factor p_i that signifies the relative political preponderance of proenvironmental forces vis-à-vis opposing interests.⁷

If each country minimizes its national total cost of emissions, the optimality conditions are derived by differentiating the objective function (1) with respect to emissions. Using the chain rule, this yields (for $L_i > 0$)

$$-\partial C_i / \partial E_i = b p_i L_i^{b-1} t_{ii}. \quad (3)$$

This is the standard noncooperative Nash solution in which countries choose their optimal emission level such that their marginal abatement costs of emissions are equal to the corresponding marginal benefits of avoided damages in their own country.⁸ Furthermore, emission reductions are contingent on political pressure p_i and are zero once the assimilative capacity of a country is not exceeded ($L_i \leq 0$).

In contrast, the cooperative solution is obtained if each individual country i chooses its emission level E_i to minimize the joint total cost of pollutive emissions in all countries. Thus, the objective function becomes

$$\min_{E_i} \sum_{j \in N} [C_j(E_j) + p_j D_j(L_j)], \quad (4)$$

with the first-order conditions for optimality

$$-\partial C_i / \partial E_i = \sum_{j \in N} b p_j t_{ij} L_j^{b-1}. \quad (5)$$

The summation sign implies that in the cooperative solution, emission exports and their damages to other countries are fully taken into account, and each country reduces emissions until its marginal abatement costs are equal to the sum of marginal benefits of avoided environmental damages caused by those emissions in all countries. This can be interpreted as the collective optimum because it would be the optimal choice of the international community acting as a unitary actor. Obviously, it implies higher marginal abatement costs and accordingly higher emission reductions as compared to the no-regime counterfactual.

The function of a regime is to overcome the collective action problem, which follows from the transboundary character of emissions and to enable countries to enter into mutually beneficial agreements (Keohane 1984; Snidal 1986). The factors explaining a regime's degree of effectiveness are not explored in this article, and there-

6. The empirical results in Murdoch and Sandler (1997) underline the importance of the domestic political component. The overall fit of their model increases dramatically after a variable representing civil liberties and political freedom is added; indeed, without it, the estimated coefficients of their central explanatory variable are insignificant.

7. Note that we have normalized the political pressure by actors concerned about abatement cost to equal 1 so that p must be interpreted as the relative political pressure of environmental pressure groups. Thus, it can be compared with the role of relative prizes in economic models.

8. In the case of a linear damage cost function, $b = 1$, and equation (3) simplifies to $-\partial C_i / \partial E_i = p t_{ii}$.

fore we are not interested in how cooperation could be sustained as an equilibrium outcome. Yet, we argue that the noncooperative and the cooperative solutions are appropriate yardsticks to evaluate a regime's effectiveness. Indeed, they are related to each other in an elegant way. Once the no-regime counterfactual has been estimated, the collective optimum—which is in principle another counterfactual representing a perfect regime based on the same preference ordering that underlies the no-regime counterfactual—can be derived straightforwardly via theoretical reasoning. Thereby, consistency of the two evaluative criteria with each other is ensured.

By assessing the relative position of the actual performance between those two points, effectiveness scores can be derived according to the general measurement concept introduced above (see Figure 1). This can be undertaken for individual countries, yielding country-specific regime effectiveness scores, as well as for the aggregate of all countries, yielding the overall effectiveness of the transboundary regime. In contrast to the compliance literature, this approach includes the effects of international regimes on nonsignatory countries. Thereby, the selection effect between signatory and nonsignatory countries—with the latter expected to be less “compliant” than the former group—is avoided. Because the group of all countries affects the environmental quality of a biogeographical region, omission of nonsignatory countries may seriously bias research findings.

THE EFFECTIVENESS OF THE EUROPEAN REGIME FOR TRANSBOUNDARY AIR POLLUTION

Building on the derivation of the measurement concept of regime effectiveness for transboundary pollution problems in the previous section, we will demonstrate its empirical usefulness with the help of an example from the various policies to reduce transboundary air pollution in Europe during the 1980s and early 1990s.

Although localized air pollution problems have been known ever since early industrialization, transboundary air pollution problems have attracted public and scientific attention more recently. In the wake of hypotheses of damages to lakes, forests, buildings, and public health resulting from acidifying pollutants, such as sulfur dioxide (SO₂) and nitrogen oxides (NO_x), an international regime was formed during the late 1970s within the UNECE to regulate the emission of these pollutants. Besides acidification, problems related to eutrophication (oversupply of nutrients), tropospheric ozone episodes, heavy metals, and persistent organic pollutants have been regulated following the 1979 Convention on Long-Range Transboundary Air Pollution (LRTAP/UNECE 1979).⁹

Of particular interest are two international environmental agreements that permit an evaluation of past accomplishments—namely, the 1985 Helsinki Protocol to the

9. For a detailed introduction to the transboundary air pollution and its regulation in Europe, see Boehmer-Christiansen and Skea (1991), Levy (1993), and Sprinz and Vaahtoranta (1994).

1979 LRTAP convention (UNECE 1985) and the 1988 Sofia Protocol (UNECE 1988). The Helsinki Protocol requires that “parties shall reduce their national annual sulphur emissions or their transboundary fluxes by at least 30 per cent as soon as possible and at the latest by 1993, using 1980 levels as the basis for calculation of reductions” (Article 2). Similarly, the Sofia Protocol requires parties to reduce their national annual emissions of nitrogen oxides or their transboundary fluxes so that these, at the latest by December 31, 1994, do not exceed their 1987 levels (Article 2).

In addition, 11 countries have signed a declaration that obliges them to reduce their NO_x emissions in the order of 30% by 1998 in comparison to any base year chosen between 1980 and 1986. Among the European members of this international regime for transboundary air pollution, some countries did not sign the Helsinki Protocol, whereas the Sofia Protocol enjoys close to universal support.

DATA SOURCES

In the following, we will explain how the various components of the measurement concept for transboundary pollution problems have been operationalized to compute actual effectiveness scores for the two protocols. A systematic summary of the data sources can be found in Appendix A.

As temporal domains, we use the base and target years of the Helsinki Sulfur Protocol (1980 and 1993) and the Sofia NO_x Protocol (1987 and 1994). Data on emissions, depositions, critical loads, transport coefficients of transboundary emission flows, and marginal abatement costs are available from the Cooperative Programme for Monitoring and Evaluation of Air Pollutants in Europe (Barrett and Berge 1996) and the Regional Acidification Information and Simulation (RAINS) model developed at the International Institute for Applied Systems Analysis (Alcamo, Shaw, and Hordijk 1990).

However, neither data about the relative political preponderance of proenvironmental pressure groups nor cross-national damage cost estimates are available (Cough et al. 1994). This problem is quite common because the cause-effect chain of emissions and environmental impacts is often insufficiently understood, and the valuation of impacts by the society is difficult to assess, especially for intangible values such as biodiversity and impacts on human health (Johansson 1993). Therefore, we have solicited expert judgments to assess emission reductions of the no-regime counterfactual.

For SO₂, country teams in Finland, Germany, Hungary, Italy, the Netherlands, Norway, Spain, Sweden, Switzerland, and the United Kingdom have conducted standardized interviews with at least one senior policy expert from each of the following three groups: governmental organizations, environmental nongovernmental organizations (NGOs), and academia. To increase the reliability of results and to include all countries in the biogeographical region (see Table 1), the same questions have been presented to a long-standing expert in the field of the LRTAP regime. If the estimates of these two sources differed, the arithmetic average has been taken. For NO_x, only estimates of the long-standing expert were available, and those results should therefore be treated with more caution.

TABLE 1
The Effectiveness of the LRTAP Regime

<i>Countries</i>	<i>Results for SO₂ Emissions</i>															
	<i>AT</i>	<i>BE</i>	<i>BG</i>	<i>CS</i>	<i>DK</i>	<i>FI</i>	<i>FR</i>	<i>DE</i>	<i>GR</i>	<i>HU</i>	<i>IE</i>	<i>IT</i>	<i>NL</i>	<i>NO</i>	<i>PL</i>	<i>PT</i>
No-regime counterfactual (NR)	80	60	30	35	60	60	65	54	-30	32	20	38	63	70	30	-20
Actual performance (AP)	82	64	31	43	65	79	66	58	-28	54	29	61	67	74	34	-13
Collective optimum (CO)	80	76	73	72	69	64	80	67	49	67	48	51	77	70	54	-8
Effectiveness score (E)	1*	0.28	0.01	0.21	0.55	1*	0.08	0.33	0.03	0.62	0.33	1*	0.31	1*	0.15	0.59
Sensitivity of score (S)	ND	0.06	0.02	0.03	0.11	0.24	0.07	0.07	0.01	0.03	0.04	0.08	0.07	ND	0.04	0.08
<i>Countries</i>	<i>RO</i>	<i>ES</i>	<i>SE</i>	<i>CH</i>	<i>UK</i>	<i>YU</i>	<i>RU</i>	<i>UA</i>	<i>BY</i>	<i>All</i>						
No-regime counterfactual (NR)	65	23	76	55	29	0	45	40	40	41						
Actual performance (AP)	68	38	80	71	35	54	52	43	41	49						
Collective optimum (CO)	65	57	76	56	71	46	53	51	81	62						
Effectiveness score (E)	1*	0.44	1*	1*	0.14	1*	0.86	0.27	0.04	0.39						
Sensitivity of score (S)	ND	0.03	ND	0.92	0.02	0.02	0.13	0.09	0.02	0.05						

<i>Results for NO_x Emissions</i>																
<i>Countries</i>	<i>AT</i>	<i>BE</i>	<i>BG</i>	<i>CS</i>	<i>DK</i>	<i>FI</i>	<i>FR</i>	<i>DE</i>	<i>HU</i>	<i>IE</i>	<i>IT</i>	<i>NL</i>	<i>NO</i>	<i>PL</i>	<i>PT</i>	<i>RO</i>
No-regime counterfactual (NR)	20	-9	21	42	5	0	5	15	25	-7	-8	6	-8	25	-75	-20
Actual performance (AP)	24	-9	21	46	10	2	7	20	31	-6	-8	12	5	28	-74	-20
Collective optimum (CO)	20	10	22	42	19	15	17	25	27	15	-7	21	14	29	-45	31
Effectiveness score (E)	1*	0	0	1*	0.35	0.14	0.17	0.52	1*	0.05	0	0.40	0.60	0.69	0.03	0
Sensitivity of score (S)	ND	0.05	1.94	ND	0.07	0.07	0.08	0.10	0.44	0.05	1.41	0.07	0.05	0.23	0.03	0.02
<i>Countries</i>	<i>ES</i>	<i>SE</i>	<i>CH</i>	<i>UK</i>	<i>YU</i>	<i>RU</i>	<i>UA</i>	<i>BY</i>	<i>All</i>							
No-regime counterfactual (NR)	-38	3	19	11	8	18	43	18	11							
Actual performance (AP)	-38	10	20	13	8	25	48	23	14							
Collective optimum (CO)	-20	19	42	29	15	18	62	37	21							
Effectiveness score (E)	0	0.45	0.04	0.11	0	1*	0.26	0.26	0.31							
Sensitivity of score (S)	0.06	0.06	0.04	0.06	0.15	ND	0.05	0.05	0.10							

NOTE: LRTAP = Long-range transboundary air pollution. Country codes: Austria (AT), Belgium (BE), Bulgaria (BG), (former) Czechoslovakia (CS), Denmark (DK), Finland (FI), France (FR), Federal Republic of Germany (DE), Greece (GR) (due to missing data, only for SO₂), Hungary (HU), Republic of Ireland (IE), Italy (IT), the Netherlands (NL), Norway (NO), Poland (PL), Portugal (PT), Romania (RO), Spain (ES), Sweden (SE), Switzerland (CH), United Kingdom (UK), (former) Yugoslavia (YU), Russian Federation (European part) (RU), Ukraine (UA), and the Republic of Belarus (BY). Figures in the first three rows are reductions in percentage points for the periods 1980-1993 (Helsinki Sulfur Protocol) and 1987-1994 (Sofia NO_x Protocol). Negative emission reductions represent increases in emissions. The definition of the effectiveness score and its sensitivity can be found in Figure 1. Countries for which actual performance entails higher reductions than in the collective optimum have been assigned the score "1*," indicating that they have done more than would have been required in the optimal cooperative solution. If the no-regime counterfactual and the collective optimum are identical, calculation of the sensitivity of effectiveness score would require division by zero. This has been marked "ND" (not defined).

Interpreting these estimates for emission reductions in the no-regime counterfactual as the Nash equilibrium of the noncooperative game, they can be used to infer marginal abatement costs at the respective emission levels and in turn also the corresponding marginal damage costs because the two must be equalized in equilibrium (see equation (3)) (Mäler 1989; Helm 1998). The resultant marginal damage costs are interpreted as “the revealed preference of the governments and parliaments for reductions in emissions of sulphur” (Mäler 1991, 81). This concurs with the inclusion of political pressure groups into the countries’ pay-off functions as elaborated in the previous section.

To determine the cooperative solution, marginal damage costs for the complete domain of emission levels are needed. Due to the absence of reliable empirical estimates, we had to make assumptions concerning the shape of the damage cost function. One possibility is to assume that damage costs increase linearly in emissions (as in Mäler 1989). Although this considerably simplifies the analysis, it would imply that the choice of optimal reduction levels was independent of the associated changes in the state of the environment. This can easily be seen by setting the exponent b in equations (3) and (5) equal to 1 so that marginal damage costs would be constant. Therefore, we suggest a quadratic functional form of damage costs ($b = 2$), which has been widely used in the literature (see Baumol and Oates 1988; Barrett 1994). This specification makes each country’s emission reductions dependent on the emission reductions of the other countries: the larger the reduction of imported depositions, the lower the incentive to reduce one’s own emissions (see equation (5)). Therefore, equation (5) has been solved simultaneously for all countries included in the analysis (see Appendix B).

EMPIRICAL FINDINGS

By applying the calculi from the previous section to the data on transboundary air pollution regulations in Europe, we arrive at the measure of regime effectiveness (see Table 1). The aggregated effectiveness score is 0.39 for the SO₂ regime and 0.31 for the NO_x regime, as compared to a permissible range of [0, 1]. By contrast to the compliance literature, which would emphasize the high degree of covariation between legal obligations and the emission reductions accomplished among signatory countries, the aggregated regime effectiveness scores are substantively larger than zero in both pollutant domains but fall short of their theoretical maximum. It is also noteworthy that the overall regime effectiveness scores for both pollutants are of a similar order of magnitude.

Turning to the country-specific effectiveness scores, it has to be noted that some countries have reduced emissions beyond Pareto-optimal levels; these results have been marked by an effectiveness score of “1*” in Table 1. Most of these countries are characterized by relatively high emission imports (e.g., Austria, Norway, Sweden, Finland, and Switzerland for SO₂). Accordingly, the additional emission reductions that other countries undertake in the cooperative solution drives them substantially down along their marginal damage cost curve, thereby providing an incentive to reduce their own abatement efforts. At the same time, these countries are not major

TABLE 2
 Quadratic and Linear Damage Functions Compared (aggregate findings)

Functional Form	SO_2				NO_x			
	NR	AP	CO	E	NR	AP	CO	E
Linear	41	49	76	0.24	11	14	45	0.11
Quadratic	41	49	62	0.39	11	14	21	0.31

NOTE: For the definition of variables, see Table 1.

emission exporters themselves, so taking into account the damages they cause to others does not require substantial additional abatement efforts from them. Particularly if marginal abatement cost curves are relatively steep, this may easily lead to the initially counterintuitive result that emission reductions in the Nash equilibrium (and in the actual performance) are higher than in the cooperative solution.

A further reason why some of the environmentally more concerned countries have reduced emissions below Pareto-optimal levels is that it would have been politically difficult for them to demand substantial emission reductions by others while doing little themselves or even increasing emissions.¹⁰ This applies particularly to an environment where interests to “level the playing field” (i.e., avoid the adverse effects of environmental regulation on international competitiveness) figure prominently. These concerns reflect the fact that the Pareto-optimal solution is blind with respect to distributive issues, which are, however, important for states’ willingness to cooperate. For a highly effective regime, it may therefore be necessary to provide mechanisms other than an inefficient allocation of abatement measures to address distributional concerns—such as monetary side payments or the distribution of tradable emission rights. Such issues of regime design are not discussed in this article.

The indicator of the sensitivity of effectiveness scores shows that the results for most countries are quite robust, and (modest) measurement errors would not lead to disproportional changes in effectiveness scores. There are some notable exceptions such as Italy and Bulgaria for the NO_x regime. Overall, results for SO_2 (sensitivity score = 0.05) are considerably less sensitive to measurement errors than those for NO_x (sensitivity score = 0.10).

In addition to our previous remarks, the overall results depend on the specification of damage functions. Table 2 clearly shows that a linear (rather than a quadratic) shape of the damage cost functions would lead to substantively lower effectiveness scores, although they would still be larger than zero. They appear less realistic because changes in the exceedance of critical loads resulting from emission reductions would be neglected.

10. For this reason, we have assumed that in the collective optimum countries do not undertake less emission reductions than in the Nash equilibrium.

Furthermore, in the formulation of the pay-off functions, it has been assumed that the regime is exclusively driven by a concern for reducing environmental problem pressure. It is, however, likely that other issues have also played an important role in the LRTAP regime, such as the mentioned interests to level the playing field.

Finally, it should be noted that the RAINS abatement costs functions have been criticized especially for the Central and Eastern European countries. A study by Rentz et al. (1995), which, in contrast to the RAINS model, also takes into account the possibility of fuel switching, efficiency improvements, and energy conservation measures, finds considerably lower abatement costs. Therefore, the very high effectiveness scores for some Central and Eastern European countries should be regarded with caution because they may result partly from overestimated abatement costs. These countries underwent major economic transitions in the early 1990s, which have not been fully anticipated in the RAINS model.¹¹

This research concludes that the LRTAP regime had discernible effects on the aggregate behavior of countries for reducing sulfur and nitrogen dioxide emissions in the 1980s and early 1990s—with considerable variation across countries. For comparison with our results, only Gehring (1997) and Levy (1993) provide more detailed assessments of the effectiveness of the LRTAP regime. Gehring (1997, 59) concludes that the most pronounced effects of the LRTAP regime consist of (1) including the East Central European countries into a regulatory structure during the time of the cold war in Europe and (2) domestic political mobilization in some nonsignatory countries of the Helsinki Sulphur Protocol (especially the United Kingdom, Spain, and Poland). The second aspect also holds for countries whose nitrogen dioxide emissions are increasing (Gehring 1997, 60). Ultimately, Gehring's argument on effectiveness rests on measures of the degree of compliance with international obligations rather than a measure of regime effectiveness.

By contrast, Levy (1993, 115-27) uses qualitative counterfactual analysis to group countries according to the degree of regime effects. In comparing his results with those presented in Table 1, both studies agree that some countries show pronounced regime effects for sulfur emission reduction (e.g., the former Soviet Union and Denmark), but there is also considerable disagreement for other countries (e.g., Portugal and Spain). Some of these differences seem to stem from (1) the lack of a systematic counterfactual for all countries along the same dimension of instrument use and (2) the omission of a collective optimum in Levy's procedure. Only if the lower and upper bounds of the regime potential for each country are developed systematically do cross-nationally comparable results become feasible.

On the methodological side, the study by Underdal (1997) resembles most closely our approach by using an explicit numerical measurement technique. In his analysis of 15 regimes and a total of about 45 phases,¹² Underdal reports "highly preliminary findings"; on average, scores of 0.69 (on a scale ranging from 0 to 1) are achieved if a

11. It has to be noted that in the ex-post estimates of the no-regime counterfactual, the policy experts took the impact of deindustrialization in Central and East Europe after 1989 on emission reductions into account.

12. The regime phases constitute the unit of analysis for the statistical evaluation.

behavioral change concept is employed and 0.41 if progress toward technically optimal solutions is assessed. However, most of the scales appear truncated and lack symmetry,¹³ and the coding procedures do not clearly show how the (inter)calibration between regimes is accomplished.

CONCLUDING REMARKS

The question of whether regimes matter has been widely discussed in the international relations literature. This article provides a systematic tool to assess the effectiveness of international environmental institutions. By carefully deriving a no-regime counterfactual and a collective optimum, the performance of international institutions can be assessed. This measurement procedure offers two major advantages in comparison to conventional qualitative studies on effectiveness. First, the method and the underlying assumptions have been clearly described, thereby confining the room for hidden subjective judgments to a minimum. Second, the standardized method lends itself to the comparison of the effect of different regimes such as international river pollution, international transport of hazardous waste, and transboundary health problems.¹⁴ Eventually, the measurement concept might even be applied to problems outside the domain of environmental policies. Comparative research would not only probe the generalizability of the measurement concept but is also of particular use to public policy: it allows scarce resources to be allocated between less effective and more effective regimes.

The research presented in this article also makes a contribution to the debate between scholars working in the neorealist and neoliberal institutionalist traditions. Because neorealist scholars are particularly pessimistic about the effect of institutions and would therefore predict an effectiveness score close to 0, neoliberal institutionalists would ideally suggest an effectiveness score close to 1. As our findings suggest for the two cases under investigation, aggregate values ranging between 0.31 and 0.39 would be sufficiently far from the ideal positions of both schools of thought. These results are also broadly in conformity with some theoretical and empirical studies from the economics discipline (e.g., Barrett 1994; Murdoch, Sandler, and Sargent 1997), which assess the potential for cooperative agreements that improve substantially on the Nash equilibrium as modest.

Future research should systematically link the degree of regime effectiveness (on the aggregate and disaggregate levels) with factors explaining its variation across substantive issue areas and time. The perhaps best-known general explanation for regime effects are the three Cs put forward by Levy, Keohane, and Haas (1993)—namely, international regimes acting as

13. All degrees of negative change of regime effectiveness are captured by one scale value (0), which allows any positive values to dominate the assessment. Ideally, such a scale would be ranging from -1 to +1 to provide a symmetrical scale. In effect, the present scale takes negative change as the reference value and shows to which degree this can be overcome.

14. An extension of the measurement concept to global environmental problems, such as global climate change or ozone depletion, can be found in Sprinz and Helm (1999).

1. enhancers of governmental concern,
2. enhancers of the contractual environment for mutually profitable agreements, and
3. enhancers of national capacity to implement and comply with the rules of international regimes.

Although these perspectives point to major explanatory routes to be found in the empirical domain, it remains to be demonstrated in a more systematic and comparable form to which degree they matter. Young and Levy (1999, 3) take a cautious step in this direction, but they “do not claim to have produced a set of empirically-tested generalizations about the sources of regime effectiveness that are valid across a range of issue areas.”¹⁵ The most systematic approach to explaining regime effectiveness has been taken by Underdal (1997), who focuses on the (1) benignity of the (environmental) problem and (2) problem-solving capacity. In his findings, Underdal highlights the explanatory power of issue-specific power—understood as a form of entrepreneurial leadership—particularly in the case of malign problems.

Future research may beneficially combine the measurement concept for regime effectiveness advanced in this article with the explanatory factors elaborated above. This would improve our understanding by focusing on broader explanations of the different degrees of effectiveness across regimes (on the aggregate level) and the particular factors influencing country-level effectiveness (on the disaggregated level).

15. In particular, Young and Levy (1999, 4-5) refer to the behavioral pathways encompassing regimes as (1) utility maximizers, (2) enhancers of cooperation, (3) bestowers of authority, (4) learning facilitators, (5) role definers, and (6) agents of internal realignments.

APPENDIX A

Data Sources

All data sources are summarized in Table A1. In a few cases, further country-specific adjustments had to be made, mainly to take account of changes of territorial borders during the implementation period. A detailed description of those adjustments is available on request from the authors.

TABLE A1
Data Sources

<i>Variable</i>	<i>Data Source</i>	<i>Remarks</i>
Emissions E_t in base and target years	Barrett and Berge (1996, 33, 36)	
Depositions D_t in base years	Barrett and Seland (1995, Appendix E)	Barrett and Berge (1996) does not contain data on depositions
Conditional pentile (5%) critical loads L_t^* , that is, pentile of cumulative density functions of all critical loads within the country conditional on the level of oxidized nitrogen (oxidized sulfur) depositions in 1993, adjusted by country area	Calculated from aggregated pentile critical loads data (computed on the basis of depositions for 1990), which have been kindly provided by the Coordination Centre for Effects (CCE) at the RIVM in Bilthoven (Netherlands) (personal correspondence, January 16, 1997)	The method to transform aggregated pentile critical loads into conditional critical loads is described in Posch (1996). Because calculations are performed on a country (rather than grid) basis, the derivation of conditional critical loads requires the assumption that depositions are spread uniformly across the area of individual countries.
Country area	World Bank (1992)	Only the European part of the Russian Federation is included.
Transboundary transport coefficients t_{ij} averaged for the years 1985 to 1995	Calculated from deposition budget matrices for SO ₂ and NO _x in Barrett and Berge (1996)	
Emission reductions for the no-regime counterfactual (the individual answers of the anonymized interview partners are available on request from the authors; averaged answers are provided in Table 1)	SO ₂ : Country teams in Finland, Germany, Hungary, Italy, the Netherlands, Norway, Spain, Sweden, Switzerland, and the United Kingdom conducted standardized interviews with senior policy experts from each of the following groups: governmental organizations, environmental nongovernmental organizations, and academia (policy analysts). SO ₂ and NO ₂ : Separate interview with a long standing expert in the field of the long-range transboundary air pollution (LRTAP) regime.	Central question asked to experts: "Looking back at the time period y year (base year) to z year (target year) and all the changes (economic, political, etc.) that took place in this period, I would appreciate if you answered the following question: "Which $x\%$ reduction policy would the government of your country have unilaterally undertaken (i.e., actually accomplished) for the pollutant w during the time period y year until z year in the absence of the LRTAP regime?"
Marginal abatement costs of emission reductions relative to the projected pathway of energy use without abatement measures during the implementation period (1980-1993)	RAINS 6.1, Official Energy Pathway 03/1992 (Alcamo, Shaw, and Hordijk 1990)	Data for 1995 had to be taken as an approximation because no data for the target years 1993 and 1994 are available.

APPENDIX B

Calculation of Emission Reductions in Collective Optimum

The specification of marginal abatement costs as stepwise increasing rather than continuous functions in the Regional Acidification Information and Simulation model necessitates a modified procedure to solve equation (5) simultaneously for all countries. First, optimal cooperative emission reductions were calculated for the case of linear damage cost functions. Because this solution does not take into account the decrease of exceeding critical loads due to emission reductions, the resultant cooperative emission reductions are too high and can be regarded as an upper benchmark (see Table 2). Second, when these (maximum) cooperative emission reductions are used to solve the quadratic version of equation (5), the decrease of exceeding critical loads due to emission reductions of other countries is overestimated, resulting in too low emission reductions in one's own country. Therefore, these (minimum) cooperative emission reductions can be regarded as a lower benchmark. Third, the stepwise increasing marginal abatement costs functions within the interval between the lower and upper benchmark have been approximated by linearly increasing functions. The simultaneous equation system now contains only linear equations, which were solved using matrix algebra subject to the constraint that a country's cooperative emission reductions are at least as high as in the noncooperative solution.

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