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Reihe Politikwissenschaft

Political Science Series

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October 2011

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Founded in 1963 by two prominent Austrians living in exile – the sociologist Paul F. Lazarsfeld and the economist Oskar Morgenstern – with the financial support from the Ford Foundation, the Austrian Federal Ministry of Education, and the City of Vienna, the Institute for Advanced Studies (IHS) is the first institution for postgraduate education and research in economics and the social sciences in Austria. The **Political Science Series** presents research done at the Department of Political Science and aims to share "work in progress" before formal publication. It includes papers by the Department's teaching and research staff, visiting professors, graduate students, visiting fellows, and invited participants in seminars, workshops, and conferences. As usual, authors bear full responsibility for the content of their contributions.

Das Institut für Höhere Studien (IHS) wurde im Jahr 1963 von zwei prominenten Exilösterreichern – dem Soziologen Paul F. Lazarsfeld und dem Ökonomen Oskar Morgenstern – mit Hilfe der Ford-Stiftung, des Österreichischen Bundesministeriums für Unterricht und der Stadt Wien gegründet und ist somit die erste nachuniversitäre Lehr- und Forschungsstätte für die Sozial- und Wirtschaftswissenschaften in Österreich. Die **Reihe Politikwissenschaft** bietet Einblick in die Forschungsarbeit der Abteilung für Politikwissenschaft und verfolgt das Ziel, abteilungsinterne Diskussionsbeiträge einer breiteren fachinternen Öffentlichkeit zugänglich zu machen. Die inhaltliche Verantwortung für die veröffentlichten Beiträge liegt bei den Autoren und Autorinnen. Gastbeiträge werden als solche gekennzeichnet.

Abstract

The study of policy change has been receiving increasing scholarly attention. Despite the growing number of empirical studies on policy change, the definition and measurement of the concept has made limited progress. In comparative environmental policy research, for instance, most existing large n studies rely on impact data such as pollutant emissions to approximate processes of policy change, often without discussing the conceptual implications of this measurement approach. Against this background, this article proposes a new measurement concept for empirically assessing environmental policy change, which conceives of policy change in terms of changes in policy outputs. We illustrate our measurement concept on the basis of an original dataset covering the evolution of clean air policies in 24 advanced democracies over a period of almost three decades (1976-2003). In a second step, we evaluate the relationship between our measurement of environmental policy change and standard emission data representing the most widely used proxy in the literature. Our findings suggest that clean air policies cannot be consistently associated with emission levels, therefore calling into question the viability of environmental impact data for the study of the determinants of policy change.

Keywords

Policy change, policy outputs, environmental policy

General note on content

The opinions expressed in this paper are those of the author and not necessarily those of the IHS.

Contents

I. Introduction

The analysis and explanation of what governments decide to do (or not to do) has always been at the heart of political science. This holds particularly true for the sub-discipline of public policy, given its focus on policy dynamics and the conditions facilitating or constraining. It is hence hardly surprising that there is an on-going debate about how to conceptualize and explain policy change (Baumgartner & Jones 1991; Radaelli 1997; Richardson 2000; John 2003; Capano & Howlett 2009; Howlett & Cashore 2009; Baumgartner et al. 2009; Howlett & Joshi-Koop 2011; Jacob & Jörgens 2011). At the same time, however, the predominant conceptual and theoretical focus on policy change comes along with limited attention paid to the question of how to empirically test the – partly quite complex – theoretical propositions. This is not to say that there is a lack of empirical assessments of policy change. However, empirical findings are rarely systematically interpreted in the light of existing theoretical and conceptual approaches. One of the most important deficits in this regard refers to the lacking critical reflection of the theoretical consequences that arise from the manner in which policy change is empirically assessed. This mismatch between theory and data used for their empirical test has been identified as an impediment to scientific progress in this field of inquiry (Howlett & Cashore 2009).

These problems are particularly pronounced in the analysis of policy change in the environmental field. Especially large-n studies on environmental policy change are characterized by a discrepancy between the way in which they measure policy change and the factors they employ to provide for a theoretically informed explanation. While the theoretically-derived causes of policy change (e.g. the number of veto players, the policy positions of the government and legislature, problem pressure, or socio-economic conditions) are expected to affect governmental action and hence changes in environmental regulations, the measurement of change often relies on environmental impacts, for example, pollutant emissions or pollution levels. The measurement approach is typically inspired by the unavailability of direct measurements of governmental action. In short, theories accounting for change in policy outputs are tested by using data on policy impacts. It is either implicitly assumed that impact data constitute a valid proxy for policy outputs or that the theoretical causes of policy output change and policy impact change are more or less the same. This approach, however, entails the risk of producing misleading findings as policy impacts are usually affected by a plethora of confounding factors.

It is the objective of this article to scrutinize this discrepancy between the measurement and the theoretical explanation of environmental policy change. To this end, we first develop an alternative approach to measuring environmental policy change, which differs from existing concepts insofar as it assesses policy change in terms of alterations in public law-making, i.e. policy outputs. Second, we test whether our measurement concept can be related to changes in environmental impacts as this represents a precondition for using the latter as a proxy for changes in environmental policy outputs. Only if there is a robust causal relationship between the two measurements of environmental policy change, we can safely state that the proxies based on environmental impacts indeed provide a viable substitute for a direct measurement employing legislative outputs.

Empirically, we focus on clean air policy, a prominent subfield in environmental policy research. In so doing, we use a new dataset compiled from changes in clean air regulations in 24 OECD countries over almost three decades[1]. Our results show that changes in clean air regulation cannot be systematically related to changes in pollutant emissions, hence questioning the validity of impact data as a proxy for testing theories of change in environmental policy outputs. More generally, our findings underline that more research efforts should be made in the collection of policy output data in order to advance our theoretical understanding of environmental policy change.

This article proceeds as follows. We first discuss existing deficits pertaining to the analysis and measurement of environmental policy change. Subsequently, we introduce our alternative measurement concept of policy change. We finally proceed to the empirical analysis of clean air policy and provide a critical reflection of the results.

[1] The data were collected in the context of the collaborative project CONSENSUS (confronting Social and Environmental Sustainability with Economic Pressure). The project has been financed within the 7th Framework Program of the European Commission. Generous research funding is gratefully acknowledged. For further details seehttp://www.polver.uni-konstanz.de/knill/forschung-projekte/confronting-social-and-environmental-sustainability-with-economic-pressure-balancing-trade-offs-by-policy-dismantling-consensus/team/.

II. Measuring environmental policy change: the dominance of impact data

Many large-n comparative studies in environmental policy research have used changes in environmental impacts as proxies of environmental policy change (Young et al. 2008). The most frequently applied indicators are levels of environmental quality or aggregate data on pollutant emissions and degradation levels of certain environmental media, e.g. deforestation rates (see, e.g., Crepaz 1995; Murdoch 1997; Midlarsky 1998; Neumayer 2003; Wälti 2004; Van & Azomahou 2007; Perkins & Neumayer 2008). However, this choice is rarely theoretically motivated, but rather the result of data availability since impact data are regularly compiled and published by international organizations like the Organization for Economic Development and Co-operation (OECD) or the World Bank.

What are the consequences of using impact data as indicators of policy change? Basically, there is a potential validity problem because intervening variables cannot sufficiently be controlled for. Even though there should, in principle, be a connection between the actual decisions taken by governments (i.e. policy outputs) and changes in environmental quality (i.e. policy impacts), this relationship might be influenced by a multitude of additional variables (see, e.g. Neumayer 2002). It is thus a demanding task to extract the net effect of governmental decision on changes in environmental quality by using control variables. The level of carbon dioxide emissions in a country, for instance, may not only depend on economic up- and downturns – a popular control variable – but also on a range of additional structural variables, such as investments in energy efficiency, shifts to more or less energy-intensive final goods, or the use of different fossil fuels and renewable energies (see, e.g., Aubourg et al. 2008).

The same objection with regard to confounding factors can be made against environmental performance indices that are sometimes used in large-n studies (see, e.g., Jahn 1998; Scruggs 1999, 2002; Esty & Porter 2005; Roller 2005). Although constructing indices is a means to overcome problems emerging from a too narrow conception of the dependent variable, it does not necessarily result in a valid measurement of policy change. Rather, the number of confounding factors may rise with the number of environmental impact indicators included in the index, hence further aggravating the ‘dependent variable problem’ (Howlett & Cashore 2009) in the study of environmental policy change.

The above discussion has shown that environmental impacts are quite distant proxies of governmental decisions. For analysts interested in examining and explaining changes in environmental policy-making, the use of impact data therefore poses serious problems of validity, which are only rarely reflected in the literature (for notable exceptions, see Neumayer 2002; Andonova et al 2007).

A further and also hardly acknowledged problem with the use of impact data in comparative environmental policy research refers to their limited reliability. Usually, the generation of this data is based on national reporting whose measurement procedures and data quality are largely unknown to social scientists. For example, there are no common principles for the production of emission data from industrial sites at the international level (Saarinen 2003; see also Styles et al. 2009). Moreover, these measurement procedures are frequently subject to changes that go unnoticed.

A final problem with environmental impact data relates to time-lags between governmental action and potential policy effects. It is hardly impossible to exactly determine how much time has to pass until, for instance, new emission standards result in lower or higher levels of pollution. Even though time-series analyses often use lagged explanatory variables, usually by one year, the exact quantification of these intervals rests constitutes an arbitrary choice rather than resting on a sound scientific calculation.

In the light of these potential problems, it is striking that many environmental policy studies, in particular macro-quantitative ones, use impact data rather uncritically to order to examine theories that refer to policy output change and the behaviour of political actors. However, as long as systematic reflections on restrictions regarding the measurement of the dependent variable are not made explicit, it is difficult to assess the explanatory power, comparability, and reliability of different research results.

III. Alternative concepts of policy change

Which alternatives exist to more convincingly approach policy change conceptually and methodologically? What are the advantages and disadvantages? This section tackles these questions. The conceptual and analytical ideas that are developed here are based on three considerations. Firstly, policy change should be measured more directly on the basis of policy outputs, i.e. governmental regulatory activity. Secondly, policy output measures should be generally conceptualized to capture events of policy change in a more detailed manner and to avoid focussing on single policies or policy instruments. Thirdly, concepts of policy change should allow for accurately assessing developments in both directions, i.e. policy expansion and dismantling.

III.1 Environmental policy change as change in policy output

In contrast to impact data, the assessment of policy outputs allows for the detailed and multidimensional assessment of political decisions and their changes over time. Policy outputs can, for instance, refer to basic principles or paradigms of political programmes, the chosen policy instruments, or the concrete settings of these instruments (Hall 1993). Environmental impact data, by contrast, cannot capture such complex structures of policy change. For example, emission data do not entail information about how, i.e. by means of which policy instruments or their precise calibrations, certain pollution levels have been achieved (Jordan et al. 2005). This is a serious limitation because the choice of an environmental policy instrument as such already represents a crucial political decision and significant trade-offs can arise in the choice of instruments (Goulder & Parry 2008).

Comprehensive assessments of changes in different dimensions of environmental policy output are predominantly provided by small-n studies (see, e.g., Hoberg 1991; Urwin & Jordan 2008; Kochtcheeva 2009; Mazmanian & Kraft 2009). In contrast, the few existing large-n studies analysing environmental policy outputs either concentrate on the diffusion of certain policy innovations, e.g. environmental impact assessments (Hironaka 2002) or new types of policy instruments (Tews et al. 2003), or on concrete regulatory settings, e.g. the maximum allowed lead content in gasoline (Fredriksson et al. 2005). By focussing on single policies or policy dimensions, however, these studies adopt a selective perspective on policy change which might lead to biased conclusions regarding the degree of change (see also Meseguer & Gilardi 2009). To date, there are hardly any studies of environmental policy change that equally deal with different environmental policies, instruments, and their concrete settings (for an exception, see Holzinger et al. 2008a, 2008b).

Closely related to the dominant focus on environmental innovations is the fact that change is typically defined as a departure from the status quo without considering the direction of change. This way, it is neglected that change is not unidirectional, but can go into two directions, i.e. expansion and dismantling. For example, the introduction of environmental taxes can be interpreted as expansion, while their abolishment would imply dismantling (Knill et al. 2009). Moreover, expansion and dismantling activities can differ across the dimensions under study. For instance, it is possible that states introduce a variety of new policy instruments, while, at the same time, lowering regulatory levels of existing instruments, e.g. the strictness of emission standards.

How can we capture policy change in both its complexity and its innate direction on the basis of policy outputs? In the following, we propose a measurement concept based on a partially modified version of Peter Hall's typology of policy change (Hall 1993). On a first, very basic level it is analyzed whether a policy for specific target, e.g. the quality of drinking water, is in place. The second category relates to the instruments used to realize the respective policy goals, e.g. emission standards or environmental taxes. The third, most narrowly specified, category of policy change corresponds to the concrete setting or calibration of the applied instruments, including the level of, for instance, emission limits, and their scope of application, that is, the individuals, organizations, or activities targeted by a specific instrument.

This way, our concept goes beyond the typology put forward by Hall as it additionally includes the scope-dimension. Moreover, the way in which we apply the measurement concept allows for a more nuanced empirical assessment of environmental policy change, since in contrast to Hall we do not limit ourselves to only pointing out instances of major or minor policy change, but characterize each event in the most comprehensive manner. Table 1 illustrates how policy expansion and policy dismantling can be measured along these three categories.

Table 1: The measurement of policy expansion and dismantling.

Category	**Policy Expansion**	**Policy Dismantling**
Policy presence	Introduction or addition of a new policy	Dismantling of an existing policy
Policy instruments	The number of policy instruments increases, e.g. information-based instruments are adopted	The number of instruments decreases, e.g. market-based instruments are abolished
Policy calibration: Instrument levels and scopes	Tightening regulatory standards or increasing the target group, e.g. by lowering emission limits	Loosening regulatory standards or decreasing the target group, e.g. by increasing emission limits

Following these categories, it is possible to give a detailed assessment of policy change. In order to avoid a selective perspective on policy change, we advocate assessing policy change in terms aggregate developments comprising all changes in state activity within a policy field. This perspective can be realized by classifying and counting all events of change according to the aforementioned categories.

Moreover, we are able to identify the direction of each event of change. For example, expansion in terms of policy presence takes place if a new pollutant becomes subject to regulatory activity. Dismantling, by contrast, would occur if a given pollutant is not regulated anymore. The number of policy instruments increases if a new measure to curb emissions of a certain pollutant, e.g. a tax, is introduced and decreases if a measure is abolished. Policy-makers can also increase or decrease the concrete calibration of policy instruments. For example, the specific levels of a tax as well as the target group of a tax can be either increased or decreased. We count the first case as an event of expansion and the latter as an event of dismantling.

III.2 Density and intensity of policy change

Even when relying on the analytical categories of policy presence, policy instruments, and policy calibration, we still have to clarify how to aggregate the magnitude and direction of changes in a given policy field. When do we speak of policy expansion and when of policy dismantling? How do we assess the degree of potential changes in one or the other direction?

For this purpose, we distinguish between two basic dimensions, namely ‘policy density’ and ‘policy intensity’. The dimension of policy density describes indicates the degree of legislative penetration and internal differentiation of a policy field. It hence explores the number of policies or instruments used within a policy field, and how this number changes over time. In a complementary vein, policy intensity aims at measuring the stringency of the adopted measures. This second dimension includes regulatory standards, such as emission limits, but also the scope of application of these regulations, i.e. those affected, such as specific industry branches.

The concepts of policy density and policy intensity thus complement one another in their assessment of policy change. In this regard, it is important to note that a densely regulated policy field does not automatically require that the respective legal provisions are very strict or far-reaching and vice versa. Nevertheless, depending on the maturity of the policy field, changes in policy density may be accompanied by changes in intensity. A development along this pattern is likely for policy fields in their early stages of development. In such a constellation, any new legislation will not only increase the density of the field, but might also increase policy intensity, assuming that the status quo was characterized by the absence of any governmental activities with regard to the policy item in question.

For more established policy fields, by contrast, this linkage is less likely. It is rather conceivable that increases in density are accompanied by decreases in intensity and vice versa. Such trade-offs might occur, for instance, if a country is obliged by international law to introduce new legislation and seeks to compensate negatively affected domestic actors (e.g. private companies) by reducing other regulatory burdens in the policy area.

Table 2 summarizes the proposed dimensions and sub-dimensions of policy change and attaches a set of indicators to them. The latter are only broadly introduced at this point, as they have to be operationalized more specifically in the relevant research context. In section four, we introduce the specific operationalization for the example of clean air policy.

Table 2: Dimensions and indicators of policy change

Dimension		**Indicators**
Policy density	Policy target density	Development of policies over time (Difference between number of adopted and abolished policies)
	Policy instrument density	Development of instruments over time (Difference between number of adopted and abolished instruments)
Policy intensity	Intensity level	Development of policy instrument strictness over time (Difference between number and/ or degree measures with increasing and decreasing effects)
	Intensity scope	Development of personal scope / substantive scope / temporal scope of a policy instrument over time (Difference between number and/ or degree measures with increasing and decreasing effects)

III.2.1 Conceptualizing policy change as change in policy density

Broadly speaking, an increase in policy density points towards policy expansion, whereas a decrease in policy density can be interpreted as policy dismantling. Changes in policy density can be measured via two indicators – the number of policy targets and the number of policy instruments within a policy field. It generally holds that the larger the number of policy targets, the higher is the regulatory penetration of a policy field. For example, in order to reduce air pollution, governments can define a plethora of regulatory goals. On the one hand, they can regulate the amount of pollutant emissions. The corresponding policy targets would then refer to the regulation of emissions of different substances. On the other hand, governments can also adopt policies that define specific air quality goals with regard to various substances, e.g. particulate matter (see, e.g. Jordan et al. 2010). If the number of regulatory issues, i.e. policy targets or items, in a policy field increases, it indicates policy expansion, whereas policy dismantling becomes manifest in a decrease of the number of policy targets.

The second indicator for measuring policy density is determined by the number of policy instruments in a given policy field. The number of policy instruments measures policy density on a more concrete level than the number of policy targets. A change in the number of instruments, however, does not necessarily coincide with a change in the number of policy targets. To reduce the amount of industrial carbon dioxide (CO2) emissions into the air, for instance, governments can rely upon a broad array and combination of instruments, including command-and-control approaches (the definition of legally-binding emission standards), economic incentives (such as environmental taxes or emission trading systems), or industrial self-regulation and voluntary agreements (see, e.g., Sterner 2002). Even if the number of policy targets in a given policy area remains constant over time, the number of policy instruments can hence increase or decrease.

In sum, expansion in the dimension of policy density is measured by any increase in the number of policies and instruments, whereas any decrease means policy dismantling. In other words, we measure the extent to which policy density expansion occurs by the addition of new policies or policy instruments and policy density dismantling by the abolishment of existing ones.

III.2.2 Conceptualizing policy change as change in policy intensity

The dimension of policy intensity captures changes in the stringency of governmental intervention in a policy field. An increase in intensity over time can accordingly be understood as policy expansion, whereas a decrease indicates policy dismantling. We measure changes in policy intensity by two indicators: the intensity level and the intensity scope. First, changes in policy intensity rest upon potential increases or reductions of regulatory standards such as the concrete level of permissible emissions. Second, changes in policy intensity refer to the scope of application of policy instruments. The intensity scope increases in conjunction with the number of cases, constellations, or addressees covered by a certain policy instrument. For instance, the scope of an emission standard regulating certain emissions from combustion plants increases once the threshold defining the size of plants covered by the regulation is lowered. In this case, more companies would be covered by the emission standard and would therefore have to comply with the respective legal obligations.

The dimensions of policy density and intensity are useful for studying aggregate changes in policy output with considerable attention to detail. In particular, they allow examining regulatory changes over entire policy fields. They are, naturally, less useful if the researcher is interested in studying regulatory shifts with respect to particular types of policy instruments, for example a shift from command-and-control instruments to economic incentives.

IV. Operationalizing changes in policy density and intensity: the case of clean air policy

How can the proposed measurement concept be applied in practical research? This section answers this question by demonstrating a specific operationalization of changes in policy density and intensity based on policy outputs for the case of clean air policy. Our operationalization is guided by the main purpose of testing whether environmental policy outputs can be empirically related to environmental impacts. As discussed, most studies in the field that employ theories of policy change to explain environmental impacts assume that there is a direct relationship between them (see, e.g., Wälti 2004; Bernauer & Koubi 2009). But does this assumption bear an empirical test based on our measurement of policy change?

For this purpose, the operationalization should, above all, allow for an encompassing measurement and comparison of developments in clean air policy-making across countries and over time. We measure clean air policy change as an aggregate count of all events of change in policy output coded according to the proposed measurement concept. In total, we study changes in clean air policy outputs in 24 OECD countries from 1976 to 2003 which have been compiled by the CONSENSUS project[2].

To begin with, table 3 lists all policy targets considered to comprehensively assess changes in clean air policy. Overall, twenty policy targets are examined that refer to both air quality and pollutant emissions as well as product standards. If one of the listed items becomes subject to regulatory intervention, we count it as one event of policy expansion. Conversely, if one of the items is not regulated anymore, we count this as an event of policy dismantling.

[2] For more detailed information visit http://www.polver.uni-konstanz.de/knill/forschung-projekte/confronting-social-and-environmental-sustainability-with-economic-pressure-balancing-trade-offs-by-policy-dismantling-consensus/team/.

Table 3: Clean air policy targets

Air quality	
1.	Air quality standards for nitrogen oxides (NO_x)
2.	Air quality standards for sulphur dioxide (SO_2)
3.	Air quality standard for carbon monoxide (CO)
4.	Air quality standard for particulate matter
5.	Air quality standard for ozone (O_3)
6.	Air quality standard for lead
Emissions (from stationary or mobile sources, product standards)	
7.	Nitrogen oxide (NO_x) emissions from large combustion plants using coal
8.	Nitrogen oxide (NO_x) emissions from passenger vehicles using unleaded gasoline
9.	Nitrogen oxide (NO_x) emissions from heavy duty vehicles using diesel
10.	Sulphur dioxide (SO_2) emissions from large combustion plants using coal
11.	Sulphur dioxide (SO_2) emissions from passenger vehicles using unleaded gasoline
12.	Sulphur dioxide (SO_2) emissions from heavy duty vehicles using diesel
13.	Carbon dioxide (CO_2) emissions from large combustion plants using coal
14.	Carbon dioxide (CO_2) emissions from passenger vehicles using unleaded gasoline
15.	Carbon monoxide (CO) emissions from large combustion using coal
16.	Carbon monoxide (CO) emissions from passenger vehicles using unleaded gasoline
17.	Particulate matter emissions from large combustion plants using coal
18.	Arsenic emissions from stationary sources
19.	Maximum permissible limit for the lead content of gasoline
20.	Maximum permissible limit for the sulphur content of diesel

Second, we consider every instrument that is used to achieve the underlying regulatory objective of each of the specified policy targets. The types of instruments examined are listed in table 4. Note that there can be several instruments of the same type, e.g. several technological prescriptions, in place to regulate a given policy target. Again, the introduction of a new instrument counts as an event of policy expansion whereas the abolishment of an existing instrument counts as an event of dismantling.

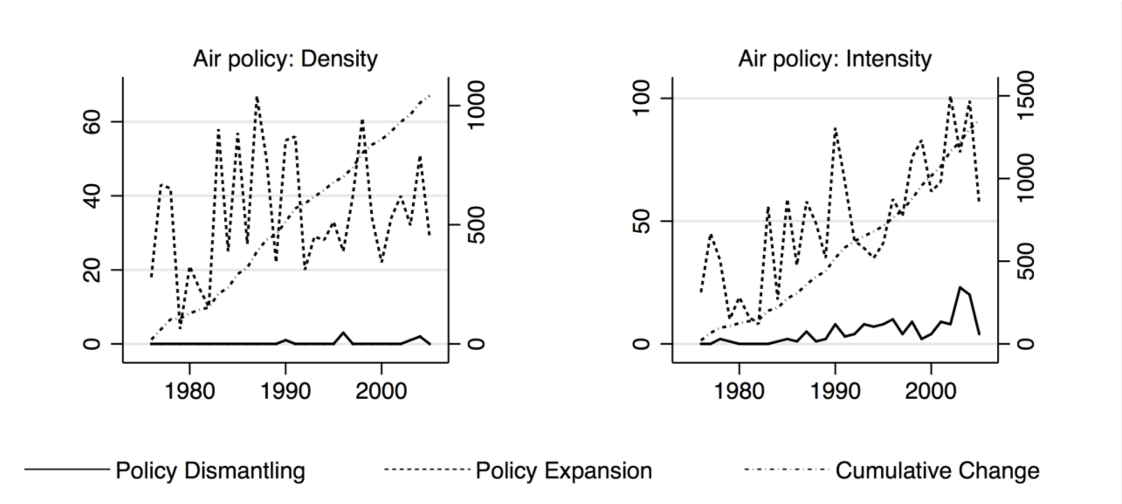
Table 4: Environmental policy instruments

Instrument	**Description**
Obligatory standard	A legally enforceable numerical standard, typically involving a measurement unit, e.g. mg/l
Prohibition / ban	A total or partial prohibition/ban on certain emissions, activities, products etc.
Technological prescription	A measure prescribing the use of a specific technology or process
Tax / levy	A tax or levy for a polluting product or activity
Subsidy / tax reduction	A measure by which the state grants a financial advantage to a certain product or activity
Liability scheme	A measure that allocates the costs of environmental damage to those who have caused the damage
Planning instrument	A measure defining areas or times that deserve particular protection
Public investment	A specific public investment
Data collection/ monitoring programmes	A specific programme for collecting data
Information-based instrument	Exchange of information between the state and polluters or among polluters
Voluntary instrument	Voluntary agreements or commitments between the state and private actors or among private actors
Permits	A permit to pollute the environment or to produce/import/export/sell environmentally harmful products

Finally, we count every change in the strictness and scope of an instrument as an event of either policy expansion or dismantling. Another possibility would have been to precisely quantify the changes in level and scope. We do not pursue this approach, however, because it is very difficult to make these changes comparable across policy instrument and countries over time. For example, we cannot readily compare the levels of taxes with those of emission standards. Another problem is represented by the fact that some jurisdictions prefer defining specific standards for individual industry sectors whilst others prefer to define universal standards that are uniformly valid for all industry sectors. We thus deem it conceptually more appropriate to adhere to an aggregate count of all events of change with expansive and dismantling effects. From this it follows that the final measure of aggregate policy change is composed of the difference between all expansive and dismantling events of change (see table 1). Density change consists of all changes in policy targets and instruments and intensity change is measured in terms of changes in levels and scopes.

Figure 1 illustrates clean air policy changes aggregated over all countries in our sample. The dashed line denotes the count of all events of policy expansion, whereas the solid line gives the count of dismantling events. The continuously rising graphs represent our principal measures of clean air policy density and intensity, i.e. cumulative change (expansion minus dismantling) over time. Dismantling activities in clean air policy-making do indeed exist for our country sample but are overall rather the exception. This leads to a continuous increase in overall policy density and intensity, i.e. policy expansion discounted of dismantling. Moreover, both expansive and dismantling activities in terms of clean air policy intensity are more frequent than density changes (see, Hall 1993). They also increase towards the end of our observational period, suggesting that policy-makers focus increasingly on the intensity dimension as policy fields grow more mature.

Figure 1: Clean air policy change in 24 OECD countries.



V. Explaining air pollutant emissions by clean air policies

This section finally examines whether environmental policy output can help to explain environmental impacts. In fact, this is the implicit, and hardly tested, assumption made by studies that use theories of policy change to explain changes in environmental impacts. Empirically, we focus on clean air policy, and in particular whether clean air policies in terms of output can be related to emission levels of pollutants.

V.1 Introducing the variables

Table 5: Descriptive statistics

Variable	Description	Mean	Std. Dev.	Min.	Max.
CO_2	CO_2 emission intensity (grams of emission per unit GDP, log)	17.839	.415	16.691	18.600
SO_2	SO_2 emission intensity (grams of emission per unit GDP, log)	12.369	.999	9.631	13.947
NO_X	NO_X emission intensity (grams of emission per unit GDP, log)	12.243	.611	10.564	13.913
Policy density	Clean air policy density	19.875	18.672	0	88
Policy intensity	Clean air policy intensity	21.590	23.019	0	114
GDP pc	GDP per capita (2000 US$, log)	9.627	.595	7.794	10.559
GDP growth	Annual GDP growth	2.930	2.599	-6.854	11.494
Population density	Population density (people per sqkm of land area)	133.978	132.285	1.826	494.411
Urban population	Urban population (% of total population)	71.802	11.921	41.2	97.22
Industry	Industrial added value (% of total GDP)	31.768	4.867	19.034	42.623
Trade intensity	Imports + exports (% of GDP)	60.607	29.224	9.102	184.742
Manufactures exports	Manufactures exports (% of merchandise exports)	68.321	21.785	9.361	96.558
Manufactures imports	Manufactures imports (% of merchandise imports)	68.932	12.437	18.315	87.438
FDI inflow	Foreign direct investment net inflows (% of GDP)	2.017	5.073	-.663	92.498

Our dependent variables are national emission intensities, i.e. emissions in grams per unit of GDP (see, Cao & Prakash 2010), of three frequently used air pollutants, namely CO_2, sulphur dioxide (SO_2) and nitrogen oxide (NO_X). The emission data were gathered from the Emissions Database for Global Atmospheric Research (EDGAR)[3]. A summary of all key variables is presented in table 5.

To begin with, our core explanatory variables refer to the effect of regulatory activity measured in terms of clean air policy density (*Policy density*) and intensity (*Policy intensity*) as introduced earlier. In general, we expect emission intensities to decrease with more dense and more intense clean air regulations and vice versa. Since the two dimensions are highly correlated for clean air policy, we build separate models around them.

We test the effect of clean air regulations on emission intensities against several control variables. First, following the reasoning of the so-called environmental Kuznets curve, demand and supply of environmental policies are expected increase at higher income levels, which might finally also lead to lower emissions (see, e.g., Selden & Song 1994; Grossman & Krueger 1995; Aubourg et al. 2008). Moreover, economic downturns can be responsible for reductions in pollution loads (Hughes & Lovei 1999). Contrary to this expectation, however, economic upswings can also boost investment in more advanced and environmentally friendly technologies. We control for these factors by including the natural log of GDP per capita (*GDP pc*) and the growth rate of GDP in our models (*GDP growth*). Apart from these general economic conditions, the structural composition of national economies should have an impact on emission levels. In particular, the industrial sector can be expected to contribute above average to overall pollution (see, e.g. Earnhart & Lizal 2008). We therefore control for the size of the industrial sector via its contribution to total GDP (*Industry*).

Another set of controls refers to the effects of international trade. First, we control for differences in overall *trade intensity*, measured as exports plus import as a percentage of GDP. Higher levels of trade can be assumed to either exert downward pressures on environmental standards (Prakash & Potoski 2006) or induce convergence towards higher levels of environmental standards (Vogel 1995). We also use two more specific controls that capture effects emanating from differential patterns of international trade, namely the share of *manufactures exports* and *manufactures imports*. Since the production of manufactured goods is known to be particularly pollution intense, the standard theory of regulatory competition would predict laxer regulation and hence higher emission intensities, particularly for countries that rely more heavily on manufacture exports. The alternative scenario, however, is that increased competition from manufacture imports and exports has positive effects if domestic firms move towards more environmentally efficient product and production technologies (Perkins & Neumayer 2008). By the same token, beneficial technology

[3] Available at: http://edgar.jrc.ec.europa.eu/

spillovers can be expected at higher levels of foreign direct investment (FDI) inflows. We therefore also include net *FDI inflow* in our model.

Finally, in accordance with previous studies, we control for *population density* and *urban population* in order to rule out confounding effects related to demographics. All independent variables, except for our output measures, are derived from the World Bank's World Development Indicators.

V.2 Discussion of main findings

We estimate the relationship between emission intensities and the discussed explanatory variables by means of standard panel analysis techniques. As customary in the literature, all explanatory variables are lagged one year. We include country fixed-effects in our analyses because we are interested in the effects of changes in environmental policy output on environmental impacts rather than in the effects of country-differences in regulatory levels. This way, we are also able to control for unobserved country-level heterogeneity and to ensure that the results are not driven by particular countries (Kittel & Winner 2005). In our first set of models, reported in table 5, we use panel-corrected standard errors with corrections for first-order autocorrelation to correct for disturbances arising from cross-sectional heteroskedasticity, contemporaneous correlation, and temporal autocorrelation (Beck & Katz 1995). We first do not include a lagged dependent variable as it is likely to absorb any trend in our dependent variables (Plümper et al. 2005).

Our first findings show that policy outputs have overall a significant negative effect on emission intensities. In other words, higher levels of clean air policy density and intensity are associated lower levels of emission intensities, suggesting that a positive effect of environmental regulatory efforts exists in the countries under study. Only in the case of CO_2 emissions, the coefficient for policy density does not reach statistical significance.

As regards the control variables, our results confirm most of the discussed expectations based on previous research. While emission intensities for all three pollutants decrease at higher levels of per capita income in a country, periods of strong economic growth increase emission intensities for CO_2 and SO_2. Moreover, as expected, higher shares of value added by industrial production significantly increase emission intensities, at least in five out of six models. Urbanization also has a significant positive effect on emissions, whereas increases in population density turn out as a significant negative predictor of SO_2 emissions only.

Table 6: Determinants of air emissions, 1976-2003. Specification in levels.

	(1) CO_2	(2) CO_2	(3) SO_2	(4) SO_2	(5) NO_X	(6) NO_X
Policy density	-0.0002		-0.0029**		-0.0017**	
	(0.000)		(0.001)		(0.001)	
Policy intensity		-0.0012***		-0.0055***		-0.0030***
		(0.000)		(0.001)		(0.001)
GDP pc	-0.7360***	-0.6844***	-1.7159***	-1.5500***	-1.1387***	-1.0507***
	(0.064)	(0.063)	(0.164)	(0.165)	(0.115)	(0.115)
GDP growth	0.0021*	0.0019*	0.0052**	0.0044*	0.0018	0.0014
	(0.001)	(0.001)	(0.003)	(0.002)	(0.002)	(0.002)
Population density	-0.0019	-0.0017	-0.0109***	-0.0101***	-0.0033	-0.0029
	(0.001)	(0.001)	(0.003)	(0.003)	(0.002)	(0.002)
Urban population	0.0261***	0.0250***	0.0446***	0.0406***	0.0388***	0.0362***
	(0.004)	(0.004)	(0.013)	(0.012)	(0.007)	(0.008)
Industry	0.0040**	0.0026	0.0337***	0.0290***	0.0128***	0.0102***
	(0.002)	(0.002)	(0.005)	(0.005)	(0.004)	(0.003)
Trade intensity	-0.0015**	-0.0013**	-0.0054***	-0.0047***	-0.0025**	-0.0021**
	(0.001)	(0.001)	(0.002)	(0.002)	(0.001)	(0.001)
Manufactures exports	0.0007	0.0007	0.0025	0.0025	-0.0013	-0.0013
	(0.001)	(0.001)	(0.002)	(0.002)	(0.002)	(0.002)
Manufactures imports	-0.0012	-0.0011	-0.0044**	-0.0038*	-0.0005	-0.0003
	(0.001)	(0.001)	(0.002)	(0.002)	(0.001)	(0.001)
FDI inflow	0.0004	0.0006	-0.0003	0.0001	-0.0000	0.0002
	(0.001)	(0.001)	(0.002)	(0.002)	(0.001)	(0.001)
Constant	23.3096***	22.9283***	25.8002***	24.6267***	20.9237***	20.3431***
	(0.555)	(0.566)	(1.362)	(1.434)	(1.045)	(1.021)
R^2	0.999	0.999	0.987	0.987	0.995	0.995
N/Countries	632/24	632/24	632/24	632/24	632/24	632/24

Notes: Panel-corrected standard errors in parentheses. Coefficients for country fixed-effects are not reported. All independent variables lagged one year. *** p<0.01, ** p<0.05, * p<0.1.

Interestingly enough, higher levels of overall trade are associated with lower emission intensities for all three pollutants. At the same time, levels of FDI inflow cannot be associated with emission intensities. Together, these findings counter the pollution haven scenario and rather point to a beneficial effect of economic integration on environmental quality, at least among OECD countries. With regard to the more specific characteristics of trade in our models, we can only find a significant reduction in SO_2 emissions at higher levels of manufactures imports, whereas exports of manufactures do not seem to have a systematic effect on emission intensities. In accordance with previous research, this finding could be explained by environmental efficiency enhancing technology and knowledge spillovers, which should be particularly high for imports of advanced capital goods (Perkins & Neumayer 2008).

We next check the robustness of our findings by including a lagged dependent variable, which is the widely used alternative to deal with serial correlation and to model dynamic processes. By including a lagged dependent variable, we effectively assume that current emission levels can be explained by emission levels in the previous period. Beck and Katz (2011: 342) also recommend this procedure for fixed-effects specifications, since it outperforms instrumental variable approaches in time-series with more than twenty periods.

The results reported in table 7 show that policy density and intensity do not turn out as significant predictors of emission intensity levels. Instead, our findings suggest that the yearly adjustments in emission intensities modelled by the lagged dependent variable approach can best be explained by per capita income and urbanization. More specifically, increases in per capita income lead to lower emission levels of CO_2 and NOX, whereas urbanization leads to increases in these emissions. As regards SO_2 emissions, the coefficients for both variables are not significant. Moreover, increasing shares of industrial production increase emission levels of SO_2 and NOX, but not CO_2. The results also moderately support that increasing trade volumes tend to limit CO_2 and SO_2 emissions.

Overall, our lagged dependent variable models cast first doubts on the causal relationship between changes in clean air policies and air pollutant emissions. In particular, periodical adjustments in emission intensities do not seem to follow directly from changes in regulatory activity.

Table 7: Determinants of air emissions, 1976-2003. Specification in levels including a lagged dependent variable.

	(1) CO_2	(2) CO_2	(3) SO_2	(4) SO_2	(5) NO_X	(6) NO_X
Policy density	0.0000		0.0002		-0.0002	
	(0.000)		(0.001)		(0.000)	
Policy intensity		-0.0002		-0.0003		-0.0004
		(0.000)		(0.000)		(0.000)
GDP pc	-0.0748***	-0.0676***	-0.0967	-0.0827	-0.1622***	-0.1597***
	(0.026)	(0.023)	(0.063)	(0.062)	(0.049)	(0.045)
GDP growth	-0.0026**	-0.0027**	-0.0029	-0.0030	-0.0041**	-0.0042**
	(0.001)	(0.001)	(0.002)	(0.002)	(0.002)	(0.002)
Population density	-0.0008*	-0.0008	-0.0024**	-0.0024**	-0.0005	-0.0004
	(0.000)	(0.000)	(0.001)	(0.001)	(0.001)	(0.001)
Urban population	0.0044***	0.0044***	0.0060	0.0060	0.0051**	0.0052**
	(0.002)	(0.002)	(0.005)	(0.005)	(0.002)	(0.002)
Industry	0.0002	-0.0003	0.0059***	0.0051**	0.0053***	0.0049***
	(0.001)	(0.001)	(0.002)	(0.002)	(0.002)	(0.002)
Trade intensity	-0.0007**	-0.0007*	-0.0014*	-0.0013*	-0.0006	-0.0005
	(0.000)	(0.000)	(0.001)	(0.001)	(0.001)	(0.001)
Manufactures exports	-0.0001	-0.0001	-0.0004	-0.0004	-0.0006	-0.0007
	(0.000)	(0.000)	(0.001)	(0.001)	(0.001)	(0.001)
Manufactures imports	-0.0001	-0.0001	-0.0004	-0.0003	0.0007	0.0007
	(0.000)	(0.000)	(0.001)	(0.001)	(0.001)	(0.001)
FDI inflow	0.0004	0.0005	0.0007	0.0009	0.0002	0.0003
	(0.001)	(0.001)	(0.001)	(0.001)	(0.001)	(0.001)
Lagged DV	0.8548***	0.8533***	0.9173***	0.9163***	0.8778***	0.8734***
	(0.032)	(0.032)	(0.028)	(0.028)	(0.027)	(0.027)
Constant	3.0556***	3.0211***	1.4303*	1.3217*	2.5790***	2.6144***
	(0.668)	(0.647)	(0.789)	(0.803)	(0.616)	(0.571)
R^2	0.986	0.986	0.989	0.989	0.988	0.988
N/Countries	632/24	632/24	632/24	632/24	632/24	632/24

Notes: Panel-corrected standard errors in parentheses. Coefficients for country fixed-effects are not reported. All independent variables lagged one year. *** p<0.01, ** p<0.05, * p<0.1.

Table 8. Determinants of air emissions, 1976-2003. Specification in first-differences.

	(1)	(2)	(3)	(4)	(5)	(6)
	Δ CO_2	Δ CO_2	Δ SO_2	Δ SO_2	Δ NO_X	Δ NO_X
Δ Policy density	-0.0003		0.0005		-0.0001	
	(0.001)		(0.001)		(0.001)	
Δ Policy intensity		-0.0004		-0.0005		-0.0004
		(0.000)		(0.001)		(0.001)
Δ GDP pc	-0.4078***	-0.4049***	-0.3225	-0.3146	-0.4805**	-0.4772**
	(0.148)	(0.148)	(0.266)	(0.267)	(0.213)	(0.213)
Δ GDP growth	0.0004	0.0004	-0.0018	-0.0019	-0.0017	-0.0017
	(0.001)	(0.001)	(0.002)	(0.002)	(0.002)	(0.002)
Δ Population density	-0.0012	-0.0012	-0.0146**	-0.0145**	-0.0073	-0.0072
	(0.003)	(0.003)	(0.006)	(0.006)	(0.005)	(0.005)
Δ Urban population	0.0151	0.0147	0.0644***	0.0633***	0.0483***	0.0477***
	(0.011)	(0.011)	(0.015)	(0.016)	(0.014)	(0.014)
Δ Industry	-0.0006	-0.0006	-0.0016	-0.0016	-0.0055	-0.0055
	(0.003)	(0.003)	(0.004)	(0.004)	(0.004)	(0.004)
Δ Trade intensity	-0.0003	-0.0003	0.0007	0.0007	0.0005	0.0005
	(0.001)	(0.001)	(0.001)	(0.001)	(0.001)	(0.001)
Δ Manufactures exports	0.0005	0.0005	0.0017	0.0017	-0.0008	-0.0008
	(0.001)	(0.001)	(0.002)	(0.002)	(0.002)	(0.002)
Δ Manufactures imports	-0.0007	-0.0007	0.0013	0.0013	0.0011	0.0011
	(0.001)	(0.001)	(0.002)	(0.002)	(0.001)	(0.001)
Δ FDI inflow	0.0006	0.0006	-0.0005	-0.0004	-0.0000	0.0000
	(0.001)	(0.001)	(0.001)	(0.001)	(0.001)	(0.001)
Lagged DV	-0.0022	-0.0014	-0.0526	-0.0505	-0.1044	-0.1047
	(0.088)	(0.088)	(0.060)	(0.060)	(0.081)	(0.081)
Constant	-0.0105*	-0.0102*	-0.0698***	-0.0682***	-0.0345***	-0.0338***
	(0.006)	(0.006)	(0.011)	(0.011)	(0.009)	(0.009)
R^2	0.118	0.118	0.103	0.103	0.105	0.105
N/Countries	607/24	607/24	607/24	607/24	607/24	607/24

Notes: Panel-corrected standard errors in parentheses. All independent variables lagged one year. *** p<0.01, ** p<0.05, * p<0.1.

Further to unit heterogeneity and autocorrelation, which we have dealt with in our first two regressions, panel analysts need to worry about nonstationarities. Particularly if the dependent variable is nonstationary, this usually creates an additional source of bias and might lead to falsely confirming relationships that are spurious. Testing for unit roots in our dependent variables by means of Im-Pesaran-Shin tests (results not reported), we confirm that our emission data are, except for CO_2, indeed subject to significant trends. Since the inclusion of a lagged dependent variable and panel corrected standard errors alone cannot deal with nonstationarities, we rely moreover on a dynamic specification in first-differences (see Kittel & Winner 2005). Results of the first-difference models are reported in table 8. We do not include country-fixed-effects, because first-differencing the variables has removed most of the cross-country variation.

The results of the first-difference models reveal that changes in clean air regulation do not remain a significant predictor of changing emission intensities. Still, with the exception of policy density in the case of SO_2 emissions, the coefficients of our clean air policy output measures turn out with the expected signs. Less robust are the results for the variables measuring a country's reliance on industrial production and trade, which both turn out with insignificant coefficients and for the most part even with reversed signs.

Judged by the first-difference models, the most robust predictors of emission intensities remain changes in GDP per capita with increases leading to lower emissions and urban population, where increases are associated with higher emissions. Yet, there is also some variation across the different pollutants under study. Per capita increases in GDP are not a significant predictor of SO_2 emission intensities anymore while urbanization does not have a significant effect on CO_2 emissions. These differences suggest that it may well matter for conclusions reached by impact data which particular pollutant is studied.

In summary, the initial analyses in this article suggest that clean air regulations, i.e. environmental policy output, cannot be easily associated with environmental impacts. Put differently, increases in clean air policy density and intensity do not translate directly into decreases of emission intensities. The downside of this finding is that theories of policy change cannot easily be tested on the basis of environmental impact data.

VI. Concluding remarks

In this article we have developed a new measurement concept of policy change that is suitable to study detailed changes in public policy-making in policy fields. To this end, we introduced the dimensions of policy density and policy intensity. The dimension of policy density is composed of the policy targets and policy instruments that populate a policy field and thereby measures the legislative penetration and regulatory differentiation of the field. Policy intensity, by contrast, refers to the strictness of these measures by considering the concrete calibrations of policy instruments in terms of their levels and scopes of application. While these dimensions imply a necessary simplification with regard to the various aspects of policy change, they advance the state of art as they enable the systematic measurement and comparison of policy reforms over various policy fields or subfields, and countries with considerable attention to detail.

We have illustrated the application of our measurement concept with an example from environmental policy research where the study and explanation of environmental impact data dominates the field. We focused on clean air policy as a particularly popular subfield. In a second step, we evaluated whether clean policy outputs can explain emission intensities of well-known pollutants. Such a relationship is a necessary, though not sufficient, condition for using environmental impact data to study theories of policy change. The results of our analysis show that clean air regulations have no straightforward effect on emission levels of major air pollutants. As a consequence, we conclude that environmental impacts are not necessarily a reliable proxy for environmental policy change.

This finding has several implications for future research. First, our findings underline the general need for a more cautious approach when measuring policy change. In particular, the theoretical consequences of selecting a specific measurement approach should be more critically reflected. In this article, we have discussed issues pertaining to environmental impact data such as uncertain reliability resulting from largely unknown data generation processes and issues of unknown lag structures. Second, and most important, the results suggest that a more promising way to study policy change and its theories is to rely on the direct measurement of policy outputs.

That said, we should also like to emphasize that a focus on environmental impacts can still be a reasonable choice if the analyst is interested in testing grand theories of change at the macro level. After all, environmental regulations should lead to decreases in pollution if implemented successfully. In other words, implementation effectiveness could be the ‘missing link’ between policy outputs and impacts in our analyses, which we did not model. Yet, a certain degree of implementation effectiveness is also a necessary condition in studies testing policy change theories by means of environmental impact data. In this context, our analyses merely show that this necessary condition cannot be simply presumed.

In fact, a systematic investigation of the linkage between changes in policy outputs and changes in policy impacts could also inspire a new generation of large-n implementation studies. So far, however, implementation studies are typically based on small-n designs, given the need for a detailed assessment of certain processes and causal factors involved during the implementation stage, and hence serious practical restrictions regarding the gathering data for a larger numbers of cases. The collection of data on policy outputs and impacts, however, could offer a promising starting point for large-n investigations of implementation effectiveness.

Finally, we would like to point out that this contribution is only an initial step to stimulate a more lively academic discussion about policy change in environmental policy research. Such a debate, while being quite advanced in social policy research (see, e.g., Pierson 2001; Green-Pedersen 2004; Clasen & Siegel 2007; Carsten 2011), has so far hardly taken place in environmental policy research. To be sure, also our approach is subject to numerous empirical limitations with respect to the size of the country sample and the restricted focus on clean air policy. Despite these limitations, however, we are confident that the direct measurement of governmental activity represents a step into the right direction for integrating the theoretical literature on policy change with large-n empirical studies.

VII. References

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Author: Christoph Knill, Kai Schulze, Jale Tosun

Title: Measuring environmental policy change: Conceptual alternatives and research implications

Reihe Politikwissenschaft / Political Science Series 125

Editor: Prof. Johannes Pollak, IHS
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ISSN: 1605-8003

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ISSN: 1605-8003