# Naming and Shaming or Compliance Tracking?

## International Water Treaties and Upstream-Downstream Monitoring along European Rivers

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**Abstract:**

International rivers are the classic case of asymmetric externalities. These externalities create divided interests over cooperation. Can downstream states reduce these negative effects by naming and shaming upstream perpetrators through scientific measurements of the water quality close to the border? Or do downstream states report measurements to keep track of upstream compliance with existing agreements? The unequal distribution of monitoring stations over space and time provokes an explanation for why certain states monitor and report more others. Theorising that there is an endogenous relationship between monitoring and cooperation, I operationalise monitoring as a directed one-mode network based on geo-referenced gauging stations and the cooperation network as a two-mode network. To empirically study between network effects, I use a Stochastic Actor Oriented Model. The results suggest that downstream monitoring increases cooperation, however, that shared treaties are not conducive for monitoring.

*To those who I admire.*

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# 1 Introduction

Upstream-downstream situations along international rivers are characterised by asymmetric externalities. Upstream states have little to gain from reducing pollution emissions to downstream states because upstream states endure pollution abatement costs while benefits accrue downstream. By contrast, downstream states gain from upstream pollution reduction but do not bear any costs (Mitchel & Keilbach 2001). But can downstream states achieve cooperation with upstream states by *naming and shaming* upstream perpetrators for the externalities they emit? Or do downstream states *track upstream compliance* after they sign a treaty to deter upstream incentives to defect from existing international water quality agreement before reaching an agreement? Understanding when and why states monitor the water quality of incoming rivers is important for policymakers who rely on monitoring measurements to decide on water quality policy instruments for the river (Voulvoulis et al., 2017) or to understand the effects of changing runoff patterns due to climate change (Malve, 2012). The European Environmental Agency (EEA) publishes yearly assessments (e.g. EEA, 2018) of the status and the pressures in European rivers. Policy recommendations in these assessments rely on the data obtained through water quality monitoring stations. Academics have used pollution measurements to understand what factors are responsible for pollution levels in rivers (Sigmann, 2002; Bernauer & Kuhn, 2010). However, the reported measurement stations of riparian countries vary considerably over space and time.

Yet, when and why states monitor the quality of rivers has thus far only received very limited attention from political scientists and international relations scholars. Except for a study by Beck, Bernauer & Kalbhenn (2010, henceforth BBK), the literature has largely neglected this question. BBK investigate if states' monitoring choices are primarily driven by environmental pressure or if political and economic factors have greater explanatory power. Their study offers an interesting starting point to investigate how states' choices to monitor their upstream peers affect choices to sign treaties and how shared treaties affect downstream monitoring. They theorise that greater involvement in international organisations (IGO) and global multilateral agreements (MEA) positively influences the number of reported monitoring stations. However, such dependence arguments can be better modelled with quantitative methodology that does not assume independent observations as conventional econometric methods do. Recent advances in statistical network modelling (Snijders, 2001; Snijders et al., 2010) make it possible to model different interdependent networks (Snijders et al. 2013; Snijders et al., 2017). This empirical approach does not presume a one-way causal relationship, but enables an understanding of the directionality and sequentiality of the relationship between water quality monitoring and international cooperation.

Based on insights from individual cases of the Danube and the Rhine, this thesis theorises that downstream states who monitor the water quality of incoming rivers from upstream states are more likely to sign treaties. While BBK presume a one-way causal relationship, I theorise and model the relationship as potentially endogenous. The underlying mechanism is that the naming and shaming of upstream perpetrators induces reputational costs upon upstream states. Upstream states may alleviate these costs by committing to water quality agreements with downstream states. However, as upstream states commit to water quality agreements, downstream states may want to keep track of upstream compliance with treaty provisions. Therefore, downstream states are inclined to monitor the quality of the incoming waters from the upstream state after they sign treaties. As the major alternative explanation, I further theorise that EU-members are more likely to enter into agreements due to lower opportunity costs from cooperation for two reasons. First, EU members have higher environmental standards due to EU environmental directives which reduce the costs of reaching water quality agreements. Second, EU members are more interconnected than non-EU members due to shared EU membership resulting in an assimilation of norms and rules which reduce the costs of future negotiations.

This thesis focuses on the European River Basins from 1971 to 2012. I cover Western and Eastern European states, excluding Scandinavia as well as the British Islands because they are not situated in clear upstream-downstream settings. The first two-mode cooperation network relies on the database gnevar[1] which integrates several existing databases into one comprehensive source of international agreements. In this cooperation network, there are two node-sets. The first node-set are states while the second node-set consists of water treaties that states have signed until 2012. A tie represents the signature of a specific state to a particular treaty. To obtain the second monitoring network, I construct a geographical information system (GIS) database that consists of the measurement stations lying within a five kilometer buffer around each side of the national border. The spatial data is based on the number of stations which countries reported to the European Environmental Agency (EEA). Layering these stations on top of European rivers allows for coding of the stations' upstream and downstream state. I then aggregate all stations close to the border to construct a directed, one-mode monitoring network where directed edges measure if downstream states monitor upstream states.

To investigate how state choices in the one-mode monitoring network and the two-mode treaty network affect each other, I use Stochastic Actor Oriented Modelling (SAOM) with multiple levels (states and treaties) and two networks (treaty network and monitoring network). Snijders et al. (2013) propose this approach as an extension to the previously implemented SAOM[2] which employ a simulation-based approach that models network evolution in continuous time. The model uses an actor-oriented perspective which is consistent[3] with the rational choice framework this thesis builds on. In this framework, states evaluate their tie choices based on the utility they can derive from changing their local network configuration (or leaving it unchanged), given their opportunities and constraints. Opportunities and constraints are shaped by the current network configuration and the specified covariates.

The results suggest that downstream monitoring of upstream states' pollution externalities makes cooperation more likely. There is, however, no evidence for a reverse causal relationship of cooperation on monitoring. While the coefficient has the expected positive sign, it remains insignificant at the 95% confidence level throughout the different model specifications. These results show that, in Europe, downstream states may engage in naming and shaming of upstream perpetrators to induce reputational costs upon them. Upstream

[1]Version from the 07.01.19. I am indebted to my thesis advisor Prof. Dr. James Hollway for generously sharing his data with me.

[2]for basic models, see Snijders, 2001; 2005, for models that allow the joint modelling of selection and influence, see: Burk et al., 2007 and Steglich et al., 2010.

[3]But also consistent with other frameworks such as for example historical institutionalism.

states alleviate these costs by joining water quality agreements with their downstream peers. However, downstream states do not report to keep track with treaty compliance of upstream states. Hence, these results are consistent with BBK who conclude that membership in MEA does not affect the number of monitoring stations close to the border.

This thesis proceeds as follows. The second section starts with a definition of externalities to then contextualise cooperation and water quality monitoring in Europe within the literature. The third section descriptively presents developments in water quality cooperation and monitoring and looks into the specific cases of the Rhine and the Danube river basins. The fourth section theoretically discusses the sequentiality of water quality monitoring and cooperation to derive testable hypotheses. Based on these theoretical considerations, I present the statistical network model in the fifth section based on which the sixth section then discusses the empirical specification including composition changes as well as time period choices. The seventh section introduces the cases analysed, discusses the construction of the cooperation network and the monitoring network to then describe the two networks. I provide evidence for the plausibility of the mechanisms from the theory in the eighth section to finally conclude with suggestions for future research in the ninth section.

# 2 Contextualising Cooperation and Water Quality Monitoring

The following two subsections define externalities and contextualise water quality monitoring in Europe in the existing literature. Subsection 2.1 clarifies the differences between externalities and free-riding. While the former is better capable of describing the pollution behaviour of upstream states towards downstream states, the latter mainly describes incentives actors have to contribute to a public good. Subsection 2.2 then presents the literature on cooperation and monitoring in Europe which argues that EU membership constrains the use of brute power of nation states. EU institutions thus make relative power a less salient explanatory factor than in other regions of the world. However, EU members have higher requirements for the adoption of water regulations such as the Water Framework Directive (WFD). The institutional network of the EU makes them less likely to treat their neighbours worse than themselves which means that they do not locate polluting industries closer border (Sigmann, 2002). But EU members pollute more than non-EU members (Bernauer & Kuhn, 2010). These findings are, however, based on measurements obtained through river basin monitoring stations with statistical models that assume independent observations. Yet, these measurements are hardly random, provoking an explanation for why some states monitor and report more than others. In the light of these considerations, it is surprising that there exists no statistical network modelling study on European water quality agreements between states. Understanding when states monitor can have a far-reaching influence on reporting behaviour. Thus, if downstream states policy-makers know that their reported measures increase chances for agreements they be more inclined to comply with the WFD which requires states to report their measurements.

## 2.1 Definitions

*Externalities* are defined as *any behaviour that affects the welfare of other actors for which they do not receive a compensation* (Buchanan & Stubblebine, 1962). However, this definition does not include pecuniary externalities which are compensated through market prices. Externalities in a Pigouvian (1929) sense create a mismatch between social and private costs of the consumption and the production of goods and services. When self-interested actors decide on the quantity they want to consume or produce, they do not take into account the full costs or benefits that their actions have for others. Therefore, externalities result in an inefficient allocation of resources which means that society as a whole could gain by charging perpetrators a price for externalities and redistribute the gains to the victim.[4]

*Free-riding* is different from externalities in the sense that it describes the behavioural incentive structures for the provision of public goods. These goods are in-excludable and non-rivalry in consumption. Each actor has an incentive to consume the public good without paying for it. Therefore, public goods are

[4]Coase (1960) argued that prefect markets do not need centralised regulation to achieve an efficient allocation. The Coase Theorem was derived from a two-actor model with perfect information and zero transaction costs operating in perfectly competitive markets where well-established property rights exist. The model shows that under these minimal assumptions, Pareto-Optimality can be reached without a central agency who coordinates an outcome by interfering into the market.

often under-provided. For example, the provision of water quality enhancing policies is a public good. While such policies are costly, the benefits are shared between all downstream states. In that sense, when upstream states externalise water pollution to the downstream state, they free-ride on downstream states' efforts to keep water clean. River water quality, which is different from national policies that are aimed at enhancing the quality of the river water, is a common pool resource (CPR). Similar to a public good, a CPR is non-excludable. The upstream nor downstream states can impede the other party from consuming the good. Different to a public good, a CPR is rivalry in consumption. Water consumed in the upstream state increases pollution for downstream states, reducing downstream possibilities to consume freshwater without receiving a compensation through the market. Whereas CPRs are usually not characterised by asymmetric interests of the users[5], the analysis of river pollution externalities is complicated by asymmetric interests: The upstream state has no interest in reducing pollution while the downstream state has a vested interest in upstream pollution reduction.

## 2.2 The Literature on Asymmetric Externalities, and Water Cooperation

In the case of *asymmetric externalities*, not all states prefer cooperative outcomes to non-cooperative outcomes. The upstream-downstream structures creates divided incentives for cooperation because perpetrators bear the cost of reducing pollution while victims obtain all benefits. Downstream states have a clear incentive to push upstream states into agreements that alleviate pollution. On the contrary, upstream states have little interest to enter into an agreement which bind them to pollution reduction commitments (Mitchel & Keilbach, 2001). By refining the arguments of Koremenos et al. (2001) that symmetric externalities produce aligned interests to reduce externalities, Mitchel & Keilbach (2001) argue that powerful downstream states may coerce weaker upstream states into agreements that reflect the preferences of the downstream state. When the downstream state is weak, exchange by issue linkage as an institutional design mechanism will be more important because issue linkage can increase the net-benefits from agreements for the powerful upstream state. Yet within the EU, power does not matter as much as in other regions of the world. While trade power is central for EU neigbourhood policy (Meunier & Nicolaidis, 2006), the use of brute power to coerce upstream states into agreements is significantly constrained by the intergovernmental and supranational institutions of the EU. This is precisely the reason why the predecessor of the EU, the European Coal and Steel Community, was established after World War II. For example, the Monetary and Economic Union constrains nation states' own authority to set tariffs or create other trade barriers – instruments which in other regions of the world may be used to coerce upstream perpetrators into cooperative behaviour. In fact, Mitchel & Keilbach (2001) exemplify their arguments with the Rhine where the downstream victim is weak and upstream perpetrators are powerful. In their case study, they show that weak downstream states use side-payments to make agreements with strong upstream states more attractive. Not only do EU institutions constrain the power of nation states, but they also have an active role in the policy process.

[5] as it is the case in Hardin's (1968) tragedy of the commons where every farmer has an incentive to overuse the common grazing ground until the resource is depleted.

The *Water Framework Directive* (WFD) enacted by the Eruopean Commission in 2000 sets a common basis for water quality management at the level of European river basins that initially affected 27 countries. Its aim is that all water bodies should achieve "good" ecological status by setting a common institutional framework for water quality management (Kallis & Butler, 2001). The WFD aims at an inter-calibration of the national monitoring schemes so as to achieve a common basis for the assessment of the ecological status of rivers and lakes by 2004 (Heiskanen, 2004). River Basin Management plans are coupled with measures to achieve environmental quality standards with an integrated basin approach addressing the management with a system's thinking approach. These measures include the characterisation of the properties of river basin districts such impact analysis of pressures or the delineation of waters (COM, 2012). Acquiring a management approach which respects the geographic boundaries of basins instead of political boundaries was deemed to be more effective for resolving negative externalities due to interdependencies of upstream-downstream water quality and quantity issues. However, water quality problems are caused by a range of different factors, including agricultural fertilisers, industrial pollution from industry production or human sewage treatment. Each of these activities are embedded in different national and legal contexts. Integrating these standards can thus cause frictions. Rigid top-down command-and-control approaches are likely to create a spatial misfit and therefore an incompatibility of governance structures imposed by the WFD. Such an approach can further cause problems of institutional adaptation (Moss, 2004; Borowski, 2008). Therefore, there has been a shift from command-and-control approaches of Directives to negotiated agreements which involve a greater variety of stakeholders. Such developments benefit the implementation in states with good negotiation capacities where they foster the involvement of public and private actors (Moss, 2004). These arguments are potential explanations for why one of the most ambitious environmental legislations by the EU has not managed to achieve its own standards (Voulvoulis et al., 2017).

Controversy also surrounds the role of *EU membership* for pollution externalities towards downstream states. Sigmann (2002) empirically examines the free-riding behaviour in international river basins around the world. She uses the pollutant biological oxygen demand as a proxy for human water pollution to test if pollution levels close to the border are higher than domestic pollution levels. She finds that upstream countries emit significantly more pollution towards their downstream neighbours than they do domestically. However, in the EU, upstream states do not emit significantly higher pollution levels close to the border of downstream neighbours which suggests that EU institutions successfully reduce free-riding. In a similar study, Bernauer & Kuhn (2010) investigate if there is an environmental version of Kantian peace[6]. For the empirical analysis, they use two different pollution variables, biological oxygen demand and nitric oxygen $NO_3$. For these two pollutants, they construct four different dependent variables. The first two measure absolute pollution levels for both pollutants close to the border and the second two measure the difference between pollution levels close to the border and domestic pollution levels. The authors find that EU members generally emit

[6]This means that democratic states that are interconnected trough trade and international institutions externalise less pollution to their neighbours

more pollution regardless if domestically or close to the border. But the results for the pollution levels close to the border relative to domestic pollution levels suggest that EU-members do not emit more pollution close to the border than they do domestically. In short, Sigman (2002) finds that EU institutions successfully reduce externalities towards downstream neighbours. But Bernauer & Kuhn (2010) find that EU members do not allocate polluting industries close to the border but pollute significantly higher externality levels domestically and to neighbours than non-EU members. Both of these studies, however, implicitly assume that pollution measurements are representative of overall river pollution.

At the same time, states' interest and incentives to correctly report pollution levels vary, especially when monitoring is not conducted centrally by independent international organisations (Abbott & Snidal, 1998). While downstream victims have an incentive to overreport as well as to name and shame upstream perpetrators, upstream states may feel inclined to position gauging stations where pollution emissions are relatively low. Statistical estimates for pollution levels based on non-random samples can lead to biased estimates and, more importantly, to wrong inference. In fact, as an interesting paper by BBK shows, states position their monitoring stations more frequently in rivers that are characterised by an upstream-downstream situation structure than along other rivers. Only economic development has a greater substantive effect on the monitoring activity of states across Europe. Astonishingly, in all of the presented models, EU membership has a large negative effect on the number of monitoring stations. This is surprising as EU members are strongly institutionalised and are therefore also more central in the network of international environmental regimes. As for example Ward (2006) argues, states that are central in the network of international environmental institutions can use their social capital to effectively deal with environmental issues. Moreover, not only is the EU globally seen an environmental leader, but the supranational institutions of the EU may themselves encourage states to adopt pioneering behaviour in the domain of environmental policy in the hope that other states follow a similar trajectory. Such forerunner strategies are also encouraged by competition over the future design of EU policies. Thus, intergovernmental and supranational institutions of the EU as such serve as channels through which environmental policies can spread (Tews, 2005; Börzel, 2002) and EU members are more likely to be exposed to such forces.

*EU accession status* changes the incentive structure for new member states to comply with EU rules (Sedelmeier, 2008; Epstein & Sedelmeier, 2013). Before the 2004 and 2007 enlargement rounds, EU institutions have effectively influenced domestic policy of potential member states by conditioning membership on compliance with EU standards. Some scholars argue that the influence of the European institutions on the compliance behaviour primarily depended on the incentive structure for countries with accession status to obtain membership rather than on a process of persuasion or learning. Therefore, they expected that new member states' compliance with EU law would decrease after accession (Kelley, 2004; Schimmelfennig & Sedelmeier, 2005). With these arguments, BBK explain the finding that EU-membership significantly reduces monitoring behaviour by arguing that "once states have joined the EU, political pressure to engage in more monitoring may, paradoxically, be smaller" (p.9). However, the authors do not control for EU-candidacy. It may be that EU membership captures part of the effect of EU candidate status as membership and candidacy are correlated. But also EU candidates have an incentive to comply with EU directives to signal cooperative behaviour (Schimmelfennig, 2010). Thus it seems theoretically sensible to expect that both EU-candidacy and EU-membership have a positive effect on monitoring. Robust evidence shows that the prospect of EU membership encourages candidates to ratify EU's preferred MEAs[7] (Schulze & Tossun, 2013).

However other scholars observe that *compliance* of new member states with EU rules is surprisingly high. For example, the transposition of EU directives into national law was higher for new member states than for old member states after enlargement. Possible explanations for this increase in compliance are threats of post-accession sanctions, the legislative capacity building of member states and socialisation to EU norms and rules. Post-accession sanctions include the possibility to penalise non-compliance financially trough the European Court of Justice or the possibility for the European Commission to take measures against countries that seriously infringe the functioning of the EU internal market (Sedelemeier, 2008). The EEA however has no enforcement capabilities and therefore cannot punish non-cooperative behaviour (WFD, 2000, Art. 4.7). Epstein & Sedelmeier (2013) conclude that, in some policy areas such as the economic and monetary union, the changing incentive structure after accession lead to decreasing compliance with EU standards, whereas in other areas, it did not. According to Börzel (2017), no other EU policy domain than that of environmental policy witnesses more violations of EU directives. But Börzel et al. (2019) also show that the compliance problems decreased over time. This discourse in the literature makes it interesting to investigate if compliance problems are really driven by EU-membership, or if EU-candidates are the group of states that are responsible for varying compliance in Europe.

Causes and effects of International agreements are *endogenous*. Agreements should be more attractive when there are some environmental problems to be solved and environmental regimes can, at the same time, contribute to problem solving. Hence, studying their relationship in isolation is inappropriate and requires a modelling approach that takes into account these interdependencies. This issue also remains unaddressed in the models by BBK. Therefore this thesis uses statistical network modelling which is capable of capturing interdependencies between the cooperation and the monitoring network. This thesis will be the first study to endogenously model the relationship between monitoring intensity and international treaty cooperation using data generated from geographic information systems. By doing so, I contribute to a more nuanced understanding of how upstream-downstream dependencies affect cooperation. Do states in a dyad sign more treaties when downstream states have named and shamed upstream perpetrators' pollution behaviour *before* cooperation or do downstream states increasingly monitor upstream states *after* having signed a treaty to track compliance with treaty provisions? These are the two major research questions that this thesis seeks to answer.

Recent advances in *stochastic actor oriented modelling* make it possible to model how tie changes in one network affect tie changes in another network (Snijders et al., 2013). These models have been de-

[7]Note that to it is generally impossible to correctly predict the directionality of in multivariate analysis (Forbes, 200) as in empirical studies one can never be sure of having included all other potentially omitted variables. For further discussion and a mathematical proof that this an erroneous endeavour when there are several omitted variables, see Basu (2008)

veloped by Snijders (2001, 2005) to combine theoretical and empirical models of network evolution (Block et al. 2019). In the tradition of Exponential Random Graph Models (ERGMs; Lusher et al., 2013), which excels at modelling how dependence between ties lead to an observed network structure, the SAOM extends this framework in an attempt to model continuous processes of network change. Later, SAOMs have been extended to model endogenous processes between actor attributes and their propensity to change network ties (Snijders et al., 2007). For example, Manger and Pickup (2016) make use of this extension to revisit the debate on the endogenous coevolution of democracy and preferential trade agreements (PTA) formation. In international relations, the model extension for the coevolution of two-mode and one-mode networks (Snijders et al., 2013) has only been used by Milewicz et al. (2018). They use this model to investigate how state choices to include non-trade issues in PTAs affect choices to include non-trade issues in multilateral trade agreements network and vice versa. Thus, this study will be the second study to apply this SAOM extension to international relations, contributing to the expanding importance of of statistical network modelling in general and in international relations in particular.

# 3 Rise and Relevance of Water Quality Monitoring and Cooperation in Europe

The following chapter provides an overview of the historical development of international cooperation and water quality monitoring in Europe. The first subsection shows trends in the number of water quality measurements stations and water cooperation. The second subsection presents two short case studies on the Rhine and the Danube to shed some light on mechanisms for how the relationship between cooperation and monitoring unfolded in these particular cases. While the Rhine case exemplifies the importance of path-dependency in cooperation, the case on the Danube stresses how reduced political and ideological cleavages facilitate multilateral cooperation. These insights should then serve as a basis to make generally testable propositions in section 4.

## 3.1 General Trends in the Evolution of Cooperation and Water Quality Monitoring

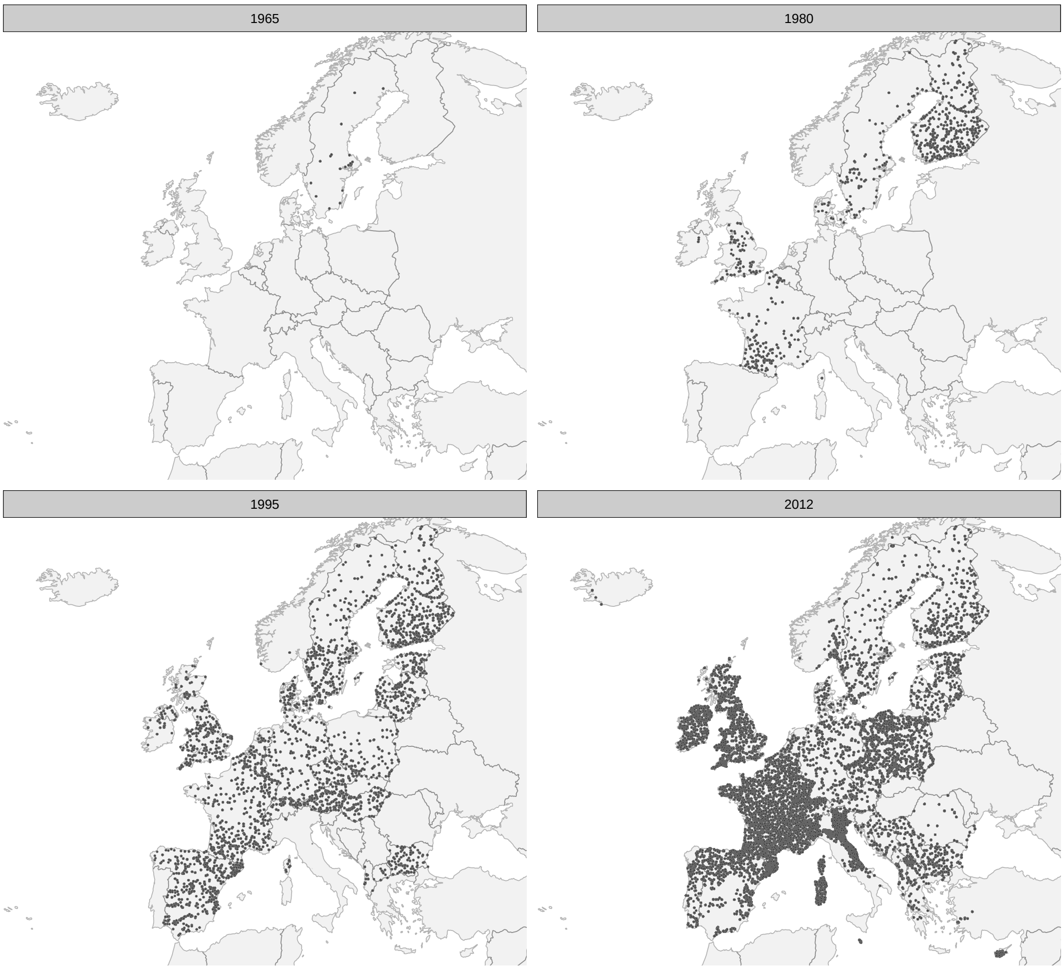
Clean freshwater is indispensable for all living organisms. The ecosystem, human communities and the economy all rely on sources of clean and fresh water. Contrary to many other regions in the world, the quality of European waters has substantively improved over the past decades. Indicators such as biological oxygen demand, ammonium, nitrate, and phosphate all show decreasing trends in European waters. However, other pollutants such as mercury are still a major problem in European rivers. These pollutants are responsible for the failure of many river basins to achieve "good status" – the major objective of the 2000 WFD. 38% of surface waters are in good chemical status and 40% are in good ecological status (European Commission, 2019). These numbers show that despite the improvement, there is still substantial room for water quality improvements. The basis of any such assessment is, however, the water quality measurements reported to the EEA by member states which is why more research on this topic is needed. The number of stations that have been reported to the EEA shows a sharp upward trend until 2012.

*Monitoring*

Figure 1 maps the number of stations in Europe for four different years. In 1965, Sweden started reporting monitoring stations. The snapshot of 1980 shows that Great Britain, Denmark and France started reporting measurements. The map shows that the station density increased substantially in the 1990s. The map also illustrates that there is some level of fluctuation in the data of Hungary which reports in 1995 but not in 2012. It is also apparent that states which witnessed sovereignty or territorial changes did not report water quality measures after these processes had been completed. In light of the communist decline in Eastern Europe, the rise of democracy and the EU expansion coupled with increased economic growth, changes in the political and economic landscape of Europe should be of importance for cooperation and monitoring.

Figure 2 shows the development of the total of number of monitoring stations in Europe. France

Figure 1: Mapping Water Quality Monitoring Stations in Europe. This figure is inspired by BBK.




reported measurements taken in Rhone Basin in 1969 and one year later also close to the border of Belgium. In 1972, the total number stations across Europe reached 248. In 1976, Finland reported the measurements of 327 stations. For 1982, measurements taken in the Federal Republic of Germany were reported to the EEA. After 1980, we see a somewhat steadier increase with less fluctuation. The sharp rise in the number of monitoring stations in 1992 can be attributed to several Eastern European states starting to report measurements. These countries are Estonia, Latvia, Lithuania, Poland, Slovakia, the Czech Republic, Slovenia and Austria. The number of stations hit exactly 3'000 in 1998. A sharp rise is again observable in 2006. This increase is mainly attributable to Italy, France and Spain. Note that after 2012 there is a drop in the data for the number of monitoring stations as reported to the EEA. According to the EEA, this drop occurred because all states failed to meet the deadlines but Spain, Germany, Austria, Finland, Denmark Latvia and Bulgaria. The EEA still waits for states to report these measurements.

Figure 2: Total Number of Water Quality Monitoring Stations in Europe over Time

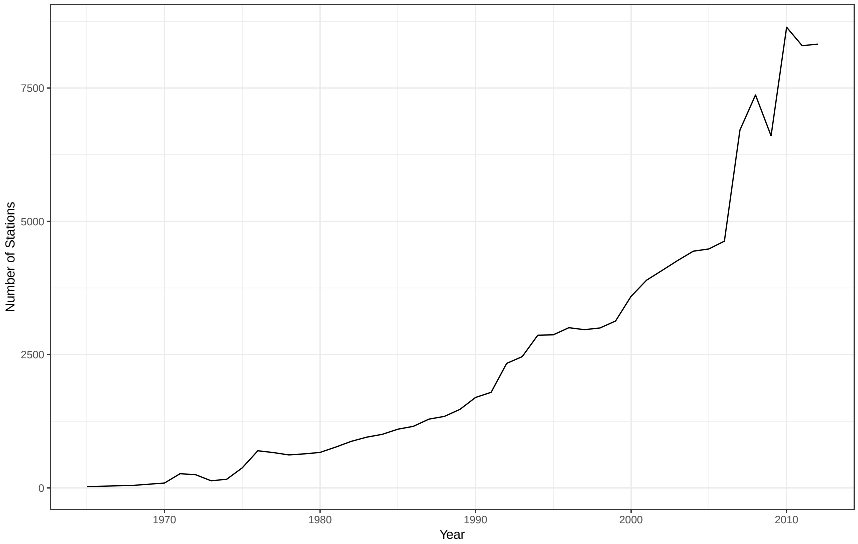
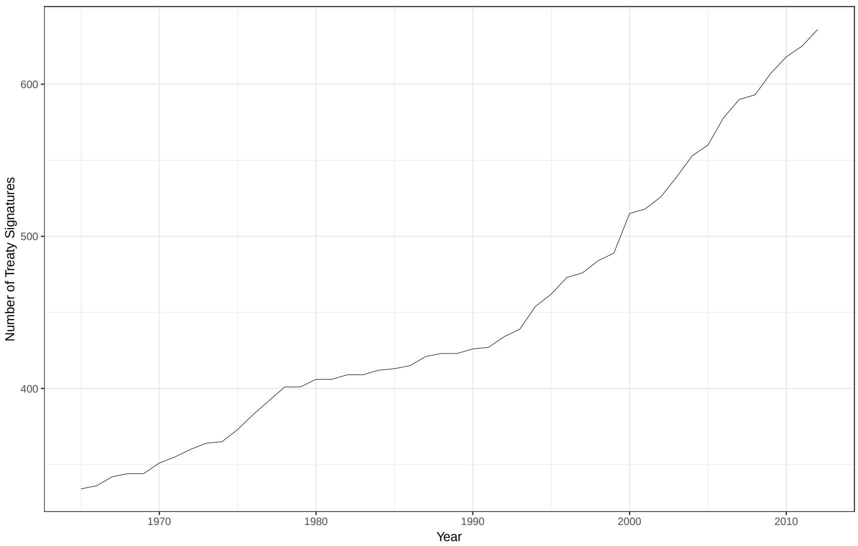


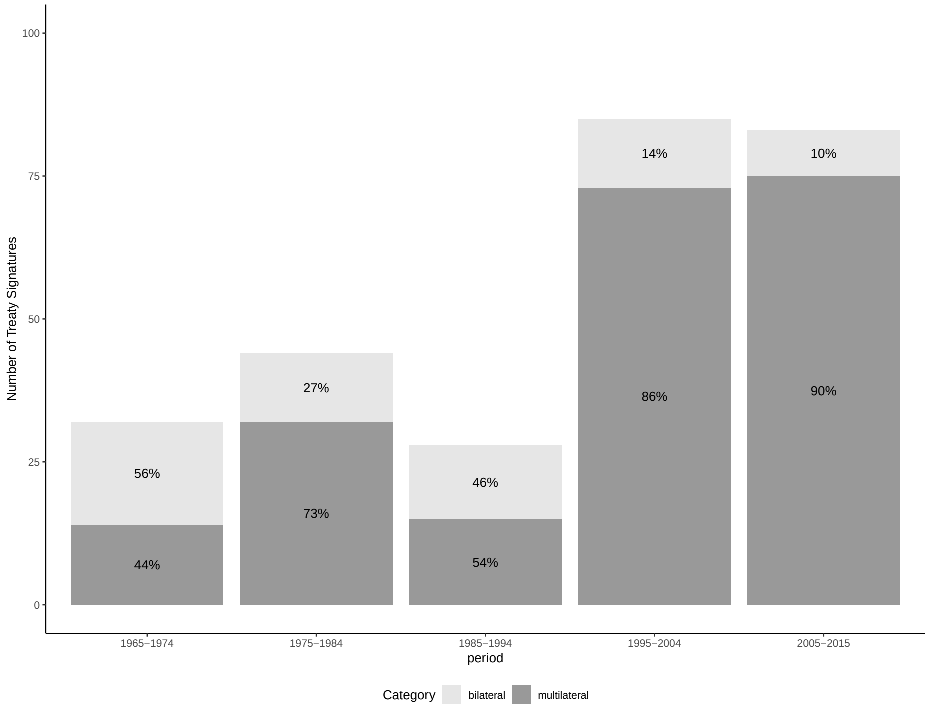

Figure 3: Total Number of Water Agreement Signatures over Time




*Cooperation*

The number of treaty signatures that states made substantially increased over time, which, especially due to an increase in multilateral cooperation lead to a higher interconnectedness. Or as Alter & Meunier (2009, p.1) notes, "the number, level of detail, and subject matter of international agreements have grown exponentially in recent decades". These developments are also apparent in the domain of international water quality agreements. Even though institutionalisation of central European River Basins started as early as 1900, for other basins, predominantly in the Eastern European States, many states signed multilateral treaties more recently. Figure 3 shows the rise of water agreements over time. In 1965, the number of signatures for water quality agreements was roughly around 350. One can see that there is a relatively sharp increase from 1965 to 1977 which then level off in the 1980s. After 1990, the number of signed agreements again steadily increases from over 420 to just below 650 agreements.

Figure 4: Bilateral and Multilateral Agreement Signatures over Time




Multilateral cooperation among all basin members is best way to curb externalities from a theoretical perspective because it reduces incentives to externalise pollution levels for all states. In the case of bilateral agreements (BLA), states that are not members of the treaty still have an incentive to pollute at the expense of cooperating parties. Special attention should therefore be devoted to multilateral agreements (MLA). Figure 4 plots the number of newly signed treaties from 1965 to 2015 for 10 year periods. The data shows a clear trend towards multilateralism in European water cooperation. In the first period from 1965 to 1974, a total of 32 treaties were signed. 18 of these 32 treaties were bilateral. In the second period, 44 treaties were singed out of which 12 were bilateral. In the period from 1985 to 1994 overall fewer signatures are observable. Note that in this period sovereignty changes occurred which means that the number of states also increased. This may also explain parts of the increases in the number of signatures from 1995 to 2004. In this period, a sharp rise in the number of signatures to a total of 85 is observable. In the last period, further 82 treaties were signed with the share of multilateral agreements again increasing to 91% of the total. These descriptives indicate that water governance is increasingly interconnected. Although institutionalisation has started very early in Europe as the following subsection will demonstrate, the institutional density keeps increasing. In the light of arguments by Andonova & Mitchell (2010) who show that the global level of environmental politics has become increasingly interconnected with the regional level, these descriptives show that this interconnectedness has also risen within the regional European level.

## 3.2 Case Evidence of the Endogenous Relationship between Monitoring and International Cooperation from the Rhine and the Danube

Cooperation along the Rhine and the Danube have comparable, yet different historical developments of cooperation and monitoring. Along the Rhine, basin-wide cooperative arrangements emerged relatively early. In the Danube Basin, multilateral cooperation which includes all states in the basin was only possible after the fall of the iron curtain. On the Rhine, monitoring was also much earlier institutionalised whereas along the Danube similar institutions could only be established when the ideological and cultural dissimilarity between upstream and downstream states started to fade.

*Rhine*

International Cooperation on the river Rhine started as early as 1885 with a treaty on the conservation of the salmons. Additionally, since the beginning of the $20^{th}$ century, several international conventions[8] have managed the transportation of poisonous substances. But those conventions did not directly regulate pollution emissions. The high salinity in the Netherlands was also noted by Dutch drinking water companies who pushed the government to take action and bind upstream states with a treaty (Dieperink, 2000). But it was in the 1950s that the Netherlands brought the problem to the agenda in the international commission established by the Treaty on Salmon Protection in 1885 (Kiss, 1985).

Eventually, in 1963, after informal negotiations (Dieperink, 2000), the Convention on the International Commission for the Protection of the Rhine against Pollution (ICRP) was signed in Berne. The signatories were the Netherlands, Switzerland, the Federal Republic of Germany and Luxemburg. The main functionality of the ICRP was the monitoring of the water quality of the river and the identification of its sources. The mandate also included the proposition of measures against pollution and the preparation of further agreement between riparians. However, because of its limited political power, the ICRP could make little progress on the latter two points (Bernauer & Moser, 1996). In 1972, state Ministers' held an annual conference on the protection of the Rhine (Le Marquand, 1977) which, according to Bernauer & Moser (1998), had positive effects on the competences of the ICRP in the process of the formulation of a convention on chemical production.

The Bonn convention on chemical pollution was signed in 1976. This convention banned substances as specified in Annex I completely but gradually over time and required the submission of total quantities of the substances listed in Annex II. Under this convention, every member state is obliged to set up monitoring networks. States then had to submit the aggregated data obtained through national monitoring networks to the ICRP (Kiss, 1985; Schwabach, 1989). In 1987, the yearly Minister Conference founded the Rhine Action Plan which replaced the Bonn Convention form 1976 partly due to its failure to protect the river from the Sandoz

[8]such as the 1900 the Convention between the Riverain States of the Rhine respecting Regulations Governing the Transport of Corrosive and Poisonous Substances or the 1902 Convention Relative to the Carriage of Inflammable Substances on the Rhine both cited in Kiss (1985)

accident in Basel (Bernauer & Moser, 1996). As a result of this accident 10,000 and 15,000 cubic meters of water, consisting of vast amounts of toxic substances[9], entered the river which had detrimental impacts on the ecosystem. It killed every living organism 300 kilometers downstream of Basel and was widely criticised because Swiss authorities took longer than 24 hours to notify downstream riparians about the accident (Schwabach, 1989). After the failure of the early warning system that had already been in place before the accident, there was a need to improve joint monitoring. When monitoring is conducted by an IGO, communication between riparians can be improved and accidents can be faster communicated to downstream riparians. This accident was a main driver for the implementation of a joint monitoring station in downstream Germany just below the city of Basel. The 1990 Agreement between the government of the Land Badenwurttemberg and the Swiss Federal Council concerning the joint construction and operation of a monitoring station downstream of Basel' was implemented. This monitoring station is still active and provides yearly measurements of the water quality of the river Rhine taking daily measures and publishing yearly reports on the amount of toxic substances that Switzerland externalises to its downstream riparians (Bundesamt für Umwelt in der Schweiz, 2017, p.30-32).

In conclusion of the river Rhine case, the evidence shows that monitoring and cooperation are hardly exogenously determined. While cooperation set the basis for common monitoring schemes, the monitoring data set the basis for suggestions on future cooperation. Moreover, the Rhine case also shows that international cooperation is path-dependent. The costs of reaching an agreement are dependent on past agreements. Past Agreements such as the treaty on the conservation of salmons provide a forum through which riparians may utter concerns on different but interrelated topics. Also, the ICRP was given the mandate to develop further treaties that tackle pollution sources that the ICRP identifies through its independent monitoring scheme. But it also shows that states take an important role with their national monitoring programs, the results of which they report to the ICRP.

*Danube*

The Danube is the worlds' most international and Europe's largest river basin. 19 countries have territory in its catchment area. But the basin has a history which is, relative to other parts of post-World War II Europe, characterised by unstable political and economic history. EU expansion, the retreat of communism in post-Soviet countries and the foundation of six new states in 1992 have systematically altered the pre-conditions for cooperation. The dissolution of the Soviet Union was followed by the formation of the Balkan states after the disintegration of Yugoslavia during the Balkan war. Communist Czechoslovakia partitioned into the Czech Republic and Slovakia after economic progress hampered in the post-war period. During these changes, the Danubian countries have turned ever more strongly towards Western alliances. (Linnerooth-Bayer & Murcott, 1996). Since 1995, all of the four last enlargement rounds of the EU comprised of at least one country which drains into the Danube. In 1995, Austria joined. Of the countries that became members

[9] amongst which were 200 kilograms of mercury.

in 2004, the Czech Republic, Slovakia, Hungary and Slovenia have territory draining into the Danube. In 2007, Bulgaria and Romania joined. In the 2013 enlargement, Croatia joined. Thus, according to the logic of collective action, cooperation should become more difficult as more actors need to agree on the rules that govern them. But this effect has been outweighed by other political and economic factors. In the course of these political changes, the Danubian states have become culturally and ideologically more homogenous. The linguistic borders of Western Slavic, Southern Slavic, Anglo-Saxon and Romanic languages have become less pertinent with the decline of nationalism and the rise of democracy in former Czechoslovakia and the former Yugoslavian states. The central authority of the EU that acted as a force to counter nationalistic tendencies has effectively set incentives for more cooperative and less isolationist policies. This development also manifests itself in international water cooperation.

Before the fall of the iron curtain, there had been no basin-wide agreement for the Danube. In the Danube declaration signed in 1985, states declared their intention to sign bilateral and multilateral agreements addressing water quality issues. Only in 1994, the states managed to reach an agreement that applied to the entire basin (Ovodenko, 2016). After the agreement had been ratified in 1998, The International Commission for the Protection of the Danube River (ICPDR) came into force which has comparable tasks to those of the ICRP. Before the decline of communism, cooperation was merely divided along political and cultural lines. Germany and Austria concluded several bilateral agreements, while Soviet influence secured strategic interests in the Danube. The most extensive multilateral agreement that entered into force before the ICPDR was the Belgrade Convention in 1948. This convention was mainly concerned with navigation and included all member states but the Federal Republic of Germany. Like its successor, the ICPDR, the treaty set up a commission. But its main task was to maintain the navigability of the river (Linnerooth-Bayer & Murcott, 1996). The ICPDR, however, includes a more diverse set of duties which includes the obligation to monitor water quality by setting up river gauging stations. But similarly to the ICRP, Art. 9 of the convention prescribes that the Danubian states shall "harmonise or make comparable their monitoring and assessment methods as applied on their domestic levels, in particular in the field of river quality, emission control, flood forecast and water balance" (p.9). Thus, these considerations show that the ideological and political division of the riparians matters for cooperation.

While the number of actors in the Basin increased, the cultural division decreased. Successful multilateral cooperation is hence unlikely to be mainly the result of geographic factors. The role of the EU in this process is not to be underestimated because it provides a framework for cooperation between its members but also between members and non-members. The presence of a central authority can stabilise a political environment by reducing (political and economic) uncertainty (Ostrom, 2015). Moreover, as the similarity in the structure and design of the ICPDR and ICRP shows, cooperation efforts along the different river basins in Europe are hardly independent. In an assessment on the success or failure of cooperation along the Rhine, it is thus important to also incorporate the influence that successful agreements may have on agreements in other basins. The conclusion that international cooperation along the Rhine was unsuccessful because it failed to prevent an unarguably terrible accident (Schwabach, 1989; Bernauer, 1996) must be balanced against the pioneering role that cooperation along the Rhine has for other less institutionalised basins. Although there is no universal set of rules that can be successfully applied to every different context (Ostrom et al., 2007; Pahl-Wostl et al., 2012), an integrated monitoring scheme across Europe can only be achieved with a certain degree of institutional convergence across basins.

# 4 Theorising International Cooperation and Water Quality Monitoring in Upstream-Downstream Settings

I develop a theory which posits that there is an *endogenous relationship* between downstream monitoring and international water quality agreements. Drawing on Mitchell & Keilbach (2001), upstream states per se have limited interest in forming agreements with downstream states because they would bear the costs of abating pollution externalities they emit to downstream states. I therefore expect that downstream states may name and shame upstream perpetrators to induce reputational costs upon them which upstream states may alleviate by joining water quality agreements with downstream states. Once treaties are signed, states may want to keep track of upstream compliance with treaty provisions. This relationship is described in Section 4.1. As alternative explanations for environmental policy output, I discuss path-dependency (Pierson, 2000), EU membership, EU candidacy, democracy and GDP.

The theory relies on a rational choice institutionalism which assumes that states optimise their choices based on cost-benefit considerations. Although this may not always be a completely correct presumption, it is a useful theoretical abstraction that enables the deduction of clear testable hypotheses. This theory broadly builds on works by Abbott & Snidal (1998, 2001) and Mitchell & Keilbach (2001) synthesising their theoretical work with findings from Bernauer & Kuhn, 2010 and BBK (2010) along with my theoretical considerations.

## 4.1 Naming, Shaming and Compliance Tracking in International River Basins

The upstream-downstream situation structure along international rivers is characterised by *asymmetric externalities* (Mitchel & Keilbach 2001; Sigmann, 2002; Lavenex, 2008; Bernauer & Kuhn, 2010; BBK, 2010). This means that downstream states suffer from upstream pollution emissions as these reduce downstream states' possibilities to use water for their preferred purposes. Costs also arise for example from negative impacts on ecosystems including reduced use of pesticides in agriculture which results in lower agricultural productivity or the installation of costly wastewater management facilities. The effects on the ecosystem are not restricted to living organisms within the river[10] but also affect the ecosystem in the river catchment area because groundwater systems and rivers together form an interconnected system[11] (Brunke & Gonser, 1997). Thus water pollution not only harms animals and plants but also human health (Khan & Gouri, 2011).

Yet, asymmetric externalities generate *divided interests* for upstream and downstream states which

[10] Geeraerts & Belpaire (2010) review the literature on the effects of pollution on the eel population in Europe. They conclude that pollution has had significant impacts on the reproduction rate of the eel species.

[11] For example in Basel, part of the water from the river Rhine enters the groundwater system where ecosystems perform a filtering functionality. This water is naturally cleaned in the meadowland called Langen Erle even to the extent that the water which initially came from the main river Rhine stream flow can later be used as the major source for drinking water in the city Basel (Rüetschi, 2004).

are amplified by the unequal distributive effects of cooperative agreements. On the one hand, downstream states strictly prefer pollution abatement as they do not *à-priori* bear any of the costs for it. But upstream states have little interest in reducing water pollution towards downstream neighbours because they bear the costs (Mitchell & Keilbach, 2001). Therefore downstream states are interested in entering agreements with upstream states that set pollution reduction targets. But upstream states have little interest in committing to such agreements. (Mitchell & Keilbach, 2001; Lavenex, 2008).

I propose *naming and shaming* as the underlying mechanism through which downstream victims may instrument water quality monitoring to leverage upstream perpetrators into agreements. Naming and shaming can be achieved with a credible scientific measurement of the ecological state of a river, especially when it is close to the border of the upstream perpetrator. Making these measurements publicly available through the institutions of the EU has several positive effects for the downstream state. First, the downstream state is perceived as environmentally friendly if it commits to policy output such as water quality monitoring. Second, it demonstrates that part of the pollution in its rivers comes from the upstream state. Third, the upstream state incurs reputational costs for externalising pollution to the downstream state. To reduce these reputational costs, upstream states can commit to an agreement with the downstream state and thereby display their commitment to high environmental standards.

> *Naming and Shaming Hypothesis: An agreement between two states is more likely when the downstream state monitors the upstream state.*

To deter incentives to defect from previously signed water quality agreements, downstream states may want to track the compliance of upstream states with treaty provisions. When upstream states know that defection will cause them to be named and shamed for their uncooperative behaviour, they have less incentives to do so. Therefore monitoring upstream states can increase compliance with international water quality agreements by monitoring upstream states. Yet, downstream states have an incentive to misreport pollution levels. However, there are two mechanisms that discourage them from doing so. First, upstream monitoring close to the border can generate comparable measurements against which upstream states can cross-validate reported measurements of downstream states. Second, in some basins, water quality monitoring is also conducted by independent international organisations such as the International Commission on the Protection of the Rhine or the Danube River Protection Convention. According to Abbott & Snidal (1998), these international institutions generate information that is more credible than measurements of individual nation states. This implies that independent international organisations take the preferences of several cooperative states into account. Nevertheless, the different measurement protocols and sampling designs of the national monitoring schemes (Parr et al., 2002), as well as the criteria for choosing the location of monitoring stations (BBK), call into question if cross-European water quality estimates are reliable. States may in fact more often report their measurements when they had previously signed an agreement because they want to track upstream compliance and deter defection from treaty provisions.

*Tracking Compliance Hypothesis: Downstream states monitor and report upstream states' pollution level more intensely after they signed a treaty.*

## 4.2 Alternative Explanations for Environmental Policy Output

When trying to reach an agreement, *EU members* face lower costs than non-EU members. There are two main reasons why EU members are more likely to sign agreements than non-EU members. First, EU members are urged to transpose EU directives into national law. Although directives such as the WFD are of non-binding legal nature (WFD, 2000) which means that EU-institutions have no enforcement possibilities, these directives result in higher environmental standards in those countries that would otherwise have adopted less stringent water policy. Countries that have a higher preference for environmentally friendly water policy may still implement them domestically. Sometimes such directives require secondary agreements for their implementation. Third, the dense institutional web of intergovernmental and supranational institutions of the EU serve as channels through which countries interact and exchange information (Rogers, 1995; Axelrod, 1997) which requires building the necessary bureaucratic capacity. Because those capacities may also be used when negotiating agreements, EU members have lower costs to implement future agreements.

International cooperation is *path dependent* (Pierson, 2000). Previous shared agreements form channels through which states may exchange information[12]. For example, the 1997 UN Convention on the Law of the Non-Navigational Uses of International Watercourses, Art. 9, paragraph 2, prescribes that "watercourse States shall employ their best efforts to collect and, where appropriate, to process data and information in a manner which facilitates its utilization by the other watercourse States to which it is communicated" (p.6). This excerpt exemplifies how treaties may reduce asymmetric information. River Basin Organisations that operate independently of states such as the International Commission for the Protection of the Rhine lend even more credibility to generated information. Based on existing shared agreements, riparians can more easily identify key challenges, foster a common understanding of them and solve them cooperatively. Existing shared agreements therefore reduce the costs for future agreements. However, if states fail to cooperate in the first place, the asymmetric situation structure along international rivers may lead to entrenched positions. Put simply, there are decreasing average costs of cooperation. Thus, I expect positive tendencies towards treaty network closure.

Based on an article by Bernauer & Kuhn (2010) which investigates the factors leading to water pollution reduction, BBK expect that EU member states are more likely to monitor water quality. EU institutions provide member states with a forum through which countries may mitigate negative water pollution externalities from upstream states. Poorer EU-members even benefit from payments and financial support for environmental policy output from EU-projects (Lifferink et al., 2009) which improves possibilities to fund

[12]especially when information exchange mechanisms are specified. Information exchange mechanisms reduce incentives to defect from water quality agreements because they can reveal non-cooperative behaviour (Stinnet & Tir, 2009)

monitoring projects. Because EU members are more strongly interconnected than non-EU members, they also have better access to the relevant scientific information that facilitates effective policy implementation. But the EU also has higher legal requirements in the domain of environmental policy than most non-EU states in Europe. The development of river basin management plans under the WFD is a case in point.

*EU accession status* changes the incentive structure for potential member states to comply with EU rules (Sedelmeier, 2008; Epstein & Sedelmeier, 2013). EU membership is conditioned on the implementation of the *acquis communaotaire*. This conditionality includes the fulfillment of certain minimal democratic norms (Schimmelfennig, 2008), but also comprises of requirements to invest in infrastructure, environmental protection and agricultural reform (Grabbe, 2002). This means that when candidates comply with EU standards, they not only generate benefits of improved environmental quality but at the same increase their chances for successful EU accession. Moreover, I expect that EU accession status increases the salience of policy competition. If one EU candidate increases the compliance with EU standards, it puts pressure on other candidates to follow them. Otherwise, laggards could face disadvantages in the accession process. If, however, none of the candidates complies with EU standards, the disadvantage relative to others is smaller.

*Democratic* states have higher environmental policy output than less democratic countries. First, citizens in democracies have more information about the state of the environment due to higher press freedom. Second, citizens can better voice their concerns. Third, citizens have the freedom to form interest groups. Fourth, politicians seeking re-election are more likely to respond to pressures and translate it into policy output. These four factors are less prevalent in non-democracies (Payne, 1995). Deacon (2009) deduces from a formal model that democratic regimes are more likely to provide public goods. Less democratic regimes' rational leaders are unlikely to decide on non-exclusive policies that create spillovers to groups whose support is unnecessary for leaders. But democratic, re-election seeking leaders are more likely to provide non-exclusive services because it serves a larger population on whose votes politicians rely to stay in power.

*Income levels* are an important explanatory factor for the demand for environmental quality. The literature around the environmental Kuznets Curve suggests that an inverted U-shaped relationship between pollution levels and income. As income levels increase, the willingness to pay for environmental quality rises disproportionately because environmental quality is luxury good[13]. Although there is some disagreement in the literature about the solidity of the theoretical and statistical foundation of these findings[14] (Stern, 2004), the expectation that the demand for environmental quality increases with income is less challenged. States with higher income levels have better possibilities to finance monitoring programs because they can afford more developed national administrations. Therefore benefits from water quality monitoring increase with per capita income levels.

These alternative explanations take into account many of the factors that also have been discussed in

[13]According to microeconomic theory, luxury goods are a specific type of goods for which demand rises over-proportionately as income rises. On the contrary, for normal goods, demand increases proportionately as income increases. Classic examples are potatoes for normal goods and expensive cars for luxury goods.

[14]for evidence for against the Kuznets Curve, see Sirag et al., 2018 and for evidence in favour Jalil & Mahmud, 2009 with evidence in favour

the cases on the Rhine and the Danube. These factors are a non-exhaustive list of alternative explanations to the two main hypothesis which I contribute. The alternative explanations are established theoretical arguments in the literature.

# 5 Stochastic Actor Oriented Models

The model extensions of the SAOM by Snijders et al. (2013) is a useful choice to test the theorised endogenous relationship between monitoring and cooperation. This section discusses the model choice, the mathematical specification of the model and its assumptions.

## 5.1 Model Selection

Statistical Network Modelling is a suitable approach for disentangling the sequentiality of the relationship of downstream monitoring and cooperation. Statistical network models do not assume independent observations. Because the upstream-downstream situation structure along rivers implies dependence due to the interconnectedness through shared agreements and shared rivers, using a model that can capture these dependencies is warranted. Deterministic statistical models[15] are hardly able to capture how local decisions lead to the emergence of macro-patterns because they assume independent observations. Three major model categories seem most attractive for this study[16].

First, ERGMs (Frank & Strauss, 1986; Pattison & Wasserman, 1999; Snijders et al. 2006; Lusher et al. 2013) are particularly useful for explaining emergent structures of a network. However, the ERGM is a cross-sectional model. Thus it is incapable of explaining network change. There exist extensions such as the temporal ERGM (TERGM) and the longitudinal ERGM (LERGM). With these models, it is however currently not possible to model endogenous processes between different networks or between networks and actor attributes. Moreover, with these methods, it is not possible to incorporate composition changes of actors in the network. Importantly, the TERGM, as an autoregressive model, cannot properly explain network change because it simply regresses the present structure on previous panel observations (Block et al., 2018).

The second model category is the Dynamic Actor Oriented Model[17] (DyNAM). There are several reasons why this model is an attractive option for international relations scholars. DyNAMs use information on the exact time-point when changes in the network configurations happened. Continuously modelling possible evolution trajectories is not necessary because all changes are observed. Consequently, the computational burden is much smaller (Stadtfeld et al., 2017; Stadtfeld et al. 2017; Stadtfeld & Block, 2017). This is an important advantage because simulating endogenous network processes can be time consuming. Yet, because the monitoring data comes in yearly observed panel data which is different from time-stamped data, the DyNAM is less suitable here. An advantage that DyNAMs however have is the possibility to model weighted ties to represent the strength of a relationship. But because the DyNAM cannot (yet) model endogenous processes between two different networks, the SAOM is most appropriate.

[15] such as Ordinary Least Squares, Fixed Effects, Random Effects, or conventional event history models.

[16] For more general recent reviews on different statistical network models, see Snijders (2011); Salter-Townshend et al. (2012); Hunter et al. (2012). For a review on SAOMs, see Snijders (2017) and for cross-sectional models including ERGMs, see Amati et al.(2018).

[17] The current version of the package goldfish is available from Prof. Dr. Hollway or the ETH Social Networks Lab. The package cannot yet be downloaded from the public R repository CRAN because it is currently being developed.

In the tradition of agent-based models[18], the SAOM links micro-level behaviour with emergent macro-level patterns (Snijders, 2001; Snijders, 2005; Snijders et al. 2010). Like DyNAMs, this model category is actor oriented. SAOMs, in the tradition of Holland & Leinhardt (1977), assume that network change is a first order Markov-Chain. This means that tie changes, which are endogenous to the existing structure of the network in time period $t$, is assumed to be only affected by tie changes in time period $t-1$. The SAOM further assumes that tie changes in the network configuration that continuously lead from one discrete network observation to another can be broken down into mini-steps. At each mini-step, only one tie change can occur. The rate function determines which actor is selected to reconsider his local network configuration. By default, the rate function is the same for all actors, which means that at each mini-step an actor is selected with a uniform probability to reconsider his local network configuration. The actor can either dissolve an existing tie, create a previously inexistent tie or can leave the network configuration unchanged (Snijders & Pickup, 2017). These mini-steps make it possible to model potential endogenous processes in continuous time. Discrete time models, such as TERGMs miss important aspects of the evolution of the network over time because they do not model what happens between observations (Block et al., 2018). An extension by Snijders et al. (2013) makes it possible to model interdependencies *between different* networks, which is the model that I will be using.

There are several existing applications of the SAOM to international relations (Warren, 2010; Manger et al.2012; Manger & Pickup; Kinne, 2013, 2014; Warren, 2016; Kinne & Bunte, 2018). However only one study by Milewicz et al. (2018) has applied the extension of Snijders et al. (2013) to international relations. Milewicz et al. (2018) model how interdependencies between bilateral and multilateral trade agreements influence the inclusion of non-trade issues into trade agreements as two interlocking networks initially proposed by Hollway & Koskinen (2016a, 2016b) for cross-sectional Multilevel ERGMs. The present study will thus be the second application of the model proposed by Snijders et al. (2013) to international relations.

## 5.2 Theoretical Specification of the Stochastic Actor Oriented Model

The SAOM uses repeated snapshots in the form of panel data to model underlying processes in continuous time as a Markov Chain process. This means that the probability of a tie change depends on the structure of the current network configuration. Current tie changes modify the network configuration based on which actors evaluate tie changes in future time steps. This modelling approach excels at modelling dependence structures over time. In the light of path-dependency in political science in general and in international cooperation in particular, this model is thus useful for modelling generic change that depends on previous network configurations (Snijders et al. 2010; Snijders & Pickup, 2017). There are two major extensions of the SAOM that make it possible to model the coevolution of decisions of actors in different outcome-spaces in the form of several different dependent variables within one model. Two extensions have been proposed.

[18] The canonical study of agent-based models is Schelling's (1971) segregation model. This model shows under minimal preferences of people of the same race living together that patterns of racial segregation emerge.

First, a model for the *coevolution of actor attributes and the network* configuration has been introduced by Steglich et al. (2010). These (nodal) actor attributes are not relational variables in the sense of two actors sharing a network connection (tie). Actor attributes can be any characteristic of an actor who is also part of a network. Actors make decisions to change local network configurations by weighing the attractiveness of ties based on the current network configuration and actor attributes. An interesting application to international relations that may serve as an illustrative example is the study by Manger & Pickup (2016). They model the coevolution (i.e. the mutual dependencies) of PTA agreement formation and democracy. The model divides into two components. The first component of the endogenous relationship, estimates the propensity of states to form ties in the network of PTAs conditional on its level of democracy as well as the level of democracy of potential PTA peers. The second component models the influence of network structures on democratisation. Thus, SAOMs are not only capable of modelling how endogenous dependencies affect the evolution of one network but may also model endogeneity of actor attributes and the network.

Second, an extension for modelling the *coevolution of multiple networks* has been proposed more recently by Snijders et al. (2013). As opposed to the first extension of Steglich et al. (2010), the model is not concerned with actor attribute changes. This model allows for an understanding of how tie changes in one network affect tie changes in a different network. The creation, maintenance and dissolution of ties in one network depend on the present network configuration of several different networks. These models thus introduce an additional level of complexity. An excellent application to international relations is the paper by Milewicz et al. (2018). In line with previous research by Manger & Pickup (2016) discussed in the preceding paragraph, they are interested in understanding what factors foster the inclusion of non-trade issues into trade agreements. Milewicz et al. (2018) model changes in the network of preferential trade agreements as being interdependent with changes in the multilateral trade-agreements network. Their findings suggest that cost explanations to non-trade issue inclusion are the most important explanatory factor. They argue that the spread of non-trade issues can be explained with path-dependent costs – once a first non-trade issue is concluded the average cost of including future non-trade issues decreases. This thesis contributes to this research strand by developing and applying the model proposed by Snijders et al. (2013) to water quality monitoring activity of downstream states towards upstream states and cooperative water quality agreements in Europe.

I now turn to the *mathematical specification* of the coevolution of a network with two node sets (states that may join agreements) and a network with one node set (states that monitor other states) network under the framework of a SAOM. Before going into the model, consider a two-mode network $Y$ and a one-mode network $X$. Let $N$ be the first node set and $A$ be the second node set. Each state $i \in N$ can engage in $a \in A$ agreements and monitor other states $i \neq j$. Thus at time period $t$, the two-mode network $Y$ contains of information about the affiliation of states $i = 1, 2, ...n$ to treaties $a = 1, 2, ...n$. Each country $i$ may be a member of each agreement $a$ and country $i$ may monitor $j$. The two-mode network $Y_{ia}$ consists of information on the ties for states $i \in N$ with agreements $a \in A$. In this network, ties between actors $i \neq j$ are not possible. The tie $Y_{ia} = 1$ if country $i$ is a member of agreement $a$ and it is $Y_{ia} = 0$ otherwise. The one mode network $X$ consists of the same node set $N$ with ties $X_{ij} = 1$ if country $i$ monitors $j$ and 0 otherwise (Snijders et al., 2013).

The SAOM models tie changes as a *continuous underlying Markov-Chain* process. This process decomposes the network evolution into so-called mini-steps. At each mini-step, only one actor is chosen with a uniform probability to reconsider her local network configuration. The waiting time until the next actor $i$ is chosen from the set of actors $N$ depends on the waiting time $\lambda_i^Y(x, y)$ in the two-mode network $\lambda_i^X(x, y)$ in the one-mode network under the constraint of the current network configuration $x$ and $y$. The waiting time until any actor receives the possibility to make a change to its local network configuration has an exponential distribution with the following value (Snijders et al., 2013, p.275).

$$\frac{1}{\lambda_+^Y(x, y) + \lambda_+^X(x, y)} \tag{1}$$

with

$$\lambda_+^Y(x, y) = \sum_{i \in N} \lambda_i^Y(x, y), \qquad \lambda_+^X(x, y) = \sum_{i \in N} \lambda_i^X(x, y) \tag{2}$$

Formula (2) shows that the waiting time $\lambda$ for any actor to make a tie change in Network $Y$ depends on the sum of the waiting time of all actors $i \in N$ in networks $Y$ and $X$.

Recall that ties have the binary values $\{0, 1\}$. Therefore, tie changes can be represented as a toggle of a tie. A change from $Y_{ia}$ to $1 - Y_{ia}$ or from $X_{ij}$ to $1 - X_{ij}$ can conveniently be noted as $y^{(\pm ia)}$ and $x^{(\pm ij)}$. As an example of what that means, if there is a tie between individual $i$ and $j$ in the one-mode network, the value of the entry in the cell row of $i$ and the column of $j$ in $N \times N$ matrix is equal to 1. With other words $X_{ij} = 1$. The only other possible state that the tie between $i$ and $j$ can have is 0 (subtracting the value of $X_{ij} = 1$ from the value of 1). If the tie does not exist, meaning that $X_{ij} = 0$, subtracting $1 - X_{ij} = 1$. Note that all other ties between individuals $i \neq h$ remain unchanged. For the one-mode network $X$, this can be expressed as:

$$x_{ih}^{(\pm ih)} = x^{(\pm ih)} \text{ for dyads } (i, h) \neq (i, j) \text{ and} \tag{3}$$

$$x_{ij}^{(\pm ij)} = 1 - X_{ij} \tag{4}$$

and very similarly for the two-mode network $Y$,

$$y_{ia}^{(\pm ia)} = y^{(\pm ia)} \text{ for dyads } (i, b) \neq (i, a) \text{ and} \tag{5}$$

$$y_{ia}^{(\pm ia)} = 1 - Y_{ia} \tag{6}$$

If actor $i$ has been chosen to reconsider his local network configuration in the two-mode network $Y$, the probability $p$ of a tie change from state $i$ to treaty $a$ is being evaluated from the perspective of the focal actor $i$ based on a comparison to all other treaties $b \neq a$ and local monitoring structures in the monitoring network $X$. How attractive the model estimates it to be to tie to a certain actor thus depends on a linear combination of the specified explanatory variables that can generate a network that is somewhat similar to the observed network based on the weighted set of explanatory variables. The probability $p$ of a tie change can be expressed as follows(Snijders et al., 2013, p.275)

$$p\{Y(t) \text{ changes to } y^{(ia)}|X(t) = x, Y(t) = y\} = \frac{exp(f_i^Y(x, y^{(\pm ia)}))}{f_i^Y(x, y) + \sum_{b \in A} exp(f_i^Y(x, y^{(\pm ib)}))} \tag{7}$$

Let $\{j \in N | j \neq i\}$ be the complete set of actors $N$ excluding state $i$. If actor $i$ has been chosen to reconsider his local network configuration in the one-mode network $X$, then the probability $p$ of a tie change from actor $i$ to actor $j$ is the following (Snijders et al., 2013, p.275)

$$p\{X(t) \text{ changes to } x^{(ij)}|X(t) = x, Y(t) = y\} = \frac{exp(f_i^X(x^{(\pm ij)}, y))}{exp(f_i^X(x, y)) + \sum_{h \in \{j \in N | j \neq i\}} exp(f_i^X(x^{(\pm ih)}, y))} \tag{8}$$

Based on the evaluation function $f_i^Y$ for individual $i$ in network $Y$, actors decide if they want to change their local network configuration or leave it unchanged. The evaluation function is a linear combination between the $k$ specified statistics $s$ and the number of parameter estimates $\beta_k$ for $k = \{1, 2, ..., k\}$. The size of $\beta_k$ is an indication for the strength of an effect for the network coevolution. (Snijders et al., 2013, p.275).

$$f_i^Y(x, y) = \sum_k \beta_k^Y s_{ki}^Y(x, y) \tag{9}$$

Similarly, in the one-mode network $X$, the statistics depend on both network configurations $x$ and $y$ because of the theorised endogenous relationship between choices in the network $X$ and the network $Y$. Consequently, the evaluation function of $f_i^X$ for individual $i$ that models tie changes in the one-mode network is defined as (Snijders et al., 2013, p.275)

$$f_i^X(x, y) = \sum_k \beta_k^X s_{ki}^Y(x, y) \tag{10}$$

Endogenous cross-network effects and structural cooperation network effects that go into equation (9) are specified in Table 1 and 2. These effects serve to model the attractiveness of tie changes assuming that states tie to those treaties from which they can derive most utility as a stochastic process.

### 5.3 Model Assumptions

It is often useful to evaluate a model against its *assumptions*, to understand if a model captures the essential components of underlying observed processes. The model should be able to abstract to *major* components of the relationships that connect the phenomenon of interest to its hypothesised explanation[19]. There are five major assumptions that the SAOM makes.

1. *Continuous-Time Markov process*. This assumption divides into three components. First, the state space of possible network configurations is finite. The network has clear boundaries which limit the possible set of treaty networks and monitoring networks which makes this component unproblematic. Second, the model evolves in continuous time. This component is also reasonable because, in principle, ties can change at any time. Third, tie changes have a Markovian property. This means that the future depends on the past only through the present. However, It is important to clarify that the developments in previous time periods have an effect on the future development by being observable in the network configuration of the present. This assumption enables modelling the sequentiality of a relationship possible making this assumption particularly useful.
2. *Condition on the first observation $Y(t_1)$ and $X(t_1)$*. This is a simplifying assumption which states that it is not possible to model the evolution beyond the first period. One cannot draw inferences on periods that were not modelled. Thus, the validity of this assumption is not of much concern.
3. *Only one tie can change*. It may seem that international agreements could be signed at the same time. A time point should be understood as an infinitesimally small period of time. With this understanding, which is modelled by the waiting time in the rate function, the assumption seems plausible.
4. *Actors control their outgoing ties*. This assumption is an as-if abstraction for analytical purposes. Agreements are negotiated and states cannot unilaterally decide to join a treaty. The assumption seems to hold fairly well for the monitoring network where states may decide to monitor the upstream state. Also, directly modelling negotiations would increase the already high computational burden.
5. *Complete knowledge.* Information costs are not modelled which come at a cost in reality. These costs are not directly modelled.

These are a useful set of minimal assumptions that make it possible to estimate useful models which allow for an investigation of questions which most other statistical techniques cannot deal with. Surely, these assumptions are not alway fully given in reality. While there exist model types of the SAOM that can model reciprocal proposal confirmation interactions, this is currently not (yet) possible for a two-mode SAOM. Also, states have to make an effort to obtain information about the structure of a network. But given that they make an effort for which states actually have enough resources, these assumptions are a useful as-if abstraction for analytic purposes. Or as Box (1979, p.202)) is famously quoted: "All models are wrong but some are useful".

[19] This may be different when the researcher is interested in prediction. For a discussion of prediction with social network models see Block et al. (2018).

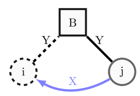
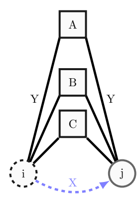
# 6 Empirical Specification

Deciding on the empirical specification of the model involves four major considerations. First, I discuss the *network effect choices* that capture concepts from the theory section. Second, I discuss *state composition changes*. Third, I present the *time periods* used for modelling in continuous time. Fourth, I describe input data for the modelling exercise. I follow instructions by Ripley et al. (2019) who provide an excellent guideline for making these choices.

## 6.1 Network Effect Choices

Starting with the cross-network effects that test the endogenous relationship between the monitoring and the cooperation network, I proceed with the discussion of endogenous network effects in the cooperation network. I provide a description, an illustration and the mathematical specification for the effect testing the cross-network relationships in Table 3 and the structural cooperation network effects in Table 4.

Table 1: Endogenous Cross Network Effects. Circles are states and squares are treaties. The dashed line indicates that the focal actor creates a tie to a treaty or a state.

| Concept | Description | Visualisation and definition |
|---|---|---|
| SHAMING TO AGREEMENT | This network effect measures if upstream state $i$ has a higher tendency to form a tie with treaty $B$ (in the treaty network $Y$) if downstream state $j$ monitors $i$ (in the monitoring network $X$). A positive effect means that downstream states $i$ naming and shaming upstream perpetrator $j$ makes $i$ commit to higher standards through an agreement $B$ with downstream state $j$. |   $s_i(x) = \sum_{j \neq B} x_{ij} w_{iB} x_{Bj}$ |
| TRACKING COMPLIANCE | This network effect measures if downstream state $i$ has a tendency to monitor $j$ (in the monitoring network $X$) when $i$ and $j$ share a treaty $B$ (in the treaty network $Y$). This effect puts a weight $\alpha$ on each additional treaty that states $i$ and $j$ share. For example if $\alpha = 0$ $A$ and $B$ do not add an additional propensity for ties in $X$ but if $\alpha = \infty$, each treaty has the same marginal effect. |   $gwespMix(i, \alpha) = \sum_{j=1}^{n} x_{ij} e^{\alpha} \{1 - (1 - e^{-\alpha})^{\sum_{h=1}^{n} w_{iB} x_{Bj}}\}$ |

This table is based on the description and mathematical definition of Ripley et al.'s (2019, p.146 & p.149). I develop the graphs based on their descriptions.

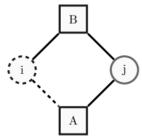
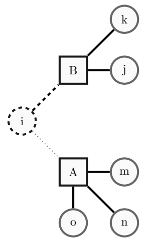
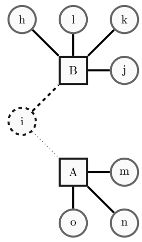
The SHAMING TO AGREEMENT effect tests if a directed tie from the downstream state towards the upstream state in the one-mode monitoring network leads to agreement of both states in the two-mode cooperation network. Such an effect has been proposed in the paper by Snijders et al. (2013). Recall the *Naming and Shaming Hypothesis* stipulates when downstream states monitor upstream states, they are more likely to sign an agreement with the mechanism that naming and shaming of upstream perpetrators induces reputational costs upon upstream perpetrators. Upstream perpetrators may alleviate these costs by showing their commitment to reducing asymmetric externalities by signing a water quality agreement with the downstream state. Thus, this effect better captures the theoretical concept than, for example, activity related effects. Activity related effects test if activity in one network, i.e. if monitoring many other states increase the propensity for a state to sign more agreements.

The TRACKING COMPLIANCE effect tests that when two states are both signatories of an agreement in the two-mode cooperation network, the downstream state should be more inclined to send a monitoring tie towards the upstream state. Recall that the *Tracking Compliance* hypothesis is the second component of the endogenous relationship between the cooperation and the monitoring network. This hypothesis posits that downstream states tend to monitor more when they already have a water quality agreement with the upstream state. Downstream states may want to track of upstream compliance with treaty provisions. Two effect variants exist here that may capture this concept with minor technical differences, however. While the first variant presumes that the effect on the downstream states' decision to monitor upstream states is proportional to the number of shared treaties, the second variant can model marginally decreasing effects of additional shared treaties. This second variant seems theoretically more appealing because when many downstream and upstream states share many treaties, independent international institutions may substitute these functionalities. For example, multilateral river basin institutions, such as the ICPDR along the Danube or the ICRP for the Rhine, may already decrease incentives for upstream states to externalise pollution to downstream states. States that are already embedded in a dense institutional network that constrain externalisation of pollution, can be expected to experience a smaller additional effect from a treaty, i.e. decreasing the marginal effect, to monitor upstream states.

Now that I have introduced the main endogenous cross-network effects, the remaining *structural cooperation network effects* are of particular interest. Recall that the difference between endogenous cross-network effects and structural effects is that the latter measures the propensity to form ties based on the existing structure within the *same* network and the former refer to endogenous effects *between* different networks. SHARED COSTS captures that costs for cooperation may be lower when states already share agreements. MULTILATERAL COSTS measures if the costs of reaching an agreement are lower when only two states are members of an international water quality agreement. And lastly, PREFERENTIAL ATTACHMENT measures the tendency of states to tie to others who already have many ties. Table 3 provides an overview with a precise summary of these effects. The framework for the terms for these effects were taken from Milewicz et al. (2018) because it offers a more intuitive way to understand how technical network effects measure theoretical concepts.

SHARED COSTS measure if states that already share agreements have lower costs of reaching a future agreement. Shared agreements may facilitate reaching future agreements because an existing institutional framework reduces information asymmetries by creating a channel for communication. Based on this channel, new challenges and pressures may be more easily identified. A positive effect means that costs are reduced

Table 2: Structural Cooperation Network Effects. Circles are states and squares are treaties. The dashed line indicates that the focal actor creates a tie to a treaty or a state.

| Concept | Description | Visualisation and definition |
|---|---|---|
| *Treaties* | | |
| SHARED COSTS | This figure illustrates a 4-cycles network configuration for actors $i$ and $j$ who are already connected through the treaty $B$. The dashed line indicates that actor $i$ prefers becoming a member of the treaty $A$ of which state $j$ is already a member. This statistic sums over all treaties A & B with which $i$ has a connection to $j$ in the network $Y$ and divides this sum by four. |  $s_i(x) = \frac{1}{4}\sum_{AB} x_{ij}$ |
| MULTILATERAL POPULARITY | This network effect measures the preference of state $i$ to tie to others with an indegree of larger than 2. A negative effect indicates that there is a negative tendency to sign multilateral treaties. This effect generates an indicator variable $I$ which is based on the sum of the members that $B$ has. For example, for the treaty $B$ it is 0 because it has 2 members and for the treaty $A$ it is 1 because it has more than 2 members. |  $s_i(x) = \sum_B I\{x_{+B} > 2\}$ |
| POPULARITY COSTS | This network effect measures if there is a tendency to tie to others who already have ties to many others. This effect sums over all members $h$, $l$, $k$ and $j$ of $B$. A positive effect indicates that states prefer tying to others who already have a high number of ties. |  $s_i(x) = \sum_B x_{iB}x_{+B}$ |

This table is based on Ripley et al. (2019, p.121, 130, 126)

if states already share an agreement. This effect is important to capture network clustering of treaties around certain actors.

MULTILATERAL COSTS captures cost considerations of states to sign multilateral treaties. This effect captures the tendency of states to sign agreements that are multilateral. Even though there has been a sharp rise in the number of multilateral agreements, this effect should be negative, meaning that states find it *à-priori* more difficult to sign multilateral than bilateral agreements. In bilateral international basins, problems may often be more efficiently solved bilaterally because the institutional design can respond to specific bilateral needs.

POPULARITY COSTS accounts for already popular treaties becoming more attractive to states. While first agreeing on a specific set of rules and design principles may be particularly difficult, once several members have joined, it may induce reputational costs on non-members. For example, multilateral water treaties under the EU may to non-signatories being perceived as environmental laggards. This effect is also known as preferential attachment[20](Barbasi & Albert, 1999).

Additionally to these structural effects that are crucial for obtaining a convergent model, the major covariates follow from the theory. Because those effects are much more similar and therefore much easier to understand, I do not discuss them in as much detail as the Cross-Network effects and the structural cooperation network effects. Recall that the major covariates are EU MEMBERSHIP, EU CANDIDACY, DEMOCRACY and GDP. The rationale for the inclusion of these variables can be found in the theory section.

## 6.2 Changes in the Composition of European States

In the time frame of 1971 to 2012, the retreat of communism, the rise of democracy and the changing incentive structure due to the prospect of EU membership lead to a different configuration of water cooperation network. Major changes have occurred at the turn of the decade from the 1980s to the 1990s. With the fall of the Berlin Wall on the $9^{th}$ of November in 1989 before the reunification of Germany in 1990, a series of major political changes started. Most of these changes occurred in 1991 when Slovenia, Macedonia, Croatia and Bosnia and Herzegovina declared independence from Yugoslavia. In the same year Estonia, Latvia and Lithuania became independent states from the Soviet Union. Close to the border of the DDR, in 1993, the previously communist Czechoslovakia split into democratic states of the Czech Republic and Slovakia. This marks a natural boundary in the time series of state composition changes in Europe. Because SAOMs model tie changes in continuous time between discretely observed snapshots, the formation and dissolution of states marks a natural point for a snapshot. Only minor changes have occurred thereafter with the independence Montenegro from Serbia in 2006 and Kosovo in 2008.

These composition changes lead to better prospects for river basin cooperation, especially on a multilateral basis. As Linneroth (1996) notes, these composition changes have fundamentally altered the preconditions for cooperation in the Danube Basin. The weakening of communist influence, the rise of democracy and the expansion of the EU have reduced the cultural and economic differences which had previously set downstream and upstream countries apart. Downstream states turned more westwards, making the first basin-wide agreement possible. Since 1995, all of the four last enlargement rounds of the EU comprised of at least one country which drains into the Danube.

Table 3 shows that in total there are 15 joiners and 3 leavers in the composition of the total of 31 states in the sample. While three states, the DDR, Czechoslovakia and Yugoslavia leave the system, there are 12 states that have been established in this territory. Additionally, the joiners include the three Baltic states Estonia, Latvia and Lithuania which declared independence from the Soviet Union in 1991. Thus, of the total of 15 newly founded states, 6 states were part of the 2004 enlargement round of the EU. This means that 66%

[20]Preferential attachment has been shown to be an important effect to explain the structure of many observed networks. The general hypothesis is that nodes with many connections increase the number of ties faster than nodes with few connections. This means that the degree distribution follows a power-law. A power law means that many actors have few ties but a few actors have many ties (see e.g. Jeong et al. 2003). Scale-free refers to the structure of the network ties being similar no matter how close one zooms into the network (Barbasi & Albert, 1999).

(6 out of 9) new members in 2004 were newly established states. Not modelling these changes could miss important effects of dependencies between cooperation and monitoring.

Table 3: Joiners and Leavers

| *Year* | *Joiners* | | | | *Leaver* |
|---|---|---|---|---|---|
| 1990 | - | | | | DDR |
| 1991 | Slovenia | Macedonia | Serbia | Croatia | - |
| 1991 | Latvia | Lithuania | Estonia | | - |
| 1992 | | | Bosnia and Herzegovina | | Yugoslavia |
| 1993 | Slovakia | Czech Republic | | | Czechoslovakia |
| 2006 | Montenegro | | | | |
| 2008 | Kosovo | | | | - |

A major advantage of process-based models such as the SAOMs over autoregressive models such as the TERGM is that it is capable of reflecting composition changes of actors in the network (Block et al. 2018). For the SAOM, there are two different possibilities to do this. The first is the method of joiners and leavers, developed by Huismann & Snijders (2003). The second possibility is the specification of structural zeros. Structural zeros simply make it impossible for states that still remain part of the network to become members of treaties. By contrast, with the method of joiners and leavers, exogenously chosen actors no longer remain part of the network. The method of joiners and leavers is more efficient and therefore preferable to the specification of structural zeros. I make use of the method of joiners and leavers and exogenously incorporate the changes in the composition of European states into the model. This makes the modelling more precise. Ripley et al. (2019, p.34-36) outline the method in more detail.

## 6.3 Time Periods

The temporal boundaries for the start and end years for the study are given by the data. After 1971, ties are stable enough for the countries in the sample to make modelling possible. For the years after 2012, the monitoring data has not yet been integrated into the EEA Water Quality database WISE (EEA, 2018). The choice of the last two periods from 1993 to 2003 and from 2004 to 2012 are mainly based on political considerations. For simplicity, I split the remaining time into equal 10-year periods as there have been fewer path-breaking political events from 1971 to 1981 and from 1982 to 1992.

The last time period is given by the *2004 Eastern enlargement*. By joining, the new EU members that fulfilled the *acquis communautaire* committed to higher environmental standards. EU membership resulted in a higher institutional interconnectedness with other EU member states. When the Eastern members joined, they completed an assimilation process to Western European states with several dimensions. First, the case on the Danube shows that there was a smaller ideological and cultural divide in Eastern Europe than before 2004. Second, the *acquis communautaire* includes minimal requirement for the democratic norms of potential members (Schimmelfennig, 2008). Third, the EU has higher environmental standards through legislation such as the WFD. These three major factors show that the year 2004 marks an important point of change. Because theory suggests that EU membership matters for cooperation, the first period in the model include the years 2004-2012.

In the year 1993, major *composition changes* in the sovereignty of European states had been completed. The process of Eastern European assimilation to Western European political and cultural norms started with three major changes in the composition of European states and the decline of communism as section 6.2 discusses in more detail. Therefore, I use the year 1993 as the second natural boundary for the choice of the time period from 1993-2003.

During the remaining time from 1971-1992, there were less profound and relevant political changes that could serve as a basis for the choice of snapshots. Therefore, I split the time into 10-year time periods. The first time period lasts from 1971-1981 and 1982-1992. The following discussion in the Appendix clarifies that this also makes sense taking the descriptive network statistics into account.

# 7 Constructing the Cooperation and the Monitoring Network

I construct two networks. The first is a cooperation network with international water quality agreements and the second is a monitoring network obtained by coding geo-referenced monitoring stations close to the border as directed ties between states.

## 7.1 Cases

To investigate how the embeddedness of states into the network of international water treaties affects their choice to monitor upstream states' emission of water quality in international river basins, I focus on Western and Eastern European states on the mainland. Because social network models do not assume independent observations, the choice of the network boundaries should reflect these dependencies. Dependencies in the network of upstream-downstream monitoring stations can only unfold directly between contiguous states. Interdependencies between clusters separated by natural boundaries are lower than within clusters. Therefore, natural boundaries such as the sea also naturally delineate network boundaries. Random samples are often inappropriate for social network analysis as they miss dependencies between the different units[21]. Thus, excluding the Scandinavian countries seems to be more appropriate than, for example, excluding Spain which shares the Ebro and the Garonne Basin with France. Figure 5 illustrates this argument in the form of map consisting of international river basins included in this study.

I cover Western and Eastern European states. I exclude the British islands because, due to their isolation through the sea, they are not embedded in a clear upstream-downstream situation structure. I do not include the Scandinavian countries because even if they share many basins, the area which lies in different countries is very skewed towards one country. Thus, they are also not characterised by clear upstream-downstream structures. I also do not include the post-Soviet states Ukraine, Belarus and Moldova because they are currently not EU candidates. Furthermore, Turkey and Russia are not in the sample because only part of their territory lies within Europe. Turkey is hardly situated in an upstream-downstream structure, hence its exclusion, even if it is a classical EU candidate. However, I do include the Baltic states because their developments with Soviet independence, rising democracy, EU candidacy and EU membership are interesting. Figure 12 in the Appendix also maps the countries in the sample together with major rivers. Note that the analysis also includes dissolved countries such as the Deutsche Demokratische Republic (DDR), Czechoslovakia and Yugoslavia because they lie within the geographic boundary of the selected states.

## 7.2 International Water Quality Agreements

The cooperation network is an undirected, two-mode network. The first node set includes 31 states. The second node set includes 190 agreements with a total of 636 signatures. A tie between a particular state and a particular

[21]For example, in a directed network with one type of node sets (or actors) in which looping ties of a unit with itself are not possible, one missing unit generates $N \times (N-1)$ missing ties.

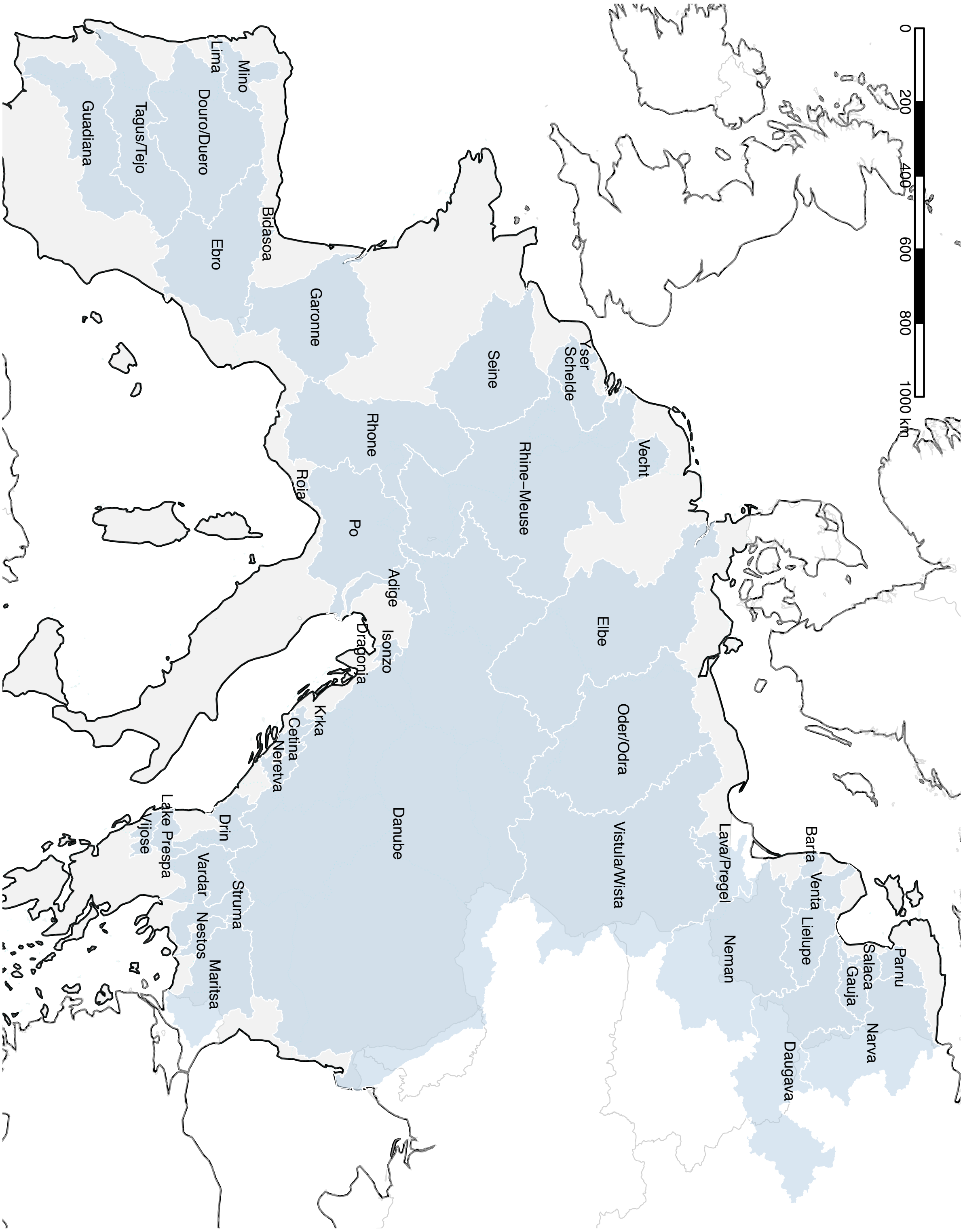



Figure 5: Basins that are included in the study are coloured in blue

treaty means that the state has signed the agreement. No tie means that it did not sign the treaty. I focus on agreement signatures because, for downstream states, the first indication that monitoring can successfully lead to cooperation is the signature of an agreement. While membership may be more important for the effects of cooperation on monitoring, the signature is the more pertinent measure for the effects of monitoring on cooperation. Moreover, because a signature is a necessary but not sufficient condition for membership, it is appropriate to first investigate if monitoring has an effect on the propensity to sign an agreement rather than the other way around. Also, investigating if monitoring leads to ratification would be an interrelated but different research question calling for a different theoretical explanation of the factors that foster ratification. For example, Hollway (2015) shows that the design features of agreements influence the ratification behaviour of signatories in the domain of international fishery agreements.

For the construction of the cooperation network, I rely on the database gnevar[22]. The freshwater agreements that are included in the database integrate several existing databases for freshwater agreements such as the Transboundary Freshwater Dispute Database (TFDD) presented by Wolf, (1999), or Mitchell's (2002) international environmental agreements. For the analysis, I focus on all freshwater agreements in Europe for which there is a title or a treaty text available in the database. Because many agreements do not only concern one issue, as the codings in the TFDD presume, a wider definition of water quality agreements seems appropriate.

I include all primary agreements treaties that contain keywords "pollut", "toxic", "waste", "protect", "environm" and "nitrat" in the treaty texts or the treaty title. Because international agreements often have more than one major purpose[23] using such a definition is more appropriate meaning that more agreements that actually concern water quality related issues can be included. This makes the analysis more complete while, at the same time, being clear replicable for other studies. Because amendments, protocols or other forms of cooperation would require a different theoretical explanation, I include only primary agreements.

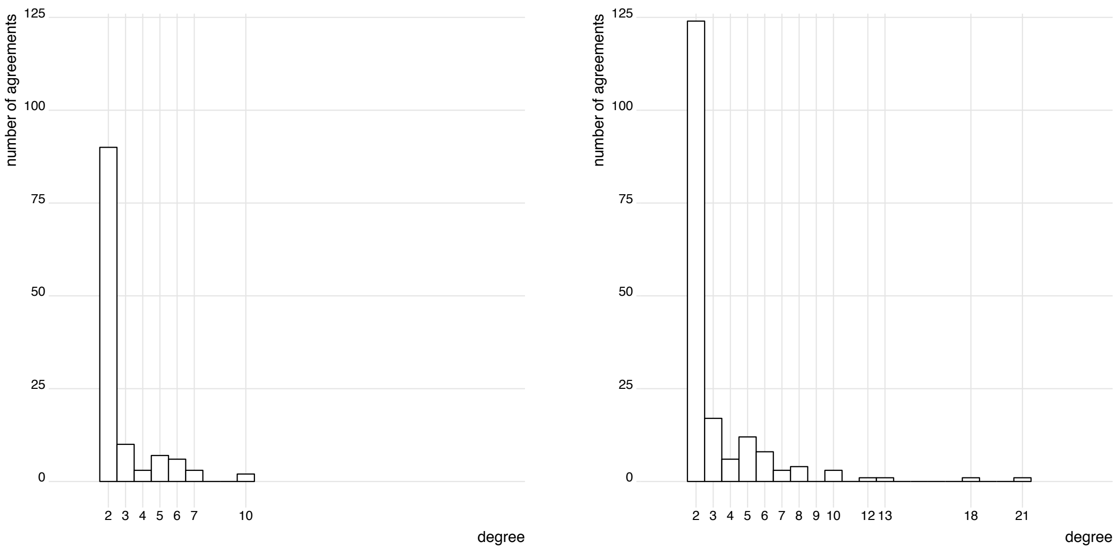
The agreements in the sample, for example, include the 1963 Agreement On The International Commission For The Protection Of The Rhine Against Pollution and the 1976 Convention On The Protection Of The Rhine Against Chemical Pollution which failed to prevent the Sandoz accident discussed in Section 2 case on the Rhine. The agreements, however, do not include EU directives such as the Water Framework Directive from 2000 because the legal nature of directives is different from international agreements. Directives are merely a recommendation of non-binding nature and therefore distinct from international agreements.

Figure 6 shows the degree distribution for the treaties for the water quality agreements in the network of 1971 and 2012. In 1971, there are a total number of 121 treaties in the network 90 are bilateral and 31 treaties are multilateral. Thus, in 1971, only 25.6% of all agreements are multilateral. There are 124 agreements with two signatories. Multilateral agreements sum to 66 agreements. Thus 34% of all agreements are multilateral

[22]Version 07.01.19 which Prof. Dr. Hollway generously provided me with. This database includes agreements in the domain of environmental policy, trade, military alliances and fisheries. The database is currently under construction and therefore not yet publicly available.

[23]This is also the case international trade agreements which increasingly include non-trade issues with the environment making up the biggest share (Milewicz et al.2018).

Figure 6: Degree Distribution of Treaties in 1971 on the left and 2012 on the right




in the year 2012.

## 7.3 Constructing the Monitoring Network

To construct the monitoring network, I use all water quality gauging stations that member states reported to the EEA from the WISE Water Quality dataset (EEA, 2018). Consistent with the literature that has previously used monitoring stations' pollution measurements to estimate pollution levels (Sigmann, 2002, Sigmann, 2004; Bernauer & Kuhn, 2010), I create a 5km buffer around the border of each country included in the analysis. The geo-referenced stations that fall within this buffer are the relevant stations to measure upstream-downstream monitoring. Using stations close to the border is appropriate because the difficulty to ascribe emissions to polluters increases with the distance of the measurement to the polluter. As new tributaries enter the main river flow, pollution levels dilute. Thus a fairly close distance to the border best captures the concept of downstream states naming and shaming upstream perpetrators. Bernauer & Kuhn (2010) use the same definition of the border length. To make the results comparable to the literature, I use their definition.

To determine if a particular station is an upstream or a downstream station, some further considerations are necessary. First, not all stations close to the border are actually upstream-downstream stations in the sense of the downstream country monitoring the upstream state. In some cases, rivers intersect with the buffer around the border but do not actually cross it. In other cases, the river demarcates the border between two countries. For example, the border between Germany and France is demarcated by the Rhine or the border between Bulgaria and Romania naturally follows the flow of the Danube. Stations of countries where the river demarcates the border are not included for the construction of the monitoring network because they do not measure pollution levels on rivers in a clear upstream downstream situation.

Figures 7 and 8 show stylised upstream-downstream situations to illustrate the coding of the WISE water quality monitoring stations. Measurement stations within a five kilometer distance on each side of the border with the river actually crossing the border are defined as upstream stations as indicated by the square shape. Rotated square shapes are downstream stations. These are defined as stations that lie within the buffer downstream of the border along a river. As the top round shape in Figure 7 illustrates, stations that lie on rivers that demarcate the border are not coded as an upstream-downstream station because they are not embedded in a clear upstream-downstream setting. Stations that lie within the buffer but which do not lie on a river at the border are also not coded as an upstream-downstream stations. This includes stations that clearly lie at a river but the river does not cross the border as Figure 8 reveals. I maximise the number of coded stations by using high resolution river shapefiles (FAO, 2009)[24].

Figure 7: Border Demarcating River

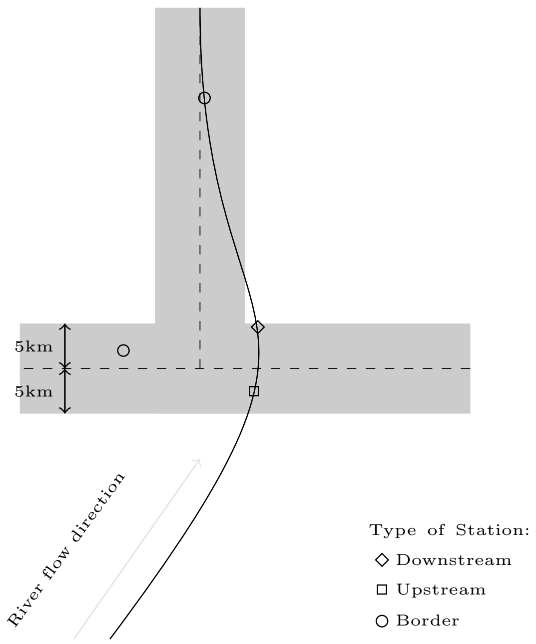
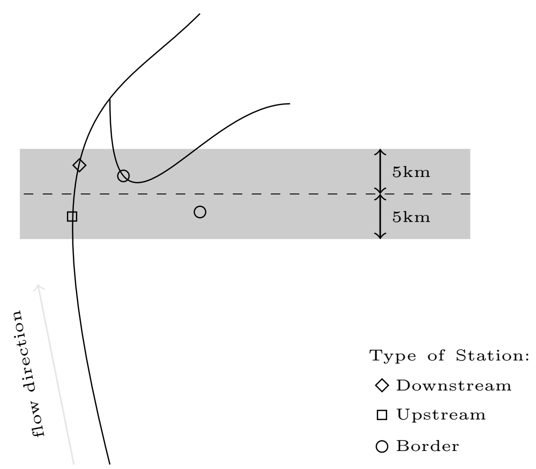



Figure 8: Border and Non-Border-Crossing Rivers




To actually code each of the stations close to the border, I construct an interactive geo-database consisting of several different layers of geo-referenced spatial objects. Additional to the rivers and gauging stations, I add river catchment systems (EEA, 2006) as a layer to my geo-database to facilitate decisions on the coding of the embedding into the upstream-downstream situation structure of a monitoring station. I code stations based on the visual inspection of the maps with interactive zoomable maps. In this way, it is possible to access information about a specific measurement station such as the gauging station identifier. With this identifier, it is possible to code observations of different years at once.

Of all the 22’123 unique measurement stations which have been reported to the EEA WISE Water

[24]I use R Studio (RStudio Team, 2018) running on R version 3.5.0. for all calculations of geo-referenced data, the plots including the statistical analysis based on the geo-referenced coded upstream downstream stations as a network.

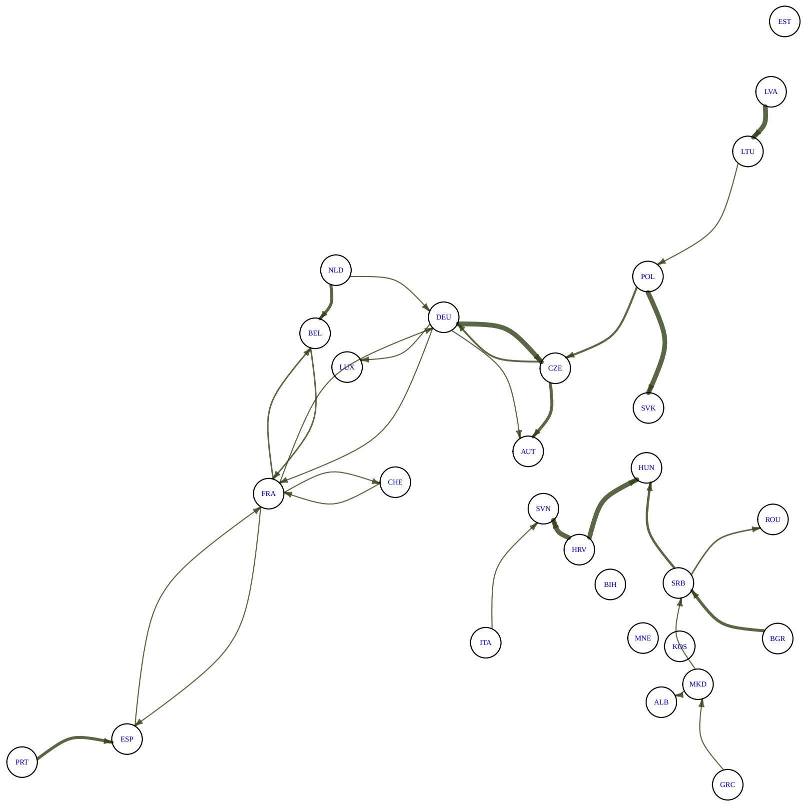
Quality dataset (2018), 18'953 stations are in states that are included in the sample. Thus, I include 86% of all stations in Europe to determine if they fall within a 5 km buffer around the border. Of these, 5.9% of the stations fall within the buffer. This is actually not a low number considering that the area of the buffer around the border sum to an overall 3.9% of the total land area and is likely an indication that externalities are important along international borders. The absolute number of stations close to the border is 1'117. Based on the coding scheme introduced above, 688 stations measure the water quality of rivers that cross the border and 796 stations in the buffer are not situated in a clear upstream-downstream setting. Thus, the total number of upstream-downstream stations account for a total of 3.6% of the stations in the sample of cases selected for this study.

Coding all stations close to the border in a dyad dichotomously could have been done easily with the data set by Furlong & Gleditch (2003). This approach would, however, have missed reciprocal relationships in dyads where both states monitor each other. Linnerooth (1990) also notes that, despite its importance for cooperation along the Danube, the upstream-downstream situation is often much more difficult to determine than it may seem in theory. The Border between Austria and Germany is a case in point. While the mainstream flows from the Black Forrest to Austria, major tributaries demarcate the border between Austria and Germany while the Danube itself crosses the border where tributaries enter. Therefore, the stations in the main flow of the Danube are not upstream-downstream stations because they also measure the flow of the tributary. While such an Austrian station does measure pollution levels from Germany to some extent, it also measures Austrian emissions[25]. Hence, the upstream-downstream situation is not clear and I do not code it as such. This is important because I want to control for the problem structure to be able to compare upstream-downstream situations in one country with similar other upstream-downstream situations in other countries. It is only when controlling for these dissimilarities that statistical analysis can yield insight into generalisable tendencies across many cases.

Figure 9 shows the monitoring network for the year of 2012. The plot shows a particularly strong monitoring activity in Western Europe. Weighted ties measure the number of stations that a state has downstream of another state. These ties can thus capture if the downstream state either names and shames other states into agreements or if it uses downstream monitoring to track the compliance of upstream states with agreement provisions. As an example of how to interpret this plot, consider the node named 'FRA' in the centre of the figure. The directed tie to 'BEL' shows that France monitored Belgium in the year of 2012. This relationship is reciprocal as Belgium also sends a tie to France. The Balkan states in the Danube basin form a separate cluster with less intense monitoring activity than the central European states. In the year of 2012, only Estonia, Bosnia and Herzegovina and Monte-Negro do not have any stations positioned downstream of any other state. Totally, there are 96 possible ties, one for each of the 96 contiguous country dyads as non-contiguous dyads cannot monitor each other.

[25]Figure 13 in the Appendix provides an example of coded upstream-downstream stations layered with the rivers used for coding.

Figure 9: The Monitoring Network in 2012. Wider edges represent more measurements




## 7.4 Describing the Networks in Different Time Periods

The dyadic correlation for the number of shared treaties and a tie in the monitoring network is 0.24. The correlation calculation includes all dyads in the monitoring network matched with the same dyads in the projected cooperation network. Thus, the correlation calculation is based on dyadic cross-network ties. Note that, for modelling, the weighted monitoring network is abstracted to a binary network which is necessary for the SOAM to be able to incorporate monitoring as one dependent network. The correlation suggests that the proposed mechanisms of the *shaming to agreement hypothesis* and the *tracking compliance hypothesis* are plausible. A plot of the cooperation and the monitoring network over time can give some more insight into the sequentiality of monitoring and cooperation.

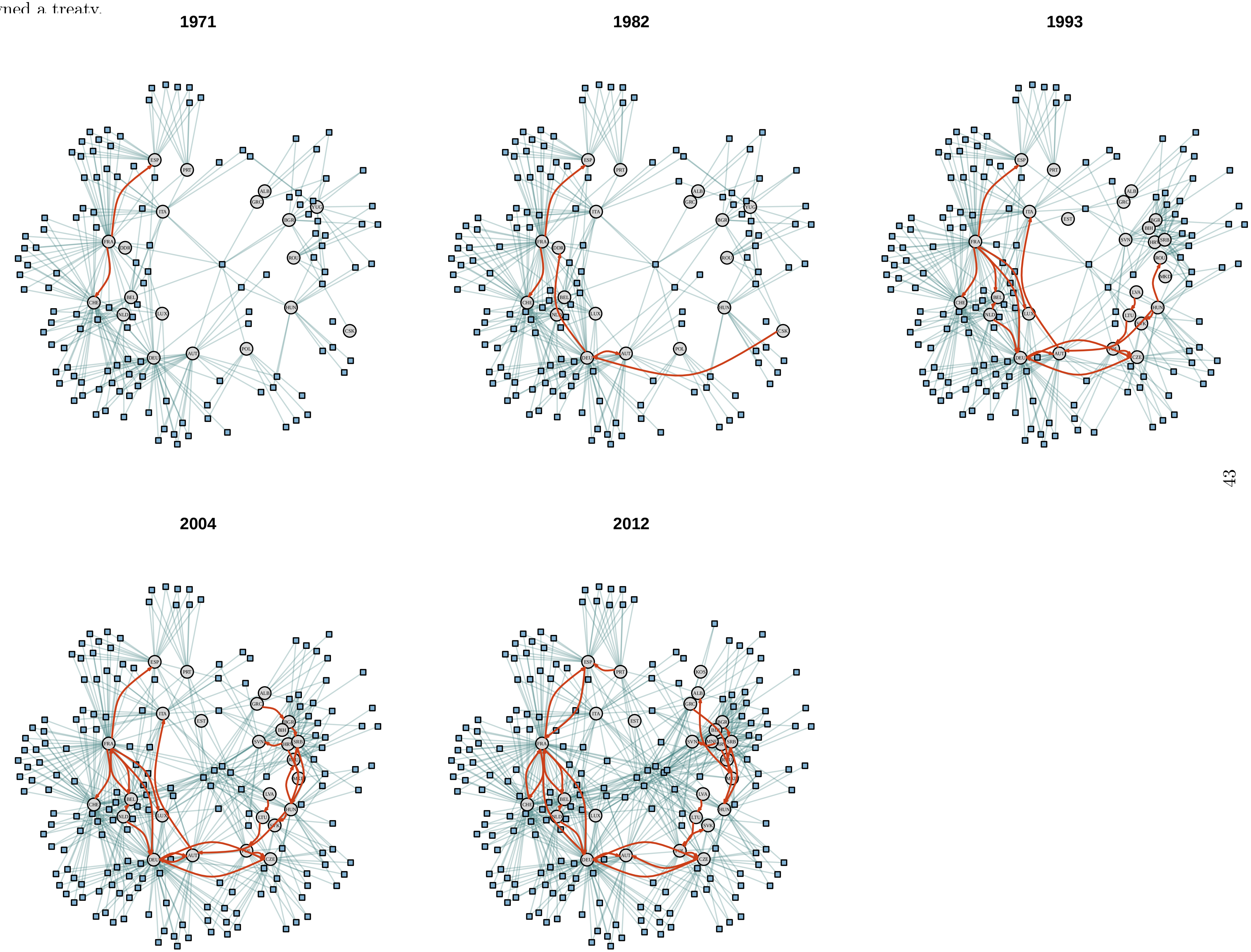
Figure 10 shows the networks over time. The main message of this plot of the two layered networks in five graphs is that most cooperation tie changes occurred *after* monitoring network changes and not *before*. In combination with Figure 4 in Section 3.1, one can see that while many treaties were signed from 1995 to 2015, Figure 10 illustrates that tie changes in the cooperation network occurred before 1993. Thus, combining insights from these two plots is a first indication in which direction this relationship likely runs.

Further, the density of this network increased substantially over time with the monitoring activity mainly clustering around states that already have many cooperative arrangements. Reported measurements for the states in the sample started in with France monitoring Spain and Switzerland. In the period of measurement, the Federal Republic of Germany started reporting pollution levels from the DDR, Austria and Czechoslovakia. Around the beginning of the 90s, major changes in the composition of states occurred. I discuss implications in more detail in Section 6.2. These states are much more sparsely institutionalised and also do not report as intensely as other states.

Before 1971, cooperation clusters mainly around bilateral treaties with multilateral treaties having mostly three members. However, from 1993 to 2012, more multilateral treaties with many members are observable in the centre of the graph. A potential explanation for this phenomenon is that shared costs in bilateral treaties reduce costs for future cooperation through multilateral institutions. In combination with the reduced political and ideological disparity, an increasing involvement of EU policies, such as the WFD, may explain this shift.

Turning descriptives of the data as used for modelling, the cooperation Network consists of $N \times A \times T$ observations, $N$ being 31 states, $A$ being 190 Agreements and $T$ being the number of waves which are 5. Thus the Cooperation Network consists of $31 \times 190 \times 5 = 29'450$ observations. The monitoring network consists of one node set. Thus the data has dimensions $N \times (N - 1)$ in each wave because self-ties are not possible. For all waves there are thus $31 \times 30 \times 5 = 4'650$ observations. A more detailed description of the cooperation and the monitoring networks used for model input can be found in the Appendix.

Figure 10: Coevolving Cooperation and Monitoring Networks from 1971 to 2012. Circles represent states and squares represent treaties. All states without ties were removed. Curved, directed edges show that a downstream state monitors an upstream state. Straight edges show that a state has signed a treaty.



# 8 Results

Table 9 provides the result for the coevolution of the monitoring and cooperation network. Parameter estimates are in log-odds. Transforming log-odds into odds ratios by taking the exponent of the parameter estimate yields interpretable quantities of interest, holding all other factors in the model constant. For structural effects, odds ratios yield the effect that is *only* due to an added tie count to *this* particular effect even if this tie may also imply a count on a different statistic.

Significance testing can be done with conventional t-tests by dividing the parameter estimate by the standard error because the distribution of parameter estimates follows a normal distribution (Ripley et al. 2019). If the obtained absolute value is bigger than 1.96, we can, under the null hypothesis of no effect, be at least 95% confident that the estimate is different from zero in a two-sided test. All reported results for varying parameters are based on simulations for which the overall convergence ratio was below 0.25 and the t-value convergence ratio below 0.1 with the Methods of Moments estimator.

The cross-network effects measure if states' choices in one network relate their embedded configuration in the other network. The effects in the cooperation network measure states' preferences to join international water quality agreements. Effects in the monitoring network capture downstream states' choices to monitor upstream states. For the interpretation of substantive quantities of interest, I focus on Model 4 because it tests all relationships in the same specification. Models 1-3 show that the parameter estimates and the significance levels are relatively similar across different specifications.

*Cross-Network Effects*

Beginning with the cross-network effects which test for the endogenous relationship between cooperation and monitoring, the evidence suggests that the relationship runs mainly in one direction but not in the other. Upstream states' choices to sign international water quality agreements are influenced by downstream monitoring but downstream states do not track upstream compliance once they have signed an agreement. As the coefficient for SHAMING TO AGREEMENT in all four different specifications reveals, the effect is positive and significantly different from zero. In Model 4, which can test for all mechanisms introduced in the theory in the same specification, the odds of joining the upstream state joining an agreement with the downstream state are 25.53 times higher when the downstream state engages in naming and shaming of upstream perpetrators. Thus the evidence supports the *Shaming to Agreement* hypothesis not only with significant results but also with a large substantive effect.

The coefficient of TRACKING COMPLIANCE which tests the mechanism that downstream states may monitor upstream compliance with international agreements after having signed an agreement is not supported at the conventional 95% confidence level in any of the specifications. These results are consistent with BBK who find that MEAs are not related to the number of stations close to the border. Their model however only tests for multilateral agreements while this study includes both bilateral and multilateral agreements that specifically refer to water quality related issues. But they find a positive and significant relationship between

Table 4: Coevolution Estimates

| | Model 1 | Model 2 | Model 3 | Model 4 |
|---|---|---|---|---|
| *Cooperation Network (Y)* | | | | |
| Density | 3.08 | 0.01 | 4.33 | 0.01 |
| | (2.50) | | (8.90) | |
| Shared Costs | 0.04** | 0.03** | 0.04** | 0.03** |
| | (0.01) | (0.01) | (0.01) | (0.01) |
| Multilateral Popularity | −2.31*** | −1.68*** | −2.32*** | −1.68*** |
| | (0.21) | (0.24) | (0.21) | (0.23) |
| Popularity Costs | −0.02 | 0.08* | −0.02 | 0.08* |
| | (0.04) | (0.03) | (0.04) | (0.03) |
| GDP Ego | | 0.16 | | 0.15 |
| | | (0.18) | | (0.18) |
| EU Candidate Ego | | −1.55* | | −1.54* |
| | | (0.78) | | (0.76) |
| EU Member Ego | | 2.46* | | 2.58* |
| | | (1.22) | | (1.22) |
| Democracy Ego | | 0.21*** | | 0.22*** |
| | | (0.06) | | (0.07) |
| *Cross Network (X and Y)* | | | | |
| Shaming to Agreement | 2.53** | 3.06* | 2.71* | 3.24* |
| | (0.92) | (1.27) | (1.09) | (1.40) |
| Tracking Compliance | 0.74† | 0.77† | 1.21 | 1.26 |
| | (0.42) | (0.41) | (0.82) | (0.98) |
| *Monitoring Network (X)* | | | | |
| Density | −1.71* | −1.75* | −2.00 | −2.01 |
| | (0.79) | (0.76) | (1.41) | (1.41) |
| Reciprocal Montoring | 0.12 | 0.11 | 0.31 | 0.32 |
| | (0.45) | (0.45) | (0.60) | (0.59) |
| GDP Ego | | | 0.52 | 0.53 |
| | | | (0.48) | (0.56) |
| EU Candidate Ego | | | −0.03 | −0.02 |
| | | | (0.90) | (1.00) |
| EU Member Ego | | | −0.93 | −0.94 |
| | | | (0.90) | (0.95) |
| Democracy Ego | | | −0.59 | −0.64 |
| | | | (1.34) | (1.56) |
| Iterations | 9807 | 10548 | 10548 | 11217 |

*Note*: the parameter for Density parameter is fixed in Model 2 and 4. I discuss implications under *Limitations*
$^{***}p < 0.001$, $^{**}p < 0.01$, $^{*}p < 0.05$, $^{\dagger}p < 0.1$

MLAs and the number of stations that are not close to the border. Therefore the existing results in the literature show that international environmental cooperation matters for states' decision to monitor domestically but not for their decision to monitor upstream states. But downstream states may engage in naming and shaming of upstream perpetrators to leverage them into an agreement.

*Structural Cooperation Network Effects*

Turning to the cooperation network, all but three coefficients across the four different specifications have the expected sign. The coefficient for SHARED COSTS which measures if states that already share membership in different international water quality agreements have lower costs for future cooperation is positive and significant across all specifications. The full Model 4 suggests that the likelihood of a state joining an international agreement with another state with whom it already shares third treaties is 3% higher. This is a surprisingly low effect considering the high amount of bilateral clustering which is visible from network plots such as Figure 10. Yet, most of these tie changes occurred before 1971. Still, these results are consistent with path-dependency which suggests that average costs of cooperation are decreasing with the number of additional cooperative arrangements making future cooperation more likely (Pierson, 2000).

MULTILATERAL POPULARITY shows that states have a negative tendency to sign multilateral agreements. Specifically, states are 81% times less likely to sign multilateral agreements. This is not surprising given that bilateral basins are geographically bound and often have only few states that drain into the same international basin. Therefore, management solutions can often be found more efficiently on a bilateral basis than multilaterally. Yet, this result is interesting given that the vast majority of treaty signatures in this study were for multilateral agreements.

POPULARITY COSTS is significant and positive once non-structural covariates are added to the Cooperation Network, but without covariates it remains insignificant. This effect captures if states wish to tie to agreements that already have many members. Model 4 suggests that for every third state who has previously signed an international agreement, the odds of a state to also sign this agreement increases by 8%. Interestingly, despite water quality agreements being traditionally rooted at the basin level with often few states that share river basins, the models that add covariates for the Cooperation Network suggests that when states decide to sign multilateral treaties there is, to some extent at least, a rich get richer phenomenon.

In summary of the structural network effects in the cooperation network, MULTILATERAL POPULARITY and POPULARITY COSTS suggest that bilateral treaties are less costly but that once a treaty has many signatories, it becomes attractive for more states to sign. SHARED COSTS show that shared third agreements make future cooperation less costly.

*Cooperation Network Covariates*

Model 4 Table 9 includes four monadic covariates additionally to the structural network effects. Starting with (log) GDP EGO, this effect remains insignificant throughout the specifications. Contrary to the expected relationship, EU CANDIDATES are significantly less likely to cooperate on water quality related issues than non-candidates[26] . The odds of not sending a tie are 79% higher than the odds of sending a tie in the cooperation network when states are candidates. This result is surprising because candidates could use water quality agreements to signal cooperative behaviour and their readiness to join the EU.

[26]The baseline category is a non-EU candidate and therefore also a non-EU member

As expected, EU MEMBERS are significantly more likely to cooperate. In substantive terms, the odds of sending a tie when the focal actor is an EU member are 13.20 times higher than when the actor who reconsiders his local network configuration is not an EU-member. These results provide evidence for two mechanisms on EU membership. The first is that the institutions of the EU provide channels for information exchange which result in lower average costs of cooperation. The second mechanism that may explain this result is that the higher requirements for environmental policy commitments are translated into observable policy output through international water quality cooperation.

The effect of DEMOCRACY is highly statistically significant. The coefficient estimate has the expected positive sign, suggesting that for every unit increase in the Democracy score ranging from complete autocracy (-10) to complete democracy (+10) states are 24.6% more likely to cooperate on water quality. These results suggest that democratic leaders are more likely to provide the public good of environmental protection. Less democratic regimes' rational leaders are unlikely to decide on non-exclusive policies that create spillovers to groups whose support is unnecessary for leaders. But democratic, re-election seeking leaders are more likely to provide non-exclusive services because to a larger population benefits on whose votes politicians rely to stay in power (Deacon, 2009).

*Monitoring Network*

All effects across all specifications for the montoring network except DENSITY in the first two specifications remain insignificant. The DENSITY effect measures the general tendency to form ties. It functions like an intercept in conventional statistical methods. The negative estimates in Model 1 and 2 in Table 9 indicate that monitoring upstream states is costly.

The insignificant results in Model 3 and 4 mean that the chosen statistics do not bear any explanatory power for the tie formation process in the observed monitoring network. Yet, for example, the theoretical explanations of DEMOCRACY both empirically (e.g. Neumayer, 2002) as well as theoretically (e.g. Deacon, 2009) is well established in the literature. Therefore, it seems unlikely that insufficient theoretical explanations are responsible for insignificant results. What seems a more pertinent explanation is that states report measurement data relatively irregularly as it is often the case with geographic data coming in unbalanced panels.

*Limitations*

Note that Model 2 and 4 have the density parameter fixed at the initial value to achieve model convergence. Convergence means that the Methods-of-Moments estimator can obtain estimates for all the specified effects that remain on a stable locus over different simulation iterations. Divergence means that the estimator tries to assign very different parameter estimates to those that are observed in the real network in successive simulation iterations. I pursued a variety of different modelling approaches, specifications and covariates to avoid a divergent algorithm. Fixing the DENSITY parameter solves these convergence problems even if it is not the preferred solution because the DENSITY parameter captures, like an intercept, the general tendency of actors to form ties. I made the following steps to avoid divergence.

First, I tried to fit a coevolution model (Snijders, 2012). This model would have estimated the endogenous relationship between domestic monitoring and states' decisions to sign water quality agreements similarly to Manger & Pickup (2016). To reduce the high number of choices that states have (190 because there are 190 agreements) I transformed (i.e. projected) the two-mode network into a one-mode network, where states, instead of joining agreements, tie to other states. Ties to other states show common membership in international treaties. But projecting the network did not improve matters because SAOMs use binary ties for modelling. The extent to which the projection reduced the amount of information made it impossible to estimate such a model of coevolving actor attributes and network changes.

Second, for the model presented here, I used different variable specifications. I added parameter estimates such as outdegree activity or truncation effects to account for differential tendencies in signing treaties. In fact, I used all the different structural effects that are available for this data algorithm combination based on theoretical considerations and the inspection of network plots so that my specification can account for salient network processes. I increased the number of simulations including the step-size which the algorithm uses to approximate the observed value of the statistics in the network. Adding or leaving out covariates also did not lead to a solution. I specified time dummies to account for differential tendencies in the data. Further, I changed the initial value for the parameter estimate to equip the algorithm with a different potential pathway by which it can approximate the true value using conditional as an unconditional estimation technique. Repeated estimations of previously unconverged (but in these cases non-degenerate) specifications also did not yield desirable simulated network properties.

Third, adding dyadic covariates such as contiguity or shared boundary length in the two-mode network which I constructed from the historical boundary dataset (Weidemann et al. 2010, Weidemann & Gleditch 2010) is not possible. Covariates have to have the same dimension as the two-mode network[27]. Therefore, estimating these covariates would only have been possible for the projected network. But for reasons outlined above, I could not use the projected network.

Fourth, using coded upstream-downstream stations as a network substantially facilitated estimation. Yet, when specifying covariates additionally to structural effects, convergence problems reappear. According to Ripley et al. (2019) fixing parameters can sometimes help with divergent algorithms. When fixed, meaning that the estimate does not vary during estimation to determine if its observed value is similar to the simulated value, no standard errors can be obtained for the pre-specified value. I also tried fixing the DENSITY estimate of Model 4 at the value of the DENSITY estimate of Model 3. This approach leads to divergence of other parameter estimates. Therefore, I fix DENSITY at the initial value of 0.01 in Models 2 and 4.

Finally, there were issues with obtaining a well-fitting model. The Goodness of Fit (GOF) statistic serves to verify if the simulated models replicate important features of the observed network. However, "the method of joiners and leavers for representing composition change [...] does not combine properly with the

[27]the two-mode network has dimensions $31 \times 190$ and the dyadic covariates have the dimension $31 \times 30$

SIENAGOF function" (Ripley et al. 2019, p.58). Therefore I analyse the goodness of fit qualitatively.

*Qualitative Assessment of the Goodness of Fit in the Cooperation Network*

To assess the goodness of fit qualitatively, I compare a typical simulated network to the observed cooperation network. This comparison has the advantage of being more intuitive and simple to understand. The goodness of fit is important to assess if the model is able to replicate important structures of the network. Yet, some previous scholars did not analyse or at least not discuss the fit of their model. For example, Manger & Pickup (2016) do not discuss the GOF for their model which estimates the coevolution of democratisation and PTA formation. I communicate the GOF of my model.

Figure 11: Simulated and Observed Network in 2012.

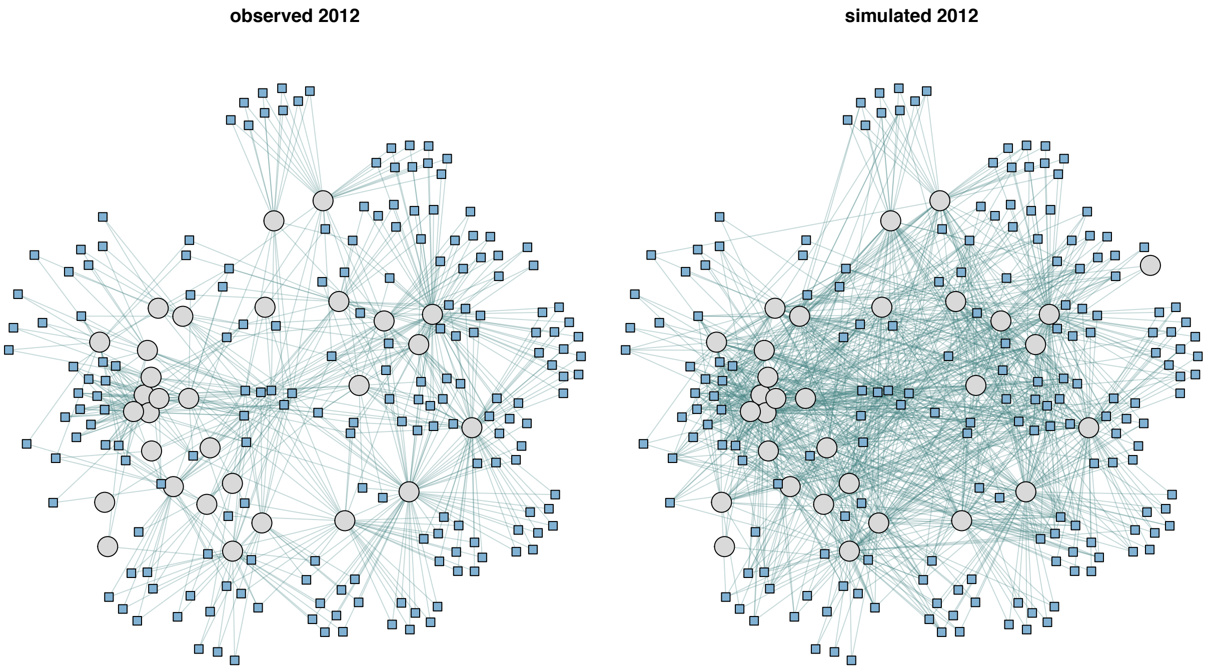



Figure 11 plots the observed cooperation network in 2012 and a typical simulated cooperation network in 2012 side by side. These plots show that there are considerable differences between the observed and the simulated network. This is problematic because the simulated network's tie structures do not resemble the observed network. I could easily have retrieved and plotted the degree distribution for these two networks or even for all simulated networks but these simple plots show the differences already quite impressively.

To improve the fit of the model I undertook a variety of steps. I fitted specifications with a different set of effects using the same and different periods for simulation. I left out the endogenous effects that actually should be testable to empirically verify evidence for or against my theory. I fixed the DENSITY parameter to a large negative value, to decrease the modelled willingness of states to form ties. A variety of different algorithm specifications also did not improve the matter. In most cases, they lead to a divergent algorithm. I systematically estimated different combinations of structural effects for the simplest Model 1 in Table 10 going through all available structural effects because structural effects can account for structural network properties. None of these steps improved the GOF but produced similar networks to the simulated network on the right in Figure 11. Important to note is that the substantive effect for the SHAMING TO AGREEMENT coefficient remained stable and significant at the 99% or the 95% confidence level.

Potential reasons for the fit of the model are the following. Although the correlation of the number of shared treaties and the number of downstream monitoring stations in the same dyad is 0.24 there is little change in the network compared to the number of existing treaties. Thus, these problems are unlikely to be caused by insufficient empirical data basis or incorrect theoretical explanations. Both, case evidence and results from a simple QAP cross-sectional logistic regression for 2012 which are, however, not shown here, provide evidence for the *Shaming to Agreement Hypothesis*.

Another explanation may be that there are many tie choices in the cooperation network compared to the number of actors making it difficult for the algorithm to model actors decisions about which treaties to join. This may be because, as one can see from equations (7) and (8) in Section 5.2, actors evaluate their tie changes based on all possible treaties they could become members of. A large number of choices can cause difficulties, especially when there are few tie changes compared to existing ties. As explained in the previous section, to reduce the high amount of information and alleviate these shortcomings, I transformed the two-mode network (states tie to treaties) into a one-mode network (only states tie to states). Weighted ties in this transformed network would then have represented the strength of the cooperative relationship between two actors. But because the SAOM cannot handle weighted ties (it may be possible to model weighted ties as a single tie in different networks, which would however only complicate rather than simplify the matter) this strategy resulted in little change in the cooperation network because European Water Quality cooperation is highly institutionalised.

# 9 Conclusion

This thesis contributes to the research of international river basin cooperation from an interdependence perspective. Based on previous theoretical (Mitchell & Keilbach, 2010), quantitative (Beck et al. 2010) and qualitative (Le Marquand 1977; Kiss 1985; Schwabach; Bernauer & Moser, 1996; Linnerooth-Bayer & Murcott 1996; ) research, I develop hypotheses on the endogenous relationship between cooperation and monitoring in Europe. Underlining previous scholars disregard for the potentially endogenous relationship between cooperation and monitoring, I develop theoretical explanations for how downstream monitoring and cooperation unfold in European river basins. Contextualising cooperation in Europe, I show the rising relevance of international cooperation for monitoring. Two succinct cases on the Rhine and the Danube discuss the importance of cooperation for water quality monitoring. International basin management institutions, created through international treaties, attempt to solve asymmetric externalities between upstream and downstream states (Linnerooth-Bayer & Murcott 1996; Ovedenko 2016). Members of these treaties engage both in shared and individual monitoring. Based on monitored water quality measurements, international river basin management institutions provide recommendations for future cooperative arrangements aimed at solving negative effects resulting from upstream-downstream externalities.

With individual insights from cases, I make two major theoretical propositions. The first is that *downstream states name and shame* upstream perpetrators by reporting downstream water quality measurements to leverage them into cooperative agreements. The second part of this endogenous relationship is that once states have signed treaties, downstream states are inclined to *track the compliance* of upstream states' treaty provisions. The mechanism that connects these constituents is that downstream reporting increases reputational costs for upstream states to externalise pollution to downstream states. *Before* an agreement is reached, upstream states can alleviate these reputational costs by committing to agreements aiming at the reduction of negative effects from upstream to downstream states. *After* an agreement is reached, downstream states could decrease incentives for upstream states to defect from treaty provisions.

I construct two interdependent networks. The first network consists of states who join water quality treaties. The second network consists of downstream states who monitor upstream peers along international rivers. Focussing on central Western and Eastern European states for the time frame of 1971-2012, I construct a directed, weighted monitoring network based on geo-referenced monitoring stations. I identify more than 1'000 stations close to the border. Layering these stations on fine-grained river shape files allows for coding of the embedding of these stations in the upstream-downstream structure along international rivers. These coded stations allow me to present a novel research design: I construct a weighted, directed monitoring network based on the number of downstream stations close to the border, from each downstream state to each contiguous upstream state.

To test if these theoretical expectations hold generally across European river basins, I make use of recent advances in statistical network modelling. I use an extension of the SAOM proposed by Snijders et al. (2013) that allows for disentangling potentially endogenous relationships across different networks. Are ties in the cooperation network more likely when a tie in the monitoring network already exists? Or, are ties in the monitoring network more likely once a tie in the cooperation network exists? The SAOM can model such endogenous relationships by decomposing network changes into its smallest components. Contrary to conventional statistical methods which assumes independent observations, the strength of this model family is that they can, consistent with theories of an interdependent world, exploit dependence instead of treating it as a nuisance. This approach can generate rich insight into the sequentiality of events, how interdependencies unfold and what they mean for cooperation and monitoring in international river basins.

The results confirm the first part of the relationship between the monitoring and the cooperation network modelled as a co-constitutive process. Downstream victims engage in *naming and shaming* of upstream perpetrators to leverage them into international water quality agreements. But downstream victims do not increase their monitor behaviour towards upstream states to *track compliance* after international water quality agreements were signed. The findings also suggest that EU membership and democracy matter for cooperation, while GDP has no explanatory power. Path-dependency in cooperation shows that previous institutional frameworks are important for successful future cooperative arrangements with bilateral treaties being less costly for water quality management. Only EU candidacy runs counter to the theoretical explanations in the cooperation network which may be a sign that EU incentives for candidates do not create spillovers to international water policy. Due to the unbinding nature of EU water policy, it may be, that instead, candidates substitute their cooperative signals to other policy domains that are of greater direct relevance for accession. The internal validity of this study is however restricted by the fit of the model which, despite major efforts, could not be satisfactorily improved.

This research offers an interesting point of departure for future studies on international river basin cooperation from an interdependence perspective. First, it will be interesting to analyse water cooperation separately across different river basins. Combining insights from specific basins with qualitative research could yield a more fine-grained understanding of causal mechanisms in specific basins. Summarising case-study oriented network models into one comprehensive and generalisable framework could then be done using a meta-analysis of the individual basin cases based on the same explanatory factors. Meta-analysis is implementable with a SAOM as well. Second, future research could look more closely into why there has been a rise in multilateralism in European river basin cooperation. Such research could reveal interesting insight into increasing interdependencies at the regional level. Is the decreasing cultural divide, for example in the Danube, responsible for increased multilateral cooperation there? How do such explanations fare in comparison to geographic explanatory factors such as contiguity or shared border length that from theoretical perspective could expected to matter?

In conclusion, this thesis has contributed to empirical and theoretical research on international water cooperation from an interdependence perspective. Consistent with the theory, I employ a statistical model that is capable of directly exploiting these interdependencies in the empirical model. Inspired by previous literature on pollution externalities (Sigmann, 2002; Sigmann, 2004; Bernauer & Kuhn, 2010, Beck et al. 2010), I present a novel operationalisation of monitoring as a directed network which I constructed by coding geo-referenced water quality monitoring stations for their upstream-downstream position. The results have implications for policymakers. In Europe more generally, downstream states should report their measurements to the EEA because naming and shaming upstream perpetrators can help to achieve cooperative arrangements along international rivers. Creating scientific measurements can increase prospects for cooperation. Because monitored pollution levels are hardly a random sample, researchers need to be cautious when interpreting findings generated from water quality gauging stations.

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# Appendix

## GIS Data Sources

The following table shows the data which I used to determine if a geo-referenced river water quality measurement station is situated upstream or downstream of an international border. The geographic data that I used for this analysis mainly consisted of spatial points, spatial polygons and spatial lines. All of these objects are composed of points which are then connected to either form a line or a polygon. If the line ends at the point where it started, the object is called a polygon. In my case, the polygons contain the delineation of two-dimensional country borders. The buffer around the border is also a polygon. Rivers are stored in the form of lines and gauging stations in the form of points. Each of these elements must contain information on the assumption of the rounding of the earth which is called projection. To minimise the distortion which arises I therefore use the European projection EPSG:3042 for calculations.

Table 5: Spatial Objects

| Geometry | Description of the data | Source |
|---|---|---|
| *Gauging stations* | Contains of geo-references monitoring stations reported by states to the EEA which integrates the data into the WISE Water Quality Database | EEA (2018) |
| *Rivers* | River shapefiles used to determine if a station close to the border is an upstream or downstream station for the construction of the monitoring network. These shapefiles are based on HydroSHEDs. | FAO (2009) |
| *River Catchments* | Used to facilitate coding of upstream-downstream stations. Used as the second decision criterion for coding upstream-downstream stations. When the stations do not lie within a five kilometre distance of an international river, river catchment polygons are used. In most cases however | EEA (2006) |
| *River Basin Delinitation* | International river basin delineations are available from the Transboundary Freshwater Dispute Database. | McCracken & Wolf (201 9) |
| *Country Delineation of Historical Boundaries* | Includes historical country delineation making it possible to retrieve country polygons at specific dates. | Weidememann et al. (2010) and Weidemann & Gleditch (2010) |

## Network Modelling Data

Table 6: Operationalisation of Covariates

| Concept | Description | Source |
|---|---|---|
| Cooperation Network | This panel network measures in each year if a given states has signed a treaty in the cooperation network. Ties are binary in this network and remain equal 1 for subsequent years. | unpublished gnevar, 07.01.19 version constructed by Prof. Dr. Hollway |
| Monitoring Network | Measures the number of stations close to the border to construct a weighted, directed monitoring network. For the modelling, I abstract to a binary relationship, because SAOMs can best handle binary ties. | EEA (2018) |
| EU member | Dichotomous variable measuring for each year if a state was EU member. EU members, due to their institutional interconnectedness face lower opportunity costs from cooperation | europa.eu[1] |
| EU candidate | Dichotomous variable measuring for each year if a state was EU candidate. EU candidates have higher incentives to cooperate. EU candidates also have higher incentives to comply with EU legislation such as the WFD | europa.eu[1] |
| Democracy | The democracy is measured based on an index ranging from -10 (full autocracy) to 10 (full democracy). Democratic leaders are more likely to be responsive to the wider public's preferences making environmental policy output more likely. | Marshal et al. (2016) |
| log GDP | GDP in current US dollars captures states preferences for environmental protection that are due to economic development | worldbank.org[2] |

[1] https://europa.eu/european-union/about-eu/countries_en#tab-0-1, accessed on the 01.04.19

[2] https://data.worldbank.org/indicator/ny.gdp.mktp.cd, accessed on the 01.04.19

## SAOM Input Network Description

*Cooperation Network*

Table 7 shows the tie changes in the undirected two-mode cooperation network. The matrix of possible observations over all time periods consists of 5'890 observations[28] in each of the four time periods. A total of 281 tie changes occurred from 1971 to 2012. As the third column in Table 7 shows, most tie changes occurred in the period from 1993 to 2003 and fewest in the preceding time period from 1983 to 1992. I do not model tie dissolution because treaty dissolution is a rarer event which would require different theoretical explanations. The distance measure is the Hamming distance which measures the amount ties that change (Ripley et al., 2019, p.20). Therefore, the Distance in Table 7 is equal to the number of formed ties in the column $0 \rightarrow 1$. The Jaccard index is an index of the similarity between networks[29]. An index above 0.3 should cause no problems for convergence during estimation. The similarity between different snapshots of the network is reasonable across different time periods. It ranges from 0.794 to 0.932 where the latter value indicates that there was little change during the 80's.

Table 7: Tie Changes in the Cooperation Networks

| | Tie Changes from $\rightarrow$ to | | | | | |
|---|---|---|---|---|---|---|
| periods | $0 \rightarrow 0$ | $0 \rightarrow 1$ | $1 \rightarrow 0$ | $1 \rightarrow 1$ | Distance | Jaccard Index |
| $1971 \rightarrow 1982$ | 5481 | 54 | 0 | 355 | 54 | 0.868 |
| $1983 \rightarrow 1992$ | 5451 | 30 | 0 | 409 | 30 | 0.932 |
| $1993 \rightarrow 2003$ | 5337 | 114 | 0 | 439 | 114 | 0.794 |
| $2004 \rightarrow 2012$ | 5254 | 83 | 0 | 553 | 83 | 0.869 |

*Note:* The calculations include all 31 states in each of the periods.

In the two-mode network, only ties from states to treaties are possible by definition. The density defined as the number of observed edges observed edges $E$ of the network $Y$ at time period $t$ divided by the number of possible edges which are given by the size of the two node sets. This means that there are a total number of $31 \times 190 = 5'890$ observations per year. The observed number of edges in 2012, the last year included in the analysis, are $E(Y_{t=2012}) = 636$ signed agreements. Thus, as calculated in equation (11), the density is 10.8%, meaning that a tenth of the possible connections are actually observed.

$$density(w_t = 2012) = \frac{E(Y_{t=2012})}{N \times M} = \frac{636}{31 \times 190} = 10.8\% \tag{11}$$

The first row in Table 8, shows the density of the network over different time periods as used for estimation with RSiena. The Table shows that the density was generally increasing. The average number of ties increased from 11.452 to 20.516 ties per state with the total number of ties increasing from 355 to 636 totally signed agreements. The network thus became denser over time.

*Monitoring Network*

Table 9 shows the number of tie changes for the directed one-mode monitoring network. The monitoring network has fewer observations in each of the four waves than the cooperation network. The monitoring network consists of the same number $N = 31$ states which by definition may tie to each other but not themselves. This network also became denser over time. Only in the period from 1983 $\rightarrow$ to 1992, the Jaccard index is relatively low at 0.235. I could not use different time periods because the Jaccard index shown in Table 7 for the same period is quite high at 0.932, while most tie changes

[28] because the study includes a total of 31 actors and 190 treaties

[29] for the mathematical definition of this index, I refer to Ripley et al. (2019, p.20)

Table 8: Cooperation Network

| period | 1971 | 1982 | 1993 | 2004 | 2012 |
|---|---|---|---|---|---|
| density | 0.060 | 0.069 | 0.075 | 0.094 | 0.108 |
| average degree | 11.452 | 13.194 | 14.161 | 17.839 | 20.516 |
| number of ties | 355 | 409 | 439 | 553 | 636 |
| missing fraction | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |

*Note:* The calculations include all 31 states in each of the periods and 190 treaties.

in the monitoring network occurred in this period, the tie changes in the monitoring network stabilise afterwards.

Table 9: Tie Changes in the Monitoring Networks

| | Tie Changes from → to | | | | | |
|---|---|---|---|---|---|---|
| periods | 0 → 0 | 0 → 1 | 1 → 0 | 1 → 1 | Distance | Jaccard Index |
| 1971 → 1982 | 925 | 2 | 0 | 2 | 2 | 0.500 |
| 1983 → 1992 | 912 | 13 | 0 | 3 | 13 | 0.235 |
| 1993 → 2003 | 903 | 9 | 0 | 17 | 9 | 0.654 |
| 2004 → 2012 | 895 | 8 | 7 | 19 | 15 | 0.559 |

*Note:* The second column 0 → 1 includes non-contiguous country pairs. The calculations include all 31 states.

Table 10 shows characteristics for the monitoring network. The monitoring network as it is specified in the SAOM before composition is also increasingly dense. The density here is calculated differently than in the cooperation network. Because ties are only possible within the same node set, in this network the number of possible ties are $N \times (N-1)$. For the time period of 2012, this means that a total of 2.9% of ties are observed in any dyad.

$$density(w_t = 2012) = \frac{E(X_{t=2012})}{N \times (N-1)} = \frac{636}{30 \times 31} = 12\% \tag{12}$$

This may seem like a very sparse network. However, accounting for the ties that I define as impossible, there are only 96 ties possible because non-contiguous states can, by definition, not monitor each other. I account for this in the model by using the method of joiners and leavers as described in the subsection on composition change. The number of ties in the monitoring network increase from 2 in 1971 to 27 in 2012.

Table 10: Monitoring Network

| period | 1971 | 1982 | 1993 | 2004 | 2012 |
|---|---|---|---|---|---|
| density | 0.002 | 0.005 | 0.018 | 0.028 | 0.029 |
| average degree | 0.065 | 0.161 | 0.549 | 0.840 | 0.872 |
| number of ties | 2 | 4 | 17 | 26 | 27 |
| missing fraction | 0.000 | 0.001 | 0.001 | 0.001 | 0.001 |

*Note:* The calculations include all 31 states in each of the periods.

## Countries in the Sample

Figure 12: States in the Sample. Coloured and labelled states are in the sample. States in grey are not in the sample. Major rivers are shown in blue.

## Example of Coded Upstream-Downstream Stations in the Danube Basin

Figure 13: Example of Coded Upstream-Downstream Stations along the Danube. This plot layers rivers, the 5km buffer around the border and the stations falling within this buffer.