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Does the Primacy System Work? State vs. Federal Implementation of the Clean Water Act

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Abstract

In the United States, environmental federalism largely relies on a system for policy implementation that allows the federal government to delegate primary program authority (or primacy) to state agencies. Although it is an ingrained feature of several major federal environmental policies, such as the Clean Water Act (CWA), there is little evidence to indicate what impact delegating authorities has on programs. In order to examine this, the authors use a synthetic control technique to determine how actual CWA program outcomes in five states compare to expected outcomes if EPA retained primary authority. Findings indicate that while there were no significant differences in Texas and Oklahoma, state primacy led to improved program outcomes in Florida, but worse outcomes in Maine and South Dakota. Conclusions suggest that primacy has asymmetrical impacts that largely depend on state implementation systems, which carries important implications for environmental federalism.

In the U.S., federal environmental policy largely relies on a federal-state partnership for implementation, a key feature of which is primacy, or primary enforcement and implementation authority. Under this system, the Environmental Protection Agency (EPA) is by default responsible for ensuring policy implementation, but states can elect to take on primacy and develop programs to meet specific challenges within their jurisdictions (Crotty 1987; Woods 2006a). Proponents argue this allows policy decisions to be made as close as possible to the people they are affecting, while national standards create a safeguard for environmental quality (Woods and Potoski 2010). Conversely, opponents argue that primacy creates principal-agent problems and states use it as a bargaining chip with the federal government (Crotty 1987: Lester 1995). Although it is an ingrained feature of major federal legislation, such as the Clean Water Act (CWA), there is little evidence to indicate what impact federal versus state primacy has on environmental policies. Consequently, we consider how environmental policy outcomes under state primacy compare to those under federal primacy.

In order to address this question, we undertake a preliminary study that takes initial steps in unpacking the complexity surrounding state versus federal leadership on environmental policy implementation. We begin by introducing the background of environmental federalism and primacy as an approach to intergovernmental policy implementation. Then, we present the CWA as a case study, and discuss the challenges of examining its implementation. Finally, we use a synthetic control technique to determine how actual reported toxic chemical releases in five states compare to expected releases if transfers of primacy from EPA to state agencies did not occur. Findings indicate that in Florida, the transition resulted in fewer reported water-based toxic releases, while Oklahoma and Texas experienced no significant changes. On the other hand, in Maine and South Dakota, the transition resulted in more reported toxic releases. While our findings face several important limitations, preliminary evidence is sufficient to indicate that primacy has asymmetrical impacts on environmental policy outputs, and warrant further inquiry.

Environmental Federalism

Facing implementation challenges with new environmental policies in the 1960s, the federal government adopted a partial preemption system that allowed EPA to delegate responsibilities to state agencies while retaining authority to monitor programs for compliance with national guidelines, and if necessary, enforce national standards. While this was initially treated as a top-down exercise, states gained significant discretion under this system by the 1980s (Woods

2006b; Woods and Potoski 2010). Contemporarily, scholars paint this relationship as a "federal-state" partnership, with states electing to take on primacy (Woods 2006b; Fowler 2014). While every state currently has primacy for the Clean Air Act (CAA), EPA still retains control over: the CWA in Idaho, Massachusetts, New Hampshire, and New Mexico; the Resource Conservation and Recovery Act (RCRA) in Alaska and Iowa; and, the Safe Drinking Water Act (SDWA) in Wyoming (EPA 2018b, 2018c). In general, primacy agencies design programs and administrative rules, and develop partnerships with pollutant dischargers (as well as with local, state, and federal agencies) (Heilman, Johnson, Morris & O'Toole 1995). If state agencies have primacy, EPA retains oversight and regularly assesses programs to ensure compliance with national guidelines. Since 2004, this chiefly occurs through the State Review Framework for Compliance and Enforcement Performance (SRF), which is designed is to create clear performance expectations and provide guidance on how to improve program outcomes (EPA 2018c).

A key argument for delegating authority to states is that decentralized policy implementation allows programs to be more responsive to local political preferences (Agranoff and McGuire 2001). This is particularly important in environmental policy, since "the policy challenge underlying pollution is balancing its environmental and health costs against its economic benefits. Achieving the optimal balance between the two is easier when the pollution is narrowly concentrated in a region…[which] allows policy solutions to be more effectively tailored to local conditions" (Woods and Potoski 2010, 723). On the other hand, critics contend that this creates principal-agent problems, systems that are more vulnerable and less resilient, and inequitable distributions in outcomes. Despite oversight tools, primacy provides states with a political tool to use against EPA in order to gain program adjustments, opportunities to pursue agendas other than environmental quality, and increases the likelihood that states respond negatively (e.g., reducing enforcement) to external stress (e.g., competition) (Crotty 1987; Woods 2006a, 2006b; Meier and O'Toole 2009). Additionally, there is evidence that primacy contributes to a "race-to-the-bottom," where states use lax regulations to attract industry; although some scholars also argue that a "race-to-the-top" has occurred (Potoski 2001; Woods 2006a, 2006b; Konisky 2007).

These contrasting arguments hint at larger trends in environmental federalism, where success or failure is largely dependent on federal-state relations. Some scholars argue that federalism rests in a collaboration matrix in which policy actors in interdependent networks negotiate and bargain with each other (Agranoff 2001; Scheberle 2004, 2005). As such, "positive" federal-state working relationships result in policy actors at different federal levels "pulling together" to manage environmental problems, while "negative" working relationships result in policy actors "coming apart" in which case, environmental programs flounder (Scheberle 2004). Wood (1988, 1991) describes a similar set of behaviors in terms of principal-agent dynamics, which, in contrast, assumes hierarchical rather than collaborative relationships where power imbalances play a significant role in interactions. In either case though, program outcomes are highly dependent on how state political and administrative systems respond to federal policies.

Adding another dimension, Peterson (1995) argues that there are two intertwined federal systems: one functional and one legislative. The functional system is predicated on well-designed power distributions that facilitate different levels of government carrying out respective responsibilities. The legislative system is tied to how national elected officials respond to political pressures by shifting responsibilities between layers of governments, so unpopular programs are put in the hands of state or local leaders (Peterson 1995). In turn, state and local leaders are willing to take on programs to gain control over regulatory regimes that they can use to satisfy political interests. To this end, several scholars contend that administrative systems tend to be designed in response to political demands rather than administrative ones, with credit claiming and blame avoidance behaviors explaining the distribution of responsibilities in the federal system better than efficient program design (Volden 2005; Wood and Bohte 2004; Woods and Potoski 2010). Despite different approaches to examining these relationships, scholars generally agree that there will likely be a range of outcomes across the nation when states are implementing federal environmental policies.

Consequently, most research into state environmental policy assumes that if states implement federal programs, then any inter-state differences in program outcomes is a result of state implementation systems, when controlling for issues such as problem severity, economic composition, and/or geography. Previous scholarship identifies dozens of variables that account for differences in state policies or administrative capacities, such as partisan control of state offices, citizen ideology, expenditures, agency missions, and bureaucratic designs (see Konisky and Woods 2012). Nevertheless, the mass of research on primacy focuses on why states choose to takeover programs, rather than how effective states are in managing those programs (Crotty 1987; Morris 1999; Travis, Morris, and Morris 2004; Woods 2006b, 2008). As such, there is little current evidence to suggest which states are likely to be successful and which states unsuccessful under the primacy system.

Lester (1995) is one of the few sources that provides predictions, arguing that not all states have the requisite administrative systems to manage these complex programs, nor the political capital to prioritize environmental problems above other interests. Subsequently, he sets expectations for four different types of states (see table 1). First, in progressive states (e.g., Florida) that are highly committed to the environment and with strong administrative capacities, state primacy is likely to improve environmental quality as state agencies have the resources necessary to implement programs effectively. Second, even though they are highly committed to environmental protection, struggler states (e.g., Maine) lack certain administrative capacities are "often structurally unable" to make programs work (Lester 1995, 54). Third, despite limited environmental commitment, delayer states (e.g., Oklahoma or Texas) tend to have strong enough administrative capacities to maintain status quos in the face of political challenges. Fourth, and most concerning, are regressive states (e.g., South Dakota) that have neither a commitment to the environment, nor requisite administrative capacities to implement programs, so state primacy "will likely be a disaster" (Lester 1995, 55). Given this is the most well-defined articulation of how states are likely to fare if granted primacy for federal environmental programs, we expect our findings to align with these assertions.

[Table 1 about here]

Clean Water Act

Programs in the U.S. that protect traditional navigable waters, interstate waters, and adjacent wetlands along with a few other special categories of waterways are collectively regulated under the CWA. A key part of the CWA is the National Pollution Discharge Elimination System (NPDES), which requires permits for all point source (e.g., discernible, confined conveyances such as a ditch or tunnel) pollutant discharges into specified waterways. In general, the goals is to limit water-based pollutants that enter surface waters through permit-based regulatory controls that assume that waterways will meet standards if discharges are at or below the maximum amount of pollutant discharges set for each body of water, known as the Total Maximum Daily Loads (TMDLs). Notably, whether states have primacy or not, they participate in the development of TMDLs (Houch 2002; Copeland 2003, 2005, 2016). To gain primacy, states must make a formal request to EPA and prove they have the necessary institutional, organizational, and workforce capacities to manage the program. Once granted primacy, states issue permits for pollutant discharges, monitor and enforcement compliance, and experiment with regulatory tools (Copeland 2003, 2006; Woods 2006b).

Primacy provides states with the opportunity to match programs to specific water quality challenges by developing regulations for prominent or emerging contaminants in their jurisdictions. Although state-led programs may include all the same administrative components as EPA-led programs (e.g., permitting, compliance enforcement), state agencies may establish more or less stringent permit limits in cases where federal laws allow for regulatory discretion in establishing permit requirements, or they may be more or less strict in compliance enforcement than EPA. Additionally, relationships between state agencies and facilities may differ from those between facilities and EPA-regional offices that are geographically or politically distanced from the realities of daily operations (i.e., more or less collaborative, compliance culture) (Fowler 2014; Copeland 2016). For instance, a research report from an environmental non-profit in Louisiana found that in 2012 ExxonMobil's Baton Rouge refinery reported no accidents to EPA, but dozens to the Louisiana Department of Environmental Quality (LDEQ), which suggests the refinery is more likely to cooperate with LDEQ than EPA (Dubose 2013).

This provides state agencies with an important advantage over EPA in that they have localized knowledge of and buy-in from facilities that can be leveraged to tailor programs to meet specific challenges that emerge. Although, it may also lead to bureaucratic capture. Regardless of how state agencies balance these interests, weak regulatory tools (e.g., permits are the only direct vehicle for enforcement) create a difficult implementation process. While compliance is compulsory and environmental organizations provide some monitoring, government inspections are rare and most violations are voluntarily reported (Adler, Landman, and Cameron 1993; EPA 2000; Houck 2002; Copeland 2005). Despite these issues, state primacy is widespread. By 1987 when the Water Quality Act (WQA) created the last major expansion of NPDES, thirty-seven states had obtained primacy, with eight more states following suit in the ensuing years (EPA 2018a) (see figure 1).[1] Notably, there were no major CWA amendments or NPDES reforms post-1987, so the period after the WQA marks the contemporary era in CWA programs.

[Figure 1 about here]

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Measuring CWA Impacts

There is mixed evidence concerning whether the primacy system has led to improved water quality in the U.S. (Adler, Landman, and Cameron 1993; EPA 2000; Houck 2002; Copeland 2003, 2006, 2016; Fowler 2014). Fowler (2014, 15) contends: "several studies, spanning from the late 1970s to the mid-2000s, conducted on the effectiveness of the CWA suggest there have been no significant trends in either direction for clean water in U.S. waterways." Some scholars argue this is because progress is uneven and some state programs work better than others, with notable asymmetries as a result of waterways exposed to point source pollution in urban and industrialized areas as well as policy innovation to address non-point sources (Lowry 1992; Andreen 2004; Hoornbeek 2005, 2011; Morris 2010; Fowler 2014). However, existing scholarship is severely lacking in considering how program performance under EPA compares to that under state agencies, so there is little indication of whether state CWA primacy has significant impacts.

A key challenge here is that there is no direct measure of how the CWA impacts water quality, due to the physical characteristics of water pollutants and geographic variability. For instance, EPA lists more than 100 regulated and monitored water pollutants. Additionally, as implementation occurs at the state-level, any measure of water quality would need to be meaningful when aggregated to that level, which precludes geographically-specific measures from being useful. To this end, while some argue that water quality should be monitored and managed on the basis of entire watershed basins, others use a waterway segment-by-segment approach. Given that state agencies tend to submit monitoring reports to EPA using the latter, segment-by-segment is the most consistent way in which water quality is monitored in the U.S. (Adler, Landman, and Cameron 1993; Copeland 2010; Fowler 2014). Although there are some indices of water quality indicators in use, these tend to suffer from issues related to internal consistency and validity. Subsequently, it is difficult to make comparisons sophisticated enough to infer a causal relationship with policy implementation (Schultz 2001; Cooter, Rineer, and Bergenroth 2010; Fowler 2014).

In response to these challenges, some scholars rely on indicators such as compliance violations or waterways listed as impaired based on specific pollutant parameters (Smith, Ye, Hughes, and Shabman 2001; Keller and Cavallaro 2008; Grooms 2015). However, in both cases, changes over time may reflect changes in interpretations by implementers rather than changes in physical characteristics of water. Additionally, states may use these measures to "game the system" in negotiating with EPA for more resources or program adjustments (Crotty, 1987; Nicholson-Crotty, 2004). Other scholars use toxic chemical releases (i.e., the physical amount of pollutants released) to assess effectiveness of environmental policies (Bacot and Dawes 1997; Bowen and Wells 2002; Sapat 2004; Konisky and Woods 2012). These scholars argue that if environmental quality is dependent on toxic chemical releases and environmental policies regulate those releases to control environmental quality, then toxic releases are a measure of the impact of environmental policies. While toxic chemical releases may be one of the most straightforward and common ways to understand how pollution varies from state to state, it has its own shortcomings in terms of reliability, accuracy, and consistency.

First, and foremost, toxics release data is not a direct measure of regulatory impacts, rather it is a proxy measure based on a specific type of pollutant (i.e., toxic chemical releases from point sources that are monitored and reported to EPA). Thus, it may not measure the complexity or magnitude of effects from CWA implementation, when considering broad implications for how regulations impact environmental management practices used by facilities controlling point sources and the many ways in which pollutants enter waterways from non-point sources (e.g., agriculture). While we know what is included in databases, such as EPA's toxics release inventory (TRI), we are unaware of the effect that missing data categories may have on measurement validity in the context of the CWA. For instance, only companies producing more than 25,000 pounds of toxic materials and fall within a few specific industries are required to report, which potentially leaves out an abundance of smaller companies producing lower amounts of toxic materials or companies in other industries which may collectively lead to non-trivial differences in water quality (EPA 2018d). Additionally, it is unclear which chemicals included in the TRI data are subject to NPDES permitting authority and which are not, given that TMDLs differ across states. TRI data also does not include other water pollutants, such as suspended solids, that have played a dominant role in fouling waterways for decades.

Second, reporting requirements to the TRI database have not been consistent over time, which limits our ability to make inferences about observed changes in reported toxic releases (Madia 2007). For instance, between 1990 and 2020, more than 300 chemicals were added or deleted from the reported requirements, with the only sustained time periods without changes being from 1995 to 2000 and then from 2000 to 2010 (EPA 2020). On its face, this raises significant questions about the comparability of reported toxic chemical releases over time, which is only exacerbated by the possibility that implementation of these changes may have been uneven by state agencies and/or EPA regional offices. Third, the CWA focuses on compliance with TMDLs, and not specifically on decreasing toxic releases. Therefore, a well-managed program could still result in increased toxic releases if new manufacturing facilities are introduced that produce discharges in compliance with TMDL requirements. Additionally, reported toxic releases are a function of voluntary reporting by facilities, and when programs are run more effectively (i.e., strict monitoring), facilities are more likely to honestly and accurately report toxic releases as compared to facilities regulated under a lax enforcement regime. Under any circumstances, there is some "white noise" surrounding exactly how and why reported toxic releases may change over time and what inferences can be made about program management as a result.

While the data that is collected and reported suffers from significant limitations that constrain the ability of researchers to make inferences about the relationship between regulatory efforts and water quality, the physical amount of pollutants being released into waterways is still one of the best approximations of regulatory impacts available. At a minimum, it provides some indication of whether reported toxic chemical releases from specific categories have changed over time, which is a basis for preliminary inferences about how federal versus state primacy may affect programs. That is, if we can reasonably infer to some degree that reported toxic releases under state agencies differ from those under EPA, then we can also reasonably infer that transferring primacy from federal agencies to states has an impact on the function and processes of CWA programs. By extension, then, one may in fact result in "better" outcomes than another. Although we must be circumspect in our inferences about what reported water-based toxic releases mean in terms of the efficacy of state versus federal programs and their impact on water quality, we can at least begin to unpack whether there is a substantive difference between primacy agencies and whether these differences warrant further inquiry.

Methods

Synthetic Control Method

We use the synthetic control method (SCM) for comparative case studies to estimate the effects of switching from federal to state CWA primacy on reported water-based toxic releases (Abadie et al. 2015; Birdsall 2017). SCM uses a data driven process to generate a synthetic control from a weighted combination of control units, which closely approximates the characteristics of the treated unit prior to treatment. In order to accomplish this, the synthetic control is constructed from a "donor pool," which is comprised of units that do not experience the intervention during the study period. From the donor pool, a weighted combination of states is selected that most closely approximates the state that experienced a transition in the period before primacy occurs according to values of reported water-based toxic releases and its predictors. The post-primacy values of water-based toxic releases for the synthetic control become the counterfactual: what water-based toxic release levels might be if the state did not experience a primacy transition. Any observed differences in values of water-based toxic release levels between the treated and synthetic states in the post treatment period become the estimated treatment effect (Abadie et al. 2015).

Dataset

Our dataset consists of publicly available state-level data from 1990 to 2016 related to environmental, economic, political, and social factors, collected from EPA, the Bureau of Economic Analysis, the U.S. Census Bureau, and the Council of State Governments' *Book of the States*.[2] This time-period corresponds to a naturally modern period in CWA programs and provides a sufficient number of primacy transfers to identify potential differences between actual and predicted reported outcomes if primacy shifts did not occur. Additionally, we supplement our quantitative data with qualitative assessments via SRF reports, which provide further insights into state programs. Under the SRF program, EPA assesses state performance and compliance every three to five years to identify areas of both strengths and needed improvements related to data management, inspections and compliance monitoring, and enforcement actions. We primarily use this data to counterbalance our quantitative findings, and provide additional evidence that may explain differences that exist between states. Assessments are staggered across states and years. To date, every state has undergone at least two rounds of assessment with the first based on data from between 2004 and 2007 and the second on data from between 2010 and 2013 (EPA 2018c). Table 2 provides a breakdown of primacy transfer and SRF assessments by year for states.

[Table 2 about here]

Dependent Variable

Despite the limitations noted above, we use total pounds of water-based toxic releases (aggregated annually at the state-level) as a proxy measure for CWA impacts on water quality. EPA requires companies classified as mining, manufacturing, utilities, waste management, or wholesale trade and that produce more than 25,000 pounds of toxic chemicals a year to report chemical releases to EPA; in turn, EPA compiles those reports into the TRI national database. Primarily, toxic releases involve pollutants directly regulated by the CAA, CWA, or RCRA and that pose a threat to human health or the environment. Reporting is specific to environmental medium (e.g., water), which allows releases to be directly connected to applicable governing legislation (i.e., CWA for water). Data is reported as annual toxic releases that we can aggregate to the state-level. However, TRI data does not include pollutant discharges from non-point sources or fugitive emissions produced during manufacturing processes. As such, this does not specifically measure water quality. Instead, TRI data only accounts for changes in reported pollutant discharges regulated by the CWA, which provides some counterfactual evidence to differences between state and federal primacy in terms of reported water-borne toxic releases (i.e., if there is no change in regulated and reported pollutant discharges, then which agencies holds primacy has no generalizable effect in this regard, *ceteris paribus*) (Ferraro 2009).

Predictor Variables

SCM uses a set of predictor variables to generate a synthetic control that closely approximates the treated unit based on those factors; selecting variables for SCM is similar to the process of selecting control variables for a traditional regression model (Birdsall 2017). Based on previous scholarship, we use nine variables associated with water-based toxic releases to generate synthetic control units similar to our treated states. Most importantly, we use gross state product (GSP) produced from four industrial sectors that are required to report toxic releases: manufacturing, mining, utilities and waste management[3], and wholesale trade. Generally, we assume there is a direct relationship between production from these industries and toxic releases (Sapat 2004; Konisky 2007). Additionally, we use two socio-economic indicators that affect interactions between people and the environment: per capita income and population density. We assume that environmental concerns in wealthy, densely populated states are different from those in poor, mostly rural states, which impacts public interests in environmental conditions and policies (Sapat 2004; Konisky 2009; Konisky and Woods 2010). Lastly, we use land area (measured as square miles) to control for how geographic size or dispersion affects monitoring and compliance enforcement (Newmark and Witco 2007).

Given the primary focus on federal versus state comparisons, it was important to identify variables that would be broadly indicative of political contexts related to administering federal program, and not specific variables that controlled for differences between state agencies. In other words, the purpose is not to control for inter-state differences, but to control for similarities between years of federal and state leadership. Consequently, we use two measures to estimate differences in political contexts: percentage of the state legislature's lower house controlled by Republicans, and state and local general expenditures. We assume these measures are indicative of a general attitude towards government that likely impacts managerial interactions with state political leaders and regulatory target populations, where administering a program in a Republican-leaning state with low government spending requires a different approach than in a Democratic-leaning state with high spending (Bacot and Dawes,1997; Potoski and Woods 2002; Travis, Morris, and Morris 2004; Konisky 2007, 2009; Konisky and Woods 2010). Table 3 shows weights assigned to each state in the donor pools used to construct the synthetic control for each primacy states. Table 4 shows how well each state matches up with its synthetic counterpart across predictors. In general, predictor balances suggest the synthetic states provide good counterfactual cases for the states gaining primacy in the period of study, although there are few exceptions. For example, both Texas and South Dakota have much lower population densities than their synthetic counterparts.

[Tables 3 and 4 about here]

Analysis and Robustness Tests

In this study, the donor pool is comprised of all U.S. states that experienced a transition from federal to state primacy prior to our study period (1990 to 2016). As Abadie et al. (2015) point out, when dealing with aggregate units, such as states or countries, a weighted combination of control units will often produce a better comparison unit for estimating treatment effects than any single control unit alone. If the treated unit and synthetic unit exhibit similar values of the outcome of interest and its predictors over the pretreatment period, a discrepancy in outcome values in the posttreatment period is interpreted as the treatment effect. While SCM and other estimation methods, such as

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difference-in-differences, traditionally use units that have never been treated, we use this "reverse approach" because there are relatively few states that have not already transitioned from federal to state primacy, making it difficult to generate an adequate synthetic control. The intuition of this reverse approach is analogous to the "late group vs. early group" difference-in-differences model (Bacon-Goodman 2018), where late treated units are compared to early treated units during the early group's post-treatment period. As Bacon-Goodman (2018) explains, units in the early treated group can serve as controls because their treatment status does not change during the period of study.

We initially performed this technique on all eight states that experienced a primacy transfer from 1990 to 2016. If predicted toxic releases are accurate during the time-period prior to primacy transfers, then we can assume that our model is also accurate in predicting toxic releases in the time-period following primacy transfer. However, our predictions of water-based toxic releases did not align with actual reported water-based toxic releases prior to primacy transfers in three states (Alaska, Arizona, and Louisiana), which suggests that our predictions after primacy transfers were unreliable. In the cases of Alaska and Louisiana, both states tend to be environmental outliers for both ecological and political reasons due to their large oil and gas industries, which may make it more difficult to generate synthetic controls from the donor pool. Consequently, we concentrated our analysis on the other five states (Florida, Maine, Oklahoma, South Dakota, and Texas) in which predicted toxic releases for synthetic controls do not deviate much from actual reported toxic releases in the period before they gain primacy.

To evaluate robustness, we use a series of placebo and "leave-one-out" tests. Placebo tests create a distribution of estimated gaps in water-based toxic release values between states that did not actually switch to primacy during the treatment period and their synthetic counterparts. If placebo tests generate gaps similar in magnitude to those estimated for our treated states, it indicates the analysis does not provide significant evidence of an effect. Figure 2 shows placebo tests for Florida, Maine, and South Dakota (where we observe large gaps).[4] The thick line represents the estimated gap in reported toxic releases between treated states and synthetic counterpart, while the thinner lines represent results for placebo tests. As the figures demonstrate, probability of obtaining a gap as large as our treated state is generally low when we reassign primacy to one of our control states. In Florida's placebo tests, a few control states do have larger gaps in the early post treatment years, but Florida maintains the largest gap for the majority of years. Figure 2 shows the leave-one-out tests, which iteratively re-estimate the baseline model, each time omitting one state from the donor pool that received a positive weight in the original estimate. Overall, leave-one-out tests indicate our results are robust to changes in the mix of states contributing to synthetic controls. In some cases, a leave-one-out estimate will deviate from our original estimate, but these deviations are generally consistent with our original results.

[Figures 2 and 3 about here]

Results

Figures 4 through 6 compare actual reported water-based toxic releases (*solid line*) for Texas, Oklahoma, Florida, Maine, and South Dakota, respectively, to those predicted if EPA retained primacy (*dotted line*), with the vertical line denoting the year of primacy transfer.

Status Quo States

First, figure 4 indicates that actual reported water-based toxic releases in Oklahoma and Texas are similar to expectations if EPA retained primacy, respectively. While a gap begins to emerge after ten years in the case of Oklahoma, this may be attributed to other factors, such as changes in TRI reporting requirements (discussed above). Interestingly, SRF assessment reports indicate that both states manage programs that largely align with EPA prescribed approaches to NPDES implementation. For instance, in both cases, the most significant areas for improvement identified are related to data management and reporting. In comparison to other states, these recommendations are rather minor, and do not indicate major deficiencies in capacity. Importantly, this only implies that CWA programs in Oklahoma and Texas align with EPA's recommended approach, and not that these programs are likely to lead to better environmental outcomes. In other words, Texas and Oklahoma are running EPA's playbook on NPDES implementation, so it should not be surprising that actual reported results are similar to expectations if EPA managed the program.

[Figure 4 about here]

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Although neither is known for environmental commitment, both states have long storied histories with the administrative challenges and political controversies surrounding water resources and regulating petroleum and mining industries, which likely contributes to institutional capacities (Lester 1995; Yergin 2008; Griffin 2011). Interestingly, SRF assessments cite administrative capacity specifically as an important explanation for why these programs are successful. For instance, EPA staff make similar evaluations of program managers in both states: "[Texas Commission on Environmental Quality (TCEQ)]'s success in administering these programs is due in large part to strong leadership through a dedicated and experienced management team and staff" (EPA 2007, 1); and, "[Oklahoma Department of Environmental Quality (ODEQ)] has operated strong compliance and enforcement programs due in large part to their leadership and their experienced management team. Their ability to retain senior managers means they have a breadth of institutional knowledge and expertise that has resulted in clear direction and support in accomplishment of the national enforcement agenda" (EPA 2004, 1). This level of praise for management is exceptional rare in SRF reports, making these evaluations outliers.

Considering the relatively low public support for the environment and strong reliance on resource-dependent industries that are key water pollutant dischargers (i.e., oil and gas, agriculture), there are significant political obstacles to pursuing a more stringent approach to program management than required by EPA (Lester 1995; Mazur and Welch 1999; Fowler 2016). SRF assessments make some indications of this by noting that both states tend to be timely in permitting and facility testing, but slow to follow-up compliance violations with enforcement actions. Although not identified as a major issue for either state, it suggests that managers may be more willing to follow EPA's guidelines when it does not lead to conflict with industry. Consequently, Oklahoma and Texas likely find that pursuing the status quo in NPDES implementation is a preferable strategy, since it allows them to avoid conflict with both political interests and EPA.

<u>Underachiever States</u>

Figure 5 indicates that while reported toxic releases in Maine are approximately four times more than what would be expected under EPA primacy, toxic releases in South Dakota are approximately 40 times higher. This suggests that although both states are struggling, South Dakota's Department of Environment and Natural Resources (SDDENR) is grappling with much more intense issues than Maine's Department of Environmental Protection (MDEP). In general, SRF assessments suggest MDEP has a strong program in many areas, but is falling short of national guidelines when determining non-compliance violations and calculating enforcement penalties, as well as with data collection and management. Although compliance determinations and enforcement actions tend to be responsive to toxic releases rather than preventive, a lack of strictness or consistency in these regards is likely to contribute to a culture of lax compliance, where facilities may adopt environmental management norms that contribute to increased toxic releases over time. These issues could be partially explained by a degree of bureaucratic capture in that MDEP may lack the resources (including political capital) to defend itself against pushback from regulated facilities (i.e., threats of lawsuits), so managers may be willing to acquiesce in order to avoid conflict. However, EPA largely commends MDEP for "excellent jobs" or "highly effective" in several areas (EPA 2015, 1).

[Figure 5 about here]

On the other hand, SRF assessments point to more systematic issues in South Dakota. For instance, EPA determine that "current staff levels make it difficult to keep up with expectations for the number of inspections expected by EPA…Inherent in this process is the need to focus on "wins" in terms of bringing enforcement actions forward where there is a clear impairment/violation recognized" (EPA 2014, 14-15). In other words, enforcement action is only likely when cases are clear and egregious; otherwise, the process of taking such action is likely not worth the effort for staff. This may be due to the rather onerous process of pursuing enforcement activities. While MDEP can unilaterally pursue enforcement actions that are later enforceable by the Attorney General, any SDDENR enforcement activities are routed through the Attorney General's office for a legal review before the Secretary can officially approve them. These additional organizational layers likely create transaction costs in the enforcement process as well as potential conflicts as agendas diverge, especially as agencies with separate missions attempt to coordinate efforts (Waterman and Meier 1998). This may partially explain EPA's finding that SDDENR struggles to follow its Enforcement Response Guidance when determining and responding to non-compliance cases.

As previous research indicates that Maine tends to be among the most environmentally-conscious states both nationally and regionally, program managers likely experience pressure from elected officials or the public when it comes to environmental conditions. On the other hand, since South Dakota tends to be at the bottom of states in terms of pro-environmental public attitudes both nationally and regionally, environmental protection is likely a low priority, so SDDENR has little political capital to push back against industry (Lester 1995; Mazur and Welch 1999; Fowler 2016). In both cases, this may indicate that primacy has led to a degree of bureaucratic capture. Given regional economies and natural resource amenities in both states, Maine likely finds that protecting the environment creates a potential quality of life draw from Northeast urban areas, while South Dakota likely sees an advantage in attracting resource extraction and heavy industry with lax environmental enforcement. In other words, although administrative capacity is an issue in both states, there may be more political advantage and willingness in Maine to enforce water quality regulations than in South Dakota. However, our data and expectations are not wholly consistent in that they do not account for the one-time spike in the year of primacy for Maine, which could be a result of changes in administration or reporting requirements, or errors in our statistical modeling. Unfortunately, data limitations do not allow us to tease this apart.

Overachiever State

Figure 6 indicates that, after an initial spike, Florida has much lower reported water-based toxic releases after primacy compared to what would be expected if EPA retained primacy. Unsurprisingly, these findings conform to expectations suggested by Lester (1995), as Florida is both highly committed to the environment and has strong administrative capacity for addressing environmental issues. Unlike other states discussed here, SRF assessments provide little evidence to suggest what administrative strategies or techniques are contributing to these successes. For instance, in initial assessment reports, evaluators noted that the Florida Department of Environmental Protection (FDEP) followed recommended guidelines for data collection and enforcement, but that FDEP also needed to revise the enforcement guidance manual. Furthermore, in the twelve assessment categories used in round 2, Florida's NPDES program only met expectations in four categories, and received no designations for "good practices." Compared to the assessments discussed above, results for Florida are much more difficult to pin down to a specific set of issues occurring in the implementation process, but we surmise three potential explanations for why SRF reports do not directly account for FDEP's success.

[Figure 6 about here]

First, deficits that EPA evaluators note may create a degree of flexibility that allows FDEP to better match resources to local challenges. In other words, FDEP is utilizing their localized knowledge and relationships to tailor programs to local conditions and create buy-in from target populations. Given that this system relies heavily on cooperation of regulated facilities, these results may indicate that FDEP has built collaborative relationships that are not readily measured in EPA's evaluations. In comparison to Texas and Oklahoma, TDEQ and ODEQ are unlikely to outperform EPA, since those agencies are following EPA's recommended course of action, but FDEP may be strategically diverging from recommendations when it allows for better program management. In fact, EPA notes that FDEP's "inspection activity suggests strong presence in the field" (EPA 2006, iv). This would indicate that inspectors are familiar with the unique challenges that exist within their jurisdiction, and work cooperatively with regulated facilities to overcome those challenges. Importantly, the game-playing that is inherent in the primacy framework may encourage FDEP to strategically diverge from national guidelines to negotiate with EPA for resources or program adjustments (Crotty 1987). However, the results from Maine and South Dakota strongly suggest that divergence from national guidelines does not always work.

Second, high capacity states may be better at "learning" how to implement policies than low capacity states (May 1999), which could explain why Florida had more success in a shorter period of time than South Dakota over a longer period of time. To this end, SRF evaluations indicate performance improvements in some states between rounds 2 and 3. This would suggest an important component to primacy is policy learning as states experiment with best practices. Finally, some factors may be outside of FDEP and the NPDES program. For instance, collaborative governance of watershed management goes back decades in Florida (Lubell 2004; Berardo, Heikkila, and Gerlak 2014). More community engagement and voluntary compliance from industry partners could be a contributor to program outcomes here, which may be part of FDEP's management strategy (Potoski and Prakash 2004). This is not to say that collaborative environmental governance does not occur in other states; it is only to suggest that collaborative governance may be a contributing factor to trends in Florida. However, the current analysis is too limited in scope to provide conclusive evidence related to either of these explanations. As such, additional research is necessary to determine why Florida's NPDES program is notably successful in comparison to expected results for EPA.

Discussion

Our SCM findings indicate that there are asymmetries in how primacy transfers impacte reported water-based toxic releases, and data from SRF assessments provides some insight into why those asymmetries may exist. However, this relies on reported toxic releases, which is limited in both what types of pollutants are monitored and consistency in reporting over time. Although water-based toxic chemicals releases serve as a good indicator of CWA impacts for our purposes here, findings should be interpreted with a degree of circumspection as there are notable and important limitations. Furthermore, SCM is a novel method of creating counterfactual evidence for "what might have happened" by using observed data to extrapolate observed trends to unobserved events, which obviously constrains the evidence that we are working with. While our robustness tests indicate SCM findings meet commonly reported and accepted standards, there are some significant gaps between predicted and actual reported water-based toxic releases before the shift in primacy, which may indicate that our predictions are not completely accurate or reliable. Additionally, there is inconsistency across states in when differences between predicted and actual outcomes begin to diverge after primacy shifts. But, in any research design, there are trade-offs to be made.

Given this, the real value in this analysis may be the comparative aspect between states. While it is debatable what reported toxic releases reflect and/or what inferences to make about changes that manifest over time, findings from both the SCM and SRF indicate that significant asymmetries exist in how states are managing CWA programs. Regardless of what assumptions we make, findings still indicate that state primacy has had different impacts in Texas and Oklahoma compared to Florida or Maine and South Dakota. Simply from this, we should be able to infer that whether states or EPA manage these programs affects federal water quality programs and, by extension, water quality. In sum, while our statistical analysis is robust enough to base inferences, it does not create evidence beyond a reasonable doubt; however, the preponderance of our preliminary evidence allows us to conclude that state versus federal primacy likely has an impact on programs, and that impact depends on the state agencies in which federal environmental programs are entrusted

Of course, there are also alternative explanations for our findings, such as changes in the reporting requirements or what toxic chemicals are monitored as well as factors that are not measured by TRI data; both of which are discussed at length above. Additionally, trends may be explained by changes in reporting behaviors, rather than polluting behaviors, by facilities. Given that TRI data reflects reported toxic releases, it is possible that while our SCM predictions are correct, they are predicting how facilities are likely to adapt reporting strategies under state agencies compared to EPA. There is some anecdotal evidence that facilities are more willing to report industrial accidents to state agencies than EPA, when state agencies are likely to be lenient in response (Dubose 2013). This may explain, for instance, results for Maine in which there was a massive spike following the state take-over of primacy (i.e., more accurate reporting). However, if we assume that our results only reflect a change in reporting behavior, then there is still evidence here that facilities are likely to alter behavior under state compared to federal primacy. If so, this is still likely a function of differences in working relationships that state agencies build with facilities producing point source pollutants.

While SRF evaluation reports provide little insights on this point or other possible explanations, they are limited to EPA's perspective, so evidence is constrained by what data EPA considers when evaluating state programs (Fowler 2019). Additionally, we have no way of accounting for policy learning here, where states that transitioned to primacy prior to the study period may potentially show improvement over-time leading us to underestimate successes of "synthetic" states or how facilities may have learned to "game the system." Thus, to fully consider how administrative functions of state versus federal agencies impact environmental management practices used by facilities requires additional research. Furthermore, arguments for decentralization in federal systems are grounded in better serving local populations, which cannot be measured by reported toxic releases alone. For instance, "better" program outcomes could also be measured in public health or economic impacts. As such, effects of state versus federal primacy may be more complex than what our analysis is able to consider. To this end, we only begin to scratch the surface of potential administrative explanations for asymmetric impacts across state agencies. Consequently, this is point of inquiry that requires further inquiry.

Despite these limitations, Lester (1995) predicted our results based on his political commitment-institutional capacities model. From this perspective, it is no surprise that some states out-performed others, especially when considering variance in both political and administrative dimensions. Given that Lester's (1995) predictions were made during the same time period that many of the states in our study were experiencing primacy transfers and are now more than two decades old, this does beg questions of whether changes in state administrative capacities or political will to support environmental goals may limit the utility of this typology to predict future results. As such, we should be cautious in applying these trends to states that take on CWA primacy in the future; although, we can likely expect that some will overachieve while others underachieve in comparison to EPA. Interestingly, Idaho (a struggler in Lester's typology) is currently in the process of taking over primacy and expects to have full control by 2021 (IDEQ 2018). Nevertheless, updating this typology is an important point of additional inquiry in as far as it will illuminate how capacities and commitment impact environmental outcomes over time, and provide additional data points to the preliminary evidence presented here.

Conclusions

In general, findings indicate that transferring primacy has asymmetrical impacts on reported water-based toxic releases that we can connect to the characteristics of individual states (see table 5). For instance, Oklahoma and Texas maintained the status quo by basing their implementation systems on EPA's national guidelines and recommendations, while Maine and South Dakota likely ran into systematic institutional barriers. Although Florida confirmed to our expectations and outperformed predictions for EPA, there is little evidence that performance is due to internal administrative capacity, so this finding requires additional research to consider more broadly the contributing factors. In sum, while some states have seen success, there is not a universal positive experience with state compared to EPA primacy, which begs important questions about the efficacy of the primacy system that is engrained in modern environmental federalism. While our findings face notable limitations, as there are few valid, reliable, and consistent measures by which to judge the impact of the CWA across states and over time, there is sufficient preliminary evidence here to indicate that state primacy likely has asymmetric impacts, which leads us to three broad conclusions.

[Table 5 about here]

First, policies that rely on intergovernmental implementation need to more directly consider state administrative capacities when delegating authorities. While allowing decision-making to be closer to the people may justify less stringent program management in some states, others may fundamentally lack the capacities to carry out those decisions. For instance, we could call state primacy in Oklahoma and Texas a success, where environmental conditions are likely no worse than they would have been under EPA primacy and this level of environmental protection likely aligns with political preferences. On the other hand, Maine highlights the potential tragedy of state primacy, where the state likely took on primacy to achieve a higher level of environmental protection but a lack of administrative capacity may have potentially had the opposite effect. As such, federal primacy may be a better alternative for states with more political will for environmentalism than capacity to administer programs. On the other hand, findings indicate that states with more institutional capacity than political will tend to have better outcomes (or at least, did not experience higher levels of reported toxic releases), which suggests that the former may be more important than the latter in predicting the impact of primacy.

Second, a theory of environmental federalism should consider under what circumstances political (i.e., determining the level of environmental quality needed) and administrative tasks (i.e., enforcing standards) are "best" managed by federal or state governments (Woods and Potoski 2010). In as far as our analysis considers, primacy chiefly involves administrative tasks, so findings suggest that not all states are fully up to the CWA's administrative responsibilities of (or, at least, differences in administrative capacities have an impact on reported toxic releases). Peterson (1995) and Scheberle (2004) begin to unpack some of this, but do not explicitly recognize the importance of administratively focused tasks as compared to those that may be politically focused, and how this may influence distribution of program responsibilities. Importantly, "best" can be defined in numerous ways and relate to both democratic (i.e., representation) and administrative values (i.e., efficiency) (Kirlin 1996). While proponents of primacy would argue that state leadership "best" serves democratic values, evidence suggests that this system does not always "best" serve administrative values. As such, there may be certain environmental policy tasks that lend themselves to federal and others to state leadership. By incorporating this into a theory of environmental federalism, scholarship can more effectively understand federal-state roles and guide policy design at a time when emerging environmental problems (e.g., climate change) are challenging traditional institutions and frameworks (Verweij, et al. 2006; Rabe 2011).

Finally, understanding how states are suited for the administrative challenges of environmental federalism may also explain the asymmetries in CWA implementation previously reported by scholars (Lowry 1992; Hunter and Waterman 1996; Hoornbeek 2012). For instance, Hoornbeek (2012) finds a notable degree of variation in state NPDES policies for point sources and policy innovations surrounding non-point sources, and some of these variations may be accounted for by how states match the CWA to their unique administrative and political circumstances. Moreover, there is far less understanding of the progress made in water quality over the last half century, as compared to other environmental programs, such as the CAA (Fowler 2014). Consequently, understanding how program organization and management affects water-based toxic releases as a component of water quality can provide significant insights for developing and refining water quality strategies across the country. For instance, if we assume that reported toxic releases are an indicator of water quality, then findings for Florida, Oklahoma, and Texas suggest that national guidelines only produce gains to a certain point, and after reaching that point, states must strategically deviate from EPA recommendations to avoid plateauing (although it is not quite clear how this may occur). In sum, the primacy system has both pros and cons, which largely hinge on state governments and their respective environmental agencies.

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NOTES:

[1] Alaska, Arizona, Florida, Louisiana, Maine, Oklahoma, South Dakota, and Texas. Utah initiated the process prior to the WQA's passage, but it was not finalized until later. In 2014, the Idaho State Legislature passed legislation approving the development of a plan to replace the EPA-led NPDES system with a state-led system by 2021 (IDEQ 2018).

[2] For missing data, the average of surrounding years is used to construct synthetic controls, which is the case for state and local expenditures for 1990, 1991, and 2003.

[3] Prior to 1997, the Standard Industrial Classification (SIC) combined utilities and waste management. After 1997, the North American Industrial Classification System (NAICS) separated those industries into two classifications.

[4] Following Abadie, et al. (2010), we exclude states with a poor fit in the pre-treatment period (i.e., states with a root mean square prediction error (RMSPE), which measures the lack of fit between the outcome variable for a state and its synthetic counterpart, two times higher than for the treated state).