The London School of Economics and Political Science

# Some Alternative Approaches to Persistent Environmental Problems and the Measurement of Inequality

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A thesis submitted to the London School of Economics and Political Science for the degree of Doctor of Philosophy.

**Declaration**

I certify that the thesis I have presented for examination for the PhD degree of the London School of Economics and Political Science is solely my own work other than where I have clearly indicated that it is the work of others (in which case the extent of any work carried out jointly by me and any other person is clearly identified in it).



I warrant that this authorisation does not, to the best of my belief, infringe the rights of any third party.

I declare that my thesis consists of 76,675 words.

**Statement of inclusion of previous work**

I can confirm that Chapter 3, *A Review of How U.S. Coal Power Plants Used Technology to Reduce Sulphur Dioxide Emissions – Lessons for the Next Energy Transition* is a substantial extension of an earlier study I undertook at Yale University. The basic domestic model presented in the chapter was initially conceived, and some of the data collected and formatted, during that prior study. The research in this thesis has undergone substantial revision and development while at the London School of Economics and Political Science.

**Statement of inclusion of conjoint work**

I confirm that all work in the Introduction, Conclusion, and Chapters 1-4 are my own. Chapter 5, *Mean-Spirited Growth*, is co-authored with my primary advisor Ben Groom at the London School of Economics and Political Science, and Eli Fenichel at Yale University. I performed the research and development of the paper. But the idea to review inequality and development comes from Ben and Eli who have since provided supervisory input throughout its development.

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To Howard Virgil Brunson,
the greatest economist to never
explicitly mention the subject.

**Abstract**

Throughout this thesis, I assume consumption is an unstoppable economic force that limits policy intervention even when the level of consumption is destructive to the environment. A result is that traditional policy interventions – caps, trading schemes, and many taxes and subsidies directly on polluting outcomes are too costly to implement in terms of political, social, or measured economic costs. However, I also assume that a limited regulator still seeks to reduce the environmental fallout. I then study some policy options that fit within the consumer's limited expectations of regulatory reach.

Deciding to use limited policy interventions is not intended to replace other efforts. I explore part of what could be a joint effort and emphasise perhaps short-term, stopgap responses to opposition. This is proposed because not all environmental problems can wait for justice or the environmental Kuznets curve to turn in their favour. Once we are free from trying to devise ways to implement first-best environmental policies, various alternatives emerge.

First, I explore environmental regulation when some pollutant is necessary. Necessity restricts policy to determining where it must occur, and the analysis becomes one of whether clustering or dispersion of an activity leads to less total damages. A revision to current regulatory approaches results. I then discuss improving recycling, a necessity to transition to a circular economy model. I find there is an optimal division of effort between producers and recyclers and propose an incentive structure to improve product design for recyclability. I then review how U.S. coal power plants use technology to reduce sulphur dioxide emissions. The objective is to derive lessons for the next great energy transition. Finally, I include a foray into economic growth, inequality, and their measurement in two chapters on the premise that there is a link between the fate of the environment and ours.

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# Introduction – Non-Cooperative Environmental Policy More Broadly Defined

*I repurpose the term Non-cooperative, or Nash Environmental Policy (NEP) to describe a decision-making framework in the same genre as second-best and satisficing environmental policy approaches.[1] I define it to use in devising and evaluating alternatives when standard environmental policies encounter insurmountable public or industry opposition. I then outline the five chapters of this thesis – three which apply NEP approaches to challenging environmental problems, and two focusing on inequality in economic growth on the basis that their combination can result in the overexploitation of natural resources.*

This research explores applications of a concept I unoriginally refer to as Non-cooperative Environmental Policy (NEP), but also Nash Environmental Policy in deference to Nash's contribution to our understanding of equilibria without coordination (Nash, 1950). The term itself has been in previous use, but I claim not to its appropriate potential. In Haupt (2006), Hattori (2010), and no doubt a few others, it refers to non-cooperative games played in international environmental agreements or explains why contributions to environmental endeavours do not add up to commitments as in Ulph (1992) and Barrett (1994). If Non-cooperative games meant only one or two sorts of application as such, Nash would hardly be a household name. In reply to what I suggest is NEP's premature designation, I non-cooperatively, unilaterally co-opt it to describe a variety of circumstances – to return it to a general concept. In doing so, I suggest the prior use of "non-cooperative environmental policy" are instead applications of non-cooperative games to understanding international accords.

To the matter at hand, informally a NEP approach is an environmental policy that does not need further buy-in from consumers, firms, and other interested parties. It is already accounted for, or "baked-into" expectations. That is, an expectation on consumption, $E[C]$ already accounts for some policy intervention. This intervention is likely limited, and perhaps there is an expectation of no change in near-term consumption resulting from it. Consumers and firms move first as they create the need for environmental policy in the first

[1] (Lipsey and Lancaster, 1956; Simon, 1956)

place in an overwhelming majority of cases. The choice by a regulator between a NEP approach and a cooperative environmental policy (CEP) – where we expect consumers to cooperate by changing their actions – then follows.[2] I justify exploring NEP more formally, though briefly, in the next section as it is the foundational ideology and motivation underlying much of this thesis.

However, I must note that it is not the purpose of this thesis to contribute to the literature on non-cooperative games. The purpose is instead to convey the underlying philosophy for exploring the research presented in this thesis. More generally, it is also a sort of advocacy for pursuing environmental policies that are sometimes limited in scope but move a conservation agenda forward despite public resistance or indifference. The non-cooperative games literature itself is quite vast and crowded. Google Scholar presently notes Nash's original work as having been cited approximately 12,400 times (July 2021). There is also a vast supply of textbooks of varying degrees of depth and complexity and many popular offerings on the subject. In comparison to many texts, the following discussion is simple. The scope is simply to provide the minimum structure necessary to convey the NEP concept in relation to CEP alternatives. So, nearly any text including non-cooperatives games material should be sufficient background, if any background is necessary at all.[3]

### I.1 A Formal Definition of NEP

Assume some expectation over consumption. Define the private goods share of consumption through the classic construct of equilibrium – restate $S_{i,t}(\boldsymbol{P}) = D_{i,t}(\boldsymbol{P})$ as $S_{i,t}\left(D_{i,t}(\boldsymbol{P})\right) = D_{i,t}(\boldsymbol{P})$, or conversely so, for some abstract general supply, $S_{i,t}$, and demand, $D_{i,t}$, responsive to prices, $\boldsymbol{P}$, impacting good *i* in period *t*. For any realisation (equilibrium) of price, $\hat{P}_{i,t}$, there is a corresponding quantity $Q_{i,t}\left(\hat{P}_{i,t}\right)$. Shorthanded, this is some level of consumption and production in the economy where for private goods denote it $C_{i,t}$. Then for any policy proposal, expectations over future private consumption are $E[\boldsymbol{C}]$ where $\boldsymbol{C}$ is the set of consumption flows of private goods over perhaps many periods of accounting. $E[\boldsymbol{C}]$ forms from current beliefs and information – what the individual and society expect over their planning horizon, *T*. Finally, I consider the expectation of the

[2] Whether they explicitly consider NEP alternatives or not.

[3] Several popular and sufficient examples are Aumann (2019); Dinar, Albiac, and Sánchez-Soriano (2008); Dixit and Nalebuff (2008); Gibbons (1992); Hanley and Folmer (1998); Harsanyi (1977); Luce and Raiffa (1957); McCarty and Meirowitz (2007); Mouline (1986); Osborne and Rubinstein (1994); Rasmussen (1989); Schotter (1981); Shubik (1983); Owen (1982); Tadelis (2013); and Vorob′ev (1977).

median voter and assume political representation votes to fit their interests – a dubious but common simplification with a fairly long history (Black, 1948; Downs, 1957). For this discussion, treat as if the median voter's realisation is sufficient and known, $E[\boldsymbol{C}] = \tilde{C}$.[4] A general treatment of the private consumption of public goods is also in order. Let $\frac{\Phi}{N}$ represent expected public goods consumption as a $\frac{1}{N}$ share of the public benefit of public policies among *N* consumers – also a dubious but useful and I think sufficient assumption.

A payoff matrix is illustrative in completing the picture of one definition of a NEP and when it is preferable. Assume the median consumer plans to consume either $\tilde{C}$ or alternative stream $\tilde{A}$, where they prefer $\tilde{C}$ when they do not account for public goods consumption. A regulator can attempt to implement either NEP or CEP alternatives. However, if the median consumer and voter does not believe they will sufficiently benefit from a CEP, it will not be implemented. Besides political opposition, repeated challenges through the legal system over an extensive period of time may come in reply to an opposed CEP approach.

Take payoffs to consumers, and similarly profits to firms, to be a utility derived from a consumption combination of private and public goods as $U\left(\tilde{C}, \frac{\Phi}{N}\right)$. Like in the distinction of $\tilde{C}$ versus $\tilde{A}$, here $\Phi$ is the public benefit from a policy that does not challenge the preferred private consumption plan – the provision of a NEP. Alternatively, let $\phi$ be the public benefit to a CEP and a resulting $\frac{\phi}{N}$ expected share. However, because the CEP is dependent on coordination, it may either succeed, *S*, or fail, *F*. The latter means the regulator pursues the policy at some cost but ultimately cannot implement it due to opposition. If we assume policy either is or is not implemented, the normal form pay-out matrix (von Neumann and Morgenstern, 1944) can condense to the $2 \times 2$ form in Figure 1 after some manipulation.

To analyse when NEP is the best action given public expectations, constraints on the parameters in Figure 1 are necessary. Assume public benefits follow $\phi_S \geq \Phi \geq \phi_F$, $\phi_F < 0$, and $\Phi = \Phi_S \approx \Phi_F$. Additionally, for $\tilde{C} > \tilde{A}$, $U\left(\tilde{A}, \frac{\phi_S}{N}\right) > U\left(\tilde{C}, \frac{\phi_F}{N}\right)$ would only result for policies where the median consumer believes the public goods pay-out is "substantial" – that their share of the pay-out is larger than their private consumption loss. Because CEP pay-outs $\phi_S, \phi_F$ are dependent on the median consumer's choice, a dimension in Figure 1 is missing. The strictly dominated set where the consumer chooses $\tilde{A}$ but rejects $\phi$ (chooses

[4] So, I am not addressing matters of expectations versus realizations nor gain/loss specifications.

$\phi_F$) is not shown. Also consider NEP *and* CEP instead of NEP *or* CEP. When the policymaker can implement CEP, I assume a NEP is no longer needed – CEP "solves" the environmental problem.[5] So, I drop row "NEP & CEP" as well as row "No Action." From the conditions on the parameters, the Nash equilibrium action set $\{NEP, \tilde{C}\}$ results unless $U\left(\tilde{A}, \frac{\phi_S}{N}\right) > U\left(\tilde{C}, \frac{\phi_F}{N}\right)$. The exception is the case when cooperative environmental policy is warranted – when the parties represented by the median consumer are sufficiently rewarded – or expect to be – due to cooperation. Examples of when this exception results in choosing action set $\{CEP, \tilde{A}\}$ are the Montreal Protocol and America's Acid Rain Program.

| Regulator \ Median Consumer | $\tilde{C}$ | $\tilde{A}$ |
|---|---|---|
| NEP | $\Phi,\ U\left(\tilde{C}, \frac{\Phi}{N}\right)$ | $\Phi,\ U\left(\tilde{A}, \frac{\Phi}{N}\right)$ |
| CEP | $\phi_F,\ U\left(\tilde{C}, \frac{\phi_F}{N}\right)$ | $\phi_S,\ U\left(\tilde{A}, \frac{\phi_S}{N}\right)$ |

FIGURE 1 NON-COOPERATIVE AND COOPERATIVE POLICY MATRIX

*Notes:* Median consumer (and their political representative executes) chooses a consumption plan $\tilde{C}$ or $\tilde{A}$ where $\tilde{C} > \tilde{A}$, and Regulator – representing wider public interests in what ought to occur – chooses to pursue either a non-cooperative environmental policy (NEP) with total public payoff $\Phi$ or cooperative environmental policy (CEP) with total public payoff $\phi_S$ when successful and $\phi_F$ when it fails. Payoff assumptions include $\phi_S \geq \Phi \geq \phi_F$ because the median consumer and voter does not support policy implementation unless they are personally better off from doing so and $\Phi = \Phi_S \approx \Phi_F$ implying NEP does not, or hardly, impacts consumption level. $U\left(\tilde{A}, \frac{\phi_S}{N}\right) > U\left(\tilde{C}, \frac{\phi_F}{N}\right)$ is the exception where consumers believe the public goods pay-out is so substantial that their individual share of it is larger than their private consumption loss under a CEP policy. The Nash equilibrium is otherwise action set $\{NEP, C\}$. Non-cooperative environmental policy is the best response to consumption expectations unless they expect the cooperative public benefit to be significant for the median consumer and voter.

### *I.1.1 Discussion of NEP Versus CEP*

Naturally not everyone wins with a NEP or CEP policy. Producers and firms dependent on dichlorodifluoromethane did not benefit from CEP policies in the Montreal Protocol, while overwhelmingly nations and consumers benefitted (Barrett, 2003). Nevertheless, it was a monumental task which took over a decade of discovery, followed by several amendments to what was and remains a win-win outcome for nearly all parties. What if an

[5] Similarly, I leave open how many NEP or CEP to implement. We could conceptually search over many mixed and pure baskets of NEP and CEP options.

environmental issue is so urgent that a decade or more of delay is unacceptable? What if we cannot ever reach a cooperative agreement because the preceding exception condition cannot be met?

The democratic, short-run feasible optimal regulatory decision is to choose NEP resulting in non-cooperative social and private payoffs $\left(\Phi,\ U\left(\boldsymbol{C}, \frac{\Phi}{N}\right)\right)$. Further, the regulator may not even be in a position to consider the cooperative outcome. Because consumption actions overwhelmingly and causally precede environmental problems, a credible commitment to $\boldsymbol{C}$ may have already occurred.[6] The game matrix in Figure 1 is then degenerate to the left column. If the regulator chooses to go against the odds and pursue CEP, the likely outcome is failure to implement it as in, for instance, America's Clean Power Plan and Waters of the United States (WOTUS) extension of the Clean Water Act (Kendall and Harder, 2015; U.S. Congress, 2015; Glicksman, 2017; Holden, 2017).[7] The regulator should consider NEP alternatives.

The argument that NEP's are sometimes preferable is not to say that CEP's are not worth pursuing. By the condition $\phi_S \geq \Phi \geq \phi_F$, when the median consumer personally benefits from pursuing the cooperative policy, CEP is promising and preferable. In the long run, we might also change public perceptions through the discovery and dissemination of information. That is, change the public's utility function.[8] An environmental amenity may also become sufficiently scarce that the marginal benefit to conservation outweighs the personal benefit of consumption. With more regulatory discretion and favourable courts, implementing CEP's may also protect the rights of future generations. So, NEP's and CEP's need not be mutually exclusive approaches either – one perspective is that a NEP preserves the environmental option value until the majority of society decides to support a CEP.

The NEP approach is an alternative for thinking about environmental policy. It presumes, foremost, limitations on the state's power – that the state acts within the public's expectations over its power. It assumes strict limitations on what the policymaker or regulator may attempt to implement as regulation. For instance, regardless of what the regulator may want to do, they must not exceedingly hinder consumption. But at the same

[6] Consumers and producers might choose $\boldsymbol{C}$ as a natural combined Stackelberg leader because they must cause an environmental problem before we are even aware that there is one to address.

[7] At the time of this dissertation.

[8] Maniates and Meyer (2010) note the difficulty (even impossibility) of getting consumers to sacrifice for environmental conservation, and then present extensive debates on how to change consumer preferences to encourage sacrifice anyway.

time, the regulator remains interested and mandated to address environmental problems. Among the constraints, it may be that no limitations are acceptable on the quantity of emissions as imposed through taxes and standards and the like. We would almost certainly preserve the environment if all our needs are already met. Realistically, we should assume that consumption occurs at an environmental cost, and with an ever-growing population, this will increase. So instead of exploring ways a policymaker can tell people what they cannot do, a NEP exploration is one of what a policymaker can do after they project that consumption needs are being addressed but not necessarily met. These are policies perhaps enacted in the short run that do not intend to change the quantity of production and consumption much, and as a result, limits the probability that policies are decided by the courts because the likely plaintiffs – consumers and producers – are not unduly harmed.

Exploring NEP is not always graceful. But taking the NEP perspective unlocks alternative policy instruments because the regulator is no longer preoccupied with proving CEP worthiness. I leave the cost-benefit analysis of CEP's to the many practitioners who will no doubt remain devoted to it – CEP has a purity that is lacking in some NEP applications. However, the need to address many urgent environmental problems is more critical than ideological victory. Impasses in addressing global climate change, biodiversity loss, and the global proliferation of microplastics all seem to suggest that society needs NEP approaches. I present some NEP-adherent alternative policy approaches to address such issues in this thesis.

## I.2 Thesis Overview

Having outlined the NEP ideology, I now outline the content of this thesis and where I deviate from NEP. In the three chapters of *Part I*, I derive NEP-sort of responses to three pressing environmental problems. I then deviate in the two chapters of *Part II* and explore economic growth and inequality. I decided to write *Part II* because addressing environmental problems without addressing poverty which leads to environmental exploitation is akin to treating the symptoms of an illness but not the illness itself.

### *I.2.1 Part I: Applying a NEP Perspective*

In the first chapter, I assume some domestic, polluting production must occur. Two applications are the storage of nuclear wastes – even if going cleanly forward, our nuclear legacy remains, and the central collection of household wastes – a nearly ubiquitous policy choice that has only become so in recent decades.[9] How can we handle this environmental problem when taxes, tariffs and related measures are not always applicable? A model of when to aggregate and disperse an environmental activity is useful. I find and econometrically explore a few applications of aggregation and dispersion policies in action.

In the second chapter, I switch to improving recycling rather than limiting production. I identify the issue of material heterogeneity within products as a remaining constraint to greater recycling rates. I use a product space to explore the issues involved and derive an efficient recycling policy given the quantity of production that occurs. The framework results in the optimal division of the burden to recycle versus innovate between recyclers and producers. This is certainly not the only issue hindering higher recycling rates, but it is one where a leap forward in innovation appears to be needed.

In the third chapter, I then explore the result of abatement technology decisions on the sulphur content of coal consumed in the U.S. energy sector. In such circumstances, we are constrained by the need to supply substantial levels of energy from existing sources while also reducing emissions levels. I outline how ambient air sulphur dioxide levels diverge from the sulphur content of fuels entering power plants due to installation of abatement technologies that capture the sulphur content. I empirically test for the scale and incentives involved in the relevant low sulphur coal versus abatement technology and high sulphur coal purchase decisions. This leads to a simple model of the purchaser's decision process and further empirical tests to validate the model. I then conclude with estimates of the scale of abatement costs involved for use in designing indirect policy incentives. While the chapter is constrained to reviewing the sulphur dioxide problem, hopefully lessons can be drawn that are applicable to other environmental problems.

While each subject area has received research interest, it is the non-cooperative perspective that I think is important and leads to new redresses. NEP originates from a view of the world I do not like, of unguided and unhindered consumption. However, I think this perspective better represents that scarcity still abounds and a variety of human needs are unmet. The result is an approach to addressing at least three environmental problems while explicitly trying to avoid worsening scarcity in the process.

[9] Stated in terms of consumption, the population wants greater consumption of safety and clean surroundings

### *I.2.2 Part II: Inequality Trends in Economic Growth*

Reducing the impact of human consumption on the environment might be undertaken more effectively by addressing root causes of overexploitation rather than treating the symptoms alone. We might observe, absurdly, that some industries (e.g. coal and oil power plants, garbage incinerators) even cause damages at rates that are several times the value added to society when environmental impacts are included (Muller, Mendelsohn, and Nordhaus, 2011). Chapters 1-3 treat exploitation by advising policy to limit the damages caused in the pursuit of consumption. But what about the factors driving the sort of consumption we pursue? Our behaviour is the result of complex processes resulting in the trade-offs we make. So, one must address the underlying causes of overexploitation, of why we might make short-sighted consumption decisions to the detriment of the environment.

Toward that end, Kuznets (1955) hypothesises what has become known as the Kuznets curve – a relationship of first rising and then falling economic inequality with per-capita income. The generally inverted-U shape of the Kuznets curve has since found application to the environment. Grossman and Krueger (1991) popularise the notion as a relation between environmental degradation and income – destruction increasing with economic development and then declining. While receiving heavy criticism (Deininger and Squire, 1998; Stern, 2004), it has intuitive appeal. One interpretation is that not only does inequality drive growth, perhaps through differential savings rates as in Chapter 5, but the environment also drives growth through transformation into capital (Dasgupta, 2021; Groom and Turk, 2021). A tension then develops in economic growth and development because the natural environment becomes divorced from daily life as we consume it away, but then becomes something we wish to have more of in our lives. Yet for a worker, we cannot conserve for tomorrow unless today's needs are met first. So, perhaps the most significant contribution to make to preserving the environment is to ensure that we can meet consumption needs. At the level of the worker that must choose between, say, illegal logging and conservation, it means ensuring a sufficient and fair share of the returns to economic development reach them. Studies of the environment, growth, and inequality are then inexorably linked.

In Chapter 4, I present a new way to estimate the inequality preferences implied by national economic growth statistics. Inequality preference estimates provide a point of comparison – it allows a policymaker to assess whether the distribution of gains from the aggregate production process match social preferences. It also allows us to assess the extent to which economic volatility impacts the distribution of gains and losses. I also extend the framework to measuring inequality trends between social groups in society. In Chapter 5, *Mean-Spirited Growth* with co-authors Dr Ben Groom and Dr Eli Fenichel, we explore economic growth and societal preferences over inequality further. A more complex model of the economic process suggests how social preferences over inequality result in different economic structures. We propose some statistics to accompany gross domestic product per-capita which highlight when society's distribution of economic growth fits or diverges from social preferences. The hope is that by aiding the design of macroeconomic policy that fits preferences – preferably during both economic booms and busts – the impact of growth on the environment will be lessened, or at least fit social preferences which seem to turn in favour of conservation after sufficient economic development.

### *I.2.3 A Note on Thesis Format*

I have presented a concept that can be explored in many ways because it is an implication of a core principle of economics – the existence and pervasiveness of scarcity. As much of our field exists to explore and address scarcity, avenues for research abound. However, "write three and let it be" is standard advice to those preparing a dissertation. I almost abide by this and emphasise three interesting but perhaps controversial applications in environmental economics: the burning of fossil fuels, nuclear storage, and the processing of consumer wastes. For readability, I sometimes reserve equations, model formalities, and contextual references for presentation as notes, footnotes, and exposition in the appendices. While this attempt at simplification is perhaps less common in a dissertation, the end goal is generally publication of each chapter, in whole or in part, in a preferably interdisciplinary journal. The approach also follows recommendations by Fawcett and Higginson (2012) and Higginson and Fawcett (2016) on equation density and citation rates, and Sen's (1970) stylistic experiment. I do try to integrate formal material where communication necessitates it but otherwise prefer the approach of debating the topics at hand.

Finally, I quote the original, 1895 LSE Prospectus to suggest this research is particularly fitting to pursue through the London School of Economics and Political Science:[10]

> *"The special aim of the School will be, from the first, the study and investigation of the concrete facts of industrial life and <u>the actual working of economic and political relations as they exist</u> or have existed, in the United Kingdom and in foreign countries."*

[10] underline emphasis added, source: http://www.lse.ac.uk/about-lse/our-history

I now begin in earnest with the subject of siting polluting activities that are necessary to society. It is a subject of both modern and historical relevance whose exploration may pay dividends as new technologies become cost-effective, transportation infrastructure modernises, and we attempt to address pressing environmental problems such as biodiversity loss.

# Part I: Applying A NEP Perspective

# 1. A Consideration of Clustering and Dispersion as Abatement Strategies for Localised Pollutants

*Clustering and dispersion of local-scale polluting activities may decrease aggregate – the sum of all local – damages and increase social welfare. I first review five areas of the literature that relate to clustering, dispersion, and local-scale regulation. I then apply structure to the organisation decision that is distinctly applicable to local-scale problems and discuss damage functional assumptions where clustering versus dispersion is preferable. While practical matters of production might bound application, the framework suggests untapped efficiencies in environmental regulation likely abound as well as unpaid recompense due for accepting damages as a service. I present a few empirical examples. I also note applications where some expectation of damages is the relevant datum such as in the storage of nuclear waste. A key takeaway is that the property right to the agenda – whether we decide first the organisation of polluting activities or quantity of emissions – impacts the level of pollution and welfare.*

Keywords: Environmental policy, industrial organisation, clustering, agglomeration, dispersion, local, damages, abatement, nuclear, waste, agriculture, prostitution, homelessness, terrorism.

JEL codes: L22, Q15, Q24, Q53, Q57, R12, R52

## 1.1 Introduction

At present, substantial attention in the press and across the sciences focusses on global-scale concerns such as climate change (see an extensive literature survey on global climate change in Dobes, Jotzo, and Stern, 2014; and discussion of Intergovernmental Panel on Climate Change (IPCC) media coverage in O'Neill, Williams, Kurz, Wiersma, and Boykoff, 2015). However, many pollutants, and subsequently policy problems, are local in nature. The Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) – another United Nations-backed agency of more than 550 environmental scientists – makes such a case (IPBES, 2018). They argue that destruction of biodiversity – often a local-scale concern – is at least as dangerous as climate change.[11] While global climate change captures headlines, these local concerns not only worsen, but are in sum an existential threat to world order. In reply, this paper contributes a discussion on the conceptual siting of polluting activities that do not individually have a global impact, yet cumulatively pose a global threat.

To be clear, I explore clustering and dispersion – the zoning of polluting activities – to mitigate environmental damages at N-localities from N-externality generating processes. I focus on local pollutants which are those that inflict damages on a limited, i.e., generally a sub-global scale. A local emphasis usually entails discussing emissions and damages at a site, community, ecosystem, or watershed scale. While each issue is small, the combined effect from many sites on social welfare is potentially large. Yet despite the small scale of each local issue, intractable issues inhibit effective policy responses. Regulatory and emission activities at some site $i$ are generally treated as irrelevant to those at other sites $j$. This disconnect factors into both the determination of optimal emissions and whether consumers care about the issue. The empowerment of policymakers and regulators may also be at a higher administrative level – say regionally or nationally – such that the concerns of a small subset of the population at local sites are crowded out. Even if policymakers develop a coordinated policy response, perverse incentives result in opposition to successful implementation. Among these, firms have a strong incentive to control the regulatory agenda and may even end up polluting more and across more sites, if successful.

[11] The extent of biodiversity loss does perennially catch the interest of the popular press, for instance Briggs (2020) and IPBES popular press coverage in Watts (2018).

There are several reasons why the traditional environmental policy framework fails to address localised pollutants. Three should be enough to make the case for this research. First, the benefits of production at some site accrue to many members of society offsite due to intensive modern production methods. Meanwhile, a few residents experience the associated damages, perhaps intensively. Rather than a setting for a Coasian solution, the scale of modern production, administrative level of regulation, and rational indifference of consumers leaves the problem unsolved. Second, within localities, it is reasonable to expect that there is a level of damages that totally destroys an ecosystem. The local destruction possibility contrasts with a global public good, such as the atmosphere, where it is likely impossible for mankind to destroy it entirely. I then introduce the possibility of full site degradation into models of damages. The result is that the policymaker may end up choosing how many sites are sacrificed to meet society's needs while preserving others. Third, implementing punitive or restrictive regulation at the scale where local damages occur is difficult. The imposition of restrictions at some site $i$ invites the transfer of production to other sites $j$ – leakage through relocation. Depending on the scale of what "local" implies, this transfer may be to the next municipality or waterway, and the result is emissions leakage without meaningful increases in transportation and other variable relocation-related costs for the polluter.

I begin by outlining a key policy result. I then show more carefully that damage functional form impacts whether polluting industries cause more harm when congregated or dispersed, even for some fixed total output. An implication is that whether environmental scientists discover the correct representation of damages – an important topic (see for instance the discussion in Paul et al., 2020), and whether policymakers use it in crafting regulation, impacts not just the scale, but the sign of policy effectiveness. I compare the resulting impact on total social damages under different industry organisational assumptions. I often assume out of convenience that the default case is a uniform dispersal of some necessary activity across available sites, and then compare damages to those under a concentration of polluting activities with special attention to the role of the total site degradation possibility.

I initially take the level of total emissions as a necessity to meet society's needs, given their technology level. One way to think about this is to discuss a future where society abates all but the most complex emissions to protect public health. Those that remain are then a necessity, and if nothing else, are the remnants of past production such as nuclear waste that takes thousands of years to become safe (Besnard et al. 2019; U.S. NRC, 2002).

There are less extreme ways to defend the framework, but this assures long-term relevance. I then introduce abatement costs into the model and show why the standard marginal abatement cost-marginal damage (*MAC-MD*) framework is ineffective in addressing *N*-locality, *N*-externality generating activity problems. The analysis leads to a revised emissions tax, fee, or penalty system to incentivise congregating to less sites or one site when called for. In doing so, I demonstrate the importance of controlling the environmental agenda. Some generally intuitive examples and empirics then suggest that firms and individuals respond to reorganisation incentives. Finally, I suggest the combined threat of localised pollutants should be taken as a call to action to develop a more complete policy framework. Some of the implications of this exercise may be surprising given the simple nature of the model introduced.

This research draws on several areas of the literature and so an expansive literature review is in order. So, before delving into the local scale damages model I review the relevant literature in five parts – on the impact of non-convexities on economic theory, the land sparing versus land sharing debate, zoning and dispersion policies, damage functional forms, and the policy literature on jointly regulating emissions based on damages and abatement costs.

## 1.2 Literature Review

The primary contribution of this paper is to redefine the local damages function to include a 100-per cent damages possibility. A second contribution is to define the local environment relative to other environments in a region and note the policy implications that result. Several areas of the literature are relevant to these simple additions. I first review the economic discussion on the implications of non-convexities in functional forms on economic theory. This has been a matter of occasional debate for the last century. I then review the hotly debated subject of land sharing versus sparing in regard to environmental protection. Often this debate is focussed on agriculture versus rural conservation, but it certainly does not need to be so constrained. Third, I survey the literature on zoning and dispersion of polluting activities for pollution control. Then, I briefly review the literature on damage functional forms. This extends beyond the economics field and briefly makes note of some comparatively exotic formulations. However, only a few generally representative functional forms are needed to make the points in this paper. Finally, I briefly review the policy-focussed environmental literature on damages and abatement costs pertaining to jointly regulating separated sites. This is the sort of regulatory problem that this paper is primarily intended to support. As these five segments of the literature are all relevant to the subject at hand, there is some overlap between them. I start with the non-convexities debate.

### *1.2.1 Impact of Non-Convexities on Economic Theory Literature*

Economists have been quite concerned with non-convexities in our functional representations for both consumers and firms for some time. Pigou (1920) recognized the issues that non-convexities would cause in finding optimal solutions. Hotelling (1935) and Samuelson (1950) briefly discuss the issue and that it leads to situations where the observer would be unable to tell whether a firm has reached an optimal solution at all. They viewed it as something to basically avoid in economic models. Koopmans (1957) then discusses the implications of ignoring nonconvexities and that the economics profession needs to develop more realistic models including those with non-convex functional forms.

Substantial advancements in addressing non-convexities then occurred in the 1959 through 1969 debate on the matter which resulted in what seemed to be a satisfactory conclusion for mainstream economists. Farrell (1959) notes that in the real world, indifference maps and productions functions will likely often not be entirely convex. After discussing when non-convexities would lead to problems, Farrell then shows that in many cases – including some with corner solutions – that non-convex representations do not impact the optimal solution. This led to a substantial debate, highlighted by a set of correspondence in 1961. Bator (1961), Farrell (1961), Koopmans (1961), and Rothenberg (1961) hotly debate how practical Farrell (1959)'s examples of non-convexities are, among other criticisms, but in general Farrell's points stand.

The debate on non-convexities in standard economic models then gets basically avoided altogether through a convenient observation – that non-convex preferences have a convex hull which will approximate them well when finding optima in competitive markets with a large number of participants. Starr (1969) gets substantial recognition for discussing the validity of such quasi-equilibrium as the Shapley-Folkman theorem in reference to correspondence with Folkman and the contributions of Shapley and Shubik (1966) on approximating, quasi-equilibrium. However, Rothenberg (1960) also discusses the approximating convexity of sums of non-convex sets, and Aumann (1964, 1966) explores the same result under stricter assumptions. Further, Diewert (1982) traces the idea to Wold (1943) as a restatement of taking the convex hull of non-convex preferences, which in turn is a much older and more general mathematical concept. Regardless of origin, the result satisfied many mainstream economists, for example Arrow (1969), and the debate pretty much ends with this convenient though not always applicable solution. Despite the issues of the convexity assumption, it is incredibly convenient. So, one approach is to rely heavily on it anyway and just note that non-convexities violate it as in, for example Montgomery (1972) as did Arrow (1969) on setting up markets for pollution control licenses. But by then the wider debate on nonconvexities in the environmental subfield is underway.

As is well known, one of the fields where the large *N* participant requirement is less often met is environmental economics where we must explicitly consider market thinness. In the case of local-scale damages as in this paper, *N* is often one or close to it. So, the issue of nonconvexities in our functional forms warrants still further discussion. Starrett (1972) gets substantial attention in the environmental economics literature for raising the debate. The issues that Starrett raises have also been raised in different settings by Baumol (1964) and Portes (1970). But what distinguishes Starrett (1972) is showing that tax-based approaches, and under some conditions artificial markets, can still work in the presence of non-convexities.

To summarise Starrett (1972), externalities can be the cause of non-convexities in production functions which may then cause the classical conditions for optimality not to be satisfied. Further, this can cause there to be no stable equilibria in generally thin markets such as for pollution licenses. This depends on whether production and pollution occur on a nonconvex segment of the production-pollution curve. Pollution-driven nonconvexities can also make setting the optimal tax rate difficult. Starrett notes that if we still try to find optimality by, say, raising an environmental tax iteratively, we may just arrive at a local, non-global optimum and not even know that we have done so. By addressing the externality problem then, the policymaker may cause a serious private problem for the firm and a headache for the regulator.

Adding to Starrett (1972), Starrett (1973) discusses the issue in more game theoretic terms. In short, when externalities or other sources cause nonconvexities, there may be no core – stable game theoretic equilibria – in markets and/or we may not be able to distinguish local maxima from global maxima at all. Laffont (1978) continues this line of thought and shows that decentralization of control, like artificial markets, is not likely to lead to Pareto efficient outcomes when nonconvexities result from negative externalities. So then, they have the result that a centrally administered, tax-based approach may be necessary in environmental policy.

Slater (1975) also makes the argument that our MAC-MD framework is only effective in finding interior optimality in the special case of downward sloping MAC curves and upward sloping MD curves. Further, Slater notes that many externalities may be best represented by MD functional forms that are not upward sloping or convex, particularly if the externality is the aggregate of several qualities impacting the polluttee. This is clearly true, but interestingly in the local-damages case there may be a non-zero optimal level of pollution and production in the aggregated case even when in the disaggregated case, say, represented by a downward sloping MD function, suggests zero pollution is optimal at each site. Then, a locality condition provides a justification for otherwise irrational types of production.

Burrows (1986) then takes a different perspective on externality-driven nonconvexities and instead asks how important these nonconvexities are – unlikely special cases or likely to be common? Burrows explores producer-producer models where one firm's externality impacts other firm's output. Even with nonconvexities in the form of a shutdown decision, they still find merit in regulation in many hypotheticals – that the value in regulating often exceeds deadweight losses of doing so imprecisely. Burrows then explores the implications of when many firms, with nonconvex MD functions, are aggregated. In short, the result is not unlike that of quasi-equilibrium in the mainstream economic literature – that we can be reasonably accurate in regulating the aggregate. Burrows also suggests the ability to regulate in the aggregate even in cases where the interior of the MD function is nonconvex. So, while regulating a firm would lead to substantial inefficiencies with imprecise regulation (in producer-producer cases), setting an aggregate for an industry polluting a commons in a way that can be described as globally convex is still possible. In the local-scale case, we must however be quite concerned about interior solutions and nonconvexities as we cannot rely on the aggregate at a local site to lead to quasi-equilibria. We can, however, with sufficient production still rely on the equilibrium at an aggregating site. But then, the issue remains of setting a tax correctly at non-aggregate sites, versus the at least approximately accurate aggregate AAD-based tax to be discussed. But this is not a new problem, rather an existing one that perhaps impacts many studies.

The matter of nonconvexities continues to find some interest in the field. Particularly relevant to this discussion, Helfand and Rubin (1994), notes that nonconvexities of the form of decreasing marginal damages – as in the case for clustering in this paper – suggest a role for policy that attempts to cluster a damaging activity into smaller areas. They identify that a standard Pigouvian tax will not necessarily have the desired effect of concentrating pollutants, but they do not note the alternative I show in this paper because they do not consider the explicitly local damages functional form of this paper either which is necessary to do so.

In the end, it seems that non-convexities are not the theory/optimality destroyer that they were first suggested to be. A century of thought on the matter has certainly made progress. Of most interest here, the aggregating tax proposed later in this paper bypasses some of Starrett's issues. I now continue the literature review by outlining the land sparing versus land sharing debate that is a particularly hot topic at the moment in the fields of environmental economics and sustainability.

### *1.2.2 Land Sparing Versus Sharing Literature*

For many environmental problems, such as those involving water and air pollution, the environmental Kuznets curve hypothesis (an adaptation of Kuznets, 1955) suggests that society can grow its way out of environmental problems. That is, first environmental quality decreases with economic development, but crucially quality then increases with sufficient development. One premise is that once a society's population is sufficiently well off, they can afford to substitute away from polluting means and pay directly or indirectly for a healthier environment. Powerful entities with a focus on development such as the World Bank have advocated for such an approach, for example in Beckerman (1992).

Unfortunately, not all environmental problems fit with the environmental Kuznets hypothesis and rather may be exacerbated by higher incomes. One of these is that higher incomes tend to lead to higher food consumption. So, with globally improving incomes, food demand has surged in recent decades leading to vast expansions in agriculture at the expense of the natural environment. This has resulted in one of the most intense and important debates in the field of conservation – whether to spare or share space between human activities and the natural environment, particularly in terms of agriculture versus land conservation for species habitat. This is the land sparing versus land sharing (LSP-LSH) debate. One particularly promising way that the local damages specification I note can contribute to the environmental field is through informing some cases of the LSP-LSH debate.

Waggoner (1996) might be credited with starting the LSP-LSH debate when they asked how much land we can spare for nature in the pursuit of feeding a global population of ten billion. But Green, Cornell, Scharlemann, and Balmford (2005) really set off the current debate. At the time, conservationists reportedly favoured LSH, while agricultural producers and development-focussed entities favoured LSP. Green, Cornell, Scharlemann, and Balmford outlined the two positions. Importantly they also show, quite approachably, when sparing or sharing would result in greater total yields and species population sizes based on concavity and convexity assumptions about agricultural yields in response to density changes – in other words in response to losses due to field-level conservation.

Fischer et al. (2008) expand on Green and colleagues by taking a case studies approach to explaining the differences between LSP and LSH approaches and outcomes resulting in a set of recommendations for when either policy is preferable. Matson and Vitousek (2006) and others also ask whether increased agricultural intensification will result in more land sparing. Their conclusion is essentially that policies to encourage intensification must be part of a larger LSP framework. Ewers, Scharlemann, Balmford, and Green (2009) further support this point empirically by observing that increases in staple crop yields only weakly increase conservation. As another angle on this debate, Chappell and LaValle (2011) ask whether agricultural production and biodiversity conservation are compatible, or whether it is strictly a matter of trading one for the other. We now know, as Chappell and LaValle suggest, that the ecosystem services received from preserving nature include positive values to agriculture when properly structured. Fischer et al. (2014) suggest integrating such values into LSP-LSH decision models, among the lessons learned during the preceding decade of debate.

Some suggest that in recent years the LSP-LSH debate has leaned in favour of land sparing policies, but the debate itself is far from over (Pearce, 2018). Goulart, Carvalho-Ribeiro, and Soares-Filho (2016) note at least 800 references to the LSP-LSH debate in the literature by 2016 and this has continued to grow. One reason for the interest is no doubt that billions of dollars are at stake when we discuss policies impacting agriculture at a national or global scale. But also, no single LSP-LSH policy can be one-size fits all. Phalan (2018) observes that there are several factors involved and that where a researcher's conclusions on the debate fall can quite often depend on their assumptions. They then see sparing as much land as possible – LSP – as the safest option. That is, policies encouraging highly intensive agricultural production on the least areas possible given all the uncertainties involved are preferred. Goulart, Carvalho-Ribeiro, and Soares-Filho (2016) also note the diverse set of factors involved. But they instead focus on the array of possibilities and assumptions about animal species – including concavity or convexity of damage functional forms – rather than focussing on agricultural production assumptions as in Phalan (2018). The result is that Goulart, Carvalho-Ribeiro, and Soares-Filho instead advocate for sharing – LSH – as their default agriculture-conservation policy. Yet Crespin and Simonetti (2019) suggest that the role of human-nature conflict is underrepresented in the literature. Their point is essentially that LSH is not likely to be an effective policy unless we research and include policies to mitigate conflict between land users and species in the natural environment.

Others suggest that no single approach, nor one conceived in a partial equilibrium vacuum, can be sufficient. Kremen (2015) reviews several quantitative studies of LSP-LSH, then advocates for a mixed solution. That is, suggesting modest land sparing be paired with modest land sharing in surrounding areas. Kremen also suggests the land in agricultural or other production be managed labour intensively rather than by using sources of capital without regard to cost. Finch et al. (2019), Finch, Green, Massimino, Peach, and Balmford (2020), and Finch et al. (2021) make Kremen's recommendations more actionable and recommend a three-tiered landscape of full conservation, high yield farming, and mixed lower yield semi-natural plots. They find in the specific cases of a few bird species that this maximises the summed benefits to wildlife and agricultural production. Basically, they find that land sparing maximises benefits across a number of indicators as long as some lower yielding land is planned around or between conservation plots. Salles, Teillard, Tichit, and Zanella (2017) focus instead on cost, specifically integrating markets, into the LSP-LSH debate at the forefront. They argue that land sparing, from an "economist's perspective" as they claim, is likely to lead to higher demand and more pressure against conservation. That is, higher yields leading to higher profits, leading to more interesting in expanding agricultural production.[12] The bigger point that these authors make is that the policymaker must consider and plan for human behaviour – policies must align incentives, which is hardly new but a critical point. This is increasingly emphasised in later research, such as Jiren, Dorresteijn, Schultner, and Fischer (2018) where careful consideration of the "social-ecological context" – ecological and institutional factors – is found necessary for effective policy. We might include, or even emphasise, getting the prices involved right, which the local-scale model I explore attempts to do.

Of course, LSP-LSH is part of a larger debate on zoning policies which in turn impacts several factors including environmental justice. Campbell, Kim, and Eckerd (2014) model

[12] Salles, Teillard, Tichit, and Zanella (2017) are also concerned as others have been that the spatial arrangement of sparing will lead to decreased efficiency of conservation areas due to issues such as pesticide drift. However, I would argue that if the species of interest is prone to pesticide damages, then a conservation area where pesticide drift can occur was not a conservation area after all and should not be treated as such nor rewards given to producers based on it. Balmford, Green, Onial, Phalan, and Balmford (2019), however, find that even with such imperfections, land sparing often still outperforms sharing in terms of species conservation.

the zoning of polluting activities in terms of the impact of environmental justice in terms of unequal health impacts on minority segments of a population. They find that proactive zoning – in terms of restricting polluting activity locations before the damage can be done – improves environmental justice measures. In comparison, reactive zoning – restricting where people can live after the damage from an activity has been done – first reduces environmental justice measures and then trends toward equivalent in the long-run. So, there are likely costs to getting it wrong and, as in most policy, to changes in policy.

Substantially missing in the debate is the local damages function I outline in this paper, and importantly, the related concept of 'regional' in relation to local sites. The local-damages framework allows the setting of conservation prices – taxation for land sharing approaches versus alternative payments for land sharing plots.[13] Regional instead informs on how maximally far apart conservation plots can be. This may be the limitation of travel distance of the species of interest and when biodiversity corridors are instead necessary for species. In general, the result is a framework supporting the implementation of land sparing approaches while offering alternative prices for land sharing if agricultural producers so choose. The subject of land use in the modern era is naturally closely tied to zoning. So, the next section briefly reviews the use of zoning and recent developments in zoning policy to mitigate the impacts of pollution.

### *1.2.3 Zoning and Dispersion Literature*

Pollution control was perhaps one of the first purposes of urban zoning. In antiquity, polluting and particularly odorous activities necessary to a city's survival would generally take place outside the city's walls. Then, as the walled system of protection fell out of fashion, polluting activities continued to be indirectly assigned to districts. Arendt (1958) notes that, until the industrial revolution, manufacturing tended to be a home-based business and so polluting activities tended to occur in areas of cities zoned for particular occupations or the social groups more often engaged in a particular occupation. Talen (2012) notes that following the industrial revolution, city planners began to zone for polluting industries to mitigate the rampant destruction of the city environment and human health that occurred with mass industrialisation. While zoning and pollution control have perhaps been linked since beginning of urban living, the prevalence and intensity of zoning

[13] Setting aside the matter of leakage which is beyond this paper. However, Finch et al. (2019), Finch, Green, Massimino, Peach, and Balmford (2020), and Finch et al. (2021) address it as a practical matter by also noting the necessity of semi-conservation areas around conservation plots.

experienced today really developed within the last century. Yet for much of zoning's history as today, the objective has remained the same –isolate a particular activity or set of activities from the population. That is, in effect, to aggregate polluters somewhere else.

Holland, Hasegawa, Taylor, and Kauper (1960) discuss how the legal framework for restricting the location of heavy polluters was still in its infancy through the 1950's. Kurtzweg (1973) notes that by the 1970's, with the U.S. EPA established in 1970, that research on safe pollution levels and spatial planning was starting to substantially increase. Like Kurtzweg, other early works such as Dajani, Jett, and Vesilind (1977) were occupied with spatially arranging pollution sources such that some safe emission ceiling was not exceeded. With perhaps a bit of path dependence, research and policy on spatial planning for pollution basically continues to primarily zone polluting activities separate from residential and other uses as single-use zones. Generally, the number and size of zones, if not taken as exogenous, is then a function of transportation costs.

White and Wittman (1982) instead consider under what conditions it is better for polluters to be established in the same areas as pollutees and instead purchase abatement technologies rather than relocate. To operationalise relocation, they consider "double pollution taxes" on both polluters and pollutees such that both parties have incentives to invest and relocate socially efficiently. They then expect that, in the long-run, polluters will tend to agglomerate and that the tax on pollutees overcomes incentives to basically live conveniently near work.

Head, Reis, and Swenson (1995) is an example of another line of research that emerged in this period. They study the incentives for firms to agglomerate even without pollution taxes being applied. Hochman and Rausser (1990, 1999) then model whether zoning is needed at all and how many zones are optimal based on a dispersion function that is either concave or convex as well as travel costs and alternative land use values. The dispersion function models how pollution from an industry is transmitted, with a convex function suggesting that pollution levels drop off rapidly with distance and subsequently implying that an aggregated, larger zoned area of polluting industry is less damaging (minimise the border between a polluting zone and surrounding areas). Rather than a Pigouvian tax, Hochman and Rausser arrive at a tax equal to the total land damages in the grid of a polluter to then use in balancing land use for polluting activities against other rent generating ones such as residential uses.

So, we can observe that zoning need not be strictly by command-and-control type approaches. A multisite approach to zoning and pollution with a bit of free will preservation is to identify the optimal level of tax applicable to each pollutant at each location and apply it such that industry organises efficiently. This is in, for example, Tomasi and Weise (1994) and is sort of zoning by taxation. (They also find that a system of Pigouvian taxes may be insufficient for reasons like those discussed by Hochman and Rausser). One can also take the location of polluting activities as exogenous and effectively permanent, then apply a tax as in Goetz and Zilberman (2000). Either approach as well as outright zoning by ordinance may be appropriate and sometimes sufficient under different circumstances. Taking location as given is, for instance, perhaps the most realistic assumption when dealing with agricultural production. In comparison, not assuming that location is fixed may be more appropriate when regulating housing developments or the location of industry with modest relocation costs.

As noted, Head, Reis, and Swenson (1995), Puga (2010), and certainly in others there are incentives for firms to aggregate without state intervention. These include information sharing/learning, development of a robust labour pool, and reductions in operating and logistic costs. Locations may also have natural advantages – known as a "first nature advantage" – that lead to agglomeration (going back to at least Marshall, 1890). One example is the build-up of (polluting) shipping activities in an area due to the construction of a port in a natural harbour. But since firms do not always aggregate, one must assume that what are known as "centrifugal" forces must also operate on industry. Particularly appropriate for the topic at hand, van Marrewijkm (2005) outlines that local-scale pollution from industry itself or collocating ones may make aggregation undesirable. It then stands that intervention may be in order when the aspects of a pollutant suggest that aggregating would be socially optimal while a centrifugal force has been naturally dominant.

Lange and Quaas (2007) further explore the centrifugal effect of pollution on industry location by adding transportation and human capital considerations. Of more novelty, increasing human capital reduces the number of workers that would be exposed to pollution for a given level of output, and subsequently offsets the effects of pollution on organisation. Kyriakopoulou and Xepapadeas (2013) presents a model where the centrifugal and centripetal forces on industry can result in aggregation, particularly around natural advantages, but also dispersion of activities with increasing regulation. Kyriakopoulou and Xepapadeas (2017) then explore the competing forces on industry location further while adding in labour location incentives. Related to White and Wittman (1982), they find that both a tax on polluters which is site-specific, and a subsidy targeting labour which is also site-specific is necessary to reach optimal allocation in their model. Both approaches would seem to require a more substantial regulatory burden than most policymakers prefer to undertake.

Arnott, Hochman, and Rausser (2008), like many of their predecessors, explore the role of transportation costs. They note that as transportation costs increase, the incentive to aggregate or zone industry away from residences decreases compared to residential-industry integration – perhaps with additional regulatory and abatement costs – and the impact on rents and prices is explored. A corrective tax of the form in Hochman and Rausser (1990, 1999) based on the total damages to a parcel from a polluting activity is then noted as bringing about an efficient spatial allocation. These and other research on the role of transportation costs have led to interesting innovations in the idea of optimal taxation versus outright zoning. However, my research sidesteps transportation costs to take a different perspective and instead assumes they are negligible in comparison to other costs involved in production and pollution. A limited interpretation is that the present paper constrains itself to particularly local – very small – environmental problems. However, the empirical applications to follow suggest that the former, more general description of transportation costs being minor compared to other considerations is more accurate.

I next present a short literature review on damage functional forms before discussing the joint regulation of polluting activities at separate sites. In zoning as well as the LSP-LSH debate, knowledge of the functional form of damages is often necessary to know whether policy to aggregate or disperse an activity is more socially desirable.

### *1.2.4 Functional Form Literature*

A brief review of damage functional forms is outlined here. It precedes the discussion of the four generally representative ones that will be emphasised later in the paper. I have already noted that the concentrating possibility is suggested in Helfand and Rubin (1994). The issue of uncertainty over damage functional form currently weighs heavily in environmental discussions. Metcalf and Stock (2017), for instance, emphasise the seriousness of this uncertainty in estimating the social cost of carbon with any confidence, and Kopp et al. (2012) suggests functional form impacts the results of many studies. Some discussions of damage form in environmental economics are in the context of global-scale concerns. But others such as Paul et al. (2020) and Dasgupta and Mäler (2003) undertake studies of functional form within the context of biodiversity and ecology. Functional form also finds extensive discussion in toxicology and, more recently, legal literatures. I find the toxicology literature to be a good place from which to initiate a discussion of functional forms.

Toxicology has a long history of discovering and debating damage functional forms at the levels of individuals and populations. These usually take the form of dose-responses – the biological response in terms or mortality, cancerous tumour growth, etc. to levels of some exposure (dosage). An extensive discussion of such functional forms can be found in Eaton and Gilbert (2008). They include discussions of concave and convex biological examples such as derived from Lassiter, Barone Jr., Moser, and Padilla (1999) (concave), and Dobo et al. (2011) (convex), but also more complex relationships such as the logistic S-shaped dose-response. An S-shaped form can be observed in many studies. A simple and, for many, relatable example is on the dose-response to wine consumption in Gilbert (2012), and a more environmentally relevant application in mining silica exposure in Utembe, Faustman, Matatiele, and Gulumian (2015). Nash and Revesz (2001) equivalently discuss concave and convex damage functional forms in the context of local and regional tradable permit schemes – that the optimal distribution depends on damage functional form.

Calabrese and Baldwin (2003) discuss the ramifications of hormesis in toxicology – that small doses of some substances are beneficial – which is equivalently that negative marginal damages initially result from some polluting process.[14] Wiener (2004) discusses linear, convex (discussed as supralinear), concave (sublinear), and also hormetic functional forms in the context of emissions trading schemes. The ramifications of the hormetic functional possibility has been further debated recently, for instance in whether polluters should emit more nitrogen oxides in urban settings to reduce ozone (see comment by Fraas and Lutter (2012) and reply by Muller and Mendelsohn (2012a)).

There are, however, many potential complications to modelling dose-responses or exposure-damages relationships well. Goodson et al. (2015) discusses that synergistic effects – interactions between different carcinogens in their case – may increase total damages. Clewell, Thompson, and Clewell (2019) find modelling complications in that there can be many "thresholds" for mutation and toxicities leading to cancers – points of sudden change in the response to carcinogen dosage. Another possibility is a relationship that is not hormetic but has a minimum threshold before any measurable response is observed. Waddell (2004) emphasises that there is a time aspect to response levels and when subjects are tested. In Waddell's case, the results of extrapolation from low-dose to

[14] The existence of hormesis has long been conjectured in folk medicine as well as folk environmentalism. For instance, exposure to small doses of allergens to address allergies, and the existence of species that thrive following limited oil spills.

high-dose effects depends on when dissection of their subject rats occur. Faustman and Omenn (2012) discuss that such uncertainties make the process of toxicological risk assessments extremely difficult, and by extension, so too other risk assessments. Controversially, the U.S. Environmental Potection Agency's response to uncertainty has been to assume linear dose-response relationships as the baseline until evidence suggests otherwise (EPA, 2005) (Abt, Rodricks, Levy, Zeise, and Burke (2010) discuss a decision process to move away from the EPA's linear assumption). One can also find several dose-response relationships not fitting the aforementioned cases, Sunstein (2010) discusses a few of these.

A couple notes remain to be made on the environmental justice component and possibility of corner solutions despite concave functional forms. An important discussion on functional form is emerging in the legal and environmental justice literature. Rowell (2012) sets off the debate by discussing how functional form and the allocation of exposure has extensive ramifications on environmental justice and the existence of potential Pareto improvements through distribution. This has immediate parallels in the joint regulation of separate sites as in, say, Muller and Mendelsohn (2009), Fowlie and Muller (2019), and this paper. It has also been noted in the literature that the MAC-MD framework can fail to arrive at an interior solution even without nonconvexities or international trade. Winrich (1982) outlines that if a damage function is instead sufficiently concave, any optimal planning will be confronted with a corner solution – an all or nothing optimal policy. We are again in need of alternative policy tools as in this paper to address production with externalities. I next discuss the literature around damages and abatement cost-based joint regulation.

### *1.2.5 Damage and Abatement Cost Policy Literature*

Preceding a discussion of jointly regulating polluted sites, it is perhaps informative to connect the subject to the international trade literature on specialisation. This is useful because people – or at least firms and political establishments – have repeatedly shown themselves to be willing to specialise in a polluting activity, raising the quantity of pollution produced locally, when their incentives sufficiently support such a course of action. Dean (1992), Azhar and Elliott (2007), Copeland (2000), Copeland and Taylor (1994), and Copeland and Taylor (1999) each show in the context of international trade that just such aggregation – in the form of spatially separating incompatible industries – can and does occur internationally. Generally, the resulting division is specialisation of the cleaner industry in a more financially advanced economy while a less fiscally advanced one specialises in polluting industries that are polluting. This certainly should not be surprising as it is a natural extension of Smithian, Ricardian, and other discussions on the division of labour and resources. Benarroch and Thille (2001) counters that transboundary pollution would muddle the picture – that specialisation/clustering of polluting activities does not work when substantial transboundary leakage is present. This is fitting with the conditions put on local damages in this paper where leakage is excluded or insignificant. However, Unteroberdoerster (2001) also suggests that specific assumptions about the form and who is impacted by transboundary pollution matter and the result of specialisation may hold.

One reason that the literature on international specialization is first noted in this section is that the literature on damage functions was in general quite focused on the transboundary pollution problem for a while (Sturm, 2003) leading to a bit of path dependence in discussing local scale-type environmental problems. One result is that the line of literature on transboundary spillovers – leakage – suffers from a lack of key characteristics of the local damages functional form outlined in this paper. So, the ability to aggregate to a single site is not, unfortunately, even within the purview of many papers outside of a gains from specialisation argument. In the preceding literature review on the LSP-LSH debate, the costs of either approach are often discussed at a farm level and sometimes summed over a landscape, fictional or literal. Here I will note some of the literature on instead jointly regulating something akin to a landscape. But it is perhaps first useful to again digress and note why we need a policy at all.

Silva and Caplan (1997), Boadway Song, and Tremblay (2013), Ogawa and Wildasin (2009), and Arbex, Behringer, and Trudeau (2017) suggest that decentralised planning can be at least as efficient as central control over environmental matters. However, Fell and Kaffine (2014) instead suggest that such results are heavily assumption driven, and rather that centralised planning is necessary for efficiency. But perhaps more important in either case, Coase (1960) outlines stringent conditions for an environmental problem, even a local one, to find redress without state intervention. Baumol and Oates (1988) also note that even when parties can reach an agreement, the "solution" is sometimes to shift an externality onto other groups not involved in the negotiations. The containment requirement of this paper's framework – no leakage from sites – makes it distinct from, say, a state "black-spot" policy as discussed in Siebert (2008). In a black-spot policy, the state balances summed damage savings against, say, international opposition and replies, and a total damage possibility is not assumed or discussed. However, there are parallels in the decision process. Cases of redress without state intervention exist of course. Ostrom (1990) chronicles hundreds of cases where successful management of the commons occurs without an exogenous power involved. Ostrom's metanalysis notes several factors that are common to successful cases. Summing them up: it is arduous work, requires close and frequent interaction between all parties, and generally expensive monitoring. Parties cannot solve many environmental problems independently as they often fail the Coase conditions for an agreement to be possible – particularly low transaction costs – or do not exemplify the Ostrom factors that make one probable. It is then likely, if an environmental problem is to be addressed at all, that a central power intervenes to coordinate intervention across local sites.

Fortunately, many environmental problems have convenient non-Coasian, non-Ostrom analytical interpretations and solutions. Generally, these begin by defining the environmental problem in terms of an externality, which is hardly a new idea. Adam Smith in 1776 gives an example of a positive externality (from education) at a time when the world's resources still seemed limitless (Smith, 1994 edition), and Marquis de Condorcet in the same year provides examples of a few negative externalities (Sandmo, 2015) albeit somewhat arbitrarily deciding who was wronged (from factory pollution). Pigou (1920) generally receives credit for formalising the idea of the externality, particularly negative ones. Pigou's concept received a lot of criticism on various technical grounds in the first decades, for example by Buchanan (1962). However, operationalising Pigou's ideas on optimal standards and taxes by Baumol and Oates (1971), and defence of Pigou in Baumol (1972), muted criticism and modernised policymaking on environmental externalities.

The most recognisable and intuitive framework for many environmental problems summarises Baumol and Oates' general equilibrium model, where the user contrasts environmental damages against abatement costs. The shortest review is that policymakers seek to minimise the sum of abatement costs, $AC(x)$, and damages, $D(x)$, as deterministic, separable, and continuous functions of emissions, *x*, as $min_{\{x\}}\, AC(x) + D(x)$.[15] The unconstrained optimality condition for an interior solution – where neither "cost" overwhelms the other in scale, is $-\frac{dAC(x)}{dx} = \frac{dD(x)}{dx}$. We often write this as marginal abatement costs, $MAC(x)$, versus marginal damages, $MD(x)$, as the point where $MAC(x) = MD(x)$. Likely the most familiar version of the MAC-MD framework incorporates externalities into Samuelson's (1954) pooled public good. But this is usually

[15] So, expressed as additive rather than compounding interactions occurring.

depicted as a $1 \times 1$ space of one polluter creating one environment problem and hardly descriptive of many environmental problems or society-wide regulatory environments.

The more general situation is to assess $N$ polluters and $M$ receivers of damages. Muller and Mendelsohn (2009) provide a heterogeneous, non-local $N \times M$ application of several polluter's emissions inflicting overlapping damages at many sites in a way that modern computing has only recently allowed researchers to quantify. The difficulty of relating damages to specific sources in the setting of Muller and Mendelsohn (2009) – of an overlapping "mixing commons" – is a sticking point for some, for example Fraas and Lutter (2012). But this uncertainty is addressed in reply by Muller and Mendelsohn (2012a) as well as by Muller (2011). The general impact of differential damages are then explored further in Hsiang, Oliva, and Walker (2019), as well as the redress framework advocated for extensively in Muller and Mendelsohn (2012b). Fowlie and Muller (2019) explore further optimal regulation in such a mixing commons. They find damages-based, differentiated tax policies are first-best – essentially the preceding separate-optimal regulatory framework – as they are not exploring a regulatory problem addressing damages of the local form. In fact, no application addresses jointly regulating $N \times M$ disjoint, local-scale polluter-environment pairs where full site destruction is possible. As implicit in Muller and Mendelsohn (2009), the "polluter" at a site represents the summed pollution of perhaps many polluters at the site. Within such a site, the standard framework of allocating among several polluters applies here too. The joint regulatory framework is instead another layer that sets the total of site level emissions, to then be allocated within the site. We might visualize the current application as a diagonal $N \times M$ matrix where $N = M$. Regulation of these disjoint and local sites, whether jointly or separately, has significant implications for both optimality and implementation of environmental regulation.

Having now reviewed five relevant areas of the literature, I proceed with a basic model of local scale environmental damages. The next section outlines the perhaps modest adjustment to damages functional form that is needed to represent local-scale problems and environments more accurately. It is a simple contribution with significant implications for how we optimally regulate many pollutants as well as social problems.

## 1.2 A General Model of Local Pollution Damages

It turns out that under certain conditions, it is always better from an environmental perspective to aggregate some types of activities. To arrive at the relevant conditions, I use a simple framework for thinking about local environmental damages. Let a per-site damage relation, $D_i(x_i)$, be defined piecewise as

$$(1) \qquad D_i(x_i) = \begin{cases} d_i(x_i), & 0 \leq x_i \leq 100 \\ 100, & x_i > 100 \end{cases},$$

such that

$$D_i(x_i): \{x_i \epsilon \mathbb{R} | 0 \leq x_i \leq +\infty\} \rightarrow \{D_i(x_i) \epsilon \mathbb{R} | 0 \leq D_i(x_i) \leq 100\},$$

and assume scaling

$$d_i(x_i = 0) = 0, and\ d_i(x_i = 100) = 100$$

where I discuss $x_i$ and $D_i(x_i)$ in percentages unless it is more convenient to use other terms. There is no reason not to consider more complex mappings, save for clarity. Scaling the input and output to percentages defines exposure $x_i$ by result $D_i(x_i)$ such that exposure, or a dose, of $x_i = 0$-per cent causes $D_i(x_i) = 0$-per cent degradation of the environment in question. Similarly, a dose of $x_i = 100$-per cent is a threshold where 100-per cent of the environmental value of a site or system is lost. That is, polluters inflict 100-per cent of possible damages. The damage specification also allows exposure beyond full degradation. In between the zero and 100-per cent extremum, $d_i(x_i)$ specifies the rate of change or transition in damages from exposure and is sometimes critical to finding when aggregation or dispersion of a polluting activity is in the public's interest. The transition and extremum cases also suggest when some value is figuratively left on the table when aggregation and dispersion possibilities are ignored. An example emphasises the analytical value of the framework. Suppose we deposit a meter of refuse atop some natural environment. Afterwards, several more meters of rubbish deposited cannot further damage the extinct ecosystem buried below. However, choosing to deposit more rubbish at the site avoids damages at other sites that could have been used for dumping too. As in the refuse example, suppose the local site is contained – or at least is intended to be – such that a no-leakage condition is included in this analysis.

We must also have definitions for the terms "local" and "regional". I define "local" endogenous to the damage specification and the related term "regional" to cost. Local is the scale where an activity can conceivably result in 100-per cent damages. This definition is pragmatic to features of the polluting activity and pollutant rather than as a geographic feature. The concept of regional complements local by placing a limit on site substitutability. It is the scale where we can discuss selection among alternative sites over a long enough time horizon – the long run – as costless. In an industrial setting, say the siting of a replacement powerplant, regional may cover several hundred square kilometres and thought of as a costless move because, by the end of a power plant's service life, related infrastructure likely must also be renewed. Regional might be a city, neighbourhood, or even a city park or similarly small setting. However, it might also be a nation or even global – I note examples of both extremes in the appendix. Throughout this paper, I then discuss choosing among local sites as though it is costless, but this is a simplification. The concept region is the financial bound where the costless movement among local sites within it is not a misleading approximation.

### *1.2.1 The Accounting of Damages Across Isolated Local Sites*

The sum of pollution damages across $N$ locations is $\sum_{i=1}^{N} D_i(x_i)$ from a total of $X = \sum_{i=1}^{N} x_i$ polluting activity. Whether the aggregate is subadditive, additive, or superadditive has implications for interior, joint-optimality. However, if $X \geq 100$ is possible for any one site, then it is sufficient to render at least one of $N$ sites valueless in supplying environmental amenities. It then follows that the joint-optimal outcome is sometimes a corner solution – the novel contribution of the preceding conceptual framework.

I first discuss industry organisation nonparametrically, perhaps unnecessarily so, to put minimal structure on the problem. To be explicit, we can expand the sum of damages as $D_N(x_N) + D_{N-1}(x_{N-1}) + \sum_{i=1}^{N-2} D_i(x_i)$ where the ordering of $x_i, i = 1, \dots, N$ is arbitrary if sites are symmetric. With site and exposure symmetry can be assumed, $D(\bar{x}) = D_i(x_i) = D_j\left(x_j\right)$ for $\bar{x} = \frac{1}{N}\sum_{i=1}^{N} x_i$ at any sites $i, j$. We can then specify a comparison of incomplete clustering as $D(M\bar{x}) + (N - M)D(\bar{x}) \gtreqless ND(\bar{x})$ where $M$ sites of activity consolidate. In the case where only two sites engage in the polluting activity, the useful simplification is to compare $D(x_1 + x_2) \gtreqless D(x_1) + D(x_2)$ for pollution levels $x_1$ and $x_2$. The two-site case clearly highlights that if the relation $d_i(x_i)$ is a twice differentiable and well-behaved function, concavity or convexity over the domain of exposure informs whether greater or lesser marginal damages occur.[16] We can appeal to Jensen's (1906) inequality for support on this matter. The framework applies to clustering and dispersion benefits in all manner of organisational structures. I focus on the value of clustering to one site versus the dispersion to many equally of all polluting activities. That is, comparing the effect of industrial structure $D(N\bar{x})$ to $ND(\bar{x})$ for clarity.

At the risk of spoiling the experience, I label three general, damages-based formulations as the cases for clustering, dispersion, and indifference. These depend on whether $d_i(x_i)$ is convex, concave, or linear, respectively. I also discuss the more pragmatic logistical damage form. The latter finds frequent application in integrated assessment modelling, natural sciences, and medicine as descriptive of a biological or ecosystem dose-response to some exposure. While I refer to it as the biological case, it is sometimes formally the convex-concave case. The next section outlines the form of Equation (1) under each of the four functional forms of $d(x)$, a brief note on other alternatives, and the implications for damage avoidance-based policies.

## 1.3 Representations of Local Damages and Joint-Optimal Implications

Having outlined the key characteristic of the local-scale damage function and damage accounting across local sites, I now emphasise four general damage functional forms. As the literature review notes, these are not exhaustive but chosen to cover a range of applicability and convey intuition. The critical point in choosing among or diverging from these is to fit the circumstances under study.

### *1.3.1 Four Basic Local Environmental Damage Relations*

In practice, the difficulty of discovering a representative functional form may be more severe in the study of localised pollutants where funding is limited. However, researchers may also have specific, informative site knowledge, and increased sensitivity to pollution may provide useful variation.[17] To aid selection, I explore the policy implications of selecting among four basic, generally representative functional forms. These may be further

[16] Another representation of whether it is better to aggregate or disperse occurs by defining $x'' > \bar{x}_i > x'$ and $x'' - \bar{x}_i = \bar{x}_i - x'$ including the case when $x' = 0$ such that total polluting activity, $X$ is unchanged. Compare fully symmetric $ND(\bar{x})$ to $D_N(x'') + D_{N-1}(x') + (N-2)D(\bar{x})$. Let the state represented by equal sharing of the pollution load be $\mathbb{D}(X)$ and the alternative be $\mathbb{D}(X)'$. If $\mathbb{D}(X)' > \mathbb{D}(X)$, concentration increases total damages. Assuming welfare is strictly decreasing in damages, thus reduces welfare, and suggests mandating dispersion as a welfare increasing policy. If $\mathbb{D}(X)' = \mathbb{D}(X)$, marginal changes in damages are either constant or symmetrically increasing and decreasing such that society prefers neither structure over the other at level X. Alternatively, if $\mathbb{D}(X)' < \mathbb{D}(X)$, concentration decreases total damages and increases total welfare suggesting clustering as a method of damage abatement.

[17] Siebert (1975) makes a similar argument in that regional regulators may be more attentive to a population's issues than federal ones (and that regional regulation may be inefficient because they do not account for changes in emissions outside their domains).

extended with caution to site-specific needs, and the literature informs on the variety of alterations available. This study, however, is unique in modelling localised pollution damages in a way that allows for total loss.

I begin with similarities. Each representation fits the piecewise damage relation of Equation (1), sharing the implication that beyond $d_i(x_i = 100) = 100$ the marginal damage from additional exposure is zero. This is equivalently known as a "ceiling effect" in pharmacology. The implication has a powerful impact on the analysis: In a closed system – one that does not allow spillovers to other environments – the sum of site damages is always subadditive at a sufficient production scale. It follows that transferring in polluting activities from other sites results in society-level welfare improvements. A policy of clustering may still be undesirable to implement without compensation to those it impacts on equity grounds. However, we may observe it in real policy preferences. For instance, activities resulting in a high degree of toxicity or danger tend to occur at highly aggregated sites. Munitions dumps, nuclear waste disposal sites, and municipal dumps all tend to be large and serve many customers outside the immediate area. They also tend to find use solely for their primary purpose as though they hold no environmental or other use- and non-use-value.

But not all environmental problems are of sufficient scale to result in corner joint-optimality. To analyse interior cases, I continue to assume that any two sites in comparison are symmetrical unless stated otherwise. In real site comparisons, users must consider different population levels, sensitivities, etc. at every site. But without site and pollutant specifics, it is not useful to focus on asymmetries at this point. I take each $d_i(x_i)$ as a twice differentiable function to discuss them in simple language and manipulate easily. I also only discuss exposure and damages as environmental "bads" and that the least damages outcome is an unexposed environment. However, the framework extends to, say, a development context. Four basic cases follow under these assumptions.

*The case for dispersion*: The first case is of increasing marginal damages. Dispensing with site notation, from initial exposure, $x$, a unit increase results in a greater magnitude of damages than from preceding units. I present this case in the marginal form in Figure 2, top-left. One burden in its use is that the policymaker must rationalise why each additional unit is progressively worse. Another is the mathematical discontinuity at 100-per cent exposure. In a biological case, justification is on the basis that full extinction suddenly occurs. But then, one must defend why full extinction does not occur at a lesser dose when the population is unsustainable. Case for dispersion sites are superadditive in damages,

$D(Nx) > ND(x)$ when $X = Nx < 100$, and any division of $X$ other than equally among the largest number of sites available results in more damages (Note 1).

Note 1 – Applying Jensen's Inequality

Theorem: if $d(x)$ is convex in $x$, total damages are minimised by dispersion. Likewise, if $d(x)$ is concave in $x$, total damages are minimised by clustering. Proof: This is immediate from the definitions of convexity, concavity, and Jensen's (1906) inequality. In the case for dispersion, Jensen's inequality is generally stated for two sites, $x_1, x_2$, and production share $\lambda \leq 1$ as the inequality $D(\lambda x_1 + (1-\lambda)x_2) > \lambda D(x_1) + (1-\lambda)D(x_2)$ while $x_1 + x_2 < 100$. This would be conversely stated for a case for clustering. A proof on why aggregation is preferable beyond 100-per cent damages is also immediate: $0 < 0 + \varepsilon$ for any positive value $\varepsilon$.

*The case for clustering*: The second case is characterised by an opposite effect – decreasing marginal damages. Each additional unit of exposure results in a smaller increase in total damages than from the prior unit. One application is when subsets of a population show greater resilience than others. I present this case in the marginal form in Figure 2, top-right. It suggests that the first increment of exposure is more damaging than any subsequent exposure – an assertion the scientist or policymaker must carefully consider. In this case, sites are subadditive in damages – minimisation of total damages always occurs through moving the activity to the least number of sites possible for any $X$.[18] This may make a case for clustering pollutant easier to regulate as there is not an interior-versus-corner optimal policy dichotomy.

[18] The feasibility of production in toxic environments has changed in the era of automation. Aggregation to higher levels of pollution paired with automation may now make feasible industrial arrangements that were limited by worker health in the past.

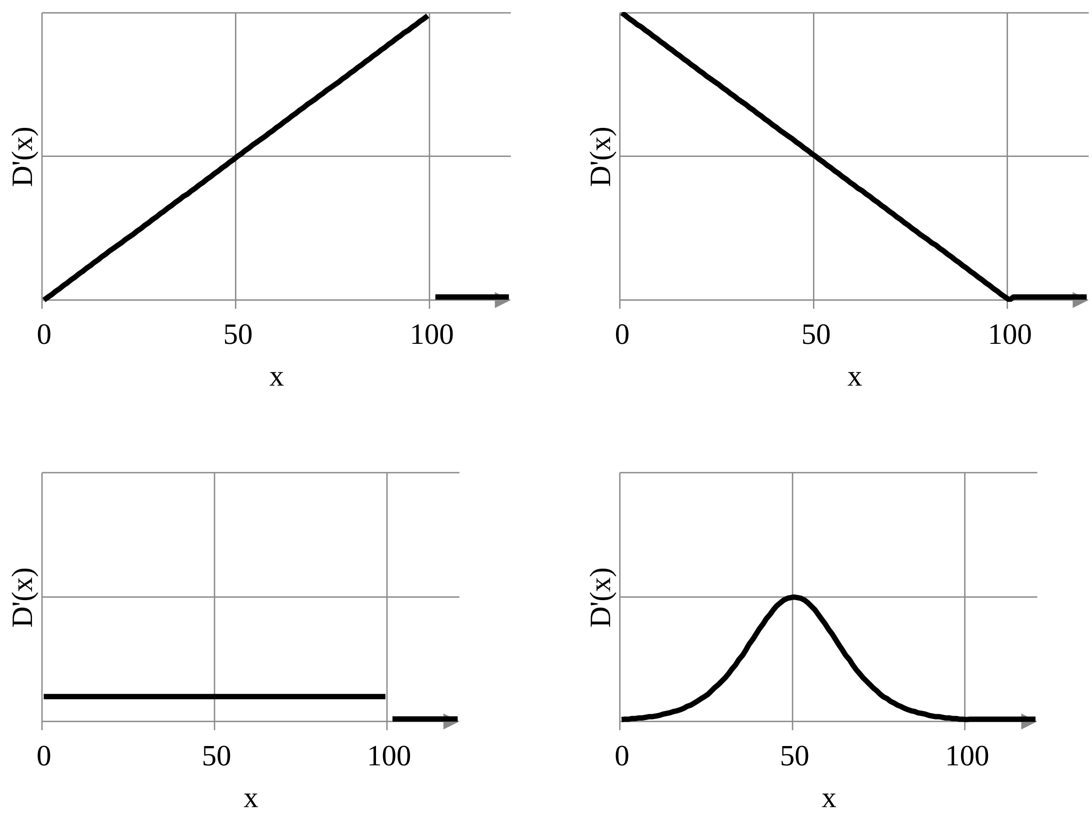


FIGURE 2 GENERAL REPRESENTATIONS OF MARGINAL DAMAGE FUNCTIONAL FORMS

Notes: Four general marginal damage functional forms meeting the conditions of Equation (1). I include plots of their total damage forms and production origins in appendix *A1.1 Damage Functional Forms*.

(top-left) The case for dispersion: $d'(x) > 0, d''(x) > 0$.

(top-right) The case for clustering: $d'(x) > 0, d''(x) < 0$.

(bottom-left) The local indifference case: $d'(x) > 0, d''(x) = 0$

(bottom-right) The biological (logistic) case: $d'(x) > 0, d''(x) \geq 0\ on\ x \leq 50$ and $d'(x) > 0, d''(x) \leq 0\ on\ x > 50$

*The case for indifference*: Constant marginal damages characterise the third case as in Figure 2, bottom-left panel. Any unit change in exposure results in an identical change in marginal damages up to the discontinuity at $x = 100$. This case may be appealing on the grounds of simplicity, but the practitioner should justify the discontinuity. As a practical matter, kinks and discontinuities may be more straw men than substantive critiques – a functional form with many kinks and discontinuities can best represent complex environments. See, for instance, Mendelsohn and Schlesinger (1999) or Win, Zin, Kawasaki, and San (2018) where damages from environmental events are divided into component damage relations. These, in turn, may sum to an unsmooth combined functional form. The essential point is to undertake careful contemplation (Pindyck, 2017). In the case for indifference, dispersal and clustering are irrelevant up to the point where total damages at a single site are possible by aggregating the activity. Additive site damages in the case for indifference can then be grouped with clustering as it is never an inferior strategy to do so.

*The biological case*: Finally, I present a more involved case of both increasing and decreasing marginal damages, depending on the level of exposure, in marginal form in Figure 2, bottom-right. The logistical form (see appendix *A1.1 Damage Functional Forms* for context) has origins in the biological and medical sciences where populations have a degree of diversity.[19] The level of diversity is the essential justification – the practitioner believes some members of the population to be substantially affected by a pollutant, while others are resilient. At an ecosystem scale, some forms of life, e.g. amphibians, brachiopods, and insects, may be harmed by small doses of a waterborne pollutant (Kerby, Richards-Hrdlicka, Storfer, and Skelly, 2010) while extraordinary doses are necessary to denude the landscape. The generic form of this case does not contain discontinuities to justify but does have an inflexion point in the aggregate form which I plot at $x = 50$. This level is discussed in the biological and medical literature as $LD_{50}$ – the "lethal dose" resulting in a 50-per cent mortality rate – an important standard in comparisons. A drawback is that the biological case requires a higher threshold of institutional knowledge to regulate well as up to the inflexion point dispersion is preferred, then clustering is preferable.[20]

We might also consider the implications of lower threshold and hormetic damage functional forms. If the number of sites is not constrained, a regulator might prefer an arrangement that allows each site to operate below the lower threshold. In the hormetic case, this can be further improved upon by operating each site at the damage minimum – the point where negative damages are maximised. Within the context of weighing the benefits of damage reductions against abatement costs, however, lower threshold and hormetic additions are likely unimportant. Rather, the optimal allocation weighs these aspects of the damage function against the costs of abatement.

In total, the four basic cases condense to two optimal arrangements – either clustering or dispersion, and it is always preferable to encourage clustering with sufficient total emissions. Limitations to these choices exist. In the introductory example, clustering in agriculture may reduce some damage, yet the necessity of dispersing over vast tracts of

[19] Verhulst devised the logistic form in 1838 to describe asymptotic population growth in response to Malthusianism (Bacaër, 2011; Verhulst, 1838). But it might also be traced to Gompertz' (1825) work describing human mortality rates. In both cases, their general forms have been widely adopted in the biological and medical sciences.

[20] There is also a special case where symmetric transfers around $LD_{50}$ can results in the case for indifference if the slope of the marginal damage function is comparable around the point and a balanced number of sites transfer pollution.

land is often necessary for production. Even so, I have demonstrated that this framework is still applicable in the planning of, say, agricultural drainage to waterways. Reorganisation to reduce damages may also be a matter that polluters can seek remuneration for as a service. Alternatively, a community willing or forced to accept an aggregated or dispersed polluting activity may have an additional basis for payments beyond redress. The scale of such payments may be large and pivotal during policy development. Next, before exploring optimality, I discuss how the framework should change when based on expectations of damages rather than certainty.

### *1.3.2 Decision Making Using Expectations: Nuclear Sites and Disasters*

Many environmental decisions must be made based on an expectation of exposure rather than stocks and flows of pollution. That is, the population accepts exposure with some probability, while expecting many sites to incur zero damage. They – the public, site managers, and policymakers – may approve activities that can result in devastating damages because the probability of exposure is small. Nuclear power production and fuel disposal are cases that immediately come to mind.

Say some exposure $x_i$ occurs at site *i* with probability $p_i$, zero exposure with probability $1 - p_i$, and the probability of exposure increases in the scale of the activity. The expectation of exposure $E[x_i] > 0$ and resulting damages $D_i(E[x_i])$ are the relevant ex-ante decision datum.[21] Such binary risk occurs in using a substance sufficiently toxic that any release renders an environment undesirable for further activity. That is, $D_i(x_i > 0) = 100$ and any additional exposure results in zero marginal damages. Binary states need not be the only application and non-use value may complicate matters. For instance, there are reports of abundant wildlife, including endangered species, within the Chernobyl Exclusion Zone and Korean and Cyprus demilitarised zones (Mycio, 2005; Billock, 2018; and Constantinou, Hadjimichael, and Eftychiou, 2020, respectively). But for the purpose at hand, I find binary states and use value analysis sufficient.

Before exposure, the calculus of site selection up to this point holds. However, once exposure occurs at a site $i$, the decision simplifies. When exposure $x_i$ occurs, we still have $x_j = 0$ occurring at all other sites and these other sites continue to operate in the world of

[21] This paper does not address damages-dependent utility functions which are assumed to be decreasing in damages, ceteris paribus. However, note that a utility function has an expected utility form if and only if (iff) it is linear, in this case iff $U(p_i D_i(x_i) + (1 - p_i)0) = p_i U(D_i(x_i)) + (1 - p_i)U(0)$ which we might assume simplifies here to iff $U(p_i D_i(x_i)) = p_i U(D_i(x_i))$. So, whether we use the expected utility form need not be determined by the damage functional form.

expectations. For pollutants with lengthy rates of decay and high clean-up costs, the exposure level at site $i$ may now be a permanent feature. Policymakers and operators face the decision to shut down the exposed site, transferring the potentially polluting activity elsewhere, or allowing continued use by perhaps using greater automation or protective measures. The consumer needs that have been driving production (including that of safety) and total polluting activity have not changed, and thus demand related to $X$ does not decline. If operations are transferred, other sites shoulder additional production and risk burdens resulting in $E[x'_{\sim i}] > E[x_{\sim i}]$ for post-disaster dispersal $x'_{\sim i}$. Alternatively, operations continue at site $i$, and in addition to realisation $x_i$, an expectation of further exposure remains. Then $x_i + E[x_i]$ is the relevant decision datum at site $i$. Figure 3 illustrates continued operation using the biological case fitting with a nuclear accident or exceptionally toxic, nondegradable chemical spill. Let the region fall within the dispersion doctrine range of the biological case at the initial scale of production. After exposure $x_i$,then $D_i(x_i) = D_i(x_i + E[x_i]) = D_i(x_i + E[X]) = 100$ and the region transitions to the clustering doctrine minimising total damages. Using site $i$ as an accumulator for the risky activity preserves all other sites in the region. One finds an example of this in the use of the Chernobyl Exclusion Zone for the storage of low and medium risk waste from other Ukrainian nuclear power plants (Laraia, 2019). We might more generally consider any nuclear production or waste site that is then expanded for waste storage.

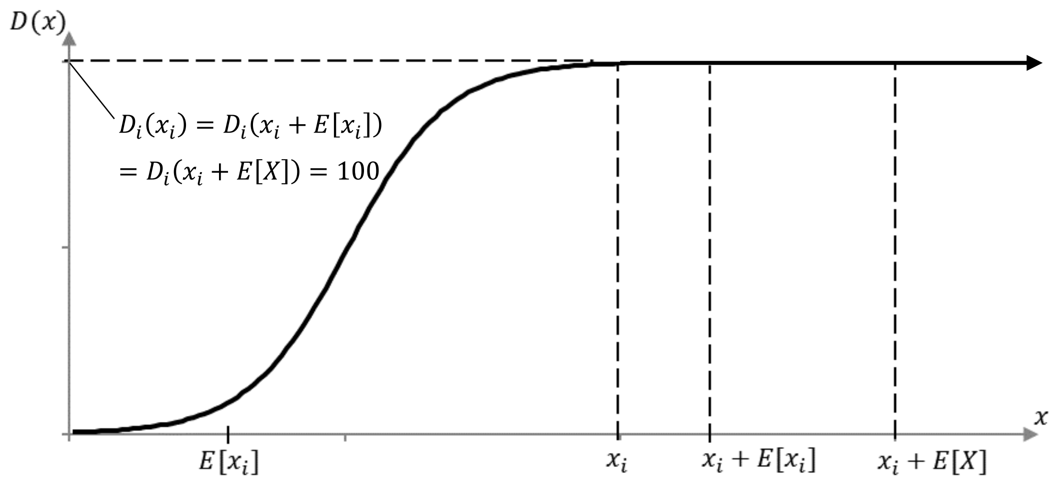


FIGURE 3 EXPECTATION OF ADDITIONAL EXPOSURE

Notes: Additional activity at an already damaged site might fall in the range of decreasing or zero marginal damages. An activity that originally warrants dispersal at the scale of production and risk in the region, may now warrant clustering at the damaged site.

Having set the bounds of the sort of damage functional forms used in this paper's analysis, I now proceed with a discussion of policy development. I also discuss issues related to whether a policy that is otherwise designed to achieve optimality will actually lead to improvements for society and the environment.

## 1.4 Local-Scale Environmental Policy Development

One application of this paper's framework is that there is a necessary pollutant that we cannot abate through technology or curtailment and instead decide to mitigate damages through industrial organisation. Other situations place a policymaker in similar circumstances. Policy development generally begins after some regulator asserts authority over the production quantity of pollution. Earlier supra-local environmental problems, for instance, sulphur dioxide emissions from power plants, have resulted in state and national regulatory scales. This geographic and political separation from the local scale constrains local governments to find options, given the level of pollution exogenously allowed. One option in the spirt of non-cooperative environmental policy is to use zoning, which often remains a local concern save for national infrastructure. Zoning is increasingly used to work around the issue of federal- and state-level environmental regulators not sufficiently addressing local-scale problems (Nolon, 2002). But local zoning can also be used poorly – to set exceptionally strict local environmental regulations because the local community and regulator only bears part of the cost when they set a higher than efficient standard. This results in states tending to overregulate versus federal standards or cap and trade systems (Williams, 2012). To be discussed, this may exacerbate underachievement of environmental goals in the aggregate due to leakage of polluting production beyond the locally regulated bounds.

So, the discussion of local-scale environmental regulation proceeds by noting a substantial constraint on policy success that I think is underappreciated. The role of the agenda – which components of a comprehensive environmental policy are decided first – puts the feasibility of the results to follow in context. Reaching an optimal solution by jointly regulating across local environments then follows as well as a section highlighting some other issues to keep in mind during policy development.

### *1.4.1 The Role of the Agenda in the MAC-MD Regulatory Framework*

The most troubling aspect of regulating $N$-environments is that control of the agenda – whether $N$ or $x_i$'s are chosen first (with $X = \sum_{i=1}^{N} x_i$ in mind) impacts total emissions and damages. It is particularly troubling because the default – choosing $x_i$ separately, or de facto separately, has the potential to result in larger *N* and more pollution.

Consider *N* emitters or groups across *N* sites. Discussing the optimal *N* under different damage assumptions has been the fundamental problem up to this point where $X$ is exogenous and then $N$ is endogenously determined to minimise total damages. That approach assumes that we have a necessity $X$, and then choose optimally $N = N^*$ from some workable interval, $N_a \leq N \leq N_d$ where $N_a$ is a regional clustering result – perhaps to one site, and $N_d$ a large dispersion.

Now, in the MAC-MD framework say we choose *N* first then each $x^*$ optimally. The symmetric sites simplification aids discussion, else choose *N* and then *X* by choosing $x_i$ for all *i* in *1:N*. The optimal, region-wide *X* for an interior solution results from $N\left[MAC\left(\frac{X}{N}\right)\right] = N\left[MD\left(\frac{X}{N}\right)\right]$ and $x^* = \frac{X}{N} = \bar{x}$; equivalently solving $MAC(\bar{x}) = MD(\bar{x})$, or more generally $MAC(x_i) = MD(x_i)$ for all $i$ in *1:N*. This separate, local sites interior problem contrasts with the pooled public good equivalent of Samuelson (1954): $MAC(\bar{x}) = N[MD(\bar{x})]$.[22] The separate-optimal result is good news in that it matches the traditional framework of environmental economics – the optimal emissions in each environment is decided independently of other environments if the regulator must solve the local sites problem separately. For example, the optimal abatement of agricultural runoff in Iowa is independent of that in Idaho.[23] But what if the level of $X$ is a necessity and cannot be met by the separate optimal condition at *N* sites – that $Nx^*$ is greater or lesser than a necessity level of pollution *X*?

Solving the interior solution jointly given necessity $X$ – and assuming all units of polluting activity can relocate in the long run – requires a different framework. First, suppose the number of sites as well as the total quantity of emissions are fixed, the objective and relevant constraint are $\min_{\{x_i,\ldots,x_N\}} \sum_{i=1}^{N}(AC(x_i) + D(x_i))$ subject to $X = \sum_{i=1}^{N} x_i$ which in the symmetric case results in optimality conditions $MAC(x_i) + MD(x_i) = MAC(x_j) + MD(x_j)$ for all $i, j$ in *1:N* (Note 2). This is equivalent to the separate-optimal framework when firms and sites are symmetric and the scale of $X$ is exactly such that $MAC(\bar{x}) = MD(\bar{x})$ can be satisfied at each site. These conditions seem unlikely to be met in many

[22] This is specifically the symmetric polluter solution which is sufficient for this comparison. The purpose of this research is not to address the Samuelson problem which has a solution of setting the sum of individual marginal benefits equal to the marginal cost of providing a public good. In this case, marginal benefits are in terms of damage reductions and marginal cost in terms of abatement costs.

[23] To be clear, they do not share a watershed. They also have different growing conditions and so for many crops are not substitutable and may serve different markets.

cases. When the joint-optimal objective allows selecting all $x_i$'s and the number of sites in $N_a \leq N \leq N_d$, then $MAC(x_i) = MD(x_i)$ can be satisfied only if $X = \sum_{i=1}^{N} x_i{}^*$ can be satisfied in the bounds of $N_a \leq N \leq N_d$. So, in either case – when optimally selecting $x_i$'s, or both $x_i$'s and *N*, it is entirely possible that the joint-optimal interior solution is different from the separate-optimal solution and is a second-best (Lipsey and Lancaster, 1956) outcome.[24]

Note 2 – Solution to the Joint Regulation of Local Sites

The symmetric as well as asymmetric case optimums can conceptually be solved by the Lagrangian method. Suppose for two sites $i, j$ there is an interior optimal solution. Then, $MAC(x_i) + MD(x_i) - \lambda x_i = 0$, $MAC(x_j) + MD(x_j) - \lambda x_j = 0$, and $\sum_{i,j} x_i - X = 0$ with shadow price, $\lambda$. Let $T_i = MAC(x_i) + MD(x_i)$ and similarly for $T_j$, then $\frac{T_i}{x_i} = \frac{T_j}{x_j}$ subject to $\sum_{i,j} x_i = X$. With symmetry, $T_i = T_j$, which is solved by $x_i{}^* = x_j{}^* = \frac{1}{N} X$ for $N$ sites, here $\frac{X}{2}$. Without site symmetry, $x_i{}^* = \left(\frac{T_i}{T_i + T_j}\right) X$ and $x_j{}^* = \left(\frac{T_j}{T_i + T_j}\right) X$, so site shares are based on their contribution to the total environmental cost across all sites.

Another issue arises when the regulator decides $x_i$'s but does not fix either *N* or *X*. Assume $x_i$'s are first set optimally at the site level. Under some conditions for the polluter such as a superadditive MAC function, $(N+1)MAC(x^*) < N * MAC\left(\frac{N+1}{N} x^*\right)$ can occur if we restrict firms to $x^*$ locally.[25] That is, an incentive can exist for polluters to expand to other locales rather than invest in abatement, and we observe leakage by expansion. Total emissions increase and possibly total damages depending on the functional form. The implication is that whoever owns the property right to the agenda – whether $x_i$'s or *N* are decided first, controls the total level of pollution, damages, and abatement. There is then a perhaps substantial incentive for industry to prefer the status quo form of regulation – choosing $x_i$'s and equivalent methods without regard to *N*, given current regulatory methods.

Unfortunately, choosing $x_i$'s first also lays the groundwork for the result that it is optimal to cluster all activities to one site and pollute without limit. The limiting result of the incentive to disperse to avoid regulation is that $N = N_d$, followed by polluters operating

[24] That is, when we cannot separate-optimal choose $x_i = x_i{}^*$ for all $i$ in *1:N*, we can still joint-optimal solve $MAC(x_i) + MD(x_i) = MAC(x_j) + MD(x_j)$ for all $i, j$ in *1:N*.

[25] At least if $\frac{N+1}{N} x^* \leq X$.

subject to regulation limiting production to $x = min\{x_0, x^*\}$ for unregulated level $x_0$. Since $N_d min\{x_0, x^*\} \geq Nx^*$, else the regulation was not binding to begin with, the region may be closer to the clustered solution being more efficient. I next discuss the difference between separate- and joint-optimal regulation when the agenda – and the general regulatory environment underpinning it – is appropriate.

### *1.4.2 Achieving Optimality*

As in the preceding sections, suppose there are *N* potential sites for some polluting activity and polluters of an inconsequential number group themselves at these sites such that we can also refer to them as *N* polluters. Let the marginal abatement cost function at site *i*, $MAC_i(x_i)$, be well behaved, downward sloping, and defined for all values of pollution, $x_i$, and alternatively abatement. In this initial discussion, let the marginal damages function, $MD_i(x_i)$, first increase in a well-behaved manner. However, as this paper models a local-scale environment, assume beyond some level of damage that pollution destroys the local environment in the sense that all environmental value is lost. That is, $MD_i(x_i) = 0$ for all values of $x_i$ beyond the point of destruction.

*The separate-optimal solution*: Suppose we regulate each site separate-optimally. A conventional approach is to set a Pigouvian tax, $\tau_i^P$, or an equivalent quantity of emissions limit, $x_i^P$ at each site, *i*, in *1:N*. Assume $MAC_i(x_i) \approx MAC(x_i)$ and $MD_i(x_i) \approx MD(x_i)$. That is, each site is sufficiently comparable. Then, Figure 4 where $MAC(x_i) = MD(x_i)$, $x_i \leq 100$, and Figure 5, left side, represent the interior optimal solution for a case for dispersion marginal damage form and a typical marginal abatement cost curve which is also assumed symmetric for all sites (the general result holds for each functional form). The result holds whether we regulate optimally by $x_i^P$, $\tau_i^P$, or measures of damages at the optimum, $MD(x_i^P)$ or $D(x_i^P)$.[26] The separate-optimal solution follows the classic framework introduced to many by Baumol and Oates (1988) to solve Samuelson’s (1954) public goods problem.

*The joint-optimal solution*: The local-scale, total damages possibility results in an alternative, joint-optimal solution. Sometimes this remains an interior solution at all sites equal to separate-optimality. But sometimes the sum of all damages and abatement costs is

[26] Muller and Mendelsohn (2009) advocate for regulation based on $MD(x_i^*)$ by, say, weighting trades in a cap-and-trade system based on the damages caused by each source rather than emissions. The solution to follow might be considered a hybrid $MD(x_i^*)$ and $D(x_i^*)$ approach – suggesting site-level regulation by $MD(x_i^*)$ (equivalent to regulation by $x^*$ or $\tau^*$ in a contained local environment) but regulating by total damages (actually by the mean) in the clustered alternative to follow.

less if all production occurs at one site – a corner joint-optimal solution. Suppose rather than the interior outcome occurring $N$ times – at $N$ sites, we ask that all polluters move to one location, site $a$ selected from the set *1:N*. Site $a$'s aggregation is represented by line $N * MAC(x_i)$ in Figure 4 as well as the right panel of Figure 5. Quantity of emissions at $a$ are $X = \sum_{i=1}^{N} x_i = N\bar{x}$. Critically, beyond some point where $a$'s environmental value is lost, any additional unit of emissions results in zero additional damages. Meanwhile, the *N-1* sites not used receive zero damages.

So, let's say the policymaker weighs the total cost of dispersed abatement and damages across symmetric sites and firms as $N[AC(x_i) + D(x_i)]$, against a clustered alternative of $N[AC(x_i)] + D(Nx_i)$. These are the sum of the private costs of abatement – their horizontal summation – plus the damages to either $N$ sites or a single site ($N_a = 1$ for simplicity). With small increases in $N$ (where each site is polluted at the optimal level), the interior solution remains optimal for all sites. However, with sufficiently large $N$, the optimal outcome is achieved by shutting down *N-1* sites and allowing polluters to operate unconstrained at one site where $X$ is emitted.

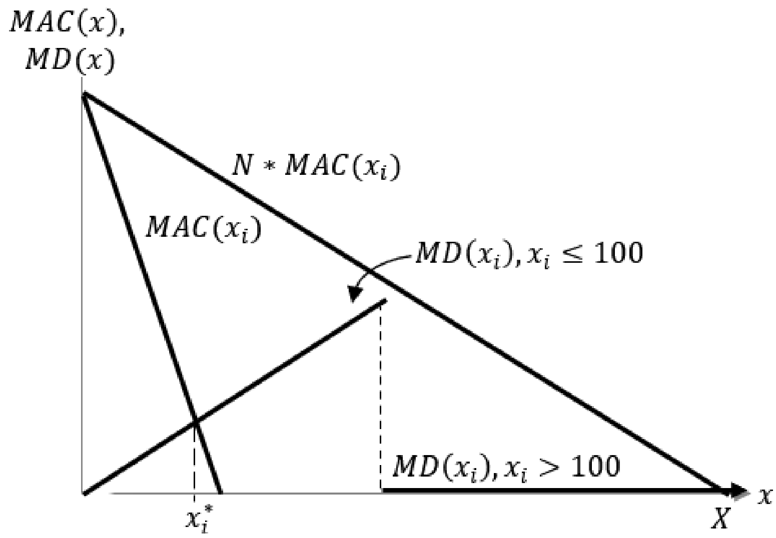


FIGURE 4 OPTIMAL SOLUTIONS IN THE CASE FOR DISPERSION

Notes: With symmetric firms and sites, selecting simultaneously $N$ and site emission levels $x_i$ optimally results in emissions $x_i^*$ at each site and $Nx_i^*$ total emissions. Once $Nx_i^* \geq 100$ for any single site and thus $D(Nx_i^*) = 100$ at that site, any additional clustering is unaccompanied by additional damages. With $N$ sufficiently large, a single-site alternative with unabated emissions reduces total damages relative to emitting separately at $N$ sites.

A mechanism to operationalise separate- versus joint-optimal regulation is presented next. The regulatory framework incentivises polluters to voluntarily choose the most efficient of the preceding industry geographic structures.

*A separate- and joint-optimal compatible incentive structure*: An issue is that, when using standard regulatory instruments of setting level $x_i^P$ or Pigouvian tax $\tau_i^P$ at several sites, clustering to one site is never desirable for polluters. However, setting the per-unit emissions penalty, or price, equal to average aggregate damages at a site $a$, $AAD(X) = \frac{TD(X)}{X}$, corrects the incentive when total emissions are sufficient. For any additional unit of emissions where the resulting marginal damages are zero, the price producers pay for emissions, $AAD(X)$, decreases. When $X$ is sufficiently large, paying $AAD(X)$ is cheaper versus paying separate-optimal emission penalties, taxes, or other abatement costs.[27] As $AAD(X)$ declines with increasing emissions, each polluter's joint-optimal level of emissions are also higher, at $x_a^A > x_i^P$, and presumably, more welfare increasing units of consumer products and services result. We need not be precise about offering one sort of regulation or the other either. Given a choice between $x_i^P$ and $x_a^A$, or $\tau_i^P$ and $\tau_a^A$ as in Figure 5, polluters can determine amongst themselves the preferable organisation, at least in an organised and informed industry.

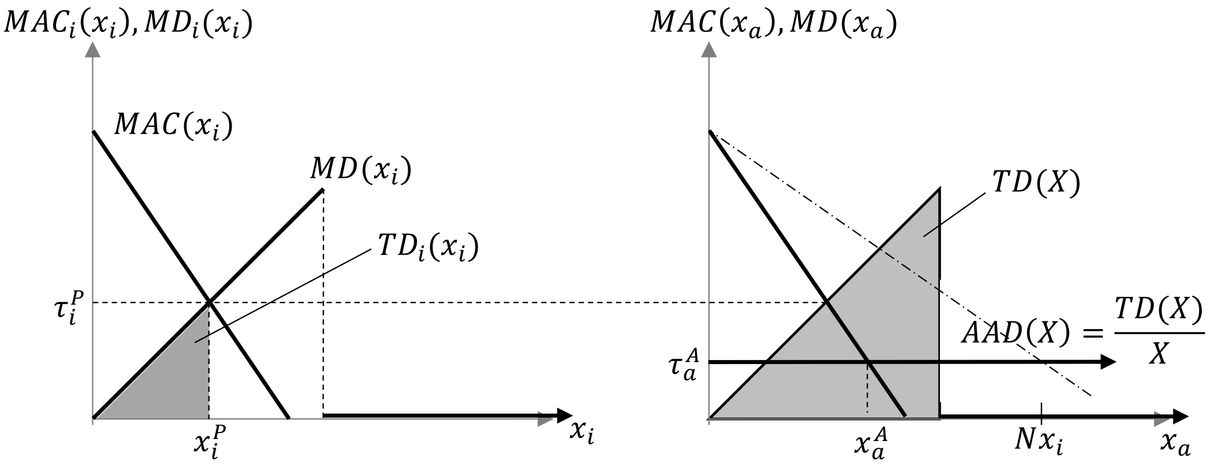


FIGURE 5 THE POLLUTER'S DECISION WITH A CLUSTERING INCENTIVE

*Notes*: The left pane identifies the individually regulated, interior optimal site result, $x_i^P$ emissions, from minimising the total cost of damages plus abatement. The optimal Pigouvian tax is then $\tau_i^P$. In the right pane, when jointly regulating sites, a different optimal decision may result. If total exposure from congregating all polluters to one location eliminates all environmental value at the site, any additional exposure results in zero marginal damages. Setting a tax, $\tau_a^A$, equal to the average damages in aggregation encourages firms to move all production to the site when it is beneficial to do so. All other sites taken out of production experience zero exposure and damages and total damages across all sites decreases. Setting the tax or standard by average damages rather than emissions achieves the result, while each firm *i* increases emissions to $x_a^A$ and any linked production.

[27] As previously noted, differences in transportation, crowding, and other operating costs – frictions – must clearly be considered in practice. However, for an issue such as nuclear waste storage, there is ideally a single transportation event followed by several millennia of inactivity.

We might refer to this policy approach as an AAD-Pigouvian choice framework or other connotation which indicates that choice remains between Pigouvian-based regulation as several sites versus average damages-based regulation at an aggregator site. To summarise the outcome, when jointly regulating local-scale environmental problems of the form outlined in this paper (e.g. with trivial leakage), the optimal tax policy may diverge from the Pigouvian recommendation. Offering firms a choice between a tax set by marginal principles versus a tax set according to an equal share of aggregate damages results in firms choosing as in Equation (2). Such a tax offers an incentive for firms to move toward the joint-optimal result. Importantly, the policy prescription is robust to the damage functional forms highlighted.

(2) $$\tau^* = min\{\tau_i^P, \tau_a^A\}$$

One cannot propose an alternative policy route, however small in difference from existing frameworks, without considering the secondary effects it might have. In the next section I briefly note some issues that may arise. They warrant particularly careful consideration when an AAD-Pigouvian choice framework is under consideration.

### *1.4.3 Some Reasons for Caution*

Applying this paper's analytical framework warrants caution so I address a few potential issues here. For one, different damage relations may exist within an ecosystem as in Mendelsohn and Schlesinger (1999) and Win et al. (2018), and almost certainly differ between sites. Even a difference in population affects how damages are disbursed. Differences in environmental value are evident in our behaviour as well – we are far more likely to tolerate crowds to visit a renowned site such as Yellowstone National Park and still benefit from it, than visit several available empty fields. We cannot then uninformedly choose between sites. Yet decades, or even centuries of intensive production have provided many devastated sites to choose from anyway, so we need not choose Yellowstone nor some farmer's field for many destructive uses.

The siting of polluting activities also has a considerable environmental justice component as there is a long history of siting polluting activities in or near poor communities. See a general overview in Mitchell (2011), discussion of social justice and zoning in Wilson, Hutson, and Mujahid (2008), examples of inequalities by different pollution types (waste, air, risk such as flooding, and landscape quality) in Walker (2012), and cases of struggles against environmental injustice in Adamson, Evans, and Stein (2002). But to ignore this framework in favour of opposing any polluting activity across all possible sites ignores the value that siting implies. The value of siting – of damage savings elsewhere with clustering or dispersion – informs whether compensation beyond repayment for damages at a use-site is warranted. These valuations might provide substantial benefit to impacted communities and knowing of their existence results in bargaining leverage which contributes to environmental justice.

I have also mostly ignored the well-developed literature on incentives in a repeated game framework and the role of reputations (for instance in Mailath and Samuelson (2006) or several other texts). The justification is simple: If a nation's environmental regulatory authorities at any scale are unable to prevent unabashed dumping, then they are likely unable to regulate the organisation of industry too. The most significant concern is that without any empowered regulatory authority, there is an incentive to pre-emptively damage a site entirely such that zero abatement remains optimal if regulation should later come into effect. Allowing destroyed sites at the time of regulatory enactment to become pollution havens incentivises pre-emptive behaviours that few advocating for intervention would prefer.

Finally, this analysis relies on the idea that the local pollutant in question does not leak – that we can maintain whatever polluting activity's emissions within the locality. When we store nuclear waste or even household waste in a landfill, containment is the objective in mind. But in applying this framework, we must consider whether containment is the case, as the intent is not to support unabashed dumping that would permeate the natural environment. The intention of this framework is quite the opposite – to carefully choose sites and contain dangerous and perhaps resilient pollutants well, even forever as in the case of fissile waste.

Having noted a few concerns – certainly not an exhaustive list of cautions to keep in mind with any potentially corrective policy – some numerical and empirical support follows. These are far from exhaustive, and the applicability of this paper's modest adjustment to the damages functional form need not be constricted to environmental regulation.

### 1.5 Empirical and Numerical Support

In this section I outline four cases of local form damage relations in action – two numerical and two empirical. I reserve a few more diverse applications for appendix *A1.2 Siting Framework Applied Beyond the Environment*. There are certainly many cases where industrial reorganisation results in reduced total damages but detecting and isolating causal relationships can be difficult. Not in My Back Yard (NIMBY)-driven policies also happen to occasionally be correct, but otherwise cloud the search for evidence.[28] There have also been “hot-spot”, or “black-spot” policies where the objective is not containment, but to allow an industry’s emissions to leak beyond the regulator’s domain and the public’s capacity to care (Siebert, 2008) – these clearly violate the requirements of this paper’s model. When this paper’s framework is appropriate, however, it also appears easier to identify a clustering change because it is easier to identify the appearance of a significant source of pollution. Even without a change in regulation, consumers might also prefer a specific polluting activity be clustered or dispersed such that public pressure without a discrete change in identifiable policy is sufficient. In other cases, such as the first agricultural example, historical usage inhibits change.

Note that I test for policy responsiveness rather than measuring the scale of the benefits that result from reorganisation. In each case, the objective is to suggest that “polluters” generally respond to reorganisation incentives. The matter of particular damages as well as abatement cost values beyond those in the hypothetical numerical examples are a matter for when policy using this framework is deliberately applied by a regulator. The assessment of damages avoided and resulting welfare benefits may also be incalculable in many cases. Rather, I suppose the changes were implemented because policymakers or the public minimally expected damages to be lower, revealing the existence of value but not quantifying the scale tied to clustering or dispersion.

### *1.5.1 Numerical Application 1: Agricultural Runoff, Drainage, and Watersheds*

A numeric, albeit artificial, application illustrates the preceding arguments. Many sorts of pollution resulting from agricultural production might fit a case where aggregation or dispersion reduces total damages to the environment. But agriculture generally requires expansive space and does not aggregate well by many measures. Yet, agricultural runoff – a significant environmental problem – can be concentrated or dispersed in how it enters waterways by changing drainage systems, and an application emerges. For a history of agricultural drainage practices in the United States see USDA ERS (1987). The sort of guidance given to reduce runoff damages includes some farm-level changes, but also a lot

[28] NIMBY responses are generally in reply to Locally Unwanted Land Uses (LULU)’s – local-scale environmental problems. A general critique of NIMBY campaigns, their frequent role in perpetuating environment injustice, and when NIMBY responses are instead justified can be found in Feldman and Turner (2010) and Feldman and Turner (2014). It is also worth considering whether NIMBY responses are a global movement as explored in Martinez-Alier, Temper, Bene, and Scheidel (2016).

of public works sort of corrections as in guidance from the USDA NRCS (2009) or more general international guidance in FAO (2017). One of the most visible forms of damages results from excess nitrogen-driven algae blooms in waterways (Biello, 2014). The total value of damages from agricultural runoff are substantial. In the U.S., these have been estimated at $157 billion/year (Sobota, Compton, McCrackin, and Singh, 2015), and at a similar scale in Europe (van Grinsven et al., 2013). The per-hectare damages inflicted often exceed the value of agricultural land uses at current production levels and prices.

I present this application “in theory” then because a complication arises “in practice”. Agriculture is generally one of the first uses of land and tends to almost uniquely own a wide range of property rights by use, if not law. In the U.S., these rights have been formalised more recently to protect agricultural producer historical rights as they and residential users come into increasing conflict. These conflicts have lead to "Right-to-Farm Statutes" in all 50 states protecting agriculture from having to redress damages (see a catalogue of these statutes at the National Agricultural Law Center, 2019). There are case studies, however, of changing property rights as secularization of the countryside occurs. As an example, Munton (1995) documents a transition from employment to an environmental focus around a coal extraction-based community (but also local resistance). But overall, until recently little precedent existed for changing these rights. Agricultural producers almost undeniably owned a right to route drainage into waterways. When the public has objected to damages from agricultural runoff, it is rarely if ever a question of whether it should occur. Instead, it is sure to be framed as an argument about farmers using “too much” of some input that then washes away – pushing their property right too far – and lending implicit support to farmer’s rights. So instead of arguing whether we can limit agricultural runoff through some quantity restriction, let us consider whether a change to where that pollution enters waterways can improve welfare.

Consider a hypothetical farming community of an initially $i_0$ set of farms in a watershed where at first the total discharge of a persistent, polluting agricultural runoff into any segment $j$ of the watershed amounts to *at most* $X_j = \sum x_j = \sum x_{i_0,j} \leq 100$. Also, let any runoff result in scaled, constant marginal damages $MD_j\left(x_j \middle| X_j \leq 100\right) = 1$ and $MD_j\left(x_j \middle| X_j > 100\right) = 0$. Suppose each farm emits 25 units of the persistent pollutant. Figure 6 presents an initially four-farm case, with farms $i_0 = a, b, c, d$. Producers route drainage ditches (black arrows) to the closest tributary composed of segments (blue arrows) in the left panel, versus arbitrarily choosing to route all drainage to the right tributary in the right panel. I present the resulting damage values in black text. Regardless of organisation, the same total damages to the watershed result at the initial scale of production. Then, consider the development of another plot in the watershed, $e$, also emitting 25 units such that set $i_1 = i_0, e$ and $X_j = \sum x_{i_1} > 100$ is possible if all $i_1$ farm discharges flow through the same segment. I present the new segment damage values in red text for each routing assumption. Now, the aggregate routing in the right panel is preferable. The rightmost tributary becomes 100-per cent damaged in its final segment – in effect becomes a drainage canal. However, the left tributary experiences no damages in any segment – the tributary returns to nature. The framework can be applied to other industries disposing of waste in waterways.

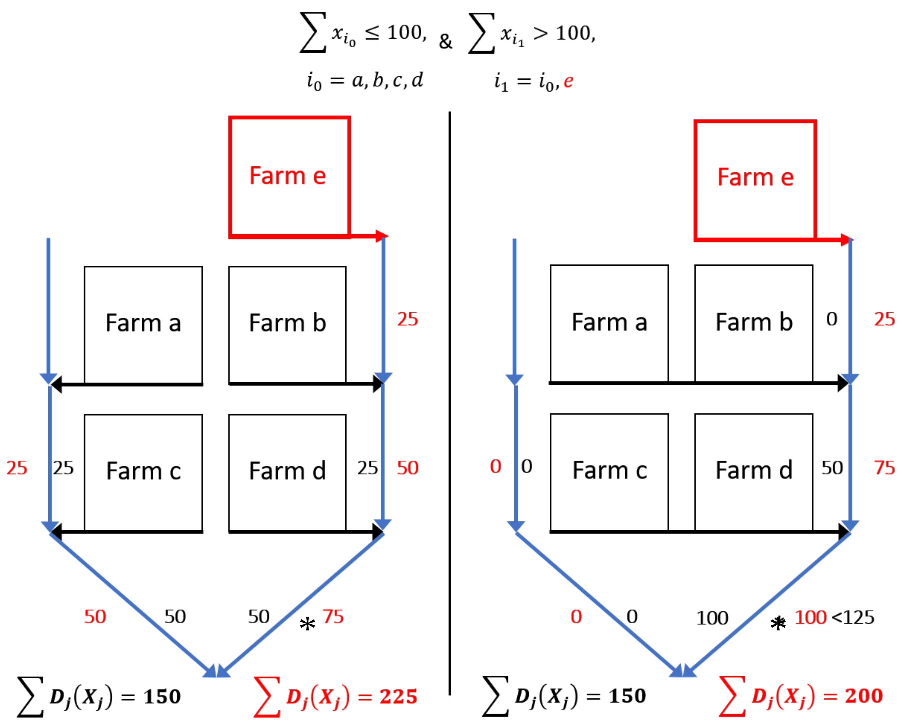


FIGURE 6 OPTIMAL AGRICULTURAL DRAINAGE WITH DIFFERENT SCALES OF ACTIVITY

Notes: Assume the pollutant in agricultural runoff results in constant marginal damages, does not degrade while in the watershed (is persistent), and the environmental value of any drainage canal (black arrows) is zero. Assume each farm emits 25 units of pollutant as runoff and each unit is normalised such that it causes one unit of damage (=1/100th of total damage potential) to any tributary segment (blue arrows) that it passes through. The total damage in both scenarios is 150 (black numbers) without the inclusion of Farm e. With the inclusion of e, the left arrangement would incur 225 total units of damage, but the right arrangement would only incur 200 units of damage (red numbers) because the segment * receives $D_*(\sum x_j = 125) = 100$ units of damage when the total environmental value of this segment is lost. The least damages drainage plan is then to route the system as in the right-side clustered arrangement.

The application need not apply to entire landscapes nor to situations where full damages occur. I provide a second numerical example to illustrate.

### *1.5.2 Numerical Application 2: Several Small "Polluters"*

Assume a local damage function in discrete units based on a change in the number of individuals engaging in an activity that generates a negative externality locally. For tractability, let the discrete marginal damage function at a site *i* be a function of *x* number of individuals engaging in the activity as

$$MD_i(x_i) = \begin{cases} 2, & x = 1 \\ 1, & x > 1 \end{cases} \tag{3}$$

where marginal damage values of 1 and 2 are for the purpose of illustration.

As an example, the sites may be neighbourhoods within a city as the region. Damages may be in percentage decreases in property value or increases in waste or incidental crime when people engage in some activity. Consider a city where five individuals or firms engage in the activity at separate sites, then total damages to the city are ten. If instead, the city council implements a policy that draws the individuals to a single site, total damages are six. An implication is that a site or neighbourhood harmed by the policy might be entitled to more compensation than just for damages from the negative externality – also a payment for providing a public service by accepting the damages. The damage savings are more than sufficient to compensate the owners or other users of the aggregation site – a potential Pareto improvement illustrated in Figure 7.

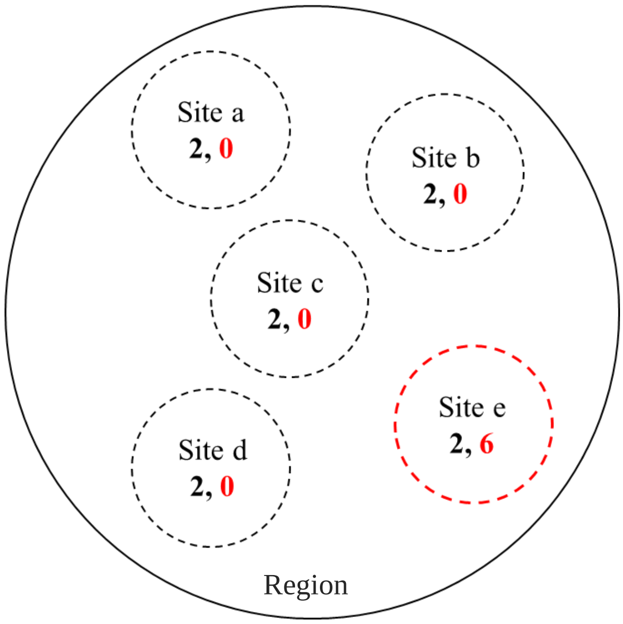


FIGURE 7 REGULATION OF SOCIAL ACTIVITIES WITH NEGATIVE EXTERNALITIES

*Notes*: Assume a discrete damage function where the first individual to engage in the activity at any site $i = a, b, c, d, e$ causes two units of damage and any additional participant inflicts one additional unit of damage at $i$. If five individuals engage in the activity at five separate sites, the total damages in the city (region) is ten units (the damage to each site is reported in black digits). If instead the activity occurs at one site, say $e$, total damages are six units (damages now reported in red digits). The damage savings at $i = a, b, c, d$ can more than compensate the owner of site $e$.

### *1.5.3 Empirical Application 1: Household Waste*

The story of household waste is of changing consumption patterns and a move centralized collection over the last century. Household waste itself is a certainty, but centralised collection away from the home is an almost entirely new phenomenon. Nearly all centralised waste management began within the last seventy-five years and was almost unheard of more than a century ago. Early exceptions occur not from discontent with the level of accumulating matter, but because some waste retained market value. A particularly vivid and well-known example is of bone-pickers and rag-gatherers in 19$^{th}$ century London as documented by Mayhew (1861). Before centralised collection, household waste was almost certainly biodegradable – food and cooking waste and the occasional wood, glass, and pottery items beyond repair. From an intensive use of the author's compost bin, I find a family's annual accumulation of these sorts of refuse amounts to nothing that would still be considered waste in the end. In comparison, the amount of nonbiodegradable, or at least non-compostable waste a family now generates in a year is enormous. For instance, the waste generated by all economic, production, and household activities in the EU-28 in 2016 amounted to an average of 4,968 kilograms per person (Eurostat, 2020).

To add rigour to the story, I use a regression kink (RK) design akin to Thistlethwaite and Campbell's (1960) regression discontinuity (RD) approach. Nielsen, Sørensen, and Taber (2010) discuss the RK identification strategy, Card, Lee, Pei, and Weber (2012) supply a theoretical basis, and Böckerman, Kanninen, and Suoniemi (2017) provide an example application. However, estimation and interpretation are intuitive when one is familiar with the RD approach: rather than a "jump", we find a change in rate of change or slope. Identification comes from the adoption rate of something changing beyond the point of implementation of the policy of interest. In this application, I use the RK design to test whether the rate that waste removal services came into existence changes with a critical change in the nature of household waste generation.

One proposition is that following the invention and widespread adoption of nonbiodegradable plastics, households using composting and household dumps encountered an intractable new problem. Rather than garbage exposure and a damage function that returns to around zero as waste decomposes, with each plastic item disposed of, the homeowner finds the quantity of garbage on hand increases. As still occurs in some developing areas, burning is an option. But doing so produces toxic smoke with uncomfortable and immediate effect on health and the relationship with one's neighbours. Unlike any preceding household waste, plastic is not a problem that the homeowner can solve acceptably on site.

To test the plastic hypothesis, I compare growth in plastic production to growth in refuse collection services across Europe. Geyer, Jambeck, and Law (2017) provide annual estimates of all plastic production globally since 1950, and I supplement with observations from Freinkel (2011) for 1939 and 1945 which is reproduced in appendix *A1.3 Plastic Production Volume, 1939-2016*.[29] Following World War II, year-over-year plastic production increases and global production rapidly becomes measured in millions of metric tons instead of thousands of kilograms. New plastic types, quality, and production capacities developed during the war drive this growth (Freinkel, 2011). However, it is almost certainly also driven by the post-war desire for more production and consumption of every sort.

For comparison data, the Orbis database on business activity provides dates of incorporation for every company in Europe including under the industry code for refuse

[29] As a matter of general interest, of all the plastic ever produced it is estimated that 9-per cent has since been recycled, 12-per cent incinerated, and 79-per cent still exists in some form.

systems, SIC code 4953 (Bureau van Dijk, 2019a, 2019b).[30] The data provides an estimate of the annual rate of change in the number of waste management firms. This may overstate the rate of change, however, as there is no data on when firms drop back out of the market. Yet the waste management sector is profitable and has instead undergone growth and consolidation. We may also note the near-universal prevalence of waste management services in Europe today – from covering nearly zero-per cent of households to around 99-per cent of EU households in 75 years. Unlike post-World War I when plastics did not become a substantial share of household consumption, following World War II the number of new waste management firms rapidly increases. The "kink" in the rates of new waste management firms and plastic production volume are illustrated in Figure 8, and I reserve the formal econometric treatment for section *1.5.5 Regression Discontinuity and Kink Methodologies and Estimates* alongside the formal estimates from the next application. A more extreme surge in waste management firm growth also occurs in the 1970's to meet the rising demand for recycling. However, the interpretation of SIC 4953 data loses the interpretation of interest in the 1970's as it becomes dominated by smaller recyclers – handlers of the same waste in a new, yet older way (Strasser, 1999).

As one metric for the plastic production-waste trends, the correlation coefficient is near unity. However, we must expect the relationship between these activities to be more complicated because, at a minimum, producers must somehow transfer plastic products to the consumers that later throw them away. Figure 8 also illustrates the change in all formal sector business activity.[31] In terms of estimating a causal effect, a problem arises because all business activity follows a similar trend – little growth preceding 1945, and dramatic growth thereafter.

[30] Appendix *A1.4 Description of SIC Code 4953: Refuse Systems* includes a formal definition of what the SIC code includes as well as annual summary data.

[31] Waste management data includes all SIC code 4953 enterprises, and all "other" firm activity is a random sample (~13,000 observations with dates of incorporation) drawn from non-4953 coded enterprises in Orbis. Each data series is summed by the year in the date of incorporation for the countries that would eventually comprise the 28 member European Union at the date of data access.

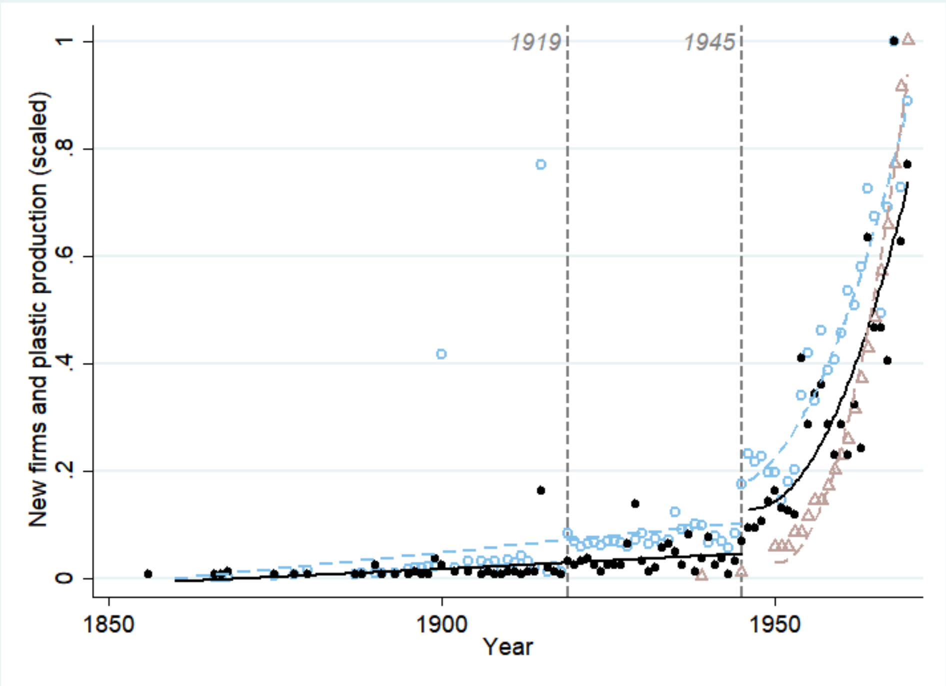


FIGURE 8 NEW WASTE AND OTHER FIRMS IN EUROPE AND GLOBAL PLASTIC PRODUCTION

*Notes*: Black with solid circle markers: Number of new waste management firms annually (refuse collection, disposal, and recyclers) by year of incorporation which is scaled by the maximum value in the period of 161 new firms in 1968.
Blue with hollow circle markers: Trend in new, non-waste management firm activity by year of incorporation – a random sample of several thousand observations representative of activity in the period, then scaled.
Red with triangle markers: Global plastic production in millions of metric tons (MMT) scaled by the maximum value in the period of 35 MMT in 1970. While data availability continues after 1970, the business code representing waste management firms changes in nature as it then includes several thousand small recyclers. Unfortunately, the impact of plastic production versus all other production cannot be separated. The two sources are collinear as general firm activity is the mechanism for the transfer of plastics to consumers.

*Source:* Author calculations based on data from Orbis (Bureau van Dijk, 2019a, 2019b), Geyer, Jambeck, and Law (2017) for plastic production data 1950-2015, and Freinkel (2011) for 1939 and 1945 estimates converted to Mmt from millions of pounds.

So, we are left to wonder: Did waste collection become nearly universal in Europe because of non-degradable waste, because all sorts of consumption and waste increased, or because of an entirely separate change in lifestyle? Collinearity prevents separation of the impact of plastic production and general firm activity. One might construct an international panel, but both the manufacture of plastic products and waste disposal involves complex international supply chains. The result is a causal change beginning in 1945 in the sense that garbage must follow consumption and disposal. However, the exact cause of the change in garbage – plastic, lifestyle, or an overwhelming volume of all waste, cannot be isolated. The waste story is as compelling as econometrically intractable. Unfortunately, it is not the only application. I next explore U.S. nuclear power generation where a single exposure event changed the entire industry's trajectory.

### *1.5.4 Empirical Application 2: Nuclear Power*

The evolution of views toward nuclear power production is a story of public discovery followed by backlash. The nuclear power sector in every so-endowed nation operates under a cloak of secrecy inherited from their origins in nuclear weapons programmes (Pearce, 2019). The transference of secrecy should not be surprising as both fields draw from the same pool of technical talent. The nuclear power story presented here is one of public discovery – of experts assuring the public that nuclear energy is marvellous and safe, followed by public outcry when something goes wrong. In the United States, the event that changed the public's perception was the Three-Mile Island incident (Pearce, 2019; Walker, 2004). In terms of radioactive release, it was minor. But in terms of media coverage, public backlash, and demand for greater oversight, the incident was substantial. It halted the expansion of nuclear power production in the United States.

The argument: Nuclear power producers enjoy a secrecy rent from lack of public awareness about the risks involved, or they are overconfident about their ability to control and contain a nuclear incident. The sector should operate aggregated to as few sites as possible because the risk of damages from an incident are enormous. For example, the 2011 Fukushima Daiichi nuclear disaster clean-up cost is estimated to be between $75.7 billion (government estimate in 2006) and $660 billion (the Japan Center for Economic Research) (Hornyak, 2018). However, without the risk being public, or even frankly acknowledged within the industry, the sector operates dispersed (at 100 or so sites in the U.S. case). When the scale of the threat does become known through some event, the public demands a transition to as few sites as possible. The limit of this is movement to a single site that does not operate but instead stores the sector's legacy. However, because of the substantial costs involved in both replacing nuclear capacity and decommissioning plants, as a practical matter the phase-out occurs over a long period. The compromise is to not build nuclear power plants except those nearly complete. The mechanics of enforcement are to make it cost-prohibitive in terms of supplying safety. The potential for nuclear power to disperse further is limited, and the natural phaseout of nuclear power occurs as plants retire, resulting in the eventual accumulation of waste to one or a few sites.

To support the nuclear power story, Figure 9 presents the number of new nuclear power building permits issued since the beginning of nuclear power production.[32] Until the Three Mile Island incident in 1979, the sector continued to expand, and regulators on average

[32] Data included in appendix *A1.5 Nuclear Power Plant Construction Permit Approvals, 1955-2018.*

issue more building permits each year. The trend since 1979 is striking in both Figure 9 and the econometric RD and RK treatment in the next section – *1.5.5 Regression Discontinuity and Kink Methodologies and Estimates*. We can observe that only recently have firms begun to consider expanding nuclear generation again and the only new construction licensed since 1979 – all in recent years – come with caveats.[33]

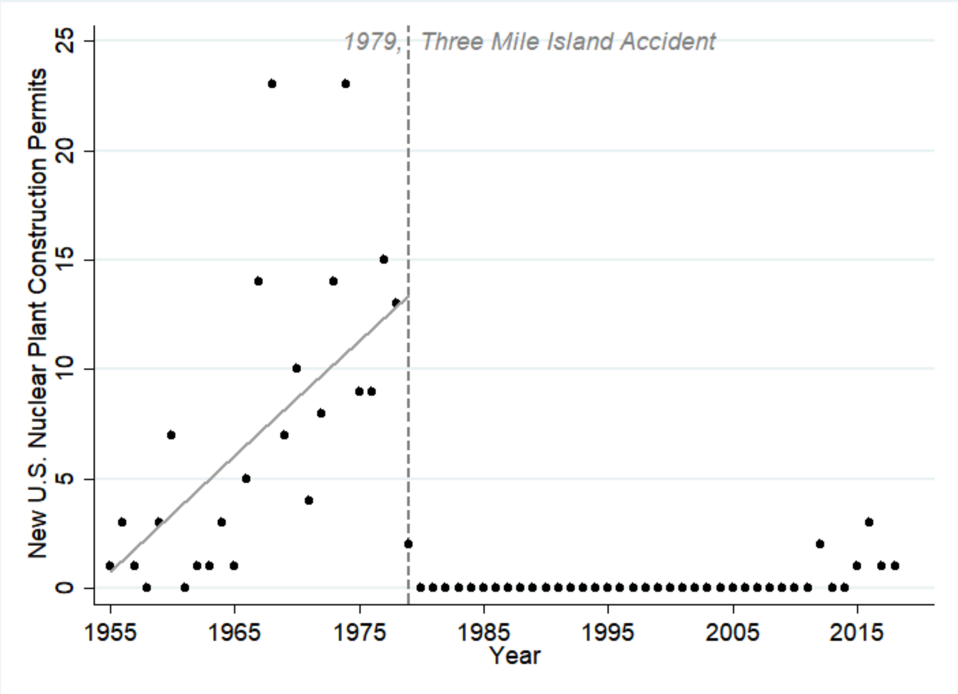


FIGURE 9 NUCLEAR POWER CONSTRUCTION PERMITS ISSUED IN THE U.S.

*Notes*: Number of nuclear power plant (generating unit) construction permits issued each year in the United States since the beginning of nuclear power generation. Following the Three Mile Island accident in 1979, no new building permit applications are completed for over three decades.

*Sources:* Author calculations based on data from the Energy Information Administration for years 1955-2011, and United States Nuclear Regulatory Commission for years 2012-2018, (EIA, 2012; NRC, 2018).

A nuclear accident provides a credible natural experiment – if it were expected, the plant operator would act to prevent it. But the weakness of the application is, perhaps, having to accept a narrative that a stoppage on nuclear expansion implies a preference for clustering. Unfortunately, other applications are also apparent. As previously noted, some additional applications are presented in appendix *A1.2 Siting Framework Applied Beyond the Environment*. These cover matters around radiation exposure, social problems, and international conflicts and terrorism. In the next section I outline a combined regression discontinuity and regression kink empirical methodology to put firm estimates to the

[33] To incentive diversification, the firms involved have been allowed to pre-charge customers for the power plants such that they are risk-free for firms to build (Biello, 2012)

household waste and nuclear power examples. This framework is also used in the noted appendix and later chapters of this thesis.

### *1.5.5 Regression Discontinuity and Kink Methodologies and Estimates*

Here I present the econometric methodology and estimates supporting the waste management and U.S. nuclear power examples. These rely on a combination of RD and RK methodologies. In the RD design the policy of interest leads to an often visually striking break in some time series of interest. Similarly, in the RK design some policy leads to a generally visible change in the rate of change of the variable of interest. The RK design is justified as a separate method from the RD design in, for instance, Card, Lee, Pei, and Weber (2012). However, the estimation strategy described by Angrist and Pischke (2009) is sufficiently flexible to estimate both sorts of effects.

In the examples of this section, let $Y_i$ be the outcome of interest in observation *i*. In the household waste example, $Y_i$ is the number of new refuse firms in year *i*, and in the nuclear sites case it is the number of new nuclear reactor construction permits in each year. In each model, let $x_i$ be an adjusted month or year, as applicable, of form $x_i = (time_i - k)$ where *time* is in standard calendar form and *k* is the moment of some critical event. As the purpose of the model is to estimate the impact of an event or policy enacted at *k*, not the influence of primitives on *Y*, the functional form for estimation is simple: Let a function $Y_i = f(x_i)$ approximate for the purpose at hand the true relation $Y_i = g(X_i)$ which has a potentially large vector of determinants of demand, $X_i$. The event, or treatment at *k* fundamentally changes underlying demand such that we can apply a binary indicator of form

(4) $$D_i = \begin{cases} 1 \; if \; year_i \geq k \\ 0 \; if \; year_i < k \end{cases}$$

and can write the conditional expectation of $Y$ as

(5) $$E[Y_i|x_i] = [Y_{0i}|x_i] + ([Y_{1i}|x_i] - [Y_{0i}|x_i])D_i.$$

Let the functional form of the pre-treatment conditional expectation be

(6) $$[Y_{0i}|x_i] = \alpha_0 + \beta_0 x_i$$

and post-treatment conditional expectation with both different slope and intercept be

(7) $$[Y_{1i}|x_i] = \alpha_1 + \beta_1 x_i$$

Then with $\alpha = \alpha_1 - \alpha_0$ and $\beta = \beta_1 - \beta_0$, the functional form for estimation is

(8) $$Y_i = \alpha_0 + \beta_0 x_i + \alpha D_i + \beta D_i x_i + \varepsilon_i$$

The coefficient $\alpha$ identifies any effect resulting in a discontinuity, and $\beta$ any change in the linear rate of change from $k$ onward. A strict regression discontinuity design results in an estimate of $\alpha$, and a strict regression kink design in an estimate of $\beta$. In Table 1, I report both RD and RK estimates for the empirical applications.

In the waste management example, the rate of new firms increases from 0.07 per year until 1945, to 0.07+4.02 per year thereafter. Reducing the span of pre-1945 observations increases the $\beta_0$ estimate but does not change the overall result. In the nuclear site example, the statistical story reverses as new building permits after the Three Mile Island accident drop to zero. Null hypotheses that $\alpha_0 + \alpha = 0$ and $\beta_0 + \beta = 0$ cannot be rejected at any appropriate level of significance which confirms the obvious – zero new permits are issued in the 30-years following the 1979 event.

In the nuclear sites example, we can take the Three-Mile Island accident as a truly random, uncorrelated treatment as the operators would avoid the incident otherwise. In the waste management example, individuals have no control over the timing of the introduction of new materials like plastic, when World War II will end and be followed by economic growth, or when a new disposable lifestyle takes hold in the rest of the populace. We might also be tempted to estimate a model using independent variables such as non-waste management firm growth and plastic production. However, general firm activity is the medium through which plastic producers transfer their goods to shoppers, and the two trends cannot be convincingly separated. One might also want to construct a panel of the EU 28 or a broader set of countries. But international supply chains in finished and intermediary goods as well as waste complicate the matter – the resulting measurement error would be prohibitive. A perhaps better methodology is to leave waste and plastic production activity in the aggregate and debate the plausibility of the story.

TABLE 1 — RK AND RD DESIGN ESTIMATES

| | Waste Management [a] (k=1945) | Nuclear Construction [b] (k=1979) |
|---|---|---|
| Date (*actual* – *k*) | 0.07*** | 0.65*** |
| | (0.02) | (0.11) |
| RD: D=1(*actual* ≥ *k*) | -5.94 | -15.53*** |
| | (6.15) | (2.08) |
| RK: Year*D | 4.02*** | -0.63*** |
| | (0.65) | (0.11) |
| Constant | 6.46*** | 15.43*** |
| | (0.94) | (2.06) |
| Observations | 89 | 64 |
| R-squared | 0.85 | 0.68 |

*Notes:* Robust standard errors in parentheses. *** Significant at the 1 per cent level, ** Significant at the 5 per cent level, * Significant at the 10 per cent level.

[a] Dependent variable is the number of new waste management firms registered in Orbis in each year up to 1970 reported in Table 24 of appendix *A1.4 Description of SIC Code 4953: Refuse Systems and Number of New Firms*. After 1970, the nature of the SIC code changes to include substantial growth in recyclers rather than general waste management firms as required in this analysis. In summary, a change in the rate of waste management firm creation coincides with widespread plastic use but also general firm activity.

[b] Dependent variable is the number of new construction permits issued for commercial nuclear reactors from 1955-2010 reported in Table 25 of appendix *A1.5 Nuclear Power Plant Construction Permit Approvals, 1955-2018*. Both a kink and discontinuity follow the Three-Mile Island incident after which zero permits are taken to completion for three decades.

*Sources:* Author calculations based on data from Geyer, Jambeck, and Law (2017), Freinkel (2011), Orbis (Bureau van Dijk, 2019a, 2019b), U.S. Energy Information Administration (2012), U.S. Nuclear Regulatory Commission (2018).

I am confident the examples I present do not exhaust applicability. Some applications occur around a premise that aggregation offers an advantage because total damages – or their potential with radiation and nuclear proliferation – are sufficient that total loss of a site's entire value is possible. For others, the issue may best be described by the interior case for clustering. Others may be a combination of both – first an interior clustering case, and then total loss potential as is expected with non-biodegradable household waste. The breadth of these applications suggest that a wide range of other applications may also exist. For instance, a stroll along many urban streams in developing countries or beaches that are unfortunate enough to become accumulators of plastic waste suggest that even the most common effluents can destroy nature. I now conclude this study before considering an approach to improving the plastic waste issue in the next chapter.

## 1.6 Conclusion

I have argued that a model of local-scale environments should allow for total environmental degradation and that important implications then result. Regulation to cluster or disperse where damages occur might increase social welfare but applying this sort of framework can have significant unintended consequences. I show that encouraging reorganisation need not require a heavy hand, but rather a change in the way regulation is applied. But without such regulation, polluters have an incentive to disperse to bypass local regulatory limits. A perverse incentive to degrade the environmental value of a site may also occur with regulation on the horizon, as once an environment is sufficiently damaged, its continued use has value as a pollution aggregator. Unfortunately, consumers, higher-level policymakers, and regulators outside the local environment have little incentive to push for regulation of local-scale pollution. Retained local authority over zoning, however, may allow the NEP alternative of regulating industrial and human organisation through zoning.

Despite various potential pitfalls, this framework expands the regulator and policymaker's toolkit at a time when such alternatives are needed to address critical issues such as biodiversity loss. It also suggests that compensation to impacted communities should exceed the damages they experience when clustering leads to avoided damages elsewhere. This provides a much-needed framework for the discussion and measurement of appropriate compensation beyond damage redresses. This paper also describes the appropriate AAD-based penalty or tax when regulating local environmental concerns in a region jointly and how providing polluters a choice between Pigouvian- and AAD-based regulation can lead to joint-optimality in appropriate cases.

I conclude with a prediction: As global transport costs fall, what we will consider local will become larger conceptual spaces.[34] As discussed, the international trade literature has already noted specialisation in polluting versus clean production. This is closely linked to the concept of local environments because it isolates areas of the global from some of the harm caused by production in others. The limit of this is globe-spanning specialisation in an increasing number of polluting processes. A problem emerges, then, in that international specialisation will be opposed by occupants of areas where the rest of the world sends their pollution. We can begin to observe this issue in where the world sends plastic waste for processing. The frequency of the issue is sure to increase unless a framework for the appropriate international organisation of pollution handling is adopted, such as the AAD-Pigouvian choice framework, and estimates of compensation to those adversely impacted are accurately made and compensation disbursed.

[34] Driven by more efficient shipping, automation, and declining borrowing rates. Transportation costs have already declined by 90-per cent in the last century and appear likely to continue to decline (Glaeser and Kohlhase, 2004).

## 2. A New Redesign Incentive to Improve Recyclability

*This paper outlines a new approach to improving the recyclability of products. By defining the optimal division of effort between producers – who can redesign products to be more cost-effective to recycle, and recyclers of consumer waste, methods of incentivisation become apparent. A tax and subsidy framework on the count of materials in products based on the optimal division of efforts may result in the full recycling of complex waste. The efficiency of this result and appropriate magnitude of the tax and subsidy are explored. As there is no precedent in the environmental sphere for the mechanism proposed, the discussion is chiefly theory-based. However, data on the complexity of common types of waste and the importance – and some values – tied to the purity of recovered waste are reported.*

Keywords: Environmental policy, recycling, sustainability, circular economy, waste purity, material recovery, industry, taxes, subsidies.

JEL codes: Q20, Q53, Q55

## 2.1 Introduction

Achieving a higher rate of consumer waste recycling has become an intractable problem in many countries. Of plastics – one of the most recyclable of all materials – only 9-per cent ever produced has been recycled (Geyer, Jambeck, and Law, 2017). Even within the regulated EU, less than one-third of plastics are recycled at present (European Parliament, 2018). Much of the plastic not recycled reaches the oceans at substantial cost. Beaumont, et al. (2019) put the value of the decline in marine ecosystem services due to plastic pollution at between $500 billion and $2.5 trillion per year.

There are many contributors to the cost of recycling. Nations with higher recycling rates such as Germany have extensive sorting and recycling systems. [35] On the demand side of production, firms may also choose new materials in production instead of recycled ones. Because of processing costs – exacerbated by sorting difficulties and the poor quality of mixed recycled materials, it can be preferable to use new materials (Brooks, Hays, and Milner, 2019). The combined effect is a generally challenging business environment for recyclers beyond a few choice materials. Difficult choices must be made, such as what to recycle or discard, and how much to invest in processing. A particularly costly aspect of processing waste is handling items made of many material types. Even if every material involved in a product is recyclable, unless the entire product is of the same material, recycling requires disassembly and sorting into separate streams at some cost. Alternatively, almost any materials used in a product – particularly packaging – has substitutes, albeit also at some cost.

The purpose of this research is to contribute an additional approach to improving the recycling system through improving product recyclability. It generally falls within the concepts of Extended Producer Responsibility (EPR) – methods to ensure producers cover the costs of wastes related to their products, and Design for Environment (DfE) – efforts to incentivise the design of products for environmental friendliness throughout the product's lifecycle. The main exploration of this paper is in finding and implementing an optimal division of effort between recyclers who disassemble and process wastes, and producers that can redesign a product to be more or less recyclable. With this in mind, I explore a mechanism that should result in improvements in product design in terms of recycling

[35] Germany has a 67% recycling rate for households and higher rates for commercial, production, and construction (BMU, 2018). However, there has been criticism of what this rate entails – whether the inclusion of plastic burned for energy, which is the end result of about half of plastic waste, should count as recycling. Wecker (2018) notes that Germany's recycling rate is based on waste collection, not the final processing and destination of waste.

efficiency. The incentive structure is intended to improve recyclability within products – changes in the composition or assembly of products (alternatively to improve the means of decomposition) – such that their whole is more recyclable. An alternative is to try to improve recyclability across products – changes in the entire stream of waste. This may, for instance, occur through efforts to standardise products and packaging. The purpose of this paper is to contribute to the former approach. It is a longstanding issue often noted in the recycling and waste literature review to follow. An approach to the latter is to instead increase the homogeneity of entire waste streams. This is discussed briefly in appendix *A2.1 Improving Recyclability Through Standardisation Auctions*. It is reserved for the appendix because it is an issue that has a perhaps more orthodox solution by today's standards – a Vickrey style auction over the right to design an industry's standard. In comparison, the within-product incentive structure in this paper is a bit less orthodox, though still likely recognisable to those familiar with the approaches of environmental economists.

First, a real-world example helps illustrate the complexity of the issue addressed. At present, around 500 billion disposable plastic drinking bottles are produced and used globally per year (Laville and Taylor, 2017). Many are of polyethene (polyethylene) terephthalate (PET), which is easily recycled. Where available, deposit-return schemes have been effective in ensuring they often enter recycling streams rather than go to landfills or into the environment (United Nations Environment Programme, 2017). Note, these programmes remain the exception, and a small share of global PET bottles are recycled. Even of the bottles sent to recycling, we generally recycle only part of the packaging. Most bottles have a separate label often made of another material – generally polypropylene (PP) or laminated PET – and a lid usually made of high-density polyethene (HDPE) or PP. Recycling lids requires more effort than bottles because lids of different materials and colours (natural versus coloured at a minimum) must be separated for the resulting recycled bulk material to have a positive value. Unseparated lid material will be HDPE *and* PP, not HDPE *or* PP, and recycled material prices are sensitive to purity. Lids and labels attached to bottles are not desirable because they must be separated, requiring processes at some expense. While sorting methods involving separation by specific gravity in a liquid are sometimes employed, it is still common that manual sorting of plastic bottles and other waste be employed at substantial cost (RDC-Environment and Pira International, 2003; Bio Intelligence Service and European Commission, 2011). This is even more true for complex wastes such as electronics and electrical parts.

No policy such as different recycling rebates for bottles with versus without caps or labels attached exists. There have been external and rather heroic efforts to recycle bottle caps through volunteer initiatives, but these are short-lived due to the economies involved – the costs versus returns expected. For instance, with careful management, the programme “500 Deckel für ein Leben ohne Kinderlähmung” (translation~ “500 Lids for a Life Without Polio”) financed substantial social benefit in the form of over 900,000 polio vaccinations through a collection of 450 million bottle caps (Deckel drauf e.V., 2019). Unfortunately, the effort ended recently when market prices for recyclable PP declined following the decline of new PP prices and a loss was feared. Unfortunately, when caps are not recycled, they become a source of ocean pollution harmful to wildlife (Boonstra and van Hest, 2017; Parker, 2019).

Even for simple products, there is a great deal of complexity to the resulting recycling problem. In the plastic bottle example, we observe that products made of 100-per cent recyclable material are often not 100-per cent recycled. When requiring that firms make products recyclable the matter can be definitional. From the perspective of a cost-minimising firm, requiring a plastic bottle and all related pieces – lid, label, and other attachments – to be 100-per cent recyclable is not equivalent to making the whole item cost-effective to wholly recycle. This issue is a symptom of misaligned incentives. One perspective on the underlying issue is that separate sorts of firms bear the alternative costs of product design versus recycling. In the bottle example, either a firm’s concern is producing bottles and the products they contain at least cost, or recycling bottles and related waste at least cost, not the least cost combination of producing and recycling. This is an issue that the EPR and DfE schools of thought sometimes seek to resolve. So, before presenting a framework to align production and recycling incentives and its efficiency, a literature review on waste and recycling is in order – particularly on EPR, DfE, and the concept of the circular economy (CE).

## 2.2 Literature Review

The literature on waste and recycling and the CE concept are based on familiar ground for economists. The underlying issue is scarcity, which is well known as a (or the) core principle of economics that we often trace to Malthus (1798). Scarcity, and in particular concerns that it will increase as resource reserves are consumed away, leads immediately to studies of sustainability, waste, and recycling. Champions of the sustainability movement often trace its origin to Boulding’s (1966) concept of a “spaceship earth” – a transition from an effectively open set of economies to a globally closed and limited economy once sufficient population and consumption demand levels are reached. But Boulding himself acknowledges George (1879) as inspiration and source material, which clearly presents the same argument as Malthus did nearly eight decades earlier.

From Boulding onward interest in waste, recycling, CE, and sustainability increases substantially. By the second half of the 1960's the concept of a globally limited set of resources is taken quite seriously, and several works predict an impending end to civilization and/or global mass starvation and so continue Malthus' tradition of failing to have predictive power. But by the end of the 1960's, technological innovation was being advocated for by the public as part of the larger green movement, and industry interest in recycling and conserving materials began to increase. Some early examples marketing to corporate interest includes Fisk (1973) and Harmon (1977). Eventually, the movement led to substantial government interest, including the establishment of the U.S. Environmental Protection Agency in 1970, and several other agencies became involved in researching and encouraging sustainability practices by the 1990's (U.S. Congress OTA, 1992). These earlier efforts emphasised reducing total material in products, reducing production waste, and reducing the amounts of hazardous materials produced overall – the low hanging fruit of the subject. Concepts such as redesigning products for recyclability and disposal were also in development but did not carry the same financial incentives for firms that reducing materials provides at this early stage. One of the key pieces of terminology to emerge from this period is Design for the Environment (DfE) and concepts within DfE such as designing for environmental processing and manufacturing, packaging, disposal or reuse, and energy efficiency.

Fiksel (1995, 1996) outlines several DfE strategies that have developed. These include reducing the number of distinct parts in products as well as using similar or compatible materials, but without consideration of the optimal improvement for firms to undertake. They also outline several recycling-relevant strategies such as designing for separability and simplifying component interfaces. These are some of the primary considerations of this paper. Here I put the concepts into an actionable economic framework and ask what the optimum number of material types is versus recycling effort and in effect make Fiksel's recommendations actionable.

One might suppose that consumer pressure through price would lead to DfE changes. That is, when consumers must pay for proper waste disposal, they will pressure firms to take DfE seriously. The use of charges for waste disposal, particularly at the residential level, have also become standard in some areas in recent decades. This subject remains an area of substantial research. Most studies have found that households and other waste producers respond little to a price on waste disposal. EPA (1990) compared weight- and volume-based approaches early on and found households responsiveness to be highly inelastic.[36] Similar values are reported across the studies noted in Choe and Fraser (1998). Bel and Gradus (2016) provide a meta-analysis of more recent studies and also find an inelastic response to the price of waste. Their value differs by location, but U.S. households, which produce a substantial amount of global waste, are particularly unresponsive to price changes on waste disposal.[37]

As firms have often already capitalised on cost saving DfE approaches, and consumers are unlikely to pressure firms further (at least in the heavily polluting U.S.), more drastic interventions are perhaps in order. The desire for further recycling progress has led to the concept of Extended Producer Responsibility (EPR) – a set of strategies or initiatives to ensure producers consider the full cost of their products on the environment. These are often mandatory regulatory, sometimes price-based, approaches to force or encourage producers to internalise environmental costs. Sometimes this results in firms physically taking responsibility for waste products but need not be so direct. It is not entirely clear where or if there is a functional difference between many EPR-branded initiatives and preceding mandatory approaches. Regardless, several attempts at encouraging EPR have emerged in recent decades.

Researchers often study policies to encourage EPR, DfE, and efficient waste processing and recycling at the producer level. Dinan (1993) studies the efficiency of taxes on new virgin materials versus a tax on material producers which is paired with a subsidy to recycled material users. They find the paired system more efficient. Palmer, Sigman, and Walls (1997) compare a paired deposit and refund system to advanced disposal fees and recycling subsidies and find the paired system again most efficient – least costly – in reaching a waste reduction goal. Palmer and Walls (1997) instead study recycled content standards but find that without a paired tax system it is difficult to reach an optimal level of recycling. They also find the information requirements of recycled content standards to be extreme as extensive information on each firm and product would be required. Choe and Fraser (1998) argue that kerbside charges, deposit-refund schemes, sales tax exemptions,

[36] They found elasticities around -0.10, suggesting a ten-per cent increase in waste disposal costs reduces waste by around one-per cent.

[37] They find an average elasticity of -0.339 from the 66 studies in their primary analysis which are generally in OECD countries. However, their mean coefficient from the U.S. studies is substantially smaller – close to zero.

and virgin material taxes may be difficult to implement in many cases. In particular, they are sceptical of how successful consumer-facing policies will be in ensuring high rates of recycling, alternatively preventing losses into the environment.

As some policies are easier to implement than others, Fullerton and Wu (1998) prepare a general equilibrium model to compare several policy options, specifically those implemented on firms versus those implemented on consumers. They argue that it is more difficult to apply something like a tax and rebate system on 100-million consumers, so it is useful to find the equivalent level of some mechanism applied on substantially fewer supplying firms. Among the policies that are suggested as being a substitute for deposit-refund schemes on consumers are, as in Palmer, Sigman, and Walls (1997), tax and subsidy systems on producers where the subsidy goes to whoever accepts the waste for recycling. Their rationale is that the most recyclable amongst existing packaging will be more widely adopted, as opposed to, say, Stillwell, Canty, Kopf, and Montrone (1991) which advocates for the redesign of packaging for recyclability.

Kinnaman and Fullerton (1999) note the rapid increase in both kerbside recycling programs as well as economist's interest in writing about recycling, but that many policy ideas have received little interest from policymakers. They also note that a consensus is forming by the later 1990's that some form of tax-subsidy combination is likely needed for many waste products to be recycled efficiently, but whether the policy is applied at the consumer "downstream", or firm "upstream" level is still a matter of substantial debate. Palmer and Walls (1999) differentiate between the concepts and incentives in Extended Producer Responsibility (EPR) versus Extended Product Responsibility. The former places the onus on firms, rather than sharing the burden to recycle along the chain of ownership. The original focus on producer responsibility involved exploring physical take-back requirements, but Palmer and Walls expand it to include a wider set of policy options. They advocate specifically for combined tax-subsidy systems on producers to encourage DfE and reduce transaction costs compared to systems involving consumers or consumer-facing stores. Calcott and Walls (2000) also explore whether upstream instruments are needed versus when downstream-focussed policies send sufficient signals to producers to encourage DfE. They argue that in most settings, transaction costs would be too high at the consumer level to send accurate signals about recyclability back to firms. Rather, Calcott and Walls advocate that at least some firm-level pricing policy is needed to encourage DfE.

Walls and Palmer (2001), like Fullerton and Wolverton (1997) and Calcott and Walls (2005), argue that multiple policy instruments may be needed when several types of externalities result from the production and consumption of goods. In particular, Walls and Palmer's argument is made in response to opinion at the time that the variety of responses to waste and recycling problems was too complex in the aggregate. They also argue that life-cycle assessments (LCA's) on goods which became popular at the time – evaluating product energy and material consumption from production through consumption and disposal – were inadequate as they lacked measures of the marginal damages component from usage. Walls (2003) then reiterates the EPR concepts and argues that it should include a breadth of policies such as DfE, not just producer take-back programs. Walls also notes the inherit difficulties of EPR, but that when recycling markets are functioning poorly – when they do not meet policy objectives on their own or substantial illegal dumping remains – that EPR-type policies are possibly the best policy addition. Slemrod (2008) too argues that when avoidance and consumer dumping are possible, that applying taxes and other policies on fewer and more easily monitored entities such as firms can be more effective.

Glachant (2004), as part of an OECD series on waste policy, also discusses policies to encourage the redesign of products to reduce waste. They explicitly view product characteristics as the result of an economic process – of price – so ensuring that the value of damages at the post-consumption stage are reflected back into the cost to firms of product design is important for DfE. Glachant also differentiates between policy requirements for DfE for nondurables – with rapid waste turnover – and durables – with long lifespans. They see the former as the realm of lightening and designing for simplicity, while the latter of reducing toxic materials and perhaps entirely redesigning products. They also note the problems of innovation – of the risks and costs inherit in revising a product's design. Because of the risks of not recovering the costs from extensive research and redesign towards DfE, it is thought that the standard tax-subsidy programs on producers has limited potential. Walls (2006) finds that such programs – and EPR in general – have been effective in reducing the weight of products and quantity of packaging materials. However, they are doubtful that more extensive product redesign will occur without more focussed EPR policies. This is a purpose of my paper. Karlsson and Luttropp (2006) also discuss the difficulties of firms in pursuing DfE, but under the term EcoDesign. They suggest that such considerations must often occur at the earliest stages of product design, which then increases the costs – and incentives required – for product change. Wang, Chang, Chen, Zhong, and Fan (2014) also explore how subsidies provided to firms at different stages –

design, production, and post-consumption, result in fairly different incentives and subsequent producer responses.

DEFRA (2011) argues that waste and recycling should have a clear goal of reaching efficient levels but that many policies in practice do not set efficiency-based goals. They also put values to different waste recovery methods in terms of tons of carbon dioxide released and show that the recycling method used can result in a substantially different environmental impact. Hennlock et al. (2014) discusses recycling policy types and effectiveness in the Nordic countries. While such nations have generally achieved higher recycling rates, Hennlock et al. outlines that these programmes have come up against familiar barriers including high administrative costs of programmes involving many participants, the difficulty of ensuring that wastes, particularly plastics, maintain a high enough purity level to be recycled into high value products, and low levels of coordination between producers and recyclers. Cimpan, Maul, Jansen, Pretz, and Wenzel (2015) evaluate the current state of material sorting and recycling technologies. They discuss that high purity requirements for recyclable wastes – typically in excess of 95-per cent – have led to substantial research and development and subsequently upscaling of recycling firms in recent years. It is not at all clear whether the scale of the necessary investments involved are efficient as the recycling sector is often symmetrically isolated from the production side of products. That is, there is insufficient information exchange on how much producers should simplify products versus how much recyclers should invest. Hennlock et al. (2014) and Cimpan, Maul, Jansen, Pretz, and Wenzel (2015) in particular have noted subjects that this paper's framework seeks to address.

Recently, the concept of the Circular Economy (CE) has gained in prominence. Ghisellini, Cialani, and Ulgiati (2016) provides an overview of its origins in the 1980's, for example in Stahel and Reday-Mulvey (1981), but that substantial growth in its use begins in the 2000's.[38] In short, CE is said to include EPR, DfE, product LCA's, and recycling innovations into a wider framework. CE is most known in simpler form as an advocacy framework for closed- or nearly closed-loop production and consumption processes. But this, and CE generally, entails a multipronged approach to environmental damages and scarcity – consumer and firm environmental awareness, advocating for cleaner and efficient production practices, extensive recycling and renewable energy development, and robust

[38] Murray, Skene, and Haynes (2017), like scholars of prior works on recycling and waste management, also trace the origin of the CE movement to Boulding (1966). It is difficult to draw a clear distinction between efforts labelled as CE advocacy versus prior work. While some authors, such as Ghisellini, Cialani, and Ulgiati (2016) depict CE as a larger umbrella movement encompassing EPR and DfE, it is not clear that CE, other than as a popular and sometimes provocative term, is a new concept.

environmental policy development. CE policy advocates generally emphasise technological advancement, including higher recycling efficiency levels, while generally acknowledging that the appropriate incentives for firms must exist. Genovese, Acquaye, Figueroa, and Koh (2017) provides an example of this lattermost issue – that at this time many closed-loop production transitions would have positive environmental impacts but would be losing propositions for firms without some form of state intervention. Ghisellini, Cialani, and Ulgiati (2016) note as does Murray, Skene, and Haynes (2017) that CE is of particular interest in China as a top-down policy objective, and elsewhere as a grassroots movement. Others, such as Orset, Barret, and Lemaire (2017), find that consumers in their study area on average have a higher willingness to pay for environmentally friendly and recycled products. So, it seems that in some cases the issue of information asymmetry remains.

Bocken, Pauw, Bakker and van der Grinten (2016) within the CE literature, like in the EPR and DfE literature, suggest the need for product redesign to improved recycling. But as with others, does not outline a sufficient incentive structure to encourage firms to do so. EEA (2017) focusses on specific issues related to product design incentives and their trends. Among these, a trend of increasingly complex product designs – and in particular more heterogeneous materials – which complicate recycling efforts. Kalmykova, Sadagopan, and Rosado (2018) also provide a guide to circular economy policy options which is decidedly lacking in voluntary, price-based incentives. So too, Savini (2019) advocates for transitioning to a circular economy but does not provide relevant guidance on incentivising a circular economy. Often, today's policy recommendations for waste and recycling problems, such as marine plastic accumulation, are still those proposed three or so decades ago (Abbott and Sumaila, 2019). While there has certainly been progress on some environmental issues, waste and recycling remain a difficult issue where further improvements appear to be needed.

Providing a new, targeted policy framework to reduce within-product material heterogeneity is the key objective of this paper. This is a variation on tax-subsidy schemes, but decidedly different in terms of design and implementation as well as limitations. It specifically seeks to improve recyclability by simplifying products while encouraging recycler-producer coordination. As shown, the issue it addresses – of material heterogeneity and complexity – has been noted in the literature since at least the 1990's yet remains a hindrance to higher recycling rates across countries. Before presenting the core model of this paper, I next provide some specific estimates on the costs and complexities involved in the recycling problem.

### 2.3 Some Select Estimates of Waste Composition and the Importance of Purity in Preserving Value

In this section I provide some estimates on the complexity of products, price sensitivity of recycled material to purity, and scale of prices and costs involved in recycling. The example products explored in this section are likely familiar to most consumers – they are the sorts of items that many of us use and dispose of daily. The products are also the source of a lot of waste we may find when exploring nature.

An issue in addressing the recyclability problem is that often we must accept that knowledge of a firm's marginal abatement cost schedule is private and guarded. Firms facing competition, taxation, or even subsidies have reason to guard this information. We can, however, derive some estimates using a revealed preference framework over recycling prices and activity for select products. I use a list of the ten or so most collected items along European coastlines (Ocean Conservancy, 2018). This list has a high degree of overlap with other beach clean-up studies such as European Commission (2017) and the consumer litter entries in Hardesty et al. (2017). I organise the list into eight items and expand the entries to include each item's associated materials (i.e., if we recover a bottle cap, a bottle also existed). The Ocean Conservancy sample is not representative of consumer waste, but rather is biased toward wastes that float and do not rapidly disintegrate. These are the most common sorts of waste observed by the public in the environment. In Table 2, I provide the derived list with estimates of material quantities ($\overline{N}$), detailed usual composition, and whether societies generally recycle the components.

TABLE 2—COMMON CONSUMER WASTES ON COASTLINES, THEIR COMPOSITION, AND FATE

| | Typical $\bar{N}$ | Typical composition | Recycled? |
|---|---|---|---|
| Cigarette butts | 3[a] | Residual tobacco, paper, and cellulose acetate filter | No |
| Food wrappers (ex. Crisp packets) | 3[b] | Layered BOPP, LDPE, and durable ionomer resins | No |
| Plastic beverage bottles and caps/lids | 4-5 | Bottle: PET/PETE, HDPE, LDPE, PS other polymers<br>Cap/lid: PP, HDPE, sometimes with a PE lining<br>Label: PP or laminated PET and/or EVA on paper | Yes<br>No<br>No |
| Plastic grocery or other bags | 1 | HDPE, LDPE, LLDPE, or PET | Sometimes |
| Disposable cup and straws or stirrers | 3-4[e] | Cup: paper with PE liner<br>Lids: PS or PP<br>Straws or stirrers: PP | No<br>Yes<br>Yes |
| Plastic take out/away containers (plus cutlery and carry bag) | 2 | Container: PP<br>Cutlery: PP<br>Bag: plastic bag (see above) or paper | If clean<br>Yes<br>Sometimes |
| Foam take out/away containers (plus cutlery and carry bag) | 3 | Container: PS or EPS<br>Cutlery: PP<br>Bag: plastic bag (see above) or paper | If clean<br>Yes<br>Sometimes |
| Glass beverage bottles and caps/lids | 4-5 | Bottle: Glass of various colours<br>Caps/lids: steel or aluminium with a PE lining<br>Label: Printed paper or laminate | Yes<br>No<br>No |

*Notes:* Reorganization of top ten list of consumer waste products collected from ocean coastlines internationally. The reorganisation includes pairing beverage bottles with lids and expanding to include complement waste (ex. including cutlery with plastic take out/away waste). In practice, whether items are accepted for recycling depends on local guidance and initiatives. Plastic acronyms used: Biaxially oriented polypropylene (BOPP), expanded polystyrene (EPS), ethylene-vinyl acetate (EVA), high-density polyethene (polyethylene) (HDPE), linear low-density polyethene (polyethylene) (LLDPE), low-density polyethene (polyethylene) (LDPE), polyethene (polyethylene) (PE), polyethene (polyethylene) terephthalate (PET/PETE), polypropylene (PP), and polystyrene (PS).

*Sources:* Top Ten list from Ocean Conservancy (2018) reorganized and expanded to include commonly associated waste and materials based on author estimates and other sources: [a] Terracycle (2019); [b] Trending Packaging (2016); [c] Margolis (2018).

On average, the waste in Table 2 is composed of three materials that are generally separable without great difficulty. Exceptions to separability are multilayer food wrappers and labels which are matters of marketing and consumer appeal rather than out of necessity. That these products are separable, however, does not mean that separation and sorting by recyclers is optimal. A recent development at fast-food franchise Taco Bell suggests the feasibility of single material packaging as 95% of the firm's new cups, lids, and straws are made of the same recyclable plastic (Danigelis, 2019). As in the bottle cap example, the difficulty is not always mechanical separability from other components, but lack of distinction such as between HDPE and PP plastic.[39] *The Plastic Bottle Cap Report* (Boonstra and van Hest, 2017) notes a great variety of cap types discovered on North Sea beaches – 25 types with 80-per cent originating from drink bottles, and several colours – roughly 30-per cent white and 70-per cent in a rainbow of variety. This may be more variety than needed to hold a product and then be discarded.

[39] HDPE/PP bottle caps float while PET bottles sink in water which influences both the prevalence of bottle cap discovery on beaches and is one method used to separate out caps from bottle material (though manual sorting is still common).

Analysis of recycling material prices emphasises the importance of material purity. Table 3 reports average prices for some types and grades of recycled materials and associated costs in the U.K. The value of recycled materials sharply declines when not separated – when it is contaminated. The extreme comparison in Table 3 is where natural HDPE bottles fetch 30 times the price of mixed plastic bottles per tonne. Plastic film also transitions from having a positive market value to a disposal cost when more than 5-per cent contamination is present, highlighted in Figure 10. A high alternative cost of disposal (a minimum of £82/tonne) bounds recyclable waste prices in the U.K. resulting in, historically, a lucrative industry exporting recyclables of dubious quality to countries with less oversight (Arkin, 2019).[40] The primary destinations for this waste – China and then others recently cracked down on dubious recyclable material imports with global repercussions. This is done primarily by raising the purity rate required to import bulk plastic waste for recycling (Katz, 2019; McNaughton and Nowakowski, 2019).[41] The problem of poor recyclable waste quality is becoming an increasingly pressing, global problem as more countries adopt China's response.

40 See also several case studies on the processing of recyclables abroad in Alexander and Reno (2012).

41 Liu, Adams, and Walker (2018), argue that regulations maintaining firm responsibility for recyclables sent abroad would improve purity rates. In practice, it seems often exceedingly difficult to hold firms accountable for waste once it has left the country of origin. They also argue that policies pushing technological transfers of waste technologies to developing countries would improve the situation – another policy that is difficult in practice with current intellectual property practices.

TABLE 3—PRICES OF RECYCLABLE MATERIALS IN THE U.K., 2018-2019

| | Price range (£ / tonne) |
|---|---|
| Plastic bottles | |
| Clear and light blue PET | 79-156 |
| Coloured PET | 25-38 |
| HDPE natural | 315-363 |
| HDPE mixed colour | 93-114 |
| Mixed | 11-58 |
| Plastic film | |
| PE Clear - Natural | 263-296 |
| PE Printed | 174-194 |
| Export 98:2 | 138-177 |
| Export 95:5 | 4-28 |
| Export 90:10 | (27)-(2) |
| Export 80:20 | (76)-(55) |
| PP Printed | 14-37 |
| Glass | |
| Clear | 15-25 |
| Brown | 13-22 |
| Green | 3-13 |
| Mixed | 7-17 |
| MRF glass | (28)-2 |
| Metal cans | |
| Aluminium | 867-938 |
| Steel | 113-129 |
| Alternative disposal | |
| Energy-from-waste site | (82)-(109) |
| Refuse-derived-fuel site | (88)-(98) |
| Landfill disposal and tax | (104)-(116) |

*Notes:* Average range of prices at recyclers in the U.K. from January 2018 through June 2019. In parentheses, red numbers indicate negative values – payments required for disposal. Prices include plastic recovery and export (PRN/PERN) value.[42] When numbers are presented in the material types column, the first indicates per cent plastic and the second the per cent of contamination.

*Source:* Author calculations based on monthly survey of prices by Letsrecycle.com (2019).

[42] Package recovery notes (PRN) and package export recovery notes (PERN) certify that materials will be recycled into new products and exported to facilities meeting UK standards. Firms can sell PRN/PERN to companies with recycling obligations, for instance in package waste compliance programmes, and so have value.

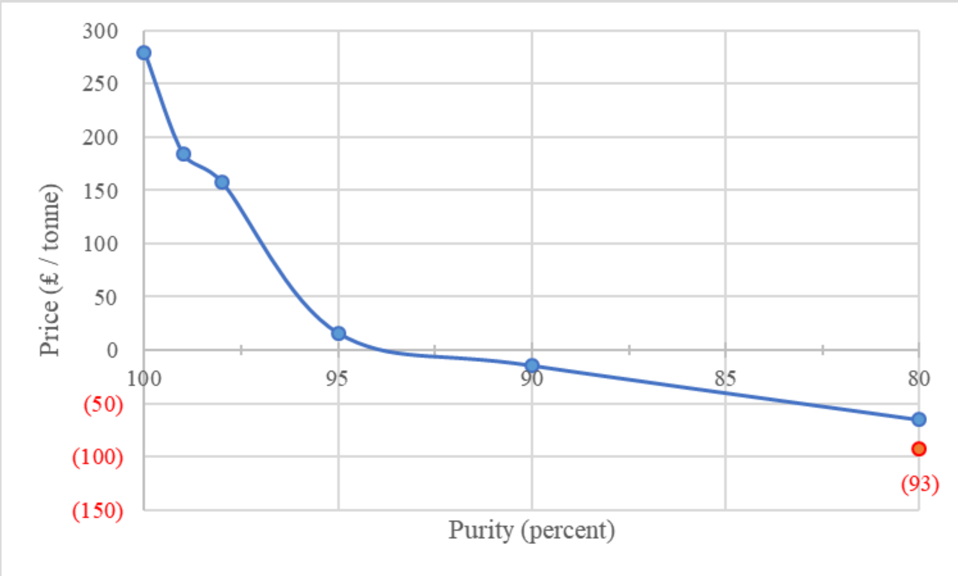


FIGURE 10 RECYCLED POLYETHYLENE MARKET VALUE BY PURITY

*Notes:* Based on mean monthly prices January 2018 through June 2019 for polyethene (polyethylene) (PE) plastic film used in packaging based on the level of contamination. Positive salvage value declines rapidly and becomes a discard price beyond 5-per cent contamination. Processing costs for the more contaminated film is bounded by an alternative waste disposal cost of on average £93/tonne.

*Source:* Author calculations based on Letsrecycle.com (2019).

The issue of waste pervasiveness in nature is becoming quite pressing yet deriving recycled materials of higher purity is costly. We can observe that when sufficient levels of purity are not met, the value of recyclables quickly becomes negatives. As this problem is resistant to existing methods of redress, new approaches are in order. Recycling requires preferably NEP-compatible policy options that are pragmatically applicable to when simplifying product composition and ease of decomposition by recyclers would improve recyclability. I next outline an instrument for when a policymaker wants to incentivize more recyclable product designs.

### 2.4 Introducing the Per-Material Marginal Abatement Cost Decision

I explore the benefits of a per-material tax on manufacturers to reduce the number of materials used in products and fund the decomposition and recycling of complex products through a subsidy. As a starting point, I present an abatement cost framework where for any $n^{th}$ component of an existing product, there are two costly options for improving whole-product recyclability. Suppose we can rank the costliness of changes to the $\bar{N}$ components from highest to lowest abatement costs. At any change in the processing of the (discrete) number of materials in a product, a cost-minimising policy satisfies

(9) $$min\{production\ change\ cost_n,\ \ recycling\ cost_n\}$$

where each choice might include a complex set of changes representable as quantities and accompanying costs. An example is a producer switching from metal to plastic fasteners in a mostly plastic product. A fastener change impacts not only the per-piece cost of fasteners, but perhaps the number of fasteners needed, the structure of the surrounding material, assembly complexity and durability, and thus consumer satisfaction with the product. Alternatively, a cost comes with separating the metal and plastic components and recycling separately. For materials that are profitable to recycle anyway, the invisible hand gets to work on the matter. Frequently, however, recycled material prices are insufficient, and recyclable materials are discarded when attached to other waste (European Environment Agency, 2019).[43]

Items entering the recycling stream are composed of one or several materials which may be separate parts or permanently combined such as laminates. For expositional purposes, suppose each part in a product entering the recycling process is somehow separable to be sorted into independent recycling streams. From an optimal recycling design perspective, two options exist to prepare any of $1{:}\bar{N}$ pieces of a discarded item for eventual recycling. Either the producer can reduce the number of material types by making parts of the same materials – make it of $N < \bar{N}$ materials – or improve disassembly (lower the cost) for recycling.

Consider if both product design and recycling are by a single firm that is required to recycle. The objective function over the potential number of pieces in a product would extend Equation (9) as

(10) $$\sum_{n=1}^{\bar{N}} \min\{production\ change\ cost_n,\ \ recycling\ cost_n\}$$

where summing over all $n = 1{:}\bar{N}$ includes the possibility to create a recyclable product of one material, but also to make no change to the initial product and rather process it by disassembly and recycling only. We might also suppose there is an option of choosing $n = 0$ materials and not make the product at all, or at least the part of it not consumed. This may be applicable to cases where the entire value of a product is exceeded by the cost of recycling it, or alternatively, the damages it imposes on the environment. This is generally

[43] Particularly difficult are recyclables attached to or containing hazardous components which this framework can also address.

assumed to be an exception, however. We might also find cases where packaging serves a purely aesthetic purpose for a target audience, e.g. placing a wrapper on a coconut.

Solving Equation (10) as if a cost minimising firm internalises recycling and disposal costs results in an efficient recycling policy. This efficient result occurs as the firm would choose the least cost combination of product changes and recycling processes. However, firms are rarely involved in both production and recycling, and a coordination problem results.[44] Producers design, manufacture, distribute, and sell a product to consumers who discard some or all of it. Other firms – recyclers – are in the business of recycling the disposed of items – processing the materials to sell for use again. Alternatively, waste may be sent out for incineration for energy, sent to landfills, or lost into the environment. Producers and recyclers face different incentives over product design. Producers, when not subject to the full cost of the environmental impact averted by an Equation (10) decision process, strictly choose the most profitable material and assembly style for each part of a product. The producer's unregulated preference is the material variety of $\bar{N}$ types. Recyclers instead prefer a uniform and easily disassembled product, all else equal, and would no doubt prefer materials of a higher recycled value since producers are footing the material bill.

Generally, one defines a marginal abatement cost curve over a group's costs as $-C'(\cdot)$ on a differentiable cost function, $C(\cdot)$. An assumption is that abatement only affects costs, not revenues, which perhaps unintentionally excludes revenue-generating aspects of abatement like public goodwill. Yet some firms use sustainability in product differentiation to great success, e.g. Patagonia, a brand whose business model and image emphasise controlling their environmental impact (Patagonia, 2018). Their business model suggests that revenues may increase when firms take responsibility for the recycling of the materials used in production, at least in specialty cases. I then derive the marginal abatement cost curve more appropriately over the entire profit function, $-\pi'(\cdot)$.[45] In a recycling context, material choices affect both revenue and costs when material choices impact quality. For producers, choosing $N < \bar{N}$ reduces profit and likely increases costs, as without regulation the firm chooses their profit maximising level of materials, $\bar{N}$. For recyclers, $N < \bar{N}$ decreases costs but has an ambiguous effect on revenue. Suppose producers initially make a product of $X$ volume of $\bar{N}$ materials, but then change their material schedule to make it of

[44] Exceptions occur when new inputs are too costly, scarce, and difficult to substitute away from to not recycle. A perhaps extreme example is the availability of frostbitten toes for garnishment in the sourtoe cocktail (NPR, 2017).

[45] If it were necessary to use both cost and profit bases in this paper, I would differentiate between them as, say, $MAC^C$ and $MAC^{\pi}$.

$N = \overline{N} - 1$ materials without a change in the total volume. Let $\overline{N} = 2$ and a recycler's initial revenue from recycling the product be $P_1X_1 + P_2X_2$ where $X_1 + X_2 = X$. After redesign, their revenue is $P_1X$. If $P_1 < P_2$, then the recycler's revenue decreases. I set this pragmatic matter aside, however, and suppose producers act efficiently and choose higher cost materials only out of necessity. We might also suppose that, generally, recycled material prices relate to producer input prices. Then, $P_1 \geq P_2$ for a recycler to keep the analysis tractable.

While the number of material types in a recyclable product is distinctly discrete, it is useful to discuss it as though it is a continuous quality so that I can use the language and notation of differentiable functions. Let the profit function of a producer be $\pi_P(N,\cdot)$, of a recycler be $\pi_R(N,\cdot)$, and let the effect on cost be greater than the effect on revenue from a change in $N$. $\pi_P(N,\cdot)$ is weakly increasing in $N$ and $\pi_R(N,\cdot)$ weakly decreasing. Define the marginal abatement cost curves for producers and recyclers by

$$-\frac{\partial \pi_P(N,\cdot)}{\partial N} = MAC_P(N) \tag{11}$$

$$-\frac{\partial \pi_R(N,\cdot)}{\partial N} = MAC_R(N) \tag{12}$$

Equation (11) is an extension of the standard definition of a marginal abatement cost curve in that it maps the impact of abatement decisions of the polluting firm on their own profitability. Equation (12) is more exotic as the recycler did not produce the polluting item. Rather, $MAC_R$ maps the cost to avoid environmental damages by recycling once a product is made and discarded. It is not a marginal damage function either, rather the marginal cost of avoiding damages. But rather than plague the world with another acronym, I refer to it as a marginal abatement cost curve for the recycler.

### *2.4.1 Encouraging Design-for-Recycling Through a Per-Material Tax*

As part of the EU Plastics Strategy, the European Commission identifies innovation as critical to reducing the quantity of plastic waste entering the environment (European Commission, 2018). This section outlines an efficient strategy to incentivise full recycling as well as product innovation toward that goal.

If an interior optimal solution exists in choosing between product change costs and recycling costs with a cost minimising objective, then $N = N^*$ achieves an efficient outcome (Note 3) when

$$MAC_P(N) = MAC_R(N). \tag{13}$$

The efficient number of remaining material types in a product, $N^*$, is the number passed to recyclers which then requires $N^* - 1$ separating processes to fully recycle. The complement $\bar{N} - N^*$ is the socially optimal reduction in material types by producers. Because it is optimal that producers reduce up to $MAC_P(N^*)$ and a recycler process $N^*$ materials, an optimal tax imposed on the producer on the number of materials used is $\tau_P^*(N^*)$. Additionally, a subsidy to recyclers $s_R^*(N^*)$ would ensure profitability in processing materials composed of $N^*$. A standard at $N^*$, alternatively, an optimal tax without a subsidy may be insufficient because the profit incentive of a recycler is not ex ante guaranteed. For an efficient interior solution, it must be that $\tau_P^*(N^*) = s_R^*(N^*)$ as depicted in Figure 11. The dashed line highlights the combination of abatement costs that minimise the cost of full recycling. An efficient recycling support strategy incentivises this combination such that no component of the product is left without an associated incentive to recycle.

Note 3 – Achieving the Efficient Outcome from a Zero Loss Objective

As perhaps the simplest approach to finding the socially optimal level, define the problem in terms of competing abatement costs from the producer and recycler with an objective to minimize loss of residual waste value into the environment, $\min_{\{N\}} Residual\ waste\ value = AC_P(N) - AC_R(N)$. Then differentiating and setting the desired loss, alternatively waste value, to zero results in $MAC_P(N) - MAC_R(N) = 0$. So, choose $N = N^*$ such that $MAC_P(N) = MAC_R(N)$. More complex objective functions can obviously be written, this one also serves as advocacy to address a particularly pressing environmental problem.

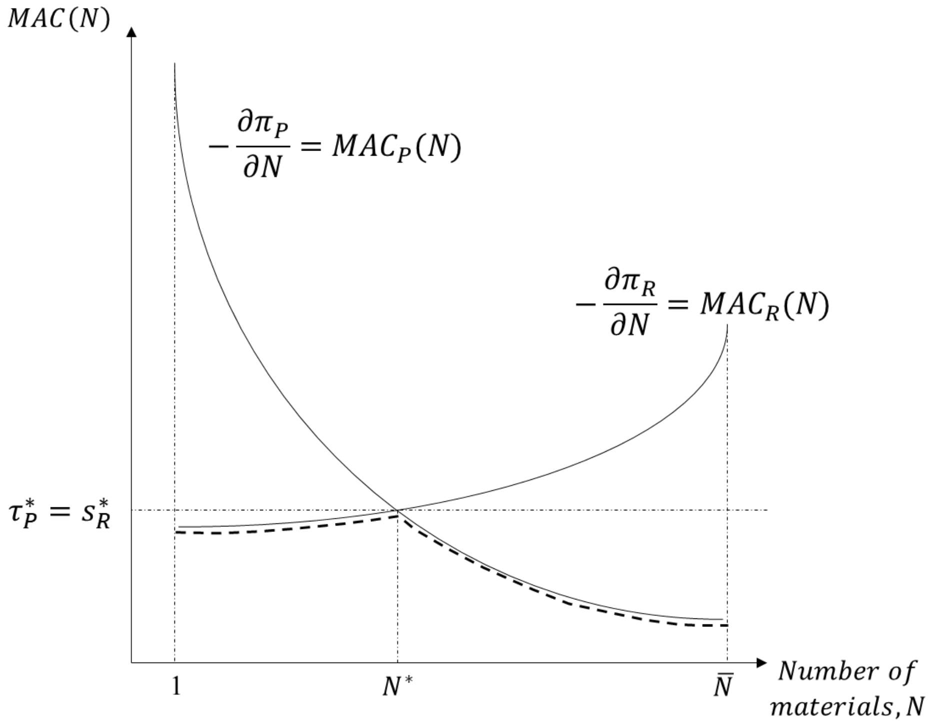


FIGURE 11. OPTIMAL $N$ AND DIVISION OF ABATEMENT EFFORT

Notes: Marginal abatement cost curve of recycling $MAC_R(N)$ maps the damage avoidance costs once a product is made and sent to recycling. $MAC_P(N)$ is the cost of reducing the number of materials to make the product easier to recycle. The least-cost combination of $MAC_R(N)$ and $MAC_P(N)$ to redesign versus decompose and recycle a product is emphasised by the dashed line. An efficient damage avoidance strategy incentivises $N$* materials in a product. Taking the curves to zero would suggest the recycler might not recycle at all, or the producer not produce at all or produce something with no waste.

Compared to optimal $\tau_P^*(N^*) = s_R^*(N^*)$, a higher level of taxation and subsidies is inefficient, and a lower level is insufficient. $\tau_P = s_R > \tau_P^* = s_R^*$ is inefficient because it incentivises overinvestment in abatement – producers seek to reduce, and recyclers to process some of the same components. It could, however, result in innovation through producer-recycler competition. In contrast, $\tau_P = s_R < \tau_P^* = s_R^*$ leaves a gap where neither invests, and some material is discarded. We might also expect bounds on each MAC. For producers, there is an alternative of not producing at all, and for recyclers of not recycling at all.

A different depiction results from a partially profitable recyclables stream. Figure 12 outlines a case where components of a recycled product up to $\widehat{N}$ are profitably separated and recycled in $\widehat{N} - 1$ processes. Even if producers, compelled by a tax, reduce the number of materials to $N^*$, a subcomponent $N^* - \widehat{N}$ is discarded without an offsetting subsidy awarded on at least processes covering $N^* - \widehat{N}$ materials.

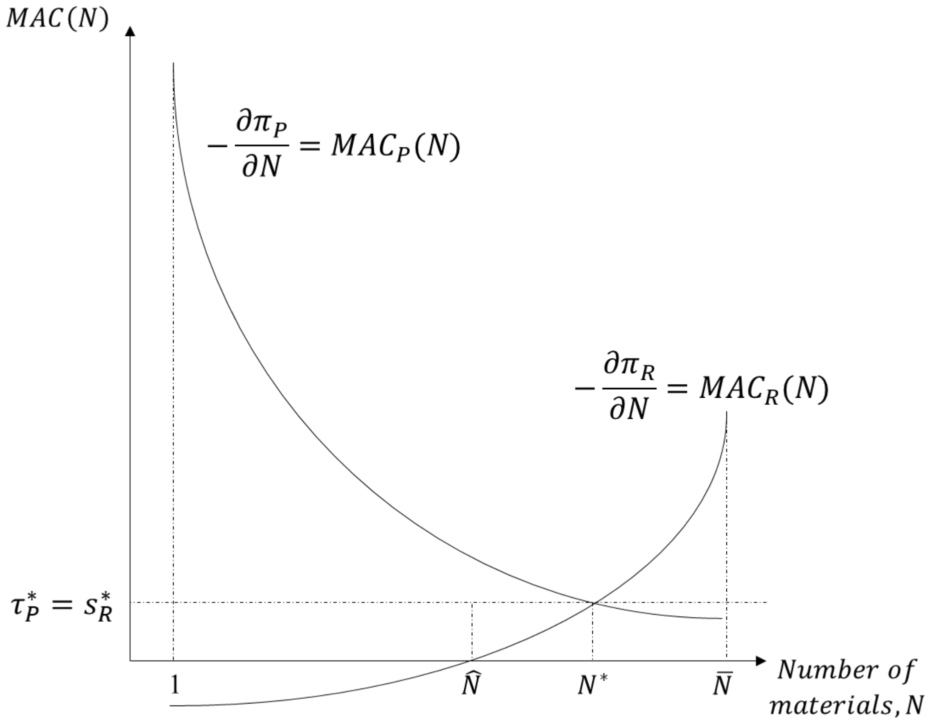


FIGURE 12. OPTIMAL DIVISION OF ABATEMENT EFFORTS WITH PROFITABLE RECYCLING

Notes: A slightly more realistic depiction of optimality – since recycling programmes already exist – notes that $\hat{N} - 1$ separating processes must be profitable. As an example, the processing of plastic polyethene (polyethylene) terephthalate (PET) bottles may be profitable while labels and caps are unwanted and discarded. $\tau_P^* = s_R^*$ compels efficient producer and recycler behaviour, but the subsidy is only necessary over components $N^* - \hat{N}$.

Firms can clearly redesign products – it is entirely within their right to do so. It is often an approach advocated for in EPR and DfE circles. In the next section I consider how the optimal tax and subsidy changes with product redesign. It explores how the tax and subsidy encourage redesign by producers to lower their tax obligation. This incentive also encourages producers to coordinate with recyclers – to seek their input on product design for recyclability. Unfortunately, I also outline that there is a cost involved in this process for the regulator – the cost of reassessing the optimal tax and subsidy following a products redesign. But surely in some cases this cost will be less than that of other EPR approaches.

### *2.4.2 Innovation in the Efficient Programme*

A leader-follower relationship is inherent in the production and recycling of goods as a producer controls their composition.[46] While the incentive for dynamic efficiency with environmental taxes is known, a second incentive to innovate occurs with a per-material recycling tax. Assume the profit function of a recycler is dependent on the design of the

[46] This is aside from, perhaps, existing cases of producer-recycler coordination. But then, without a legal framework requiring producers to act on a recycler's recommendations coordination is voluntary, and producers maintain a dominant position.

item and summarised in a parameter that indicates the ease of processing, $\delta_P(\cdot)$.[47] The recycler profit function is then $\pi_R(N, \delta_P(\cdot))$. Consider if, in each iteration of product design, the regulator recalculates the optimal tax. A producer has an incentive to confer with a recycler to make products easier to recycle because it reduces the tax burden on the producer. This change, however, also reduces the matching subsidy to recyclers.[48] I refer to this as iterative efficiency as it occurs when producers make improvements in each new iteration of a product, followed by evaluation of the tax and subsidy policy. Taking the effect of $\delta_P(\cdot)$ on a recycler as a scalar multiplier, I present the iterative efficiency incentive in Figure 13. With a decrease in processing difficulty, $\delta_P(\cdot)$ decreases from $\delta_P^1(\cdot)$ to $\delta_P^2(\cdot)$. Savings to a producer are the shaded area resulting from iteratively reducing the tax level to match the efficient level and shifting the responsibility to recyclers from $N^1$ to $N^2$. While the efficient subsidy also decreases, because the amount of business for a recycler increases, the effect on recycler profitability depends on the producer and recycler marginal abatement cost functions.[49]

[47] The sub-P notation reminds which party – the producer – is in control of the parameter.

[48] In the limit, a producer might make their product fully profitable to recycle.

[49] We might also take interdependence further and suggest that recyclers impact producer profitability through the effect on material prices. In some cases, when the volume of recycling increases, we might expect the price of recycled materials to decrease.

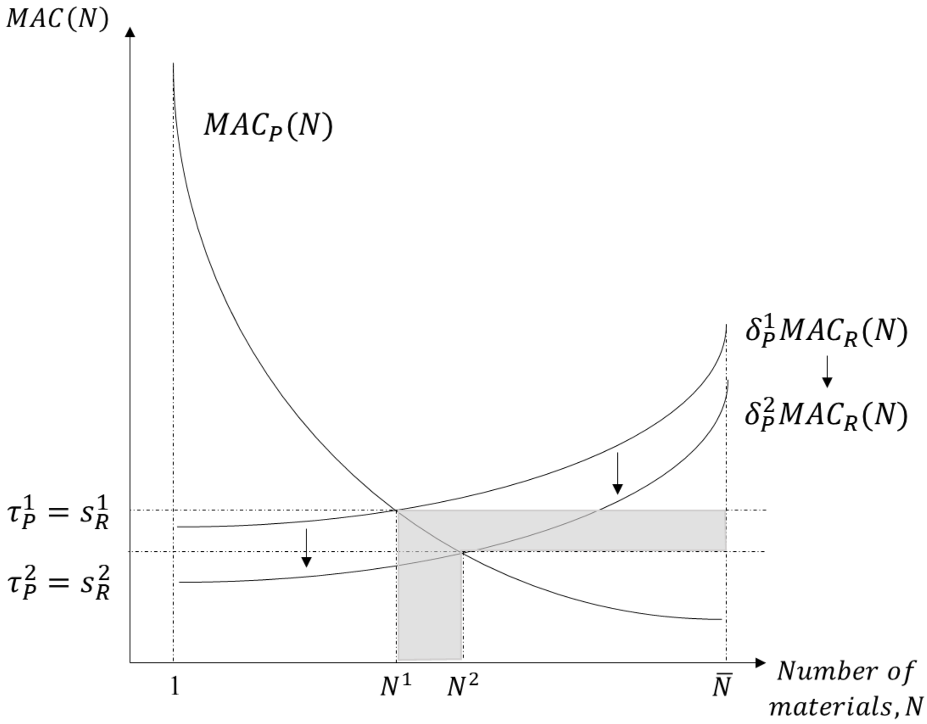


FIGURE 13. ITERATIVE EFFICIENCY INCENTIVE TO MAKE PRODUCTS MORE RECYCLABLE

Notes: Dynamic efficiency leads to improvements within the producer's marginal abatement cost function, $MAC_P(N)$, while iterative efficiency results in improvements in a recycler's $\delta_P MAC_R(N)$ as new iterations of a product are made easier to recycle. I highlight iterative improvements which reduce both the tax on producers and the number of material reductions that are efficient for a producer to undertake. Recyclers, who act given a producer's decisions, receive a lower subsidy per-material after programme re-evaluations. However, the number of materials a recycler processes increases, and the effect on the recycler's profit as well as net revenue or cost of the programme tax and subsidy is ambiguous.

In practice, I have been discussing what would be a micro-tax for many products. Consider the motivating example of plastic-packaged beverages which are generally composed of four or five materials. The optimal tax is likely to be in the few-cent range and possibly a fraction of a cent, since the entire packaging cost is at most a few cents per bottle (Economist, 2014). With sticky prices, this increase may not result in a consumer-facing price change (see Note 4). Yet at a production scale of a half trillion bottles globally per year, the tax results in an incentive for firms and industry to seek innovation. Given the scale of the issue that this framework seeks to address, I next discuss a few complications that may arise in its use. Any policymaker using this approach almost certainly would not intend to worsen the impact of the problem at hand by failing to consider secondary effects.

Note 4 – When Firm Behaviour is NEP-Consistent

It has been suggested that I include a theory of firm behaviour that connects the chapter's model to the NEP framework. Such a model follows; however, it is immediate from existing research.

Suppose firms sell a differentiated product, the industry is defined by sufficient demand substitutability, and firms compete by setting prices. Hotelling, Salop, and Spence (1976), and Dixit and Stiglitz (1977) provide relevant frameworks. I include that output prices as well as input contracts are sufficiently sticky that a penalty must be paid to change prices, $\varepsilon|p_{i,t} \neq p_{i,t-1}|$. Let the initial period ($t = 0$) price for firm *i* be $\underset{p_{i,0}}{\operatorname{argmax}} \pi_{i,0}(p_{i,0}, E[\boldsymbol{p}_{j\neq i,0}])$ given *i*'s expectations of prices for all other firms, $E[\boldsymbol{p}_{j\neq i,0}]$. For all other periods, firm $i$ sets price according to $\underset{p_{i,t}}{\operatorname{argmax}}(\pi_{i,t}(p_{i,t}, E[\boldsymbol{p}_{j\neq i,t}]) - \varepsilon|p_{i,t} \neq p_{i,t-1}|)$. A NEP-compliant tax and subsidy works within the bounds where $\varepsilon|p_{i,t} \neq p_{i,t-1}| \geq \pi_{i,t}(p_{i,t}, E[\boldsymbol{p}_{j\neq i,t}], \tau_P^*) - \pi_{i,t-1}(p_{i,t-1}, E[\boldsymbol{p}_{j\neq i,t-1}])$. Firms might still oppose a NEP-sized material tax on the grounds of reducing profitability but find it insufficiently burdensome to adjust prices and consumers are unaffected.

### *2.4.3 Some Complications and Cautions*

Because of the variety of materials involved and the complexity of recycling processes, several complications may arise in implementing the outlined framework. We might divide the complications into modelling and implementation constraints, but the distinction hardly matters in practice.

First, standard applications of the MAC-MD framework generally have the quality that all units of emissions can be assumed the same e.g., each ton of carbon dioxide emissions is uniform. When a unit of emissions is of uniform quality, the order of what unit of emissions is plotted against a unit of damages and abatement does not matter. But plotting $MAC_P(N)$ against $MAC_R(N)$ warrants caution. Consider the $\bar{N}^{th}$ material in any of the preceding figures. The implicit assumption is that the change that is the least costly for a producer to make is also the costliest for a recycler. This assumption is almost certainly a special case. If using the producer's marginal abatement cost curve as the base, the recycler's marginal abatement cost curve may not be upward sloping, and thus, there may not be an intersection of the two curves and, therefore, no equalisation of the optimal tax and subsidy. The social objective in Equation (10) still holds, however, and so for any $N^{th}$ material, either a product change or recycling processing is preferable.[50]

[50] Or at least not less preferable in a tie.

Second, $MAC_P(N)$ and $MAC_R(N)$ may not be feasible over all $N$. That is, some reductions in the material may not be possible for a producer, and some materials might not be separable and recyclable. For example, the composition of crisp packets is of many layers of plastic that cannot be separated by any practical means.[51] In other cases, specific materials are necessary for one part of a product while infeasible in others (e.g., soft silicone seals in a necessarily rigid product). Additionally, as the nature of product materials is discrete, $MAC_P(N) = MAC_R(N)$ might not exist. Equation (10) still holds, but the optimal tax and subsidy may not match.

A concern in a standard MAC-MD framework is that $MAC_P(N)$ or $MAC_R(N)$ may be strictly higher. That is, it might always be cheaper for a recycler or producer to act. If it is always cheaper for recyclers to process up to $\overline{N}$, collecting a producer tax can still finance recycling. But setting $\tau_P^*(N^*) = s_R^*(N^*)$ – the interior-optimal policy – is not helpful. Instead, setting the tax to balance subsidisation in total is sufficient and still fosters innovation. Alternatively, if it is always cheaper for a producer to reduce material variety, zero matching subsidy would be paid out, but its availability might still incentivise innovation as a prize for developing more efficient recycling methods.

Fourth, the materials that producers choose matter. An assumption has been that all materials are in some way recyclable or have a recyclable alternative, but surely there are exceptions. A less extreme perspective is that materials have different recycled value. Then, which materials producers choose to replace in a product impacts $MAC_P(N)$ in more ways than ease of separability and processing. A producer might reduce the number of materials in a product, but by eliminating those with the highest recycling value. A product that was at least partially profitable to recycle becomes a loss without subsidies, and the policymaker has made recyclers dependent on public support rather than encouraging innovation. Different materials might also have different toxicities, lifespans, and environmental impacts (though we might expect a more toxic material e.g., fissile waste, to be more costly to recycle). For those items lost into the environment despite our best efforts, a reduction in material types could then increase or decrease environmental damages.[52] Yet, making a

[51] There have been efforts to recycle crisp packets that involve special collection and pelletisation of the plastic for generally lower grade uses. Smithers (2019) reports the low success rate of a crisp manufacturer's collection program – roughly 0.01-per cent of the manufacturer's current production volume is being recovered through their effort which includes thousands of collection points and a mail-in option.

[52] Yet it is likely still preferable to have, say, a bottle made from plastic rather than glass which has a lower environmental impact after it is lost into the environment. Gray (2018) notes that a glass soft drink bottle costs about 1-cent more to make than a PET bottle, but due to its considerably heavier weight, a glass bottle results in substantially higher emissions during transportation (Gray, 2018).

product fully recyclable might increase the probability that users attempt to recycle it because they believe it will be recycled rather than partially discarded during processing.

Fifth, dynamic and iterative efficiency requires monitoring. Just as firms have been accused of ineffectually, even cynically using carbon offsets (Song, 2019), whether a product has become more recyclable requires review. And as with carbon offset operations, recycling operations often occur in less regulated countries where oversight may be costly. However, checking whether a product has become more recyclable can occur at the production facility gate or in the market in consultation with experts.

Sixth, dynamic efficiency and iterative efficiency are competing incentives. Producers may choose between pursuing dynamic efficiency – reductions in their abatement costs, and iterative efficiency – reducing recycler abatement costs and awaiting revaluation. An efficient, timely revaluation process improves the position of iterative efficiency. However, any combination of the two sorts of efficiency improvements are beneficial, and I assume producers will choose efficiently.

Seventh, producers and recyclers might seek to freeride by pushing the matter onto households. Single versus multi-stream recycling is in effect a debate over whether the gains from pushing recyclable sorting onto consumers is sufficient since sorting at collection points is remains inevitable. I also note that most plastic bottle recycling programmes prefer that consumers only deposit the bottle. They communicate this preference through the design of rebate programmes that only pay for the bottle, not whether consumers return the cap for which no rebate price is offered. It is likely not socially negative to have consumers involved in the recycling process, but it is uncertain whether doing so results in a net benefit to society.

Finally, I discuss the framework on a product scale. But this might only make sense if either a product so regulated is separable from others in the recycling stream or is applied to a sufficiently large variety of products. It might not be sufficiently beneficial if a programme improves the recyclability of some items, but in general waste remains too contaminated to sort and recycle profitably. I next conclude the paper by discussing it applicability.

## 2.5 Discussion and Conclusion

In this paper, I discuss a regulatory framework to improve product recyclability. I assume, as the literature suggests, that composite waste items pose an obstacle to recycling because they are costly to separate. I then derive a framework of a paired tax and subsidy on the count of materials in items to ensure there is a profit incentive for waste to be entirely recycled. The concept is simple – when the incentives are set properly, either the recycler or producer accepts accountability for each component in a waste item. I then observe that there are practical obstacles to setting the paired tax-subsidy well, but trial and error are likely to be effective. I also outline the possibility of iterative efficiency – producers would have an incentive to coordinate with recyclers to design new versions of products that are more recyclable. I have not, however, addressed creating a mechanism that dynamically tracks and automatically adjusts to iteratively changing optimal tax and subsidy levels. There is no obvious way to create this. It would likely need to depend on a process that gathers and merges confidential production and recycling information and then uses it in a decision process to arrive at an optimal price without excessive state intervention. This is an avenue that requires further research.

Unfortunately, when trying to address an issue as intractable as achieving exceptionally high consumer waste recycling rates, one policy approach is not sufficient to address all issues. In an appendix I outline how a Vickrey auction can be used to encourage greater waste stream homogeneity (*A2.1 Improving Recyclability Through Standardisation Auctions*). But we might also consider whether the vast array of plastics and other materials at our disposal are even necessary. For example, across a wide range of household products, the choice to use HDPE versus PP by producers appears random as these materials are perfect substitutes or nearly so. Among the seven most common plastics and dozens of less used ones, I speculate the benefit to differentiation is negligible in many applications. Committing to less plastic types across all industry and society could have value by increasing recycling efficiency and thus reducing waste. Like the two mechanisms I propose, it increases recyclability potential because it decreases the cost of recycling and increases the value of recycled materials by increasing their purity. Of course, firms producing products from the plastic to be eliminated would argue against the value of differentiation and may require compensation to adapt. To choose among plastics to phase out, we should compare the expected damages if materials are still lost into the environment, capacity for repeated recycling, and lifespan of the material when making such far-reaching choices. Clearly there are several avenues for further research on recyclability.

In this as in many proposals related to the environment, the issue of existing property rights almost certainly poses a substantial obstacle to implementation. Within many legal systems, firms would have the standing to oppose these schemes on the basis that the required changes dilute the value of intellectual property licenses granted to them. However, suppose the alternative is that the polluter must pay the full externality cost when they do not enter the scheme. That is, suppose they are offered the alternative of covering the full cost and responsibility of ensuring their product is recycled – the EPR approach, but also, critically, the cost when their defection interferes with the recycling of others. Firms may then choose their efficient solution – relinquish a license which is resulting in an increasingly concerning externality, or truly and fully internalise the externality. In either case, they address the underlying problem to society's benefit. But being required to make such a choice, too, will almost certainly be opposed by powerful interested parties.

Substantial progress is being made in recycling rates in some nations. Recycler processes have improved, and producers have, in some cases, become interested due to material cost savings and public image. However, most materials are not recycled and instead lost into the natural environment where they cause damages over decades that are difficult to measure but certainly exceptionally large. As society seeks to recycle a larger share of waste and move toward a circular economy organisation, more nuanced approaches are needed. I provide a basis for some approaches, but practical field evaluations remain necessary.

# 3. A Review of How U.S. Coal Power Plants Used Technology to Reduce Sulphur Dioxide Emissions – Lessons for the Next Energy Transition

*The objective of this paper is to advise current and future energy transition advocates on the workings of the last great transition to occur in the U.S. energy sector – a substantial reduction in sulphur dioxide emissions. This is not a new area of study, but one that I examine carefully for additional insights. I review how some U.S. coal power plants used technology to reduce sulphur dioxide emissions. To do so, I use historical coal and natural gas purchase decisions at power plants and find power plants with sulphur dioxide abatement technologies demand dirtier fuel. This results in a divergence in domestic demand by abatement technology which is robust to how the problem is approached. Along the way, I develop a couple new analytical methods, derive new fuel price demand elasticity estimates for the U.S. energy sector, and put prices on some of the incentives involved.*

Keywords: energy, power plants, almost perfect substitutes, inter-fuel substitution, dynamic linear logit, regression discontinuity, hedonic price model.

JEL codes: E13, E23, Q38, Q41, Q48, Q52, Q53

## 3.1 Introduction

This paper adds to the literature exploring reductions in sulphur dioxide emissions from coal power plants in the United States. As new energy transitions are underway – from coal and fossil fuel-based power altogether – it seems an appropriate time to take another look at the last great energy transition. The approach taken is to review the supply side of sulphur dioxide reductions – analysing fuel purchase decisions between high sulphur coal and cleaner alternatives – and how abatement technology decisions impact purchases. This generally occurs at the power plant level, but also sometimes within power plants and at the national, aggregate scale. I restrict the main analysis to the domestic market as this remains the highest binding level of regulatory reach. To achieve the objectives of the study, a substantial set of econometric techniques are developed and then implemented on a large and specifically developed dataset. However, in the end the objective is not historical reflection, but rather to develop lessons, frameworks, and tools for application in current and future energy transitions.

Undertaking this study is obviously an exercise in searching for new insights along a well-trodden path. So, I first present a literature review focussed on aspects of the U.S. energy sector. This includes the history of efforts to address sulphur dioxide emissions – an alarming issue in its time, research on abatement technology and fuel selections, and relevant empirical techniques. I then visually present the scale of sulphur dioxide reductions nationally, followed by evidence that coal power plants with flue gas desulphurization (FGD)-type abatement technologies continue to consume higher sulphur coal while their non-FGD counterparts reduce the sulphur content of their coal purchases. I explore these competing trends in detail – at the power plant level, within power plants at the boiler level, and using causal inference techniques around the point of FGD installation within the available timeframe of data. The data itself is both new and old – including historical figures that have been well-explored, and newer, more diverse data resulting from more recent enhanced reporting requirements on power plants by the Energy Information Administration (EIA). Across the data, scope, and methodology employed in this paper the message is quite clear – power plants that use FGD systems then consume dirtier fuels more cleanly. This potentially leaves less dirty fuel available for other uses.

I also explore whether power plants are switching coal sources or purchasing from the same mines which could indicate changing pre-treatment regimens. As noted, I find that the difference in sulphur content of incoming coal to power plants is substantially higher for FGD equipped power plants and that this change often occurs with FGD installation.

This generally results in power plants switching to a new set of higher sulphur coal suppliers. When power plants install FGD systems is also important, and it appears that firms do not necessarily jump into using such costly, new technologies following regulation as alternatives exist. Rather, implementation across the sector occurs gradually which is more fitting with the economic calculus of firms which are balancing several cost considerations.

I then develop a simplified model of the power plant-level decision between low and high polluting input alternatives which are roughly perfect substitutes. The model begins with a regulation in place mandating emission reductions, then allows the availability of an abatement technology which enables cleaner consumption – less emissions – from dirty inputs. I develop and implement a new method to test the perfect substitutes hypothesis. I then modify existing methods to derive the national, net effect on energy sector fuel demand in the form of own- and cross-price elasticities separated by whether power plants used FGD systems. Another implication of the model is that there are at least four variable abatement costs relevant to FGD selection and usage. I then derive estimates of the price on the two types that are less studied in the literature. This puts prices on some of the incentives involved to inform what scale of intervention might be required to increase the speed of an energy transition while maintaining firm choice.

The result of this research is an expansive but not exceedingly complex study. The focus is on the impact and incentives surrounding abatement technology decisions related to sulphur and coal demand in the U.S. energy sector. To maintain clarity, I sometimes intersperse the modest theory in this paper with the relevant empirics which attempt to derive clear support from a messy and complex sector.

To retain control over the scope of the investigation, I have chosen a set of related questions to explore: Do firms in the U.S. energy sector that adopt abatement technologies demand dirtier fuels than those that do not? Does this decision only occur at the power plant level, or does it extend to boiler fuel choices when not all boilers at a power plant have FGD systems installed? With the installation of abatement technologies, does the sulphur content of fuel change at power plants? And are the source mines for a power plant somehow retained when an FGD system is installed? Resulting from abatement technology choice, do interfuel substitution elasticities differ for the shares of the market with and without abatement technology? Finally, what type and scale of variable abatement costs occur with FGD systems, and should we consider their manipulation to be realistic options in support of domestic environmental objectives? I now proceed with a literature review to place this research in relation to the substantial body of work that precedes it.

### 3.2 Literature Review

This literature review covers some of the history and aspects of the Clean Air Act (CAA), Clean Air Act Amendments (CAAA), Acid Rain Program (ARP). The focus is on the impact of regulatory as well as preceding firm decisions on fuel and abatement technology choices. It also notes research relevant to the estimation methodologies in this paper exploring abatement system incentives and power plant fuel switching activities.

A significant body of research explores the response of power plants to regulatory changes, particularly around sulphur dioxide abatement efforts mandated by the Clean Air Act (CAA). One reason for this is that we hope to learn lessons that will be valuable in the next great energy transition of decarbonising the energy sector. So, while the current paper studies changes related to sulphur dioxide emissions, the subject may have wider interest as similar regulatory methods may be applicable to carbon dioxide emissions as well. Metcalf and Weisbach (2009), for one, note that a properly functioning tax related to carbon capture that specifically targets U.S. power plants could lead to the capture of 80- to 90-per cent of emissions. Importantly, they argue for a tax on power plants – as has been done with sulphur dioxide regulations – as the simplest regulatory approach as it would involve taxing a few thousand entities rather than a consumer-facing programme which would need to track the emissions of several million consumers. Additionally, it is important to be reminded that regulatory changes do not happen in isolation. For instance, Muller (2018) defines GDP less environmental damages as a measure of augmented GDP and subsequently an augmented measure of environmentally adjusted value added as EVA growth. They find that the thirty U.S. states that installed FGD systems in their period of interest – 2005 to 2011 – had augmented growth rates on average 0.12-per cent higher than in states that did not install FGD systems. This is fitting with EIA (2011) which notes differences in nett sulphur dioxide emissions between FGD and non-FGD equipped power plants under the same regulations. Muller’s augmented GPD measure resulted in FGD systems having a benefit-cost ratio of 20:1 to 53:1, depending on state. Such substantial returns were due to the substantial public benefits and comparatively minor firm costs involved with FGD systems. Such returns will hopefully eventually be the case with carbon capture technologies as well.

But to the purpose of this review, the regulatory environment around sulphur dioxide emissions in the U.S. is a rather complex set of sometimes competing regulations. Ellerman (2003) provides a very brief overview of the federal CAA in regard to sulphur dioxide emissions starting in 1990:

> "The acid rain provisions of the 1990 Clean Air Act Amendments, included in Title IV, required fossil-fuel-fired electricity generating units to reduce sulfur dioxide ($SO_2$) emissions by 50% in two phases. In the first, known as Phase I and extending from 1995 through 1999, generating units of 100 $MW_e$ of capacity and larger, having an $SO_2$ emission rate in 1985 of 2.5 lbs. per million Btu (#/mmBtu) or higher, were required to take a first step and to reduce $SO_2$ emissions to an average of 2.5 #/mmBtu during these transitional years. Phase II, which began in 2000 and continues indefinitely, expanded the scope of the program by including all fossil-fuel-fired generating units greater than 25 $MW_e$ and increased its stringency by requiring affected units to reduce emissions to an average emission rate that would be approximately 1.2 #/mmBtu at average annual heat or Btu input in 1985-87, and that would be proportionately lower for increased total fossil-fuel fired heat input. The nation-wide Phase II cap on SO2 emissions is 8.9 million tons, which is approximately the product of total baseline (average 1985-87) heat input and the emission rate target of 1.2 #/mmBtu."

Ellerman also notes that, because the units that had to comply with sulphur dioxide reductions under Phase I were the largest emitters, the bulk of emission reductions were the result of changes at Phase I power plants. These were often in the Midwest, and so nearly 75-per cent of early sulphur dioxide emission reductions took place in the region – larger in share than the region's energy production. They also note that sulphur dioxide allowance costs are generally smaller than other start-up costs for new power plants, such as permitting and siting related costs. So, for power plants under Phase II – generally newer and cleaner – permit prices are a smaller consideration. Compliance costs are also substantially lower than initially anticipated due to exogenous factors such as rail prices as well as improvements in FGD technologies. Ellerman's line of research then naturally lead to Ellerman and Dubroeucq (2006) where they ask whether it is better to modernize – "clean up" – existing power plants or replace them with newer units. From 1985 through 2002 they find that the clean up of old, existing power plants made the largest contribution to sulphur dioxide reductions rather than replacement by clean (in terms of sulphur dioxide) natural gas power plants, particularly since the introduction of the ARP's cap-and-trade program. What newer natural gas power plants have instead done is reduce the cost of compliance over the period. That is, as the emissions cap hadn't changed, the quantity of abatement needed among the remaining high sulphur coal (HSC)-consuming power plants is reduced when natural gas power plants are used.

Schmalensee and Stavins (2013) note that earlier railroad deregulation – the Staggers Rail Act of 1980 in particular – contributed substantially to the lower-than-expected total costs of CAAA/ARP compliance that have been realised. This is due to Powder River Basin (PRB) coal becoming affordable for many Midwest power plants.[53] They also note that public benefits from sulphur dioxide abatement have been substantially greater than expected due to reductions in small particulate matter in downwind communities. Schmalensee and Stavins also discuss how conservative politicians now oppose cap-and-trade approaches even though their predecessors championed them. As a result of modern opposition and subsequent judicial efforts, the sulphur dioxide trading markets have effectively collapsed, and similar efforts will likely face greater opposition. Schmalensee and Stavins (2019) then provides an overview of adjustments to the CAA over the last 50 years with a focus on the success of the various policy instruments chosen. They then note that the CAA, once dynamic, has increased in complexity over the years and has become impossible to update since becoming a partisan political issue. Subsequently, they argue that implementing a cap-and-trade approach to address climate change will likely be impossible in the U.S. as long as the current political environment remains. Important to this research, they then argue for practicality in new regulatory approaches and research, for example "giving greater attention to suboptimal, second-best designs".

Several other papers have studied components of the sulphur dioxide regulatory environment. Bellas (1998) studies incentives to improve FGD performance in the early years of regulation through 1992. They suggest that direct regulation on emissions levels provides less incentive to innovate. Bellas finds that costs of FGD systems were generally constant until the end of their period of interest when regulatory incentives began to change. At that point, regulation switched to a permit market, cap-and-trade system perhaps leading to more innovation. However, a decade later Bellas and Lange (2008) find that the cap-and-trade system did not lead to more efficient, less costly FGD systems on its own. Rather, market deregulation occurring in the period was necessary to allow innovations to occur and be implemented. They find the combination of incentives and leeway to innovate

[53] The PRB is an area of Montana and Wyoming of substantial U.S. energy policy interest due to vast, high-quality, low sulphur coal reserves with low extraction costs.

resulted in substantial (40-per cent) reductions in the parasitic load of FGD systems. Arimura (2002) also studies Phase I of the CAA's Acid Rain Program (ARP) which began in 1995. Phase I's implementation generally fell under the direction of public utility commissions (PUCs) – state-level regulatory bodies. PUCs often initially allowed cost recovery, for example by surcharges onto customers for compliance with the CAA. This incentivises continued high sulphur coal (HSC) usage at power plants in states with HSC mines. Arimura also suggests that uncertainty over PUC regulations lead to more fuel switching and blending than allowance purchasing. It is important to note, however, that the study was conducted given the FGD equipment installed – after the installation decision was already made – so does not include switching to FGD-connected production.

A confounding policy change to note is that deregulation of the energy sector occurred in many U.S. states during the early years of the CAA's ARP (deregulation was particularly concentrated in 1995 through 2002). Borenstein and Bushnell (2015) find that deregulation, from vertical monopolies/investor-owned utilities (IOUs) operating under cost-of-service regulation, to nonutility generators and small producers selling to resellers, did not result in expected price drops for customers. Rather, this resulted in rent shifting and prices now based on retail price instead of cost which turned out to be similar. Borenstein and Bushnell instead find that any rate changes were driven by generation technology advances and natural gas price fluctuations. Fowlie (2010) looks at differences in technological investment activities at power plants to reduce smog pollutions. Critically, they compare power plants in states where the energy sector underwent deregulation/restructuring versus those where PUCs remained in control. Fowlie finds that deregulated firms spend significantly less on capital intensive abatement technologies. Bushnell and Wolfram (2005) also find that fuel efficiency improvements were modest after deregulation, restructuring, and public divestiture. They note that both divested and non-divested power plants showed similar fuel efficiency improvements. They make the case that ownership itself has little impact on performance, but rather that changes in incentives drive change – not who owns a power plant. Chan, Fell, Lange, and Li (2017) look at restructuring from 1991 through 2005, comparing IOUs in states with and without restructuring. They find a 1.4-per cent fuel efficiency improvement, 8-per cent decrease in cost per heat unit, and also a lower capacity factor – the percentage of time a power plant is operating at 100-per cent capacity. They find that the nett effect was a 15-per cent reduction in operating expense as well as a reduction in emissions of around 7.5-per cent.

Naturally, the impact of new regulations as well as restructuring and deregulation extended beyond power plants. Di Maria, Lange, and Lazarova (2018) study the energy sector's restructuring and find that it increases uncertainty about cost recovery (compared to PUCs allowing cost recovery, etc.). This results in coal purchase contracts with more emphasis and rigidity on price adjustment mechanisms. Basically, they find that power plants try to offset increased downstream uncertainty by reducing upstream risk – shifting risk upstream. The upstream result is a greater emphasis on efficiency and productivity at coal mines contracting with restructured plants – about a 17-per cent productivity improvement. However, the authors note that transaction costs related to contracting may also increase. One of the methods written into these contracts are automatic fuel adjustment mechanisms (FAMs) which were previously discussed in Gollop and Karlson (1978). These allow passing through changes in fuel costs only. Gollop and Karlson note that allowing fuel cost passthrough may reduce efficiency as power plants as their operators would more often choose to adapt to changing conditions through fuel changes rather than other inputs whose cost is not automatically, contractually passed on to other parties. Related to CAA implementation, Douglas and Wiggins (2015) find that the ARP led to mine closures or reduced output across Appalachia and the Illinois coal basin which are sources of HSC. Specifically, they find a negative elasticity of coal mine output to sulphur allowance prices for mines producing above the 77$^{th}$ percentile in sulphur content of coal – those mines producing coal with the most polluting potential in terms of sulphur dioxide emissions. Of additional relevance to this paper, they also note that the regulators and legislators of HSC-producing states implemented policies to encourage the installation of FGD systems at power plants specifically to encourage the purchase of more local HSC.

Kneifel (2008) studies the impact of fuel contracts on compliance costs related to the ARP's cap-and-trade program. They find that fuel contract lock-in is an additional source of compliance costs as they restrict a power plant's response to regulatory changes. They also note that contracts increase the incentive to install FGD systems at plants that have traditionally purchased HSC, and as a result, would explain why some boilers installed FGD systems when Phase I of the ARP began. They also suggest that existing contracts for LSC reduce the incentive to install FGD systems. Related to my research, they also argue that a power plant-level decision model is more appropriate than a boiler/generator-level one. Among the reasons that the models would differ is that, when using a cost minimisation objective, a power plant-level model allows choosing between boilers of different designs and fuel usage. This would also imply that a power plant operator decides whether to install an FGD system based on the characteristics of all boilers/generators at a power plant, not just the one(s) that an FGD system will be integrated with.

Jha (2015) argues that regulators are less willing to pass through sudden, higher coal purchase costs to consumers resulting in more of power plant input purchases via long-term contracts. These long-term contracts are generally more expensive – roughly 3-per cent higher for procurement and storage – than the minimum of spot prices available to power plants at a given moment. That is, they find that firms under regulation are willing to in effect pay for lower price variance. They find that the more variance there is in spot prices, the longer the duration and larger the quantity stipulated in contracts signed are. They find that power plants will on average trade a 0.22-per cent increase in mean procurement costs for a 10-per cent reduction in the variance of costs. Yet another study on the impact on contracting, Lange (2012) notes that before the ARP, contracts didn't generally differ on price within the contracted sulphur dioxide content bounds required for coal deliveries. But after the ARP's implementation, contracted coal prices are generally stipulated to be reduced for coal approaching the contracted sulphur upper bound – so now coal prices became responsive to sulphur content changes within contracts.

Kozhevnikova and Lange (2009) find that increasing the number of alternative input possibilities results in reduced contract durations. They argue that reforms reducing rail rates, environmental regulation becoming more flexible through market-based approaches, and deregulation of electricity markets increased options (as would FGD installations). Alternatively, larger quantity contracts or being located very near a mine increases contract duration. Kosnik and Lange (2011) analyse long-term contracts for coal delivery and the "shock" of implementation of the 1990 clean air act amendment. However, they argue that previous research that considered inefficiencies caused by being stuck in long term contracts is incorrect. Rather, they claim that many – even most – contracts were sufficiently flexible that when the CAAA passed, power plant operators and their supplying mines were able to renegotiate. Additionally, Kosnik and Lange note that with flexible contracts, firms have less need to renegotiate when an exogenous shock occurs. The flexibility of contracts is specifically measured in their study by the flexibility of contract price adjustment mechanisms, number of years to expiration, quantities under contract, and the distance between the parties involved.

It is well known that one of the most substantial innovations within the CAAA is the ARP's cap-and-trade system of tradeable permits allowing the emission of sulphur dioxide from power plants. Considine and Larson (2006) provide another overview of the U.S. cap-and-trade system. They note that considerable substitution between emissions, fuels, labor, and capital occurs at power plants. In the short run – with fixed permits and capital, changes in lower sulphur coal (LSC) price have the largest impact on input decisions. In general, they also find that changes in fuel prices are more important than changes in emission permit prices in both the short and long-run in regard to factor substitution. However, cap-and-trade programs do generally reduce the cost of meeting environmental goals and reduce risks through enabling flexibility as well as signalling prices for factor use. This is particularly important for basically irreversible capital investments such as in choosing an FGD system. Considine and Larson also do not find a significant elasticity between emissions and permit stocks which makes sense as fuel costs are the primary determinant of production choice.

But adding a twist to the regulatory story, Frey (2013) notes that the power plants regulated under Phase I, those with more stringent state regulations, and those with lower expected installation costs – including newer units owned by firms that have already installed FGD systems (learning by doing) – are more likely to install FGD systems. They also note that sometimes both cap-and-trade/market-based regulations overlapped with command-and-control regulations in such a way as to restrict a state's power plants to a smaller set of possible abatement strategies such as through specific abatement approaches being required. The result is that the most stringent regulation drives the abatement decision, sometimes undermining the flexibility that a cap-and-trade system is meant to provide. Related, Knittel, Metaxoglou, and Trindade (2019) find that power plants which were not in restructured markets remained more responsive to price – having a cross-price elasticity of coal consumption with respect to natural gas prices of 0.28 versus 0.14 for power plants in restructured/deregulated markets. This may also be due to power plants in restructured markets undertaking different investments that locked in systems to specific fuel types or by operating under different production and profit models.

Creamer (2012) uses a cost minimisation model of the abatement decision – that firms will choose a strategy or combination of strategies to comply with environmental regulations. Creamer compares buying tradeable permits for sulphur dioxide emissions, switching to LSC, and adopting abatement technologies such as FGD systems. They note that cost minimisation behaviour seems to hold even with regulatory market distortions and external shocks such as changes in local or state regulations and subsidies, contracting related costs, and railroad influence. Creamer also finds that the installation and operating costs of FGD systems impacts their adoption with a lag (perhaps due to planning period lengths), and that again with a lag, price changes are passed on in electricity prices. They also note a decline in the use of long-term coal purchase contracts with regulation as well as increasing prices and demand for bituminous coal which has a higher heat content per unit as well as per sulphur content. They also find substitutability between LSC usage and FGD installations – that increasing LSC prices lead to more FGD adoption as does increases in FGD efficiency. Yet, they find that FGD systems and the price of cap-and-trade permits for sulphur dioxide are uncorrelated. Yet Insley (2003) analyses the decision to install FGD systems versus purchasing emission allowances and finds that higher permit prices provide more incentive to install FGD systems. Insley also suggests that greater permit volatility leads to a lower probability of FGD installation – the alternative value of an FGD installation is less certain – including the pausing and cancelling of FGD installations that are underway. One reason for this result is that power plants rely more heavily on LSC when prices drop and mix LSC with HSC more often when permit prices drop. Along the cost minimisation theme, Knittel (2002) finds that regulatory programmes linked to power plant performance and fuel cost passthrough are related to greater efficiency improvements at power plants. Importantly, fuel cost incentive programmes must result in some accountability to power plant operators for cost changes – both positive and negative. This again highlights the importance of a cost-minimising decision framework for studying the behaviour of power plants. Hite, Forster, and Rausch (1999) instead study the disposal of FGD by-products, particularly from wet scrubber-type systems in their geographic area of interest of Ohio. At that time, FGD by-products were often disposed of in landfills resulting in a sizeable waste stream. They note that there are substantial incentives – public and private – to finding alternative, perhaps beneficial uses for FGD by-products which have been partly but not entirely realised in the present.

Another popular theme is innovation incentives in sulphur dioxide abatement technologies. Gagelmann and Frondel (2005) note that in the earliest years of the CAA that emission reduction requirements were too lenient to encourage innovation. One of the secondary selling points of the ARP's cap-and-trade programme was that it would encourage innovation, but Gagelmann and Frondel note that no consensus in the literature is reached on whether the programme actually encouraged more innovation than other regulations. Yet, they note that the ARP was profoundly effective as it "ended a decade-long stalemate on acid rain legislation" enabling more stringent regulation, and in turn, more innovation in the 1990's and 2000's than would have been possible under other regulatory approaches. Others such as Spulber (1985) find that emission taxes and tradeable permits are more efficient in an industry allowing the free entry and exit of smaller firms – as in the case of the energy sector after deregulation/restructuring, compared to standards which may result in substantial fixed costs related to technology requirements. Karp and Zhang (2012) review the abatement decision with asymmetric information and strategic behaviour. One result is that they find that more durable capital investments as well as more effective (efficient, less costly) abatement technologies all favour a tax-based policy rather than a quota in efficiency terms. Increasing external investments, then, also result in favouring a tax-based approach. These investments also increase spillovers as in secondary benefits to innovation. Keohane (2003) simulates the decision to install an FGD system under different regulatory regimes – comparing the cap-and-trade based ARP outcome against a hypothetical command-and-control "prescriptive" uniform standard. They conjecture that the command-and-control method would have resulted in about 33-per cent more FGD installations. The FGD systems would have had higher operating costs and been installed at power plants with higher costs of switching to LSC as an alternative. The aggregate result is estimated to be up to a 25-per cent reduction in aggregate compliance costs under the cap-and-trade program.

Taylor, Rubin, and Hounshell (2003) also study the relationship between government policies and innovation in FGD and related technologies. They find that CAA regulations resulted in greater abatement innovations than state sponsored innovation programs would have accomplished alone. They note that the specifics of a regulation also guide innovation efforts toward a specific path. They also find that there are technological spillovers – in this case that abatement innovation efforts have led to efficiency improvements and cost reductions elsewhere within- and between-power plants. Taylor, Rubin, and Hounshell conclude that "the stringency, flexibility, market size, and time allowed to achieve mandated emission reductions" are drivers of the rate of technological innovation in abatement technologies and show this for both sulphur dioxide and nitrogen oxides abatement.

Popp (2003) uses patent data to analyse innovation in FGD design around CAA implementation. Prior to 1990 – when the CAA relied on command-and-control regulation – FGD innovations generally lowered their operating costs rather than increasing system efficiency. These FGD systems met, but didn't generally strive to greatly exceed, the 90-per cent efficiency requirement of the time. However, with a switch to the market-based-approach of the ARP's cap-and-trade programme, FGD innovations then focussed on improving system removal efficiency. (They also note that the rate of innovation decreased after 1990, perhaps since more abatement alternatives to FGD systems became cost effective). Thus, it is important to consider the long-term innovation incentives that may result from a policy. Jaffe, Newell, and Stavins (2004) also study the diffusion of innovations related to pollution abatement. They find that technological advances – in rate and direction – are influenced by both the market and regulation. Subsequently, they argue that market-based incentives will be more effective, in price, efficiency, etc. than command-and-control approaches. Yet, they expect that investments in abatement technologies will be suboptimal due to weak environmental protections and spillover effects in many cases. They then suggest that one strategy is a policy that encourages experimentation – that implementing and judging several approaches by similar standards is a productive method to search for innovations. Their recommendation fits well with the objective of this paper.

Hoel and Karp (2002) continue a long line of research stemming from Weitzman's (1974) analysis of prices versus quantities as regulatory methods in the presence of asymmetric information.[54] They study when environmental damages depend on the stock of a pollutant rather than a flow and again assume asymmetric information between regulators and polluters on abatement costs. As before, a steeper marginal abatement cost curve favours the use of taxes. Hoel and Karp also find that taxes become preferable as the discount rate or the decay rate of the stock increases or the decision period of those involved decreases. This is likely the case for sulphur dioxide emissions, and the authors argue that taxes would also be preferable for the regulation of greenhouse gas emissions. Nordhaus (2007) also argues for tax-based (and presumably subsidy-based) approaches rather than quantity approaches to environmental regulations. They argue that tax-based approaches allow an easier integration of the economic costs involved into a firm's calculations and that taxes are particularly preferable when uncertainty and learning exist. Nordhaus also argues that quantity limits can lead to market volatility and that a tax can be used to offset losses from removing other distortionary taxes and encourages less rent-seeking. However, Nordhaus also notes some popular and political opposition to taxes as well as distrust of political entities to set taxes effectively and fairly – without loopholes. Yet, Papadis and Tsatsaronis (2020) argue, as many have, that some sort of global carbon tax will almost certainly be

[54] Other continuations of Weitzman's work include Yohe (1977) – regulation of multiple firms or products and intermediate products, Adar and Griffin (1976) – marginal damages uncertainty does not have the same issue and addition of auctions to the comparison, Watson and Ridker (1984) – additional functional forms and an interesting empirical application, and Stavins (1996) – correlated abatement cost and damages uncertainty.

required to address the current pressing issue of climate change but that several barriers to an implementation exist including frequent opposition to tax-based approaches.

In this paper I derive several empirical estimates. Where I estimate variable costs, I assume fuel and abatement technology costs are additively separable into parts, and so estimates fall within the hedonic price model (HPM) methodology of Rosen (1974). Uri (1982) uses a HPM to explore the usage – energy shares – and price of competing fossil fuel inputs and notes the extent of their substitutability. Stanton and Whitehead (1995) also use a HPM to estimate the implicit price of sulphur in coal. Their results suggest a higher price on sulphur in the eastern U.S. but are otherwise comparable to others in the literature and this paper as they suggest the price estimate is roughly the removal costs associated with sulphur in coal. Busse and Keohane (2007), and He and Lee (2016) provide related HPM applications by attempting to discover the specific cost components of coal at U.S. power plants. Busse and Keohane are concerned with policy implications related to PRB coal. He and Lee use privileged data to estimate price discrimination practices by coal mines based on known sulphur by-product production decisions at power plants. I adjust their frameworks to explore the costs passed on to mines related to using marginally dirtier coal in energy production, and through cost incidence (Carver, 1924), roughly estimate the full variable abatement cost involved. One issue is that Gerking and Hamilton (2010) note that changes in regulation sometimes lead to only small substitutions between HSC and LSC even though these inputs should be highly substitutable. The issue, they suggest, is that "spatial monopolists" – railroads in particular – favoured LSC transportation as these generally involved longer distances and higher profits. This may suggest that there would be a delay in when a power plant substitutes from one fuel to another. Yet fuel prices are something that power plants can adjust to over the long run.

Lange and Bellas (2007) also estimate an implicit price of sulphur in coal using contracted coal purchase order data and a HPM methodology. Specifically, they compare the contracts made by firms immediately preceding and after inclusion in Phase I of the ARP. Their result is substantially lower than other estimates and suffers from a smaller sample size but is an interesting approach. Lange and Bellas (2005) study FGD innovation and compliance cost reductions over the long run following the ARP's 1990 implementation. Importantly, the ARP both expanded the market for FGD systems – the number of power plants required to reduce emissions – and increased competition from other emission reduction strategies. They find that both operation and construction costs had a one-time price drop following the 1990 implementation, but they also report that rates of change then followed the preceding pattern. They suggest this result implies that innovation provided for "a great leap forward in cost reductions, but not necessarily subsequent ongoing progress."

Gerking and Hamilton (2008) also study the determinants of greater PRB LSC usage in the 1980's and 1990's. They note, as had Busse and Keohane (2007), that declining rail rates following deregulation as well as decreasing mining costs in the PRB led to increased demand for PRB coal. So, again relative prices appear to drive power plant decision making. Gerking and Hamilton note too that the market power of railroads in setting freight rates for LSC is important – railroads are balancing their market power against competing abatement prices. Ellerman and Montero (1998) are also interested in how declining rail rates in the period 1985 through 1993 lead to higher utilisation of PRB coal across the Midwest energy system and subsequently less sulphur dioxide emissions rather than being reduced by CAA mandates. Additionally, they find that some states enacted stricter state-level sulphur dioxide emission regulations while others enacted legislation to enable the continued use of HSC. Closely related, Rose, Taylor, and Harunuzzaman (1993) study how some states supported the continued use of HSC, sometimes by allowing (sometimes upfront) cost recovery surcharges on customers relates to CAA compliance.

I also explore interfuel substitution at power plants from a slightly new perspective. Others, including Atkinson and Halvorsen (1976) also explore interfuel substitution between coal, oil, and natural gas. They find that power plants behave as if coal and oil are homothetic, but not necessarily for coal and natural gas. This makes sense as coal and oil are more often burned in the same system – often with oil as a start-up fuel for coal-consuming boilers. They also find power plant returns to scale (RTS) that are slightly increasing. Their RTS result is not substantially different from Nerlove (1963), Christensen and Greene (1976), and Bernstein and Parmeter (2019), each finding that intermediate-to-large coal and natural gas power plants operate near-constant returns to scale (CRTS). Bernstein (2020) also estimates the return to scale specifically at natural gas power plants as 0.94. (Bernstein also finds increasing power plant efficiency – as a means of cost reduction – to have occurred with deregulation). Returning to interfuel substitution, Stern (2012) provides a meta-analysis extending to industry beyond the U.S. energy sector. He generally finds demand for competing fuels outside the energy sector to be more elastic than those within the sector.

In this paper I specifically study interfuel substitution based on the literature implementing a dynamic linear logit model as a system of thoroughly defined equations describing a market, here the U.S. energy sector. This necessity comes from the characteristics of energy sector demand – any instrument at this level would almost certainly fail the instrumental variable exclusion restriction because of the pervasive, direct influence of the energy sector on all production and quality of life. To develop the estimation procedure, I rely on Considine and Mount (1984) who derive the basic linear logit model, further developed in Considine (1990), and transformed into a dynamic model in Jones (1995) where its use in analysing interfuel substitution in large markets is demonstrated. EIA (2012) provides guidance on usage with finer time-incremented data, and Steinbuks and Narayanan (2015) show an implementation of the model over a finer geographic scale. While the latter study addresses interfuel elasticities internationally, I apply it to the U.S. to account for market differences between North American Electric Reliability Corporation (NERC) regions.[55] I then uniquely use the method for estimating demand elasticities to compare FGD and non-FGD power plant "markets". I also find it necessary to estimate the impact of abatement technologies on fuel demand as a nett effect repeatedly. Ryan (2012), and Fowlie, Reguant, and Ryan (2016) utilise a log-log estimation of elasticities of demand including a set of covariates, and instrument to address endogenously determined price. They estimate, however, a downstream market from coal demand for energy where instruments are more valid.

Other relevant research is briefly noted here. LaCount, Haeuber, Macy, and Murray (2021) note other features that they believe contributed to the success of the CAAA's "Good Neighbor" provision which prohibit emissions that substantially damage neighbouring state air quality. These include greater transparency through near-continuous emissions monitoring and public access to more data than most programmes, compliance directly related to emissions reduction requirements, and automatic application of penalties – all contributing to high levels of compliance. Löffler, Burandt, Hainsch, and Oei (2019) explore the issue of stranded assets, specifically in regard to the ongoing renewable energy transition. They expect that stranded assets will be substantial, primarily in the coal and natural gas share of the energy sector. They argue that policy implemented with substantial foresight can reduce the amount of stranded assets substantially. In the EU with a 2050 focus year, the amount of stranded assets could be reduced by 75-per cent, perhaps comparably in the U.S. The authors then compare this outcome to a short-sighted stranding of likely 200-billion euros of energy infrastructure by 2035. Raff and Walter (2020) analyse

[55] The role of NERC regions, general structure of U.S. energy production, and some basic definitions are included in the appendix (*A3.2 Glossary of Energy Sector Terminology and U.S. Organizational Chart*).

spillover effects at U.S. power plants due to reduction efforts in at least one of the six pollutants of the CAA's National Ambient Air Quality Standards (NAAQS) following a power plant's nonattainment designation. They find that following a non-attainment designation for sulphur dioxide or carbon monoxide, power plants also significantly decrease emissions of carbon dioxide and nitrogen oxides as these can be jointly emitted and abated pollutants.

In total, the literature suggests sulphur dioxide abatement decisions have an impact on emissions which then influence input decisions at U.S. power plants. However, the market and regulatory environment where this occurs has an incredibly complex history. To limit the scope of this study to something manageable I note that one of the decisions to consider is whether to use FGD systems as part of the power plant's sulphur dioxide abatement approach. This, in turn, appears to be cost-based when the regulatory environment allows choice. So, I next explore the U.S. energy sector transition to a less polluting system in terms of sulphur dioxide. Then, I contribute to themes in the literature by exploring FGD system choice. This leads to putting estimates on some of the prices tied to the performance of FGD systems.

### 3.3 Exploring the U.S. Energy Transition

This section explores the U.S. energy sector transition of declining sulphur dioxide emissions. I begin by contrasting trends in national ambient air sulphur dioxide levels against the sulphur content of coal entering U.S. coal power plants – the primary source of ambient sulphur dioxide levels. Next, I breakdown the sulphur content side of this comparison into that entering power plants with FGD systems versus those without. I then put estimates to the scale of the difference before proceeding to evaluate differences within power plants that have both boilers with and without FGD systems. I then apply a regression discontinuity and kink (RD/RK) design to evaluate whether sulphur content changes with an FGD system installation to evaluate whether the preceding results are based on endogenous factors. I also evaluate the retention of mine sources before versus after FGD installation and the timing of installations. After this fairly expansive exploration, I present a simple model of the fuel purchase decision.

### *3.3.1 Energy Transition Outcomes and Plant Level Heterogeneity*

An exploration of U.S. coal consumption and emission trends suggests that abatement technology adoption leads to the consumption of dirtier fuels but cleanly.[56] In Figure 14, the U.S. trend of ambient air sulphur dioxide concentrations – driven by coal power plant emissions – decreases over the last 35 years, while the average sulphur content of coal entering U.S. power plants first decreases, and then increases. Over the same period, total coal consumption increased 60-per cent from 1980-2005, and then declined to a net 14-per cent increase by 2015 in part due to the shale gas boom (EIA, 2019). Substitution away from coal likely contributed to reduced ambient air concentrations in later years. However, there are at least three factors involved: substitution to cleaner coal, substitution away from coal altogether, and adoption of abatement technologies while consuming dirtier coal. Further scrutiny is necessary.

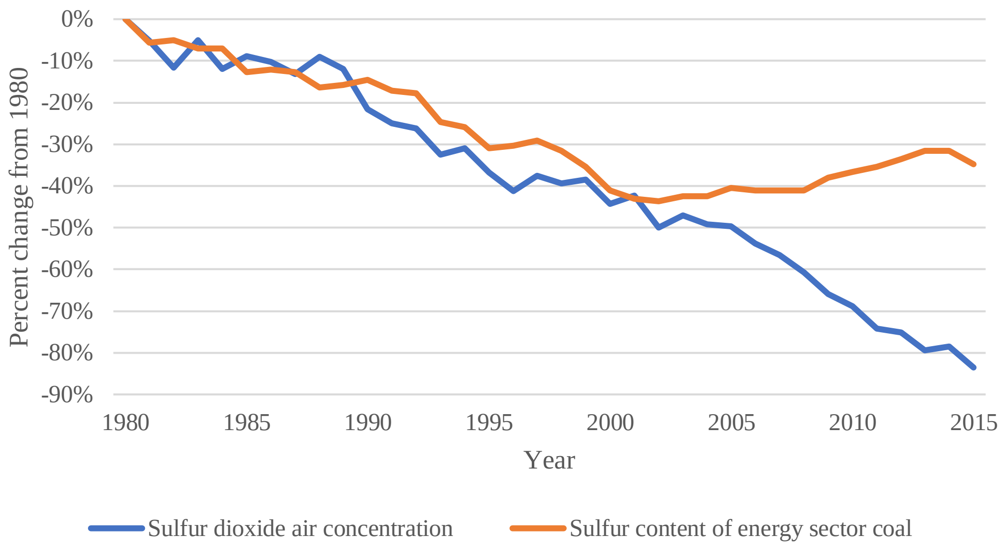


FIGURE 14 SULPHUR DIOXIDE TRENDS IN AIR AND U.S. POWER PLANT COAL

*Notes:* Orange line (top): Data from the U.S. Energy Information Administration (EIA) indicates decreasing sulphur content in coal delivered to U.S. power plants relative to a 1980 baseline of approximately 1.6 pounds of sulphur dioxide content per million BTU's (pounds/mmBTU) of coal. Blue (bottom) line: From 1980 to 2015, U.S. Environmental Protection Agency (EPA) monitoring of sulphur dioxide mean daily concentration across the U.S. indicates a substantial decline from around 155 to 20 ppb $SO_2$ during the period of EPA observation. Reductions closely mirror each other until around 2000, when they sharply diverge. The period is characterised by both substantial abatement technology adoption, and importantly, the rollout of new phases of EPA's Acid Rain Program. From 2000, more power plants were required to limit emissions or buy emissions allowances, substantially increasing the market for tradable permits at a time when technological advancement allowed substantial reductions in power plant emissions through abatement technologies. From 2005, we also see a decline in coal demand, yet by 2015 total coal consumption is still up 14-per cent over 1980.

*Sources*: Author calculations based on data from EIA and EPA (EIA, 2008, 2016c, & 2019; EPA, 2016b).

[56] I define "cleanly" as still meeting environmental regulatory limits.

In Figure 15, I divide the coal power plants in the U.S. energy sector into those with and without FGD systems. Those without FGD systems must rely primarily on lower sulphur coal to meet emissions requirements affordably. Not only are levels different, but industry trends diverge. While FGD power plants maintain roughly historical levels of sulphur content in fuel purchases, non-FGD power plants have sought lower sulphur coal alternatives. There is a drop following the introduction of the Acid Rain Program in 1995, accompanied by a jump in sulphur content at FGD enabled power plants as structural changes in demand – less non-FGD power plants buying higher sulphur coal – precede changes in coal supply. Operators build power plants to match the specification of the coal they expect to use, which is historically locally sourced. However, non-FGD power plants have had to change sources resulting in, for instance, higher exploitation of Wyoming's Powder River Basin low sulphur coal reserves and a subsequent decrease in demand for higher sulphur Appalachian coal.

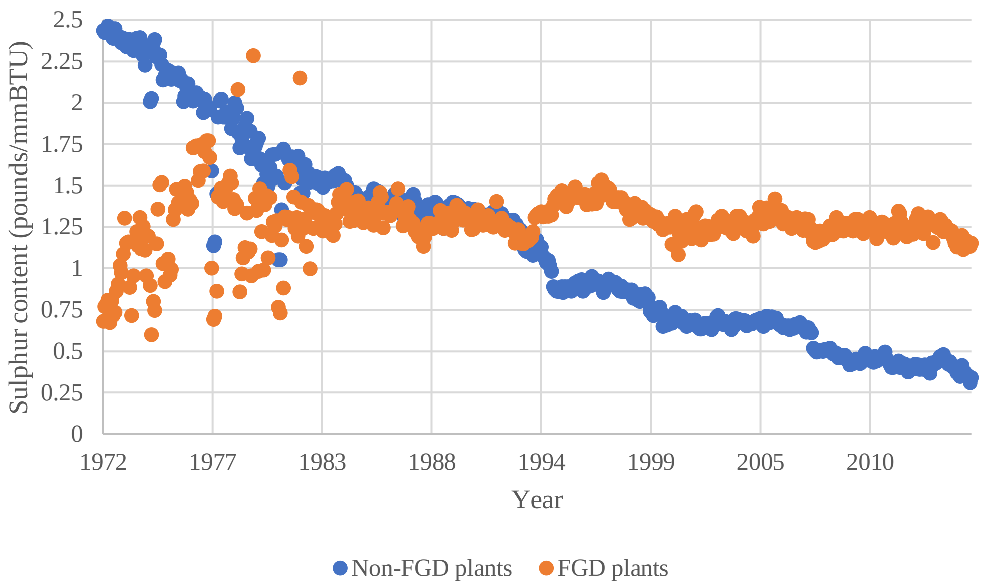


FIGURE 15 COAL SULPHUR CONTENT AS DELIVERED TO U.S. POWER PLANTS

*Notes:* Trend of mean sulphur content in pounds per millions of BTU heat units (pounds/mmBTU) at U.S. power plants using sulphur dioxide abatement technology (initially top line in blue) versus power plants without (initially bottom line in orange). Both the mean sulphur content is substantially different and their trends diverge. The non-FGD share of the sector has had to continually reduce sulphur dioxide in inputs while FGD plants have been able to substitute technology for input reductions. The number of retiring non-FGD plants exceeds that of newer FGD power plants built resulting in differences from Figure 14.

*Sources*: Author calculations based on EIA data (EIA, 2008, 2016c).

But there are many qualities to coal, so I present some of these separated by power plant abatement technology in Figure 16. We observe that coal sulphur content is higher at power plants with FGD equipment. It also appears that FGD installation is endogenous to power plant operating features – specific characteristics of power plants and fuels influence the abatement technology decision. Power plants located near higher sulphur coal sources choose to install abatement technology as the heat content of the fuel is advantageous. Avoided transportation costs also offset abatement technology operating costs and other liabilities from continuing to use high sulphur coal. U.S. power plants with FGD installed then appear to demand higher sulphur coal due to operational and logistical advantages.

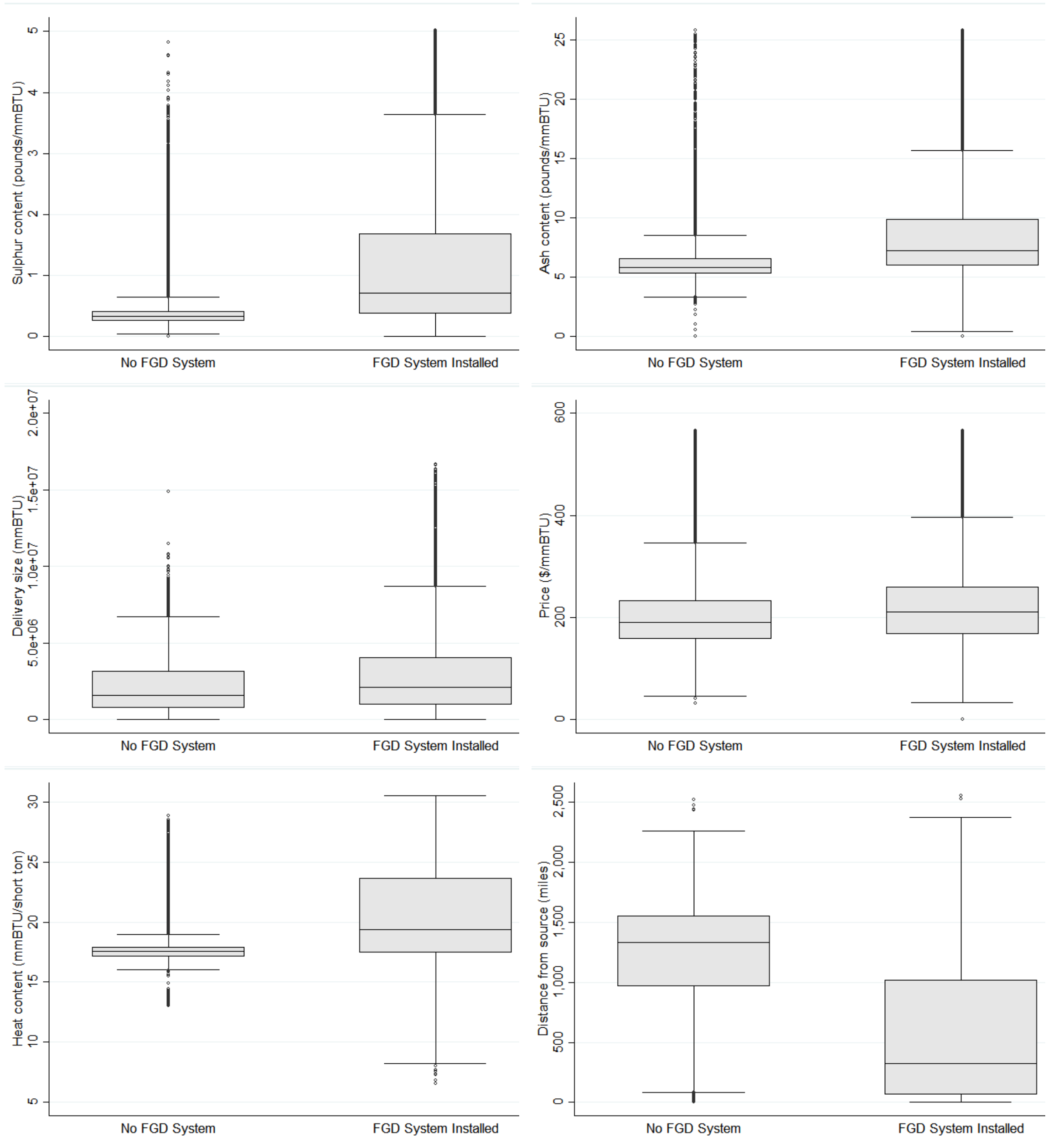


FIGURE 16 COMPARISON OF COAL QUALITIES USED AT FGD AND NON-FGD POWER PLANTS

*Notes:* Coal characteristics, separated by whether coal power plants (N≈300) use FGD technology (=1 if used). For years 2008-2015 from approximately 110,000 coal purchase observations. These years provide more extensive data than preceding ones due to an expansion of power plant reporting requirements. Descriptions and possible interpretations follow:

(top-left) Sulphur content: FGD power plants choose dirtier fuel on average.

(top-right) Ash content: FGD power plants also receive slightly dirtier fuel in ash content as this is correlated with sulphur content (coefficient of 0.20).

(centre-left) Delivery size (a capacity proxy): Power plant orders are about the same size, but the FGD plant set has larger outliers.

(centre-right) Price per million British thermal units (mmBTU) at the gate: Delivered price is about the same on average, but with considerable heterogeneity. We might expect these prices to be about the same in a market equilibrium with differences in operating costs factored in.

(bottom-left) Heat content per short ton: Power plants using higher sulphur fuels are also tapping into higher quality in terms of heat content (correlation coefficient of 0.52)

(bottom-right) Distance from source: FGD power plants avoid substantial transportation costs resulting from adaptation and instead get more heat content per-dollar.

*Sources*: Author calculations based on EIA data (EIA, 2016a, 2016c).

I now derive empirical estimates fitting with the preceding exploration. Because the abatement decision is endogenous to aspects of the production environment that alter production costs, these result in factor collinearity and attempts to estimate primitives will result in more frustration than insight. However, we can estimate the net effect of FGD installation on sulphur demand, conditional on noting that abatement equipment is not randomly assigned. I then contrast an ordinary least squares (OLS)-based estimate against a difference-in-difference (DiD) sort of estimate (Snow, 1856; Ashenfelter, 1978; and Ashenfelter and Card, 1985) to note the level of decision endogeneity.

As in the preceding graphics, I use a large dataset accumulated from U.S. government sources that I combine into a more comprehensive picture of decision making in the U.S. energy sector. I use Energy Information Administration (EIA) data from 2008 through 2015 collected through mandatory reporting requirements at U.S. power plants on forms *Annual Electric Generator Report* (EIA-860) and *Power Plant Operations* (EIA-923), and for years 1972 through 2007 based on *Cost and Quality of Fuels for Electric Plants* (FERC-423).[57] Of the roughly 1.5 million fuel purchase order observations available, nearly 900,000 involve coal purchases which I combine with data on FGD installations at receiving power plants.[58] One matter is the identification of the appropriate measure of fuel sulphur content – I use the metric pounds of sulphur per-million British thermal units (mmBTU) and outline the data preparation steps in appendix *A3.1 Data Preparation*.

I first estimate by OLS the demand for coal sulphur content by whether an FGD system is in use plus a time trend. This estimate is of the net difference in sulphur demand without attempting to discover the complicated relationship between power plants and supply primitives. The coefficient on sulphur dioxide abatement technology installed, $SO_2$Controls in Equation (14), cannot be considered a treatment effect as FGD installation is not by random assignment. Instead, it is a coefficient estimate of differences in sulphur demand at coal power plants that have FGD systems.[59] I use power plant-level purchase data – deliveries, *d*, at a power plant, *p*, in a month, *t*, and estimate the net effect from

$$(14) \quad \ln\left(sulfur_{d,p,t}\right) = \beta_0 + D_1 SO_2Controls_{p,t} + \beta_y Yeartrend_t + \epsilon_{d,p,t}$$

57 Data collection via FERC-423 was superseded by EIA-923 in 2008. While the newer reporting requirement provides a greater variety of details, the earlier data is sufficient for much of this analysis.

58 Power plants are included in the dataset if they have a total capacity of one megawatt (MW) or greater and are connected to the grid. Exceptions are power plants in Hawaii and Alaska where all power plants connected to the grid are included. Generators range in capacity from 0.1 MW to 1,440 MW with a mean and median of 58.2 MW and 5.6 MW, respectively.

59 Note individual year or year-month controls versus a continuous trend variable does not qualitatively change the coefficient of interest on $SO_2$Controls, and I report using the simpler annual trend.

The result is a conservative estimate on sulphur feed into boilers with FGD equipment because a power plant may have many separate boilers with and without ties to FGD systems at a large site. It measures a policy-relevant effect, however, of having at least some access to abatement technology capacity and allowing within-plant substitution. The estimate – I find a roughly 40-per cent higher level of sulphur content at FGD-enabled power plants – is the mean difference we observe in Figure 15.

I then separate the estimate into 5-year periods by interacting $SO_2$Controls with period vector $\boldsymbol{\theta}$, while retaining the annual trend,

$$(15) \qquad \ln\left(sulfur_{d,p,t}\right) = \beta_0 + SO_2Controls_{p,t}\boldsymbol{\theta}^T\boldsymbol{D}_{periods} + \beta_y Yeartrend_t + \epsilon_{d,p,t}$$

Due to high retrofit costs and power dynamics between the sector and the state, we observe a long period of adaptation, primarily of non-FGD power plants transitioning to low sulphur coal and higher rates of retirement among aging non-FGD plants. The pre-1980 difference in sulphur demand is around 15-per cent, and more recently, over 50-per cent. As we observe in Figure 15, the divide is driven by the average sulphur demand among remaining non-FGD power plants falling.

Finally, I highlight the endogenous nature of the adoption decision by using power plant fixed effects, $\boldsymbol{\Phi}$, and time fixed effects, $\boldsymbol{T}$, resulting in a quasi-DiD estimate as[60]

$$(16) \qquad \ln\left(sulfur_{d,p,t}\right) = \beta_0 + D_1 SO_2Controls_{p,t} + \boldsymbol{\Phi}_p + \boldsymbol{T}_t + \epsilon_{d,p,t}$$

This specification identifies the effect of sulphur dioxide controls on sulphur demand after accounting for constant power plant-level decisions, e.g. location. I do not suggest a rigorous causal interpretation of the coefficients, but rather the decision's endogenous basis. I find half or so of the effect of FGD installation can instead be attributed to location, e.g. availability of high versus low sulphur coal nearby. Table 4 reports the estimates for Equations (14)-(16) correspondingly in columns (1)-(3).[61]

[60] In effect, the estimation is as a DiD on all power plants recentered on when their $SO_2$ controls are installed. This is because DiD treatments are as if the entire group is treated at the same time. Angrist and Pischke (2009) may be referenced for a proof on the relationship between DiD and fixed effects estimation of this form.

[61] The normality of the fitted residuals in these and all estimates in this paper were found satisfactory based on visual inspections.

TABLE 4— SULPHUR DEMAND BY ABATEMENT TECHNOLOGY

| | Net effect | Separate periods | Plant-level FE |
|---|---|---|---|
| $SO_2$controls[a] | 0.409*** | - | 0.222*** |
| | (0.0628) | - | (0.0518) |
| $SO_2$*(1980 and earlier) | - | 0.157 | - |
| | - | (0.163) | - |
| $SO_2$*(1981 to 1985) | - | 0.305*** | - |
| | - | (0.104) | - |
| $SO_2$*(1986 to 1990) | - | 0.252*** | - |
| | - | (0.0961) | - |
| $SO_2$*(1991 to 1995) | - | 0.360*** | - |
| | - | (0.0835) | - |
| $SO_2$*(1996 to 2000) | - | 0.473*** | - |
| | - | (0.0812) | - |
| $SO_2$*(2001 to 2005) | - | 0.438*** | - |
| | - | (0.0841) | - |
| $SO_2$*(2006 to 2010) | - | 0.478*** | - |
| | - | (0.0640) | - |
| $SO_2$*(2011 to 2015) | - | 0.520*** | - |
| | - | (0.0764) | - |
| Annual trend | -0.0249*** | -0.0265*** | - |
| (1972 base year) | (0.00166) | (0.00177) | - |
| Constant | 0.398*** | 0.424*** | 0.315*** |
| | (0.0396) | (0.0427) | (0.0314) |
| Plant level and year fixed effects | No | No | Yes |
| R-squared | 0.144 | 0.147 | 0.552 |

*Notes:* Based on 835,802 observations from 1972 to 2015. Robust standard errors in parentheses clustered at the power plant level: *** Significant at the 1 per cent level, ** Significant at the 5 per cent level, * Significant at the 10 per cent level.

[a] equals one if FGD present

*Source:* Author calculations based on data from EIA (2008, 2016c).

The empirical exploration need not end at comparing power plants with versus without sulphur dioxide abatement technologies. One question immediately arises – where is the change in fuel composition occurring? Is it at the power plant level, or are there differences within power plants – at the boiler scale? In the next section, I refine the exploration to viewing how fuels differ within power plants where some boilers are connected to FGD systems while other are not.

### *3.3.2 Exploring the Transition at the Boiler Level*

It seems important to discover where fuel decisions are made at power plants – at the refined scale of boilers (drivers of generators), or at the power plant aggregate perhaps containing many boilers? Doing so informs on how asset owners make decisions about pollution strategies. In this case, does it matter whether a technology is regulated at the facility level, or at a more refined scale? To explore this question in the context of the U.S. energy sector, I extract another dataset from the substantial volumes of EIA data relied on in the preceding section. Unfortunately, the time scale is a limited for the sort of data needed to explore boiler-level decisions. Starting in January 2008 through the end of the data prepared for this study, selected US power plants had to provide fuel consumption data at the boiler level on a monthly scale. Boiler level data on a monthly increment is the smallest scale that we can reasonably expect for such a study. I then link the boiler consumption data to information on which boilers within power plants are connected to FGD systems. I focus on the several hundred power plants in the U.S. that have both boilers connected to FGD systems and boilers without. However, the result is quite similar when I add back in data on power plants that have exclusively non-FGD or all-FGD enabled boilers, which is about half of the power plants in the dataset. The result is that we observe differences in the sulphur content of fuel occurring at the boiler level within power plants. Figure 17 provides a graph of mean sulphur content, weighted by fuel quantities, of coal going into boilers with FGD systems versus without FGD systems connected. Clearly, boilers with connected FGD systems are fed substantially higher sulphur coal. And while the time scale of analysis is limited, the difference in sulphur content appears to be widening. I also note that power plants with both sorts of arrangements in place tend to substantially favour FGD-connected systems. In total, the quantity of fuel fed into the FGD-connected boilers is 9.6 times greater than that into their non-FGD boilers. Another implication is that observing power plants at the facility level likely results in conclusions similar to observing at the smaller, boiler scale where data is available for fewer years.

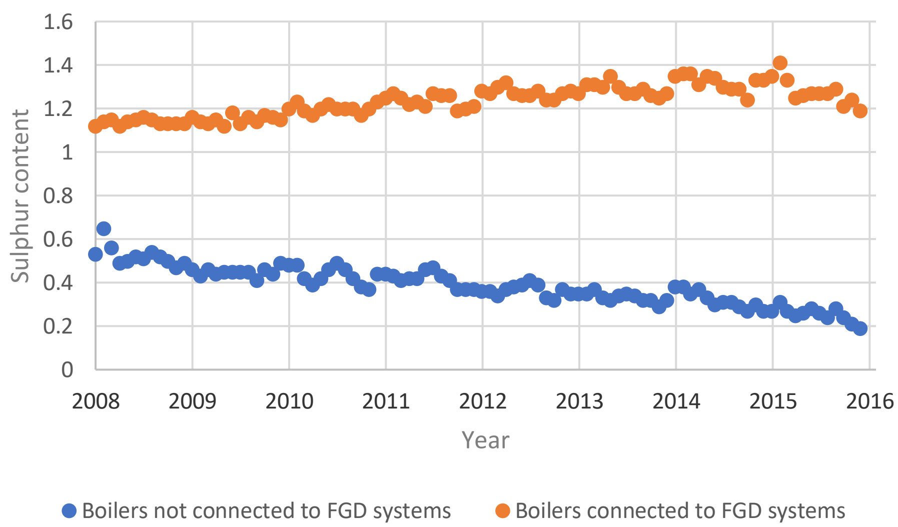


FIGURE 17 COAL SULPHUR CONSUMPTION AT THE BOILER LEVEL AT U.S. POWER PLANTS

*Notes:* Dataset is the U.S. power plants that have both FGD and non-FGD connected boilers within the same facility. The measure in each month is the mean sulphur content in pounds per mmBTU consumed within the relevant group where boiler-level measures are weighted by the quantity consumed. FGD connected boilers also consume substantially more in quantity at these power plants. We observe the sulphur content into FGD connected boilers is about twice that of non-FGD boilers and the difference appears to grow over the period of observation. Note that FGD boilers account for about 9.6 times more coal consumption at these power plants than do non-FGD boilers. The implication is that observing the data at the power plant level results in equivalent conclusions to observing at a smaller scale where data is only available for a subset of years.

*Sources*: Author calculations based on EIA data (2016a, 2016c).

We can then compare sulphur content differences within mixed power plants, to the difference between power plants specialising in either FGD or non-FGD boilers. Figure 18 reports the difference between specialising power plants which is quite similar in both scale and trend over time. The reason for this consistency is that power plant emission rules generally apply at the stack – what we can equate to the boiler level in this analysis. So, there is not an incentive to regulate differently at power plants that specialise versus mix.

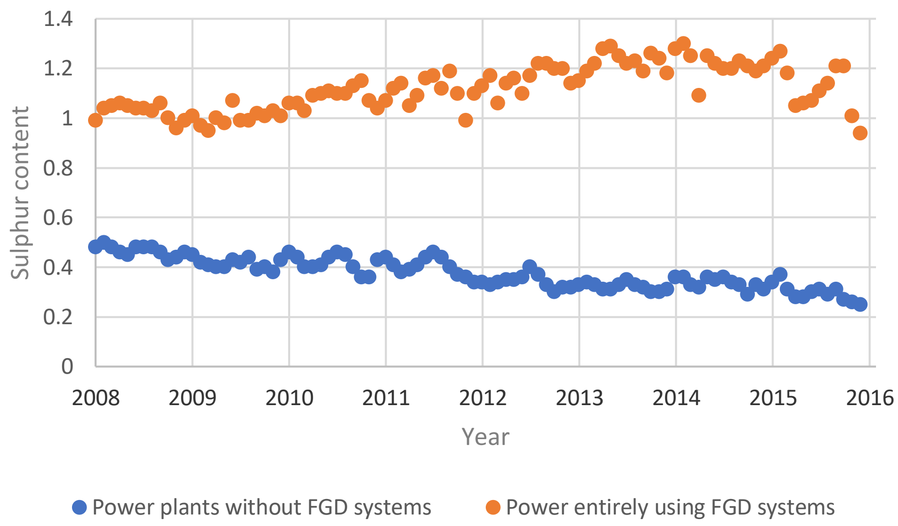


FIGURE 18 SULPHUR CONTENT AT POWER PLANTS ENTIRELY WITH/WITHOUT FGD SYSTEMS

*Notes:* Dataset is the U.S. power plants specialising in operating all boilers with or without FGD systems connected. The measure in each month is the mean sulphur content in pounds per mmBTU consumed within the relevant group where boiler-level measures are weighted by the quantity consumed. Following the national trend toward desulphurisation, the FGD connected power plants account for 3.6 times more consumption than their non-FGD counterparts on average. As we previously observed, the non-FGD power plants are on average hauling coal a substantially greater distance – generally from Wyoming – to meet their generating requirements.

*Sources*: Author calculations based on EIA data (2016a, 2016c).

I conclude this section on the boiler scale of power plant coal consumption by putting OLS estimates to the trends noted in Figure 17 and Figure 18. These estimates are derived by a basic specification on the presence of $SO_2$ controls (FGD system) and monthly sulphur content data, $sulfur_t = \beta_0 + D_1 SO_2Controls_t + D_2(SO_2Controls_t * Monthtrend_t) + \beta_m Monthtrend_t + \epsilon_t$. I report on power plants that mix FGD and non-FGD connected boilers, then also report on those that exclusively choose one or the other production method. Table 5 reports very similar differences in sulphur levels and time trends between FGD and non-FGD systems and power plants.

TABLE 5— BOILER AND SPECIALISING POWER PLANT SULPHUR TRENDS

| | Mixed power plants, boiler-level | Specialising power plants |
|---|---|---|
| $SO_2$controls[a] | 0.738*** | 0.765*** |
| | (0.0178) | (0.0169) |
| $SO_2$controls*Monthly trend | 0.0094*** | 0.007*** |
| | (0.0004) | (0.0004) |
| Monthly trend | -0.008*** | -0.005*** |
| (January 2008 base) | (0.0004) | (0.0003) |
| Constant | -0.609*** | 0.748*** |
| | (0.0169) | (0.0126) |
| R-squared | 0.9889 | 0.9848 |

*Notes:* Fitted on the monthly summary data by group (FGD/non-FGD boilers, FGD/non-FGD power plants) resulting in 192 observations each analysis. Robust standard errors in parentheses clustered at the power plant level: *** Significant at the 1 per cent level, ** Significant at the 5 per cent level, * Significant at the 10 per cent level. We observe very similar levels and trends in each study suggesting that it is the boiler level that determines power plant results, fitting with the level that regulations are applied at U.S. power plants.

[a] equals one if an FGD system is connected

*Source:* Author calculations based on data from EIA (2016a, 2016c).

While I have shown that the fuel sulphur content into U.S. power plants is related to the abatement strategy selected – FGD or non-FGD connection at the boiler level – questions remain. Some of these are again related to causality. For instance, do power plants install FGD systems to burn dirtier coal given a standard, or do regulators tighten a standard and power plants respond by installing FGD systems to continue burning locally sourced coal? In the next session, I address this question by first using a regression discontinuity and kink design, followed by an assessment of supplier turnover.

### *3.3.3 An RD/RK Exploration of Fuels Switching with FGD Installation*

Observing that fuel sulphur content differs between power plants with versus without FGD systems is interesting but not causally interpretable. For one, power plants that install FGD systems might be located near higher sulphur content sources. Such a result is fine in that it still informs policy. But what is required is some sort of evidence of how firms are changing their purchasing decisions around the FGD installation decision. Observing differences in price, heat content, and other characteristics of high versus low sulphur coal, plants may install FGD systems and source coal from entirely new sources. However, power plants might also seek to continue burning locally sourced coal. We might also expect that long-term supply contracts would lead to such a result. However, installing an FGD system is itself a long-term proposition allowing for reconsideration of a firm’s entire operating strategy. Either way, more causally interpretable results are needed.

In this section I provide some evidence on the purchasing decisions of power plants around the FGD installation decision. One issue is that the U.S. energy sector is incredibly diverse for a sector that supplies a relatively homogeneous good to households and industry. Energy producers of substantially different scales, production technologies, and ownership arrangements provide power to a massive, interconnected, and diversely regulated grid. So, any result needs to both report some sort of mean and suggest the heterogeneity of responses. We should expect that some power plants do in fact seek to retain local suppliers, which may be owned by the same authority. At the same time, others will be operating strictly based on cost minimisation while others are operating under less competitive strategies. So, I implement a combined regression discontinuity and regression kink (RD/RK) design as outlined in the first chapter of this thesis (*1.5.5 Regression Discontinuity and Kink Methodologies and Estimates*). Preceding the statistical results, however, I map them graphically to demonstrate both the heterogeneity involved and robustness of the mean result.

I select the joint RD/RK estimation strategy as a particularly flexible approach to suggesting a causal relationship. It allows estimation of both changes in magnitude and trend around some critical event. In the case of sulphur dioxide abatement technology, the event is the installation and start-up of an FGD system at a power plant. As noted in the preceding section, using the power plant level allows the use of a longer period of observation as boiler level data is available for a limited period. The prior section also notes that power plants that install FGD systems also primarily produce power using FGD connected boilers and so this is a sufficient scale of comparison, though it almost certainly underestimates the change somewhat. The RD/RK estimation methodology follows. It relies on a combination of RD and RK methods – the RD design capturing some magnitude change in the time series of interest, and the RK design capturing any adjustment in the rate of change of the variable of interest. As noted in the first chapter, while the RD design is justified as a separate method in Card, Lee, Pei, and Weber (2012) and many applications, Angrist and Pischke’s (2009) description of the estimation strategy is sufficiently flexible to estimate both sorts of effects.

Diverging from the RD/RK design in the first paper of this thesis, power plant FGD systems are installed at different points in time. The point of this critical event is denoted $k_i$ for power plant *i*. The dataset built for this study includes information on power plant sulphur consumption, as a component of coal, on a monthly scale. It also includes the year of FGD system installation. Given the time discrepancy, I design the estimation strategy such that it would underestimate an effect by comparing sulphur consumption starting at the year after installation to the sulphur consumption up to the end of the year of FGD installation. I also only consider the first installation at power plants, as some subsequently upgrade or add more FGD units. The resulting dataset includes all months in the years 1972 through 2015. Within this period, 231 power plants had a first FGD installation completed. Adding to the robustness of the results to follow, the diverse timing of these installations lessens the statistical impact of singular exogenous events such as other policy changes and geopolitical occurrences. For each power plant, let $x_i$ be an adjusted month of form $x_i = (time_i - k_i)$ where *time* is in standard calendar months. The hypothesis is that the event – or treatment – at $k_i$ fundamentally changes underlying demand such that a binary indicator can be used in the model of form

(17) $$D_i = \begin{cases} 1 \; if \; year_i \geq k_i \\ 0 \; if \; year_i < k_i \end{cases}$$

Then, let $Y_i$ be the outcome of interest in observation *i*. In this case, let $Y_i$ be the natural log of total sulphur content of fuel entering a power plant *i* versus the amount entering at event time $k_i$. In practice, the total sulphur content at $k_i$ is the quantity at the last fully operational month up to and including in the year $k_i$. The reason for this measure is so that the consumption decisions at power plants of substantially different sizes can be compared because we are comparing consumption changes at each power plant relative to itself. The resulting estimate approximates a percentage change in sulphur consumption around the $k_i$-timed event of an FGD system installation. Developing the measure based on total amounts also allows for changes in total production capacity toward more FGD-involved consumption and away from boilers not connected to FGD systems, giving a net effect at the power plant level.

A visual result of the framework is presented in Figure 19. Both axes are in effect adjusted to zero baselines – of sulphur consumption level immediately preceding FGD installation on the y-axis, and time of installation on the x-axis. Each dot represents an $i^{th}$ power plant's fuel purchase sulphur content from the 231 power plants in the sample. Given that these occur on a monthly scale, there is substantial heterogeneity in the points both before and after treatment. Trend lines pre- and post-treatment are included. While these are significantly different, it is also interesting to see the magnitude of outliers – that some power plants substantially increase their sulphur consumption following FGD installation.

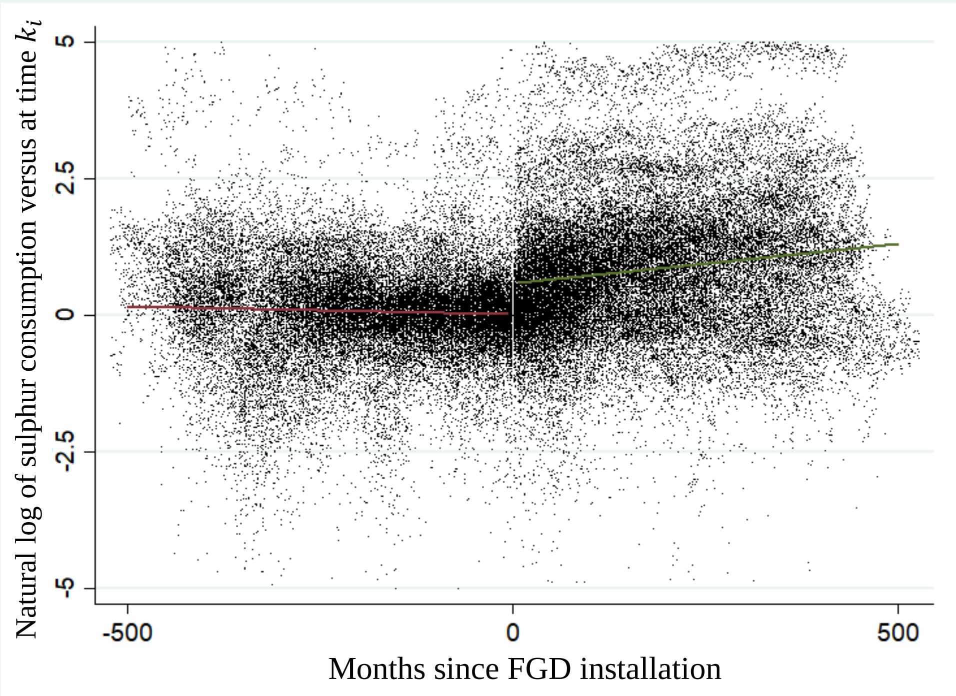


FIGURE 19 RD AND RK PLOT AROUND U.S. POWER PLANT FGD INSTALLATIONS

*Notes:* Based on the first installation of FGD equipment at each power plant among the 231 in the period of interest of 1972 to 2015. The x-axis is adjusted to the end of the year of FGD installation, and the y-axis adjusted to the quantity of sulphur in the fuel purchased immediately preceding FGD activation.

*Sources*: Author calculations based on EIA data (EIA, 2008, 2016a, 2016c).

The more formal framework and results follow. As the purpose of the model is to estimate the impact of an event or policy enacted at $k_i$, not the influence of primitives on $Y$, the functional form for estimation is simple. Let a function $Y_i = f(x_i)$ approximate for the purpose at hand the true relation $Y_i = g(X_i)$ which has a potentially large set of determinants of demand, $X_i$. The conditional expectation of $Y$ can be written as

(18) $$E[Y_i|x_i] = [Y_{0i}|x_i] + ([Y_{1i}|x_i] - [Y_{0i}|x_i])D_i.$$

As before, let the functional form of the pre-treatment conditional expectation be

(19) $$[Y_{0i}|x_i] = \alpha_0 + \beta_0 x_i$$

and post-treatment conditional expectation with both different slope and intercept be

(20) $$[Y_{1i}|x_i] = \alpha_1 + \beta_1 x_i$$

Then with $\alpha = \alpha_1 - \alpha_0$ and $\beta = \beta_1 - \beta_0$, the functional form for estimation is

$$Y_i = \alpha_0 + \beta_0 x_i + \alpha D_i + \beta D_i x_i + \varepsilon_i \qquad (21)$$

The coefficient $\alpha$ identifies any effect resulting in a discontinuity – the RD effect, and $\beta$ any change in the linear rate of change from $k$ onward – the RK effect.

I present two sets of estimates for robustness in Table 6. The first is estimated on changes in sulphur content per mmBTU of coal consumption to fit with preceding estimates in this paper. I then estimate on the dependent variable of changes in total sulphur consumption as discussed in this section. The results are statistically significant and similar. Focussing on the total-based estimates, post-FGD installation the mean quantity of sulphur content increases by 30.7-per cent on average and the rate of change adjusts from slightly decreasing to a 0.84-per cent annual increase in sulphur content.[62]

TABLE 6 — RK AND RD ESTIMATES ON FGD INSTALLATIONS

| | ln(sulphur content/mmBTU) | ln(total sulphur) |
|---|---|---|
| Date (*actual* – $k_i$) | -0.002*** | -0.0003 |
| | (0.0002) | (0.00024) |
| RD: D=1 (*actual* ≥ $k_i$) | 0.288*** | 0.268*** |
| | (0.0450) | (0.0611) |
| RK: Month*D | 0.0009*** | 0.0010*** |
| | (0.0003) | (0.00036) |
| Constant | -0.252*** | 14.483*** |
| | (0.0543) | (0.0929) |
| Observations | 87,714 | 87,714 |
| R-squared | 0.79 | 0.70 |

*Notes:* Dependent variables relative to levels immediately preceding FGD installation at $k_i$. Based on the first installation of FGD equipment at a power plant among 231 power plants in the period of interest of 1972 to 2015. Robust standard errors clustered at the power plant level in parentheses. *** Significant at the 1 per cent level, ** Significant at the 5 per cent level, * Significant at the 10 per cent level.

*Sources:* Author calculations based on data from EIA (2008, 2016a, 2016c).

It immediately follows to ask whether coal of different qualities is purchased from the same source as prior to FGD installation – after perhaps undergoing less pre-treatment, or instead sourced from mines accessing higher sulphur content coal reserves. Source mine data at the power plant scale is, unfortunately, only available from 2008 onward on a subset of power plants under the EIA's enhanced reporting requirement. This is, however, a decent sample of power plants from which to speculate – 48 with an FGD installation in the period and 133 without. I first make a list of power plants that have FGD systems installed in the period, then divide it into pre-installation and post-installation – the year after FGD

[62] Changes of 30.7% ≈ $(e^{0.268} - 1) * 100$ and 0.84% ≈ $12 * \left(e^{(0.001-0.0003)} - 1\right) * 100$, respectively.

installation and onwards – groups. I then make a list of the power plants that do not install FGD systems and divide this into first and second half of period groups. In both the FGD installation and no change groups I then compare the lists: pre- to post-installation, and first half to second half groups. I arrive at mean and median supplier retention rates and develop relevant histograms which are reported in Figure 20.

On the left side of Figure 20, the distribution of the share of coal suppliers retained after an FGD installation is reported. The mean and median supplier retention rates are 46-per cent and 40-per cent among 48 power plants, respectively. Additionally, the modal response is to retain 20-per cent to 30-per cent of suppliers. For comparison, the right side of Figure 20 reports the distribution of the share of coal suppliers retained by power plants without FGD installations in the first versus second half of the period. The mean and median retention values are 63-per cent and 67-per cent on 133 power plants. The modal response is to preserve all existing contracts.

The results appear to refute, at least in the 2008 through 2015 period, the alternative proposition that power plants are primarily installing FGD systems to retain their existing long-term contracts. This is likely because the power plants in this sample are already under regulation and primarily burning lower sulphur coal to meet emission caps. It would not be particularly beneficial to both install an FGD system and retain the same suppliers.

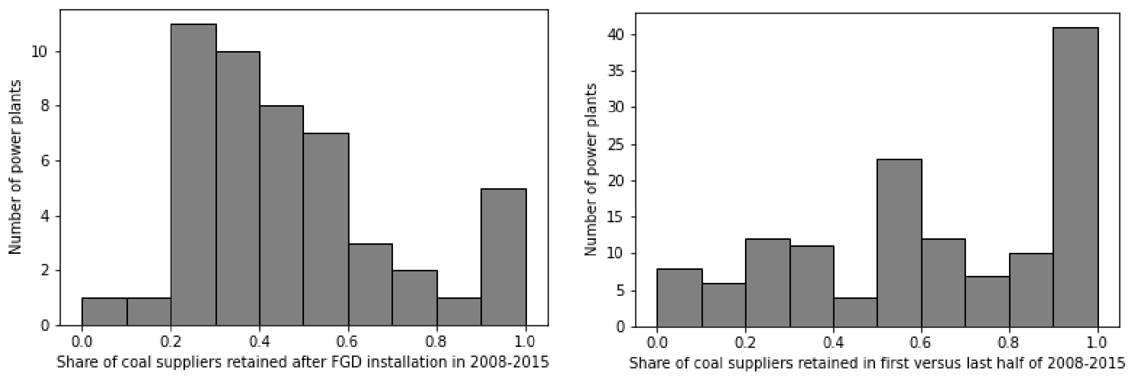


FIGURE 20 COAL SUPPLIERS RETAINED WITH AND WITHOUT FGD INSTALLATION

*Notes:* Based on lists of the suppliers to coal power plants among the sample of power plants required to submit source data for the 2008-2015 period of observation. For the power plants that install an FGD system, I divide their list of suppliers into pre- and post-FGD installation. As a comparison group, I use the reporting power plants that did not install FGD systems in the period and divide their supplier list into first and second periods and again analyse the amount of supplier switching.

(left) Share of coal suppliers retained after an FGD installation in the period 2008-2015. The mean and median supplier retention rates are 46-per cent and 40-per cent among 48 power plants, respectively.

(right) Share of coal suppliers retained in the first versus second half of the period 2008-2015. The mean and median values are 63-per cent and 67-per cent on 133 power plants.

From a comparison of their retention histograms, the power plants that installed FGD systems retained less than half of their coal suppliers after installation – most often retaining 20-30-per cent of suppliers. In comparison, power plants that did not install FGD systems usually made fewer changes to their supplier list – most often none.

*Sources*: Author calculations based on EIA data (2016a, 2016c).

This section concludes with a short note on FGD installation timing around environmental policy changes. As discussed in the literature review, substantial national changes in sulphur dioxide limits occur in the U.S. in 1971 with the Clean Air Act, the 1990 establishment of the Acid Rain Program (ARP), 1995 deadline for compliance for the ARP's Phase I power plants, and 2000 deadline for the more expansive set of Phase II power plants. Comparatively minor adjustments to sulphur dioxide emission regulations occur as required in the interims and at smaller geographic scales. Figure 21 presents the count of FGD system installations in each year against the backdrop of tightening sulphur dioxide emission restrictions. At each stage, the impending requirements as well as the problem they sought to address were well-known to both industry and the public.

Despite well-publicised information on when sulphur dioxide regulations would come into effect, we observe that FGD installations often occur in the interim between policy changes. This occurs for several reasons. For one, we know that the cost of an FGD system is substantial – a later section of this paper notes mean and median FGD system prices of $105.9 million and $28.3 million (2016 dollars). Against the price of FGD systems, power plants can contrast the cost of cleaner fuel alternatives and purchasing emission allowances. So, we should not be at all surprised to find that power plants install FGD systems often off policy change years. Rather, as a natural course of business, they balance the competing costs involved and install systems when it is appropriate to do so given the age and replacement schedules of existing equipment. It is even possible that an FGD system is installed earlier out of convenience but bypassed until needed. Unfortunately, this natural progression of FGD system installations limits the econometric techniques that can be successfully applied as we cannot expect a policy shift to result in clean changes in FGD installation rates.

The argument on timing having been made, we can observe an uptick in FGD installation rates between adoption of the ARP program and the Phase I implementation deadline – from 1990 to 1995. The power plants under this phase were the 110 largest sulphur dioxide emitters at the time, and so economies of scale likely play a role in their abatement technology choice as the returns to an FGD system installation could be substantial. The Phase II implementation from 1995 to 2000 brought all remaining power plants with a boiler size above a certain minimum capacity (75 megawatts of electricity output), so included more power plants where the fixed cost requirements of an FGD system are prohibitive. These may also more often be reserved for peak demand periods when the return to operating is higher. The conclusion is that sulphur dioxide restrictions lead to FGD installations, but at a pace and timing that fits within greater power plant considerations around cost.

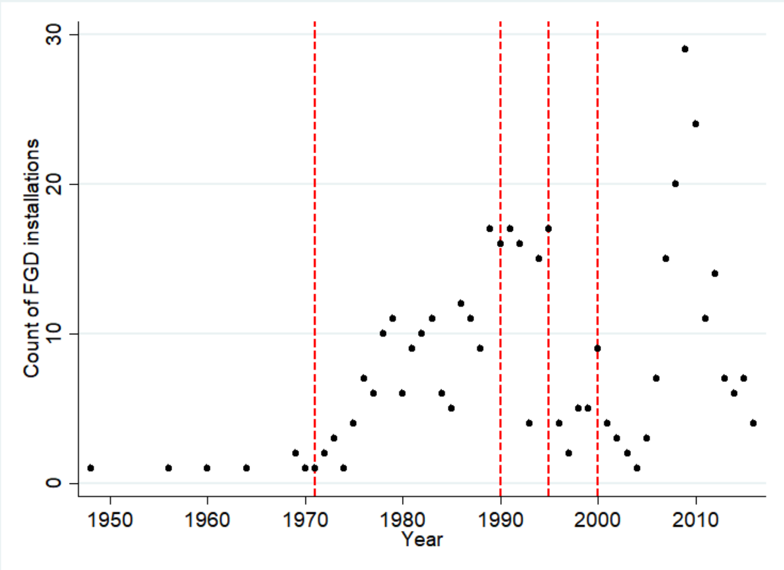


FIGURE 21 FGD INSTALLATIONS AND KEY REGULATORY EVENTS

*Notes:* Plot of FGD system installation frequency by year, with important sulphur dioxide policy change years denoted. These include the 1971 Clean Air Act adoption, 1990 establishment of the Acid Rain Program (ARP), 1995 deadline for the ARP's Phase I set of power plants to comply, and 2000 deadline for the more expansive Phase II set to enter compliance.

*Sources*: Author calculations based on EIA data (2008, 2016a, 2016c).

So far, this paper has reviewed the field's relevant literature and put some estimates on the effects of abatement technology installation on power plant sulphur demand. These estimates have included the net effect across the industry, the power plant response at a within-facility level, and changes in which mines are contracted to supply coal. Having provided a body of evidence that firms do increase fuel sulphur content after FGD installation, and tend to primarily specialise in higher sulphur coal with FGD installation versus lower sulphur coal without, I precede with a structured model of the power plant decision-making process.

### 3.4 A Simple Model of Regulated Power Plants

In this section, I compose and validate a model for thinking about the domestic power plant-level decision-making process. In doing so, I develop some tools to explore market structure despite the industry's complexity and which are general enough to be applicable beyond the U.S. energy sector. The domestic U.S. energy sector is the limited scope of this paper as the national scale retains the final say on domestic policy despite contemporary movements toward international cooperation on environmental issues. I begin with a model of demand at domestic sites for inputs of differing polluting characteristics, then add the impact of policy-driven technological intervention. I continue to anchor the discussion in the context of demand for power plant fuels of differing sulphur content and their polluting potential. The impact of abatement technologies on demand for high versus low sulphur fuels are observed.

The model in this paper describes a regulated sphere where there is an incentive to adopt an abatement technology at power plants under regulation. For comparison, an unregulated sphere has a lack of regulatory incentives and power plants are assumed to unreservedly respond to price when choosing quantities of inputs to production. I keep the decision whether to buy clean or dirty inputs simple by also assuming it is price-based, conditional on regulatory limits, I structure the discussion as choosing between coal with some sulphur content, and a cleaner alternative such as coal with less sulphur or natural gas that contributes zero sulphur.

The model will highlight the importance of abatement and transportation costs and indicates that the fuel decision occurs in two stages: whether to consume at all, then how much of the polluting version given some emissions limit. It also suggests that policymakers focus on the power plants using dirtier fuels, as those only consuming cleaner versions would be worse off by switching back. The implications for fuel demand from placing a pollution cap, followed by availability of abatement technology, guide the discussion.

*The production function:* I take the input selection decision as separate from the output decision as fitting with many industries. In energy production, the necessity to contract the delivery of fuels and stockpile to avoid discontinuities in service separates the spheres of purchase and consumption. The person making the two decisions may be the same or separate as the institutional knowledge required differs. An implication is that the literature on output decisions at power plants is not directly applicable to this analysis of input decisions. Instead, the link is thought of for expositional purposes as occurring at some weekly power plant operational meeting – the team that makes output decisions, staffed with an economist or two, meets with purchasers to coordinate. Some discussion of expected production given emissions limitations occurs, then purchasing plans are adjusted to ensure inputs will be available. By allowing a weak link between output and input decisions, I can assume a cost minimisation purchasing objective subject to some output requirement.

For tractability, firms –slightly inaccurately power plants – select between two sorts of fuels which are dirty and clean substitutes. Dispensing with engineering details which are outside the scope of the model, output in mmBTU's are converted to electricity. In terms of output – accounting for differences in power plant efficiency – demand for coal and natural gas are perfect substitutes and the end-user cannot tell which fuel is used to generate their electricity.[63] As a simplification, perfect substitution is useful – output, *E*, is the sum of outputs in mmBTU from dirty, *D*, and clean, *C*, inputs as $E = C + D$ (see Note 5). An implication is that in the context of fuel purchases, *C* and *D* should be studied in terms of heat potential in units of mmBTU to match output *E*. Relaxing this demand functional form, however, does not negate the fundamental model implications. I do note that the literature suggests many intermediate to large coal and natural gas power plants operate near-CRTS (Nerlove, 1963; Atkinson and Halvorsen, 1976; Christensen and Greene, 1976; Bernstein and Parmeter, 2019; Bernstein, 2020). While I challenge the interpretation of CRTS in appendix *A3.3 Returns to Scale in Electricity Generation*, the net effect of the matter is as if CRTS is in effect. These simplifications imply that firms make production decisions based on real output, and that this output is generally stable. Equivalently, the firm has a constant expectation.

Note 5 - Deriving $\boldsymbol{E = C + D}$

Output, *E*, in mmBTU, from consuming volumes of dirty input, $\underline{D}$, and clean substitute, $\underline{C}$, at efficiencies $a_D$ and $a_C$, is converted to electricity in processes outside the model. Modelling as strictly additive, $\underline{E} = a_D \underline{D}^{\beta_D} + a_C \underline{C}^{\beta_C}$ which we can describe in terms of output as $E = D + C$. Alternatively with more assumptions required, when power plants consume $\underline{D}$ and $\underline{C}$ in the same physical system, the efficiency in terms of mmBTU is roughly the same to match plant capacities, so $a_D = a_C = a$. Since many intermediate and large coal and natural gas power plants operate at or near CRTS, $\beta_D = \beta_C = 1$. Rather than short tons and cubic feet of volume, for coal, $\underline{D}$, and natural gas, $\underline{C}$, put in terms of mmBTU availability as *D* and *C*. Accounting for power plant efficiency as $E = \frac{E}{a}$, then $E = D + C$.

[63] The perfect substitutes assumption is often used in analyzing the U.S. energy sector, for example Hoel (2009) and Long and Stähler (2018). But it also receives criticism, for instance in Long (2014) and van der Ploeg and Withagen (2012) on the grounds that it does not represent well the substitution of biofuels and other new alternatives against coal and natural gas.

*Budget and emission constraints:* Prices and lack of monopsony influence are assumed. That is, the total cost is also additive in source price-quantity pairs, and at the power plant level prices are taken. Then, for dirty and clean input prices $P_D$ and $P_C$, the expenditure, or budget requirement, is $B = P_D D + P_C C$. Additionally, assume emissions, $\mathcal{E}$, can be equated back to batches of dirty inputs as $\mathcal{E} = \alpha D$ for constant $\alpha$, where $0 < \alpha < 1$. If $\alpha = 0$ no dirty alternative exists, and as a result, no relevant emission occurs. In the other extreme, $\alpha = 1$ suggests the input's only purpose is to pollute. The constant $\alpha$ assumption is generally realistic as power plants must expect fuel meeting a standard, else continually adjust operations at some cost.

The essence of the model is that three well-known cases result:

(i) The clean option is cheaper: A corner solution of all clean inputs is optimal and zero emissions result. This result is not particularly interesting to study because there is no environmental problem to address.

(ii) The dirty option is cheaper: A corner solution of all dirty inputs is optimal for the firm, and $\alpha D$ pollution occurs. Manipulating this case drives the analysis.

(iii) Prices of inputs are equal: The optimal choice is any combination of clean and dirty inputs resulting in output *E*. This case is unstable. For instance, enacting an emission standard with any compliance cost whatsoever results in the dirty option becoming more expensive than cleaner ones and the situation transitions to case (i).

It would be careless to continue without first testing whether this model is descriptive of the U.S. energy sector. In the following section, I devise and implement a test of whether power plant purchasing patterns suggest a perfect substitutes relationship. Afterwards, I proceed by adding regulation in terms of an emissions cap and then abatement technology that effectively relaxes that cap.

### *3.4.1 Testing the Perfect Substitutes Description of U.S. Energy Producers*

As discussed in a preceding section, there are several U.S. power plants – about half of those reporting data in the relevant period – that have both FGD connected boilers and non-FGD alternatives on premises. These power plants primarily produce energy via their FGD connected systems but also buy coal with a lower sulphur content on average for their non-FGD systems. This setup frequently results in received coal prices at the power plant level being available for both high sulphur coal and low sulphur coal. I design a testing methodology and use this data to explore how closely the perfect substitutes assumption fits U.S. coal power plants.

A defining quality of the perfect substitute production and additive budget model is that firms dramatically, rather than incrementally, change fuel input composition in response to price changes. Without regulation, these choices are between full specialisation in high or low sulphur coal or natural gas. With regulation, the optimal choice is either full specialisation in low sulphur coal or natural gas, or a mix with quantities of high sulphur coal. I test the applicability of the perfect substitutes model by computing own- and cross-price elasticities of demand grouped by the relationship between prices $P_D$ and $P_C$. We should expect to find little response to price changes at most ratios as the perfect substitute relationship suggests all switching activity occurs as one fuel transitions from being more to less expensive. That is, we might expect firm price elasticities of demand to appear very inelastic except around $P_D = P_C$, when they should instead be very elastic, as depicted in Figure 22. In comparison, an imperfect substitutes relationship would result in a more distributed switching activity. In practice, we might instead observe fuel switching around $P_D + \mu = P_C$ due to unobserved differences in operating costs, $\mu$. For instance, $\mu \neq 0$ might be some unobserved net marginal abatement cost tied to selecting the polluting input. Such divergence reveals the scale of unobserved abatement, production, and switching costs involved, or at least the share not passed to consumers and other parties.

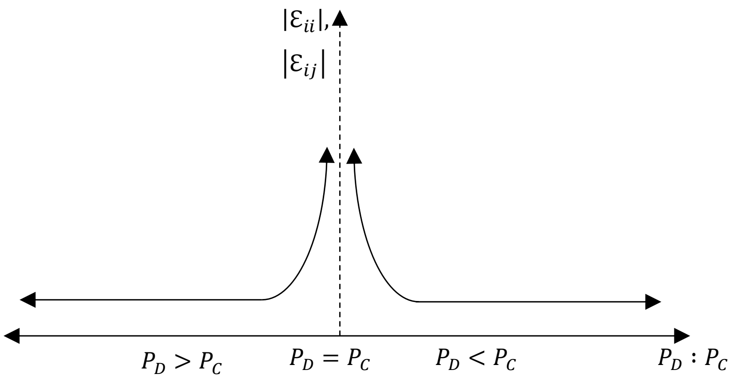


FIGURE 22 EXPECTED ELASTICITY ACTIVITY AROUND POINT OF PRICE EQUALITY

*Notes:* A perfect substitutes model of firm behaviour suggests that when the firm's objective is cost minimisation, specialisation in one input to production results. In the U.S. energy sector, we usually expect either full specialisation in input C or a stable mix of C and D. Input switching occurs when prices change such that the more expensive input becomes the cheaper one. An indicator of this transition will be own- and cross-price elasticities that are higher around the point of price equality. Without hidden costs, the transition point will be when $P_D = P_C$, otherwise when $P_D + \mu = P_C$.

To test whether the perfect substitutes model is appropriate, I estimate elasticities by groups defined by the relationship of input *i*, to alternative input *j*, as $\frac{P_i - P_j}{P_j}$ which is the price advantage of input *i*. I compare (*i, j*) pairs of high sulphur coal and natural gas (*HSC, NG*) and low sulphur coal and natural gas (*LSC*, *NG*) and iterate over price ratio groups beginning where coal prices are 50-per cent less than natural gas prices. Because coal power plants often use natural gas during start-up, we often observe prices and quantities for both fuels. I estimate own- and cross-price elasticities by a log-log transformation of prices and quantities to find the net effect of a price change on quantity.[64] I use the coal data outlined in the introduction, plus over 500,000 observations on natural gas purchases. However, in each estimation group, estimates are on only several tens-of-thousands of observations where the same power plant purchases both coal and natural gas.

I present HSC-NG own-price and cross-price elasticities in Figure 23, and LSC-NG estimates in Figure 24. Fuel switching behaviour at high sulphur coal power plants provides the more striking result as one would expect. Group elasticities are inelastic and often statistically indistinguishable from zero while $P_C < P_D$. However, the highest elasticities observed suggest that fuel switching occurs in the range of HSC being 15- to 30-per cent more expensive than natural gas. There can be many causes of this point, for example, additional power plant benefits to coal use, substantial costs to scaling up natural gas consumption, and expectations that natural gas prices will not remain lower. Contracts on fuel and by-product provision as well as coal shutdown costs may also make it challenging to switch, and meanwhile natural gas prices continue to decline beyond equality in the transitioning market.

[64] This may be an even more effective test for purchase changes that do not also require changes in technology.

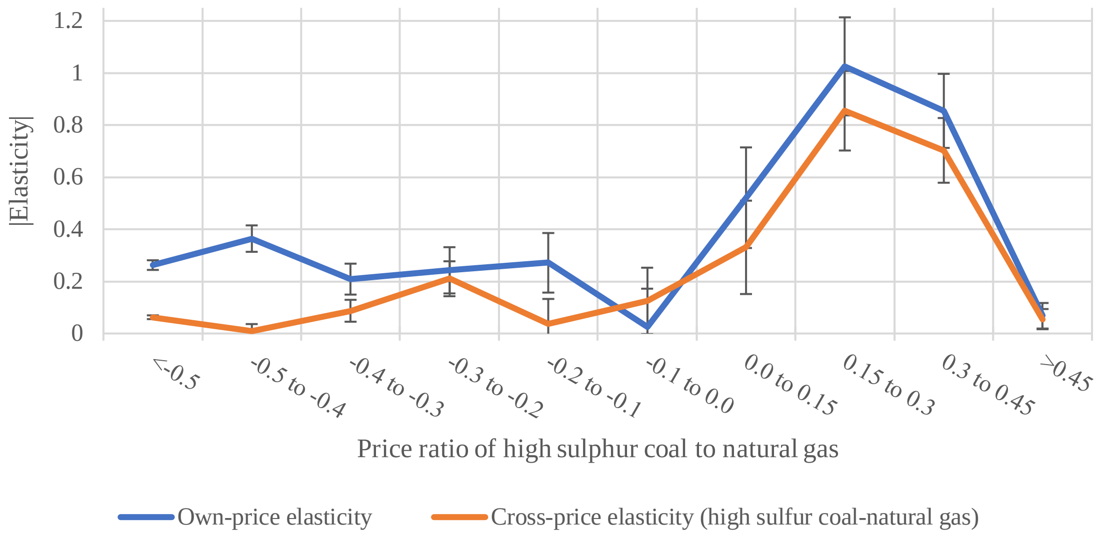


FIGURE 23 HIGH SULPHUR COAL ELASTICITIES BY OPPORTUNITY COST

*Notes:* Own- and cross-price elasticities for high sulphur coal grouped by the price ratio against the clean alternative of natural gas with around 2,000 observations in each group. Estimated by log-log specification using power plant monthly price and quantity data for power plants where both coal and natural gas are used. I include 95-per cent confidence interval bars. The perfect substitutes model suggests that more fuel switching activity will occur at some ratio than elsewhere. Without differences in operating and hidden costs, this switching point would be around price equality. Elasticity estimates suggest peak activity instead occurs when natural gas is at least 15-per cent cheaper than low sulphur coal suggesting hidden switching or other costs.

*Source:* Author calculations based on data from the EIA (2016a, 2016c).

The LSC-NG comparison provides less support for the perfect substitutes model in such cases. We still observe the highest elasticity estimates when LSC is 15- to 30-per cent cheaper than natural gas, but also observe a more gradual increase in elasticities. Power plants, then, may act as though they have perfect substitute preferences between clean and dirty fuel, but consider clean alternatives more substitutable.

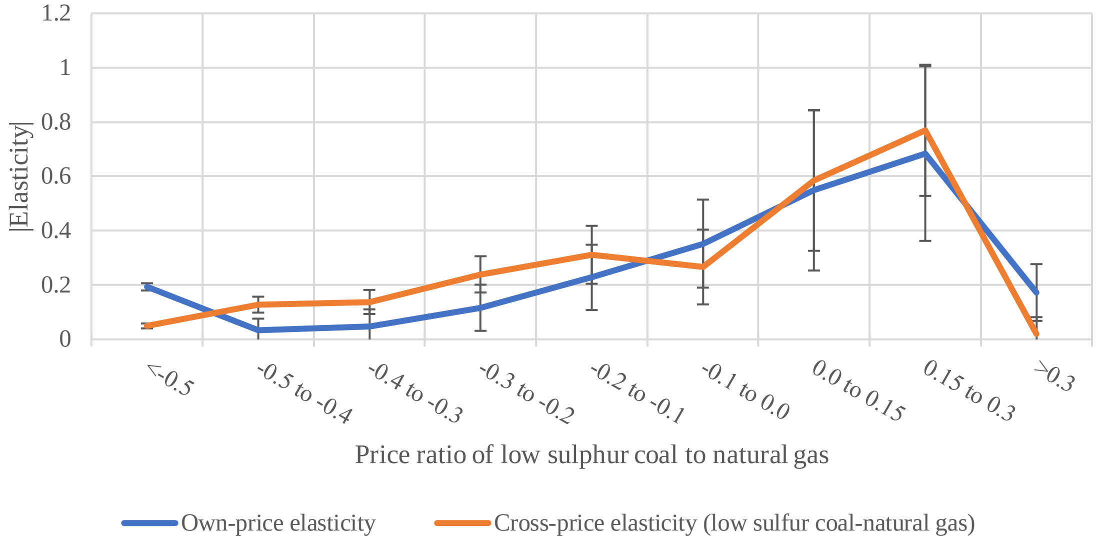


FIGURE 24 LOW SULPHUR COAL ELASTICITIES BY OPPORTUNITY COST

*Notes:* Own- and cross-price elasticities for low sulphur coal grouped by the price ratio against the clean alternative of natural gas. Estimated by log-log specification using power plant monthly price and quantity data for power plants where both coal and natural gas are used. I include 95-per cent confidence interval bars. The perfect substitutes model suggests that more fuel switching activity will occur at some ratio than elsewhere. Without differences in operating and hidden costs, this switching point would be around price equality. Elasticity estimates suggest peak activity instead occurs when natural gas is at least 15-per cent cheaper than low sulphur coal. The perfect substitute effect for low sulphur scale is less pronounced than in the high sulphur coal versus natural gas case.

*Source:* Author calculations based on data from the EIA (2016a, 2016c).

While not indisputable support, I find the perfect substitutes relationship plausible, particularly at high sulphur coal-consuming power plants. I now proceed under this not entirely fictional assumption and add regulation and abatement technologies to the model.

### *3.4.2 Adding Regulation and Abatement Technology*

From the introductory model and preceding empirics, we can expect that fundamental market shifts are required to drive changes in fuel makeup. Another implication of stability is that when an emissions cap is set, $\bar{\mathcal{E}}$, it can be expressed in terms of the dirt input, $\bar{D} = \frac{\bar{\mathcal{E}}}{\alpha}$. Adding an emission standard has different impacts on energy producers depending on their initial case. If the clean fuel is initially cheaper – case (i) – no change occurs. However, if a producer is in case (ii) and the standard is binding, a change in equilibrium quantity occurs.[65] The net emissions content of the fuel and emissions lowers from $\alpha$ to $\frac{(0)C+\alpha D}{C+D} = \frac{\alpha D}{C+D} = \frac{\alpha D}{E}$. In coal for energy production, this is in pounds of sulphur per mmBTU. Figure

[65] As long as the net effect is that the dirty fuel with compliance costs remains cheaper than the clean alternative, else the power plant has an incentive to fully switch fuels.

25 illustrates the decision with a standard in place. This is the starting point for the analysis of abatement technology implementation.

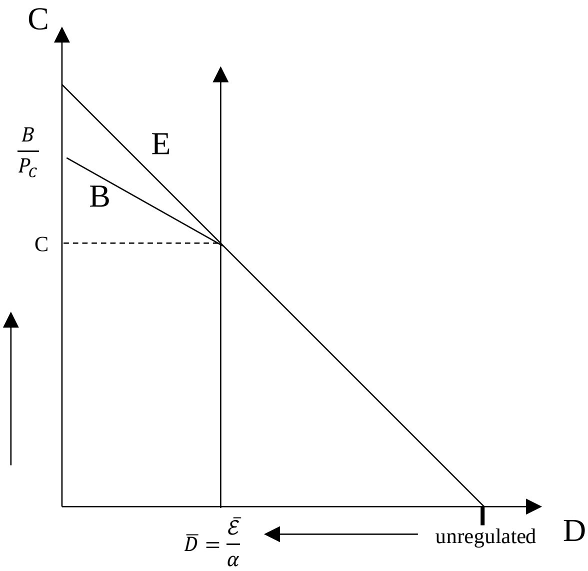


FIGURE 25 FUEL SELECTION WHEN THE DIRTY INPUT IS CHEAPER AND A STANDARD IS SET

Notes: Price of clean input, C, is $P_C$ which is higher than the price, $P_D$ of dirty input, D. The binding constraint is in terms of quantity of dirty inputs, $\bar{D}$, which derives from the emissions cap $\bar{\mathcal{E}} = \alpha D$ for polluting share α. From the perfect substitutes demand and contracted output (quantity *E*) assumptions, the emissions cap fully explains the combination of dirty and clean shares selected and total expenditure level, *B*. From an unregulated origin, the cap results in a reduction in dirty fuel which is offset by an increase in the clean alternative.

We observe a standard dilutes the emissions content of the average unit of fuel. The power plant operator does not need to purchase only cleaner coal or natural gas – though these are options – but rather might mix sources. The average quantity of pollutants in pounds of sulphur per-mmBTU decreases and emissions decrease relative to without a standard.[66] Much policy to date relies on setting a standard or comparable tax in a domestic market. However, there is the potential for offsetting as dirtier inputs are now free to export. From a regulated sphere, dirty inputs may enter unregulated ones – emissions leakage.[67]

*Adding abatement technology:* If an emissions cap reduces demand for *D*, then we expect a relaxation of that cap to increase demand. The use of FGD systems effectively does this by operating at some removal efficiency such that $\bar{D}' = \frac{\bar{\mathcal{E}}}{\delta\alpha}$ where $\delta = 1 - efficiency$ is

[66] The behaviour of the single interior result in case (ii) results from cost minimisation subject to meeting the output expectation or commitment, and the emission constraint, as $\min_{D,C} P_D D + P_C C$ subject to $E = f(D,C) = f(D) + f(C)$ and $\mathcal{E} \geq f(\alpha, D) = \alpha D$. While it is a simple framework, I explore the robustness of the interior result in appendix *A3.4 Exploring the Interior Result of the Input Selection Model*.

[67] In Phase I of the U.S. Acid Rain Program, provisions were made through a tracking requirement to avoid within-U.S. emissions leakage from the power plants under the new program to those still excluded from it (Napolitano et al. 2007).

the passthrough share. Less is emitted per unit of fuel – only share δ of the polluting potential counts against the emissions cap and sulphur dioxide into the production process diverges from the emissions out of it into the atmosphere. We should also find that the effective price of the dirty input increases due to abatement technology operation and the slope of the budget line $\frac{P_D}{P_C}$ then increases. However, increasing *D* displaces costlier *C*. With the standard and abatement technology in place, net emissions are $\frac{(0)C+\delta_e \alpha D}{C+D} = \frac{\delta_e \alpha D}{C+D} = \frac{\delta_e \alpha D}{E}$ where $\delta_e$ denotes operating the abatement system at some efficiency. This efficiency is up to, but perhaps less than, the system's rated capacity as another matter to consider. The net effect must be cost savings versus the power plant operator choosing to meet the cap without abatement technology under the cost minimisation assumption.

A starting point is to consider changes in abatement technology as costless. The firm is unambiguously better off by a decrease in the budget required to *B'* in Figure 26. However, firms using the clean alternative are not enticed into the dirty input market as it remains cheaper to use the clean alternative.

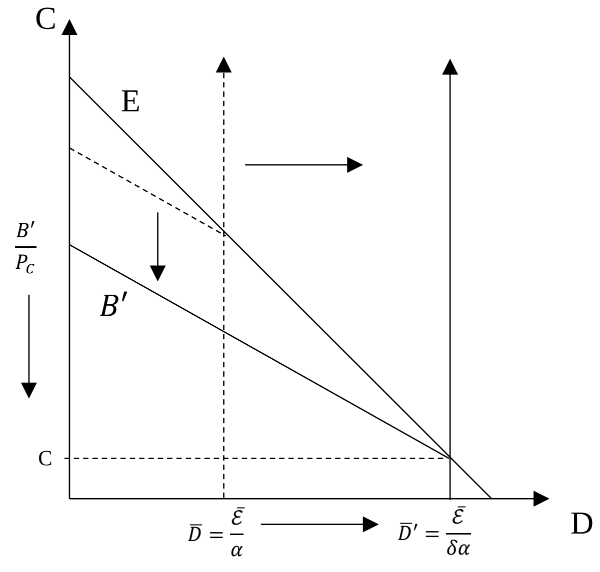


FIGURE 26 FUEL SELECTION WITH COSTLESS ABATEMENT TECHNOLOGY

*Notes:* Relative to the emissions cap, power plants that prefer the dirty input *D* because the price $P_D$ is lower, prefer a larger input share in *D*. An abatement technology operating at an $efficiency$ such that $\delta = (1 - efficiency)$ results in an effective cap of $\mathcal{E} = \delta\alpha D$. The share of the dirty input increases to $\bar{D}'$ and costly *C* decreases. Total expenditure then decreases to $B'$ relative to meeting the emissions cap without abatement technology.

With a cost to FGD system operation, the budget requirement increases relative to a costless technology, but an upper limit exists due to the alternative of not operating the abatement system at all and instead purchasing more of the costly alternative. Even if the system is designed to operate at some specific efficiency, bypassing it part of the time is equivalent to reducing efficiency.[68] The efficiency then is up to, but perhaps less than the system's rated capacity. Figure 27 suggests the complication.

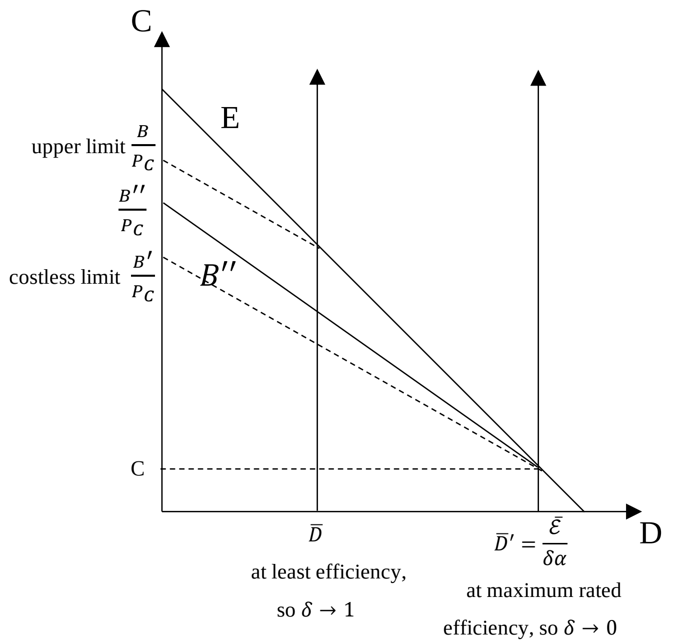


FIGURE 27 FUEL SELECTION WITH VARIABLE ABATEMENT TECHNOLOGY AT SOME COST

*Notes:* Abatement technology that has some operating cost limits when a firm will adopt or use it at full efficiency. Here some abatement operating cost can result in the operator selecting back out of using the abatement technology and instead meet the cap by mixing dirty and clean alternatives if $B''$ exceeds $B$.

While I have explored the firm-level abatement decision, the policymaker's perspective is to harness the cumulative power of large swaths of the industry transitioning to satisfy domestic emission reduction requirements. From firm decisions to market demand requires an aggregation process. Here I suggest industry aggregates from a set of perfect substitute demanders to market demand that is downward sloping in price but otherwise unspecified. This change in relationship with aggregation results from differences in the price at which firms switch between clean and dirty inputs due to differences in transportation, mining, operating, and transaction costs. I again pause to test the model, now whether the resulting market-level price elasticities of demand for firms differ by their chosen abatement strategy.

[68] See an extensive analysis of the bypass decision Rubin and Nguyen (1978).

### *3.4.3 Testing Whether Price Elasticities Differ by Abatement Method*

As in the preceding figures, demand at power plants with FGD systems are more stable than those using the dirty input without technology improvements due to flexibility. Differences in elasticities occur as it would take a increases across all available $P_D$, alternatively decreases in $P_C$, for FGD power plants to instead save by specialising in input *C*. This is because an FGD system preserves the choice of using coal as more types of coal can be used while still meeting emissions constraints. It would require changes in $P_C$ across the sector. FGD equipped power plants should have more inelastic own-price elasticities of demand in coal, and subsequently alternative fuels, than plants without FGD.[69] We can test this prediction by estimating market-level own- and cross-price elasticities separated into groups of power plants with and without abatement technology. The prediction is that both sets of elasticities will be more inelastic among FGD-enabled power plants at the market level.

I estimate market-level own- and cross-price elasticities grouped by abatement technology – whether FGD systems are in use – by using use the dynamic linear logit model of Jones (1995) which builds on the linear logit framework of Considine and Mount (1984) and constraints on homogeneity of degree zero, symmetry, and adding up in Considine (1990) such that the system of demand equations add up to total expenditure and only real prices matter.[70] I then update the method with additions from Battese (1997) to consider zero cost-share observations, EIA (2012) to use monthly-scale observations, Steinbuks (2012) to include a general market trend, and from Steinbuks and Narayanan (2015) a method to account for country differences with fixed effects which I apply to U.S. NERC regions as I aggregate to that scale as in EIA (2012).

I then diverge from the literature with three additions. First, I divide high and low sulphur coal into separate consumption shares and derive own- and cross-price elasticities between high sulphur coal, low sulphur coal, and natural gas. Second, I estimate on fuel data restated in heat content in mmBTU's, which is the primary unit of quantity in power plant input decisions. Third, I estimate own- and cross-price elasticities by power plants with and without abatement technology to address the central empirical question. The recent decline in U.S. natural gas prices results in additional exogenous variation and estimation basically

[69] In particular HSC and LSC own-price elasticities and HSC-LSC and HSC-NG cross-price elasticities.

[70] However, this form has a longer history. Theil (1969) suggests a precursor by Warner (1962) and then proceeds to develop an estimation form based on Warner that is close to Considine and Mount (1984). However, McFadden (1974) traces it back to several precursor methods published throughout the 1950's and 1960's.

takes the approach of attempting to describe the market well. Because any candidate instrument for an instrumental variable approach almost certainly fails to satisfy the exclusion restriction in estimating energy elasticities, other alternative approaches are limited.

Fortunately, the dynamic linear logit model provides elasticity estimates that are both adherent to theory and flexible to cost functional form. I use the notation convention from Jones (1995) and add a monthly indicator variable following EIA (2012). I estimate on a system of equations for all fuels except one which I drop and use to normalise those that remain. An underlying assumption of the methodology is that we can describe each input as a share, e.g. high sulphur coal from total fuel consumed. For quantity $Q_{i,t}$ of fuel *i* at time *t* and price $P_{i,t}$,

(22) $$\frac{P_{i,t}Q_{i,t}}{\sum_{j=1}^{N} P_{j,t}Q_{j,t}} = S_{i,t} = \frac{e^{f_{i,t}}}{\sum_{j=1}^{N} e^{f_{j,t}}}$$

$$where\ f_{i,t} = \eta_i + \sum_{i=1}^{N} \phi_{i,j} lnP_{j,t} + \gamma lnQ_{i,t-1} + \boldsymbol{X}_{i,t}\beta + \epsilon_{i,t}$$

where I suppress the unit subscripts identifying observations as at the U.S. NERC region scale and include matrix $\boldsymbol{X}$ of covariates. Desirable constraints of homogeneity of degree zero (HDO) and symmetry (Slutsky) are imposed by model formulation, respectively, $\sum_{i}^{N} \phi_{i,j,t} = 0$ and $S_{i,t}\phi_{i,j,t} = S_{j,t}\phi_{j,i,t}$ for input pairs *i* and *j*, and I assume adding up restrictions: $\sum_{i}^{N} \eta_i = 1$, $\sum_{i}^{N} \beta_i = 0$, and $\sum_{i}^{N} \phi_{i,j} = 0$ (see Note 6). The log-log, static form of the model derived by Considine and Mount (1984) (also see Theil, 1969) is then

(23) $$ln\left(\frac{S_{i,t}}{S_{N,t}}\right) = (\eta_i - \eta_N) - \left(\phi_{i,i+1}S_{i+1,t} + \phi_{i,i+2}S_{i+2,t} + \cdots + \phi_{i,N}\left(S_{i,t} + S_{N,t}\right)\right) ln\left(\frac{P_{i,t}}{P_{N,t}}\right) + \left(\phi_{i,i+1} - \phi_{i+1,N}\right)S_{i+1,t}ln\left(\frac{P_{i+1,t}}{P_{N,t}}\right) + \left(\phi_{i,i+2} - \phi_{i+2,N}\right)S_{i+2,t}ln\left(\frac{P_{i+2,t}}{P_{N,t}}\right) + \cdots + \left(\phi_{i,N-1} - \phi_{N-1,N}\right)S_{N-1,t}ln\left(\frac{P_{N-1,t}}{P_{N,t}}\right) + \left(\epsilon_{i,t} - \epsilon_{N,t}\right)$$

where I derive a separate equation for each $i \neq N$ fuels (all except one). A dynamic component allowing estimation of long-run elasticities is appended following Jones (1995) as $\lambda ln\left(\frac{Q_{i,t-1}}{Q_{N,t-1}}\right)$, and controls for the general market trend, month effects, and year effects are added following Considine and Mount (1984), EIA (2012), and Steinbuks (2012), respectively, as $(\alpha_{i,t} - \alpha_{N,t})\ln(G_t) + \sum_{m=2}^{12}(\beta_{i,m} - \beta_{N,m})M_m + \sum_{y=2009}^{2015}(\gamma_{i,y} - \gamma_{N,y})Y_y$. Battese (1997) suggests indicator variables for when a cost-share is at or near zero to account for corner solutions. I specify these as $D_{i,t} = 1\ if\ ln\left(\frac{S_{i,t}}{S_{N,t}}\right) = 0\ and\ D_{i,t-1} = 1\ if\ ln\left(\frac{S_{i,t-1}}{S_{N,t-1}}\right) = 0$. I add NERC region fixed effects as in EIA (2012) to account for regional differences in interfuel substitution. The fixed effects follow from a method to account for country differences in interfuel substitution in Steinbuks and Narayanan (2015). I aggregate to NERC region data where the price of fuel *i* in period *t* is the mean of individual power plant purchase prices weighted by quantities.

Note 6 – Constraints of HDO, Slutsky Symmetry, and the Value of the Adding Up Restrictions

Assume in a time, t, income, I. To show $\sum_i^N \phi_{i,j,t} = 0$, let $E(P_1, P_2, \ldots, P_N, I)$. From the Euler Theorem, $kE(\cdot) = \sum_{i=1}^N\left(\frac{\partial E(\cdot)}{\partial P_i}P_i\right) + \frac{\partial E(\cdot)}{\partial I}I$ for homogeneity of degree k. Divide by $E(\cdot)$ and note $k = 0$, then $0 = \sum_{i=1}^N\left(\frac{\partial E(\cdot)}{\partial P_i}\frac{P_i}{E(\cdot)}\right) + \frac{\partial E(\cdot)}{\partial I}\frac{I}{E(\cdot)} = \sum_{i=1}^N \phi_i + \phi_I$. Assume that $\phi_I = 0$ from the preceding model – E is mandated or contracted – and let N include cross price elasticities. To show $S_{i,t}\phi_{i,j,t} = S_{j,t}\phi_{j,i,t}$, note from the Slutsky equation the total price effect is $\phi_{i,j} = \frac{S_j}{S_i}\phi_{j,i} + S_j(\phi_{j,I} - \phi_{i,I})$ from substitution and income effects, respectively. Let $\phi_{j,I} = \phi_{i,I}$ (equality of the income effect), which I extend to $\phi_{j,I} = \phi_{i,I} = 0$ by the prior assumption on E, then $S_i\phi_{i,j} = S_j\phi_{j,i}$. The adding up restrictions ensure that the system of demand equations add up to the total expenditure, $\sum_{i=1}^N S_i = 1$: $\sum_i^N \eta_i = 1$ ensures they are equal to full expenditure even if no change occurs, $\sum_i^N \beta_i = 0$ that there is a net zero effect from changes in other preferences, and $\sum_i^N \phi_{i,j} = 0$ that only real prices matter (also met by $\sum_i^N \phi_{i,j,t} = 0$).

For the fuels high sulphur coal (HSC), low sulphur coal (LSC), and natural gas (NG), the dynamic linear logit system becomes a system of two equations where I normalise by natural gas.[71] Estimates for HSC and LSC are derived using the system[72]

[71] Petroleum fuels account for a small share of modern power plant demand, particularly in per-BTU terms, and I drop them from estimation. However, EIA (2012) finds petroleum fuels as a complement in coal power plants as a start-up fuel.

[72] Estimates are invariant to which fuel is dropped and the remaining fuel's estimates are then inferred. The high-versus-low sulphur coal definition is taken from EIA (n.d.) fitting standard conventions.

$$(24) \qquad ln\left(\frac{S_{HSC,t}}{S_{NG,t}}\right) = (\eta_{HSC} - \eta_{NG}) - \left(\phi_{HSC,LSC}S_{LSC,t} + \phi_{HSC,NG}\left(S_{HSC,t} + S_{NG,t}\right)\right) ln\left(\frac{P_{HSC,t}}{P_{NG,t}}\right) + \left(\phi_{HSC,LSC} - \phi_{LSC,NG}\right)S_{LSC,t} ln\left(\frac{P_{LSC,t}}{P_{NG,t}}\right) + \lambda ln\left(\frac{Q_{HSC,t-1}}{Q_{NG,t-1}}\right) + \left(\alpha_{HSC,t} - \alpha_{NG,t}\right)\ln(G_t) + \sum_{m=2}^{12}\left(\beta_{HSC,m} - \beta_{NG,m}\right)M_{HSC,m} + \sum_{y=2009}^{2015}\left(\gamma_{HSC,y} - \gamma_{NG,y}\right)Y_y + \sum_{NERC=2}^{8}\left(\gamma_{HSC,NERC} - \gamma_{NG,NERC}\right)Y_{HSC,NERC} - \left(d_{HSC,t} - d_{NG,t}\right)D_{HSC,t} + \left(d_{HSC,t-1} - d_{NG,t-1}\right)D_{HSC,t-1} + \left(\epsilon_{HSC,t} - \epsilon_{NG,t}\right)$$

and

$$(25) \qquad ln\left(\frac{S_{LSC,t}}{S_{NG,t}}\right) = (\eta_{LSC} - \eta_{NG}) - \left(\phi_{HSC,LSC}S_{HSC,t} + \phi_{LSC,NG}\left(S_{LSC,t} + S_{NG,t}\right)\right) ln\left(\frac{P_{LSC,t}}{P_{NG,t}}\right) + \left(\phi_{HSC,LSC} - \phi_{HSC,NG}\right)S_{HSC,t} ln\left(\frac{P_{HSC,t}}{P_{NG,t}}\right) + \lambda ln\left(\frac{Q_{LSC,t-1}}{Q_{NG,t-1}}\right) + \left(\alpha_{LSC,t} - \alpha_{NG,t}\right)\ln(G_t) + \sum_{m=2}^{12}\left(\beta_{LSC,m} - \beta_{NG,m}\right)M_{LSC,m} + \sum_{y=2009}^{2015}\left(\gamma_{LSC,y} - \gamma_{NG,y}\right)Y_y + \sum_{NERC=2}^{8}\left(\gamma_{LSC,NERC} - \gamma_{NG,NERC}\right)Y_{LSC,NERC} - \left(d_{LSC,t} - d_{NG,t}\right)D_{LSC,t} + \left(d_{LSC,t-1} - d_{NG,t-1}\right)D_{LSC,t-1} + \left(\epsilon_{LSC,t} - \epsilon_{NG,t}\right).$$

From the resulting coefficients, the short-run cross-price elasticities are developed in Considine and Mount (1984) and Considine (1990) and reported in Jones (1995) as[73]

$$(26) \qquad \varepsilon_{i,j}^{SR} = \left(\phi_{i,j} + 1\right)\bar{S}_j \ for \ i \neq j$$

Coefficients $\phi_{i,i}$ used in deriving own-price elasticities follow from the homogeneity restriction as $\phi_{i,i} = \frac{-\left(\sum_{j\neq i}^{N}\phi_{ji}\bar{S}_j\right)}{\bar{S}_i}$, and thus

$$(27) \qquad \varepsilon_{i,i}^{SR} = \left(\phi_{i,i} + 1\right)\bar{S}_i - 1.$$

Table 7 presents own- and cross-price elasticity estimates for the combined market and separated by abatement technology while the full set of coefficients are reserved for appendix *A3.5 Coefficients from Linear Logit Estimation*. Very inelastic own-price elasticities suggest FGD technology results in lock-in to high sulphur coal, and coal in general. Power plants without abatement technology instead are dedicated to cleaner fuels

[73] Long-run cross-price elasticities follow Jones (1995) as $\varepsilon_{i,j}^{LR} = \frac{\varepsilon_{i,j}^{SR}}{(1-\lambda)}$ but are not needed in this study.

and have less inelastic demand for coal. An implication is that if the regulator wants to increase dirty fuel consumption domestically, power plants with FGD installed are somewhat inflexible on the matter once set up. That is, once power plants install FGD equipment the effect is resilient. The regulator may then want to consider policies that incentivise non-FGD power plants using coal to install FGD systems and switch to dirty coal consumption. In the U.S. energy sector, we arrive at an implementation issue – it is a slow process to retrofit FGD systems onto power plants – taking up to 36 months, with up to six months of the power plant offline (EPA, 1974; Chaaban, Mezher, and Ouwayjan, 2004; Kulshrestha, 2018). But this sort of delay will not be the case for all markets and environmental remedies. I now briefly explore the implications of a few modifications to the model before evaluating the manipulation of abatement costs as policy options.

TABLE 7— SHORT-RUN PRICE ELASTICITIES BY SULPHUR ABATEMENT TECHNOLOGY

| | All power plants | | with FGD | | without FGD | |
|---|---|---|---|---|---|---|
| Own-price | | | | | | |
| High sulphur coal | -0.37 | (0.042) | -0.16 | (0.055) | -0.85 | (0.062) |
| Low sulphur coal | -0.49 | (0.046) | -0.19 | (0.068) | -0.63 | (0.061) |
| Natural gas | -0.31 | (0.036) | -0.79 | (0.165) | -0.26 | (0.025) |
| Cross-price | | | | | | |
| High sulphur coal-low sulphur coal | 0.22 | (0.040) | 0.13 | (0.054) | 0.29 | (0.048) |
| High sulphur coal-natural gas | 0.16 | (0.037) | 0.03 | (0.009) | 0.56 | (0.051) |
| Low sulphur coal-high sulphur coal | 0.17 | (0.031) | 0.17 | (0.067) | 0.06 | (0.010) |
| Low sulphur coal-natural gas | 0.32 | (0.036) | 0.03 | (0.009) | 0.56 | (0.061) |
| Natural gas-high sulphur coal | 0.09 | (0.020) | 0.42 | (0.145) | 0.05 | (0.004) |
| Natural gas-low sulphur coal | 0.18 | (0.019) | 0.47 | (0.140) | 0.05 | (0.005) |

*Notes:* I derive elasticity estimates from the coefficients of the dynamic linear logit model which I estimate on 576 observations at the NERC region scale. Standard errors in parentheses – all elasticities are statistically significant at the 1-per cent level or better. I present the full set of coefficients used to derive these elasticities in appendix *A3.5 Coefficients from Linear Logit Estimation*.

*Source:* Author calculations based on data from EIA (2016a, 2016c).

### *3.4.4 Some Extensions: Emissions Taxes, Permits, and By-Product Sales*

I note the effects of some model extensions: nonlinear cost functions, taxes, emissions permits, and by-product disposal or sales. A nonlinear cost function, e.g. quadratic in one or both inputs, weakens the result. But if a corner solution in either input occurs initially with a nonlinear cost function, the analysis is likely unchanged. If instead, costs are such that a mix of inputs is preferred and the constraint nonbinding ($D \leq \overline{D}$), then the emissions cap and relaxation through technology will not lead to a response anyway. But if the regulation is nonbinding, the circumstances are not particularly interesting to consider. If instead, the demander prefers a mix of inputs such that the cap is binding ($D \geq \overline{D}$), then installation of abatement technology still effectively relaxes it.

Adding a per-emissions unit tax changes the slope of the budget line – steepening it – and increasing the budget requirement. Consider a transformation of the budget function to $B = \tau_{\mathcal{E}}\mathcal{E} + P_D D + P_C C$ where $\tau_{\mathcal{E}}$ is the per-unit emission tax. Since $\mathcal{E} = \delta\alpha D$, the revised expenditure is

$$B = (\tau_{\mathcal{E}}\delta\alpha + P_D)D + P_C C \tag{28}$$

If all firms were to face the same price on the dirty input, the effect of an emissions tax would be either no effect at all on emissions or an entire effect of all firms switching to the clean alternative. However, firms face different prices for inputs, e.g. transportation costs may differ. Realistically, firms face bundles of other fuel price determinants and transportation costs, and we may think of energy and polluting qualities as a function of distance – choice increasing with transportation distance. However, a limited set of options are cost-effective for each firm versus switching to a clean alternative with a different delivery cost. The effect of the per-unit emissions tax is that firms with dirty input prices close to that of the clean alternative may switch, while those with a considerable price advantage continue to favour the dirty input. Within an international context, these costs result in emissions leakage – with higher emissions taxes, more regulated firms shun the dirty fuel, and more is available to export to less regulated markets.

Adding by-product sales has the opposite effect of a tax and influences the efficiency decision of abatement technology.[74] With by-product production, the firm's profit function also includes the cost of purchasing any by-product inputs, but also a potentially profitable price and quantity for the by-product through sales to industry and agriculture. This capacity may incentivise operating at higher efficiency, or installing more efficient technology than necessary to meet a regulatory cap. As an example, He and Lee (2014) find 39-per cent lower sulphur dioxide emissions at power plants following the marketing of products made from the by-product process. Encouraging by-product production may be a useful policy as it incentivises the clean consumption of dirty inputs and potentially induces negative leakage.

[74] We might ask whether by-product sales are a common occurrence. He and Lee (2014) give an overview of this market. Keeping in mind that there is some alternative disposal cost, they model a full firm profit function including the by-product decision. In summary, it is becoming quite common and around the year 2010, the share of total gypsum sales from synthetic means surpassed that from mining in the United States. This was driven by gypsum derived from $SO_2$ capture which accounted for 5 million metric tonnes of the 10 million metric tonnes from synthetic sources and 18 million metric tonnes in total.

Finally, consider tradable permits with an initial allocation.[75] Scale allotments such that one permit, $\bar{Q} = \overline{\mathcal{E}} = \alpha\bar{D}$ (the output with a cap alone). With trade, the polluter can choose to purchase permits beyond their cap and require $D - \bar{D} = e$ and thus $e(\overline{\mathcal{E}}, \delta, \alpha, P_D, P_C)$ when those with abatement technology can abate more than their cap. Then firms may increase abatement efficiency or attempt to acquire inputs with lower polluting potential and sell or buy permits if cheaper to do so. Tradable permits may then result in positive or negative emissions leakage, and perhaps both, due to geographic nuances. But an abatement system may also allow greater leeway in selling excess permits under a tradeable permit system. But then permit quantities must be set such that it is beneficial to abate at a higher efficiency. For systems operated at full efficiency, there is no remaining capacity for this.

Of course, the U.S. and any other modern economy is not an isolated system – a closed economy – all are open to trade to some extent and local effects may be transmitted abroad. The good or bad news, depending on perspective and direction of domestic changes, is that changes in U.S. policy alone do not appear to influence the emission levels from coal-based energy producers in international trade partners. Exploring this statement is beyond the scope of this paper, so the argument and link to other research in this area is instead included in appendix *A3.6 Discussion of Why Domestic Policy May Not Impact Trade Partners*. In the following section I instead return to the state of the U.S. energy sector with a cap in place and consider some policy levers through which a regulator might attempt to influence nationally aggregated emission levels.

### 3.5 Finding Policy Levers: Four Variable Abatement Costs

For output, *E*, the selection of emissions cap, $\overline{\mathcal{E}}$, FGD system passthrough, $\delta$, and polluting content of the dirty input, $\alpha$, determine the effective emissions cap in units of *D*. Within an emissions permit market, the cap also becomes a matter of allotments, *A*, and permit purchases, *e*, as $\bar{D} = \frac{\mathcal{E}(A,e)}{\delta\alpha}|_E$. The output decision at a power plant – the selection of *E* – influences emissions and is well covered in the existing literature. The social planner can conceivably manipulate emissions and output indirectly such as by interfering in transmission networks – manipulate congestion (Bjørndal, Jörnsten, and Rud, 2010; Gao and Sheble, 2010) which is beyond the scope of this paper. Permit prices adjustments and the permit allocation process are other forms of variable costs which are well-researched

[75] In an ideal permit market, anyway. In practice several regulatory and legally mandated changes complicated matters during the U.S. Sulphur dioxide tradable permit program. See a discussion of these in, for instance, Schmalensee and Stavins (2013).

(see for instance Boutabba, Beaumais, and Lardic, 2012 on manipulating permit prices, and Schmalensee and Stavins, 2013 on the allotment process). The two remaining variable costs of abatement, related to consuming fuel with higher sulphur content net of other costs, and of implementing an abatement technology with greater efficiency, are of interest here.

I use a hedonic price model to put some scale on these potential targets of policy leverage. That is, if a policymaker wants to encourage investment in technology that alters abatement decisions, what scale of investment must they aim for? The basis of the price estimates is that for a power plant to consume dirtier fuel, costly operating and facility changes must occur. These costs may be present in the form of higher-priced equipment to process higher sulphur coal, but also higher operating costs which are partially passed on to source mines as well as end-users according to cost incidence. I first estimate through the hedonic price assumption the sulphur abatement costs per mmBTU passed onto mines, and then through calculating incidence, conjecture on the costs incurred by power plants. I then turn to the installation costs of FGD systems and again use a hedonic price framework to estimate the price of system efficiency differences at the time of installation.

For data, I begin with the fuel purchase information used in the prior analyses and append 5,400 power plant observations on installation and status of sulphur dioxide abatement equipment (EIA, 2016a), and data linking coal purchases to mines of origin for roughly 6,900 unique power plant-mine relationships using U.S. Department of Labor, Mine Safety and Health Administration (MSHA) data (2016a, 2016b). The depth of information on fuel purchases needed for this analysis is only available on 2008 and newer observations due to enhanced power plant reporting requirements on Form EIA-923 (EIA, 2016c). I then use data from the EIA on the full history of power plant installations of FGD systems in EIA (2016a) – over 500 installations since 1948 – to decompose system costs. As before, I carefully consider units of measure and perform all comparisons in terms of per-heat content of fuel in mmBTU's as discussed in appendix *A3.1 Data Preparation*.

In the spirit of Rosen (1974) and building on Busse and Keohane (2007) and He and Lee (2016), I first estimate costs related to sulphur content, $\alpha$. I assume the price of fuel deliveries, $p$, are additively separable. The baseline set of coal descriptors, $\boldsymbol{X}$, are sulphur and ash content in pounds per mmBTU as primary environmental concerns, heat content in mmBTU per ton and delivery distance as primary concerns of production, and total order quantity in mmBTU included to account for bulk discounts and market power. Year, $\boldsymbol{Y}$**,** and month, $\boldsymbol{M}$, indicators account for market trends and cyclical events. I also use a set of year indicator variables interacted with sulphur content as a proxy for pollution permits, and power plant fixed effects, $\Phi$, to account for differences in management and negotiating power, among other more consistent power plant factors. A set of interaction, $\boldsymbol{I}$, and own-quadratic terms, $\boldsymbol{Q}$, account for complex trade-offs between fuel characteristics and price.[76] Coalmine state of origin indicators control for supply-side regulation, though these may account for much of the same variation as distance (their interpretation is not of immediate interest). I also take the opportunity to add an indicator for by-product production following He and Lee (2016), and $SO_2$ controls (FGD system) usage following Busse and Keohane (2007). They argue that, with FGD system and by-product production, firms have committed – are locked in – to higher sulphur coal and mines are aware of these statuses and price discriminate. I have suggested in this paper that it is really about flexibility – being able to accommodate a wider selection of coal and the negotiating power this results in. Finally, I account for the countervailing matter of increased operating costs passed to mines when using higher sulphur coal with FGD systems in place using interaction term *$SO_2$Controls*Sulphur*. I discuss this interaction as the abatement variable cost (AVC). It is the share of the increased operating cost that mines incur as a discount on the fuel price during negotiations. From abatement variable cost and incidence estimates, I then derive a back-of-the-envelope confidence level estimate for the scale of offset that would encourage an average power plant to double sulphur content. Following Carver (1924), the mine cost incidence is $\frac{\varepsilon_p}{\eta_m+\varepsilon_p}$ for power plant input-price elasticity of demand, $\varepsilon_p$, and mine price elasticity of supply $\eta_m$. I then weight by its inverse to estimate the total abatement variable cost.[77] I estimate the full specification as

$$(29)\quad P_{d,p,t} = \beta_0 + \beta_1 SO_2\mathrm{C}ontrols_{p.t} + \beta_2\left(SO_2\mathrm{C}ontrols_{p,t} * Sulfur_{d,p,t}\right) + \boldsymbol{X}_{d,p,t}\beta + \\ +\boldsymbol{Minestate}_d\beta_s + \beta_b\boldsymbol{Byproduct}_{p,t} + \boldsymbol{I}_{d,p,t}\beta_{xi} + \boldsymbol{Q}_{d,p,t}\beta_{xq} + \boldsymbol{Y}\beta_y + \boldsymbol{M}\beta_m + \\ \left(Sulfur_{d,p,t} * \boldsymbol{Y^T}\right)\beta_{ys} + \Phi_p + \varepsilon_{d,p,t}$$

[76] For Distance in 100's of miles, BTU's in mmBTU per short ton, Quantity of order in mmBTU, and Sulphur and Ash content in pounds per mmBTU; interactions, $\boldsymbol{I}$, are Distance*BTU's, Distance*Quantity, Distance*Sulphur, Distance*Ash, BTU's*Quantity, BTU's*Sulphur, BTU's*Ash, Quantity*Sulphur, Quantity*Ash, Sulphur*Ash; and quadratics, $\boldsymbol{Q}$, are squared values of Distance, Sulphur, Ash, BTU's and Quantity.

[77] I also note that $1-\frac{\varepsilon_p}{\eta_m+\varepsilon_p}=\frac{\eta_m}{\eta_m+\varepsilon_p}$ is the sum of power plant plus end user incidence. But is this how the power plant operator views it, or do they negotiate based only on what they will incur? As power plants are an intermediary that negotiates on both supply and demand, they might instead incur share $\frac{\eta_m\varepsilon_c}{(\eta_m+\varepsilon_p)(\eta_p+\varepsilon_c)}$, denoting $\varepsilon_c$ as the end user price elasticity of demand. For a public, not-for-profit oriented utility, we might expect the former estimate, and a wholly private and loosely regulated utility the latter. This extension, however, does not impede our ability to roughly estimate total abatement costs from that incurred by mines.

for fuel delivery, *d*, to a power plant, *p*, in a month, *t*. I then estimate a full set of comparison statistics in the form of a specification chart on the sulphur dioxide AVC. The goal is to assess the stability of the estimate to changes in functional form. The baseline form is

$$(30) \quad P_{d,p,t} = \beta_0 + \beta_1 SO_2Controls_{p.t} + \beta_2\left(SO_2\text{C}ontrols_{p,t} * Sulfur_{d,p,t}\right) + \boldsymbol{X}_{d,p,t}\beta + +\boldsymbol{Y}\beta_y + \Phi_p + \varepsilon_{d,p,t}$$

and $2^6 = 64$ combinations drawn from ($\boldsymbol{Minestate}_d\beta_s$, $\beta_b\boldsymbol{Byproduct}_{p,t}$, $\boldsymbol{I}_{d,p,t}\beta_{xi}$, $\boldsymbol{Q}_{d,p,t}\beta_{xq}$, $\boldsymbol{M}\beta_m$, $\left(Sulfur_{d,p,t} * \boldsymbol{Y^T}\right)\beta_{ys}$, {}) are appended to make the specification chart set.

The full and baseline specification estimates are reported in Table 8, followed by the specification chart as Figure 28. Baseline and full specification estimates are average variable costs of -14.7 cents and -13.3 cents per mmBTU, and the mean of the $2^6$ estimates in the specification chart is -14.1 cents per mmBTU. This is a roughly 6-per cent discount from the mean and median delivered prices of 250.5 and 230.8 cents per mmBTU and does not include any offsetting by-product sales. These are the mine incidence (discount) with a roughly doubling of coal sulphur content as FGD-enabled power plants on average use coal with two pounds of sulphur content per mmBTU rather than one. Estimates across all $2^6$ specifications are also statistically significant, though there is a great deal of variability in the heterogeneous energy sector. The preceding linear logit-based market elasticity estimate was roughly $\varepsilon_P \approx -0.4$, and national estimates of $\eta_m$ can be found in the literature. Zimmerman (1977) and Zimmerman (1981) provide estimates of 2.03 and 3.58, respectively, from cost analyses approaches.  Boeters and Bollen (2012), and Knittel, Metaxoglou, Soderbery, and Trindade (2019) find estimates of 3.92 and 3.6 to 3.7 by market simulations, and EIA (2020e) reports that a value of 5.0 is used in agency market analyses. The estimated variable abatement cost range is then $13.3/\left(\frac{0.40}{2.03+0.40}\right) = 80.8$ cents per mmBTU to $14.7/\left(\frac{0.40}{5.0+0.40}\right) = 198.5$ cents per mmBTU. These are in the ballpark of estimates of 146.5 to 234.5 cents per mmBTU in Srivastava and Jozewicz (2001), but lower than much earlier estimates in Devitt, Yerino, Ponder, and Chatlynne (1976) and Tilly (1983) in present value terms which may be the result of learning and innovation.[78]

[78] Estimates in Devitt, Yerino, Ponder, and Chatlynne (1976) of 167.3 to 458.4 cents per mmBTU, and Tilly (1983) of 140.7 to 378.1 cents per mmBTU (not CPI adjusted). In the former, the estimate is converted from annual dollars per kilowatt, and in the later from mills per kilowatt-hour (including a conversion factor of 1 kwh=0.003412 mmBTU).

TABLE 8— HEDONIC MODEL OF DELIVERED COAL PRICES

| | Baseline | Full |
|---|---|---|
| $SO_2$ abatement variable cost[a] | -14.71*** | -13.34*** |
| | (4.97) | (4.48) |
| $SO_2$ abatement fixed cost[b] | 7.24 | 7.49* |
| | (4.63) | (4.39) |
| Heat content[c] | 14.90*** | 4.57 |
| | (1.06) | (18.17) |
| Total order quantity[d] | -3.34e-06*** | -2.07e-06 |
| | (7.74e-07) | (7.55e-06) |
| Sulphur content[e] | -9.58** | 25.73 |
| | (4.59) | (23.84) |
| Ash content[e] | 2.30*** | -3.15 |
| | (0.49) | (5.04) |
| Distance[f] | 0.79 | 9.80 |
| | (0.51) | (6.21) |
| Distance squared[f] | - | -0.13*** |
| | - | (0.04) |
| By-product production capacity[g] | - | -8.88 |
| | - | (6.66) |
| Constant | -106.10*** | -63.61 |
| | (26.99) | (235.50) |
| Additional control variables[h] | | |
| Plant level fixed effects | Yes | Yes |
| Year effects (2008 base) | Yes | Yes |
| Quadratic terms | No | Yes |
| Interaction terms | No | Yes |
| Year*sulphur content | No | Yes |
| Month-year (January 2008 base) | No | Yes |
| Coal mine state of origin (22 states) | No | Yes |
| Observations | 94,559 | 94,559 |
| R-squared | 0.72 | 0.74 |

*Notes:* The dependent variable is the price of coal per mmBTU. Robust standard errors in parentheses, clustered at the power plant level. *** Significant at the 1 per cent level, ** Significant at the 5 per cent level, * Significant at the 10 per cent level.

[a] FGD in operation*sulphur content. [b] Equals one if FGD in operation. [c] In mmBTU/short ton. [d] In mmBTU. [e] In pounds/mmBTU. [f] In 100's of miles. [g] Equals one if capable of by-product recovery.

*Source:* Author calculations based on EIA (2016a, 2016c) and MSHA (2016a, 2016b).

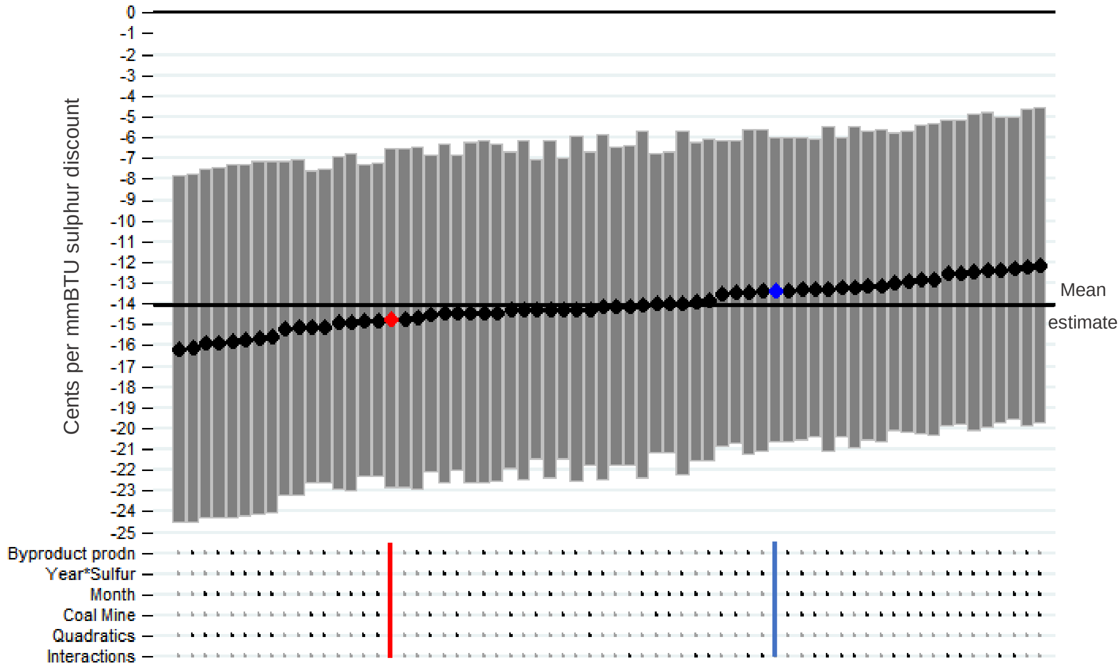


FIGURE 28 SPECIFICATION CHART FOR HEDONIC MODEL OF FGD VARIABLE COSTS

*Notes*: Estimates based on $2^6$=64 combinations of dependent variables in addition to base specification in Equation (30). The baseline estimate is denoted with a red marker and the full specification estimate with blue. The dependent variable is the price of coal per mmBTU and estimates are based on 94,559 coal deliveries at 321 power plants in years 2008 to 2015, with standard errors clustered at the power plant level.

*Source*: Author calculations based on data from EIA (2016a, 2016c) and MSHA (2016a, 2016b).

The second marginal abatement cost I explore is associated with a change in FGD system designed efficiency at the time of installation. This requires a longer planning horizon to utilise for geopolitical purposes. I use FGD installation prices and estimate the cost associated with the parameter $\delta = (1 - eff)$ by estimating on changes in system designed efficiency directly. As the dependent variable, I used power plant reported FGD installation prices from EIA-860 operation reports (EIA, 2016a) which are consumer price index (CPI) adjusted to 2016 dollars as installations occurred over the period 1948 to 2016. I again assume the price to be additively separable into component costs as fitting with Rosen's (1974) framework. I use gas exit rate as a proxy for system capacity, installation year, sulphur and ash content per cent design specification, and an indicator for whether the system produces a by-product such as calcium carbonate. Additional controls are vectors to separate by FGD system type, manufacturer, and sorbent used. Finally, the coefficient of interest is the design efficiency of the system, *Efficiency*. The full specification is

$$(31)\quad FGDcost_i = \beta_0 + \beta_1 Efficiency_i + \beta_2 GasExitRate_i + \beta_3 Sulphur_i + \beta_4 Ash_i + \beta_5 Byproduct_i + \boldsymbol{FGDtype}_i \beta_u + \boldsymbol{Manufacturer}_i \beta_v + \boldsymbol{Sorbent}_i \beta_w + \beta_y InstallationYear_i + \varepsilon_i$$

for FGD installation *i*. Some power plants have more than one installation over the period (as many as eight), but these are often installed several years apart. The average is two installations per power plant and the mode is one. The result is then that a power plant fixed effect is not particularly informative compared to the amount of variation discarded.

I again build a specification chart, here on the baseline form

$$(32)\quad FGDcost_i = \beta_0 + \beta_1 Efficiency_i + \beta_2 GasExitRate_i + \beta_y InstallationYear_i + \varepsilon_i$$

and, coincidentally, $2^6 = 64$ combinations drawn from ($\beta_3 Sulphur_i$, $\beta_4 Ash_i$, $\beta_5 Byproduct_i$, $\boldsymbol{FGDtype}_i \beta_u$, $\boldsymbol{Manufacturer}_i \beta_v$, $\boldsymbol{Sorbent}_i \beta_w$, {}) appended. Table 9 reports baseline and full specification estimates, followed by the specification chart in Figure 29. I estimate \$1.1 million, \$1.4 million, and \$1.3 million per-per cent increase in system efficiency from the average as the baseline, full specification, and specification chart mean estimates, respectively. For context, mean FGD system cost is \$105.9 million (2016 dollars), and the median is \$28.3 million – indicating positive skew in the distribution, and mean and median removal efficiencies are 88.0-per cent and 90.4-per cent, respectively, with a standard deviation of 10.8. These values suggest a substantial upfront investment to commit to changes in a system that remains in operation for decades. If a regulator's interest is in reserving efficiency as an international policy instrument, one could imagine subsidising greater efficiency at installation. But then, the regulator must face the issue of whether it makes sense to bar operating systems at full efficiency until needed.

TABLE 9— HEDONIC MODEL OF FGD INSTALLATIONS AT U.S. POWER PLANTS

| | Baseline | Full |
|---|---|---|
| Sulphur removal efficiency[a] | 1.10*** | 1.43 |
| | (0.43) | (0.89) |
| Gas exit rate[b] | 0.09*** | 0.10*** |
| | (0.01) | (0.02) |
| Specification of sulphur[c] | - | -2,564 |
| | - | (5,787) |
| Specification of ash[c] | - | -523.9 |
| | - | (1,419) |
| By-product production[d] | - | 3,259 |
| | - | (27,111) |
| FGD installation year | -511.90 | 1,927 |
| | (626.60) | (1,305) |
| Constant | 932,154 | -70,921 |
| | (1,276,006) | (178,927) |
| Additional control variables | | |
| FGD type controls | No | Yes |
| Sorbent type and FGD manufacturer | No | Yes |
| Observations | 568 | 400 |
| R-squared | 0.32 | 0.56 |

*Notes:* Dependent variable is the price of FGD installation in 2016 dollars. Estimates include installations at 294 power plants in years 1948 to 2016. Robust standard errors in parentheses. *** Significant at the 1 per cent level, ** Significant at the 5 per cent level, * Significant at the 10 per cent level.

[a] An engineering specification. [b] In cubic feet per minute. [c] In per cent by weight. [d] Equals one if the installed equipment is capable of by-product production.

*Source:* Author calculations based on data from EIA (2016a, 2016c) and MSHA (2016a, 2016b).

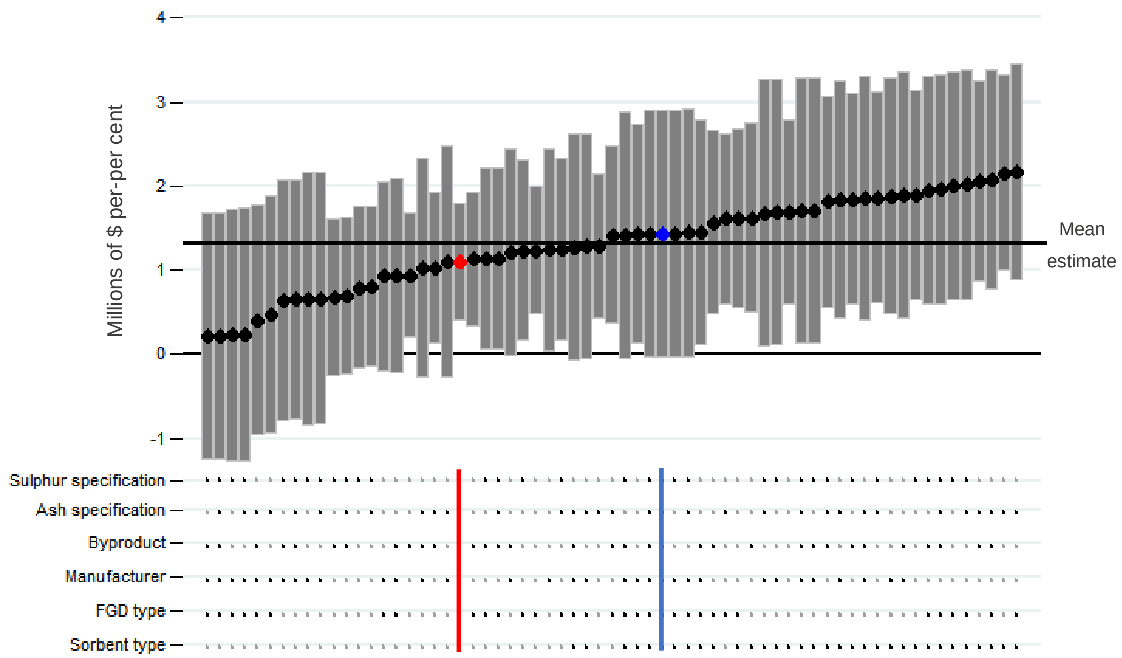


FIGURE 29 SPECIFICATION CHART FOR HEDONIC MODEL OF FGD INSTALLATIONS

*Notes*: Estimates based on $2^6$=64 combinations of dependent variables in addition to the base specification in Equation (32). I denote the baseline estimate with a red marker and the full specification estimate with blue. The dependent variable is the price of FGD installation in 2016 dollars, and estimates are of the per-per cent FGD efficiency price change in millions of dollars. Estimates based on 400 to 568 installations at 294 power plants in years 1948 to 2016 based on the specific run's specification.

*Source*: Author calculations based on data from EIA (2016a, 2016c) and MSHA (2016a, 2016b).

Having put estimates to the highlighted energy sector abatement costs, I now conclude the discussion by summarising the results.

### 3.6 Conclusion

In this paper I have explore the U.S. energy sector's transition away from being a source of substantial sulphur dioxide emissions partially by using abatement technologies. This transition was undertaken to address what was perhaps the most pressing environmental concern in the U.S. for a time. As several nations begin another great energy transition – away from fossil fuel-based energy production as we know it altogether – it seems an important moment to reflect on and explore in depth the mechanisms and incentives behind the last transition. This paper has focussed entirely on the sulphur dioxide issue from the input side – how changes in the sulphur content of coal and method of power plant consumption led to changes in resulting emissions. But it is not to be taken entirely retrospectively, rather to inform the policy development that lies ahead.

To summarise the econometric results, this paper shows that sulphur demand at U.S. power plants differs by the abatement strategy chosen. The abatement decision is, however, endogenous – the result of power plant and supply characteristics. Even so, the contrast in sulphur demand between FGD-enabled power plants and those without is substantial. I show that differences in the consumption of coal sulphur content also occur within power plants – at the boiler infeed level which is the most refined scale both relevant and possible. These within-power plant differences mirror those between power plants that specialise in production with or without FGD systems installed. I then show that the mean sulphur content of coal consumed, as well as the set of mines supplying a power plant, change with an FGD system's installation. The timing of FGD installation does not strictly coincide with national scale policy changes for a variety of power plant operational reasons. Regardless, it is the installation and start-up of abatement technology that allows power plants to consumer higher sulphur coal cleanly and thus increase demand for it.

I then present a simplified model of U.S. energy producers and find some empirical support for the resulting proposition that they tend to treat high sulphur coal versus low sulphur coal or natural gas as perfect substitutes. During this process I find an approximate scale for heterogenous hidden costs or other factors impacting when power plants choose to switch fuels. I then show that market level differences between FGD and non-FGD power plants are apparent in their market demand elasticities. I ex ante expected that power plants with abatement technology make fuel purchase decisions based on high sulphur coal prices, and those without are more responsive to the price of natural gas and low sulphur coal. However, I also find that power plants with FGD systems have more inelastic demand because they have less incentive to substantially change fuels. Finally, I identify at least four forms of variable abatement costs that power plants incur with FGD types of abatement technology, and I put estimates to the two least studied ones. These are related to changes in coal sulphur content and abatement system efficiency. Such estimates inform on the scale of intervention needed to incentivise fuel changes in support of policy objectives.

Avenues of further exploration abound despite the breadth of the existing literature and this contribution. These include further consideration of the abatement adoption decision under risk, uncertainty, and learning. An appendix to this paper outlines why the domestic effects explored here are unlikely to lead to international implications – itself a growing field of study. Application of the domestic model to other nations and pollutants as well as greater integration with models of negotiation and agency must also be considered. Finally, any estimates of elasticities or prices – such as those in this paper – must periodically be updated to reflect changing market dynamics. Hopefully this analysis supports such inquiries, newer energy transitions, and the regulation other pollutants.

# Part II: Inequality Trends in Economic Growth

## 4. An Approach to Measuring Inequality Growth Vectors

*I bridge the divide between theoretical and empirical studies of inequality by presenting a method to estimate a coefficient that is not quite the elasticity of marginal utility of consumption, $\eta$. Calling it implied-$\eta$, I discuss how it compares to true measures of $\eta$, methods to estimate it, and how it reports on inequality in the same units as used in some theories of inequality preferences. I provide national long-run estimates for 135 countries based on four datasets. I also discuss using the volatility of implied-$\eta$ over time as another measure of interest and derive some intra-society estimates in the same units as $\eta$.*

Keywords: Inequality, economic growth, distribution, the elasticity of marginal utility, preferences.

JEL codes: D63, O43, P17, P27

## 4.1 Introduction

Estimates of social inequality or economic growth imply a social preference. This preference results from bargaining between society's members over the structure of society. This connection is at least true if we are to believe that representative government represents. Yet in the study of social preferences over inequality and several other matters, an entirely different, theory-based statistic is used rather than any of our many measures of inequality in practice. This statistic is generally the elasticity of marginal utility of consumption, $\eta$.[79] In this paper I do not estimate $\eta$ in the traditional sense – as a stable and perhaps shared social preference. Instead, I estimate the $\eta$ implied by inequality trends in economic growth. In doing so this paper bridges the two literatures. I refer to the resulting estimate as *implied-$\eta$*.

Implied-$\eta$ informs on the net effect of economic and social policies involved in economic growth. It enables a determination of whether these policies lead to distributional results that match social preferences. But an implied-$\eta$ estimate is not the same as a true measure of $\eta$ because there are stochastic, and at least exogenous, elements impacting economic growth. Over long periods, however, when the highs and lows of economic growth even out and we estimate on the average trend, implied-$\eta$ is comparable to $\eta$ estimates. When estimating over shorter periods, such as on year-over-year growth, this new method instead tells us something unique, in comparison to traditional $\eta$ estimates, about the structure of the economy. That is, if the practitioner can overcome matters of measurement error in their data. As will be explored, both long- and short-run estimates almost certainly have implications for the study of inequality, instability, and redresses to the rise of populism.

### *4.1.1 A Brief Literature Review*

As an introduction, there are many ways to estimate $\eta$ which is generally taken, for better or worse, as a constant or a stable average of society's preferences over time. Dasgupta (2008), Atkinson and Brandolini (2010), and no doubt others note that $\eta$ is almost certainly not constant across the income distribution but is taken as such for analytical tractability – as an average. Estimates generally, but not exclusively, fall within $\eta = [1, 2]$ and rarely deviate from $\eta = [0.5, 3]$. Higher values imply a preference for a more equal society – less

[79] Often defined as $\eta = -\frac{\bar{c}(t)u''(\bar{c}(t))}{u'(\bar{c}(t))}$ for some clearly twice-differentiable function representing the utility of consumption.

tolerance for policies that increase inequality, among other interpretations, and conversely so.[80]

Evans (2005) provides a list of revealed preference methods and estimates on 20 OECD countries based on Stern's (1977) national tax data approach. It has become a benchmark in the literature. Groom and Maddison (2019) provide a useful comparison of the five revealed preference methodologies in action for the U.K. – the variety most comparable in principle to the method in this paper. There are several other examples of $\eta$ estimates based on tax, demand, consumption, or other preference revealing bases, individually or through public choice. Among these are estimates on 17 Latin American countries in Moore, Boardman, and Vining (2020), six transitioning Eastern European countries in Seçilmiş and Akbulut (2019), and individual country estimates in Banks, Blundell, and Lewbel (1997), Blundell (1988), Blundell, Pashardes, and Weber (1993), Cowell and Gardiner (1999), Evans (2004a), Evans (2004b), Evans, Kula, and Nagase (2014), Evans, Kula, and Sezer (2005), Evans and Sezer (2002), Evans and Sezer (2005), Kula (1984), Kula (2004), Moore, Boardman, and Vining (2013), Percoco (2008), Stern (1977), and no doubt others.[81] The reader will find implied-$\eta$ estimates are generally higher than tax-based or other revealed preference approaches because we are measuring something different.

As further comparisons, Layard, Mayraz, and Nickell (2008) derive estimates of $\eta$ based on happiness surveys and arrive at a lower mean value of 1.26 across 50 countries compared to the mean of the preceding contributions of 1.38 across the 40 national averages. In the other direction, stated preference-based estimates can be substantially higher. In, for example, Carlsson, Daruvala, and Johansson-Stenman (2005), Johansson-Stenman, Carlsson, and Daruvala (2002), and Pirttilä and Uusitalo (2010) estimates approach a high of three. But this is not consistently so, for example Amiel, Creedy, and Hurn (1999) find estimates well below what Weitzman (2007) considers to be the lower bound that any reasonable economist would consider of $\eta = 1$. However, Pirttilä and Uusitalo (2010) demonstrate that stated preference approaches are highly sensitive to how hypothetical questions are posed, in their case resulting in estimates of $\eta < 0.5$ and $\eta > 3$ for the same group depending on how the question is presented.[82]

[80] Other uses of the elasticity of marginal utility of consumption include as a representation of risk aversion (across states of the world), prudence (aversion to downsides), fluctuation aversion (preferences for lifetime consumption smoothing), and in practical matters of long-run public planning.

[81] These estimates are available in appendix *A4.1 Comparison of $\eta$ and Implied-$\eta$ Estimates* alongside the applied-$\eta$ estimates from this paper, where available.

[82] These questions are generally based on Okun's (1975) concept of a leaking bucket and one's willingness to tolerate a loss in the transfer of assets. Such a notion can certainly be presented and interpreted differently.

Estimating $\eta$ by various methods occurs somewhat frequently when sufficient privileged data is available. Unfortunately, such estimates are undertaken as an entirely distinct endeavour from the measurement of inequality. Yet societal preferences summarised in $\eta$ result in social policies impacting inequality.[83] It is not then the purpose of this paper to compare methods of estimating $\eta$, but rather to reconcile them through an alternative that measures the result of competing preferences – the societal net effect of the choices that drive all estimates of $\eta$ and the progression of inequality.

In fact, existing methods are all fine and follow a pattern I do not exceptionally deviate from: In return for accepting some structure, the practitioner receives estimates of $\eta$, here implied-$\eta$, from not quite equivalent perspectives. Generally, users compare estimates derived from competing methods and use them, for instance, in the Ramsey Rule (Ramsey, 1928; see also an intuitive discussion in Gollier, 2012). This use has a profound impact on social and, in particular, intergenerational planning. The Ramsey Rule provides an expression for the risk-free social rate of time preference. This is the social discount rate used by several nations, including the U.K., and organisations such as the International Panel on Climate Change. A more thorough discussion of its uses and interpretations can be found in Groom and Maddison (2019). Critically, an $\eta$ of one versus two – both common estimates – has a substantial impact on public planning. This is particularly important in questions involving substantial periods before the returns to effort are realised, such as on global climate change mitigation expenditures and decisions on the handling and storage of fissile waste. For instance, in Tol (2010), the social cost of carbon (SCC) is sensitive to assumptions about the rates of risk and inequality aversion, with a higher $\eta$ implying a preference for a substantially higher SCC. Yet current estimates of $\eta$ come with significant limitations.

### *4.1.2 Issues in the Measure of $\eta$*

One issue that hampers the use of $\eta$ in social planning is that estimates are based on tax or demand data, happiness reports, or hypothetical surveys on a limited subset of the population (too often on university students in the latter case). These bases each offer a partial, limited picture of the economy and we do not observe the whole, net effect of competing preferences. It can very well be that individuals prefer a more equal society –

[83] For instance, the social discount rate (SDR), sometimes known as the Ramsey rule (based on Ramsey, 1928) and the social rate of time preference (SRTP), is $SDR = \delta + \eta g_t$ for the pure rate of time preference, $\delta$, and growth rate, $g$, over time horizon $t$. If $g_t > 0$, then future generations are better off than earlier ones. In equilibrium we would then seek to discount the future more to shift benefits to earlier, poorer generations. For a larger $\eta$ – implying greater inequality aversion – we discount the future more.

suggested by some stated preference estimates, but also prefer greater personal income to use toward finding happiness – suggested by happiness-based estimates, and also support a national tax policy that balances these as well as other interests, perhaps unequally, in determining the state's direct impact on consumption and inequality. Another issue that hampers the estimation of a global picture of inequality preferences is that each existing estimation procedure requires some level of access to privileged data. One must, tautologically, have access to national tax data to estimate on national tax data as in Stern's (1977) approach.

In response to these issues, I develop a method based on publicly available economic growth statistics to derive the $\eta$ implied by the distribution of economic progress. These estimates are in comparable units to the indicators of preference derived by revealed, stated preference, and other approaches, but do so by observing the result of the whole economic process. In another application of the framework, I derive measures based on the volatility of economic growth. So, while preferences may be stable, as my predecessors and I generally assume in $\eta$, the preference structure I find implied by inequality trends are rarely stable as short-run implied-$\eta$ estimates deviate from long-run ones. An implication is that most economic growth policies do a poor job of matching social preferences during economic booms and busts.

I next outline the limited structure we must accept for the estimation strategy to yield meaningful results. I then estimate long-run implied-$\eta$'s for several – 135 – countries using four datasets. These are followed by a discussion of short-run estimation and the sensitivity of the approach to variation in the data. Finally, I extend the discussion to comparable-valued estimates of changes in inequality between social groups within society.

### 4.2 Methodology

The underlying method is not particularly complex, nor the estimation strategy itself insightful, and I unabashedly refer them as ad hoc. In effect, an economy is "black-boxed" – we jump to the conclusion of the economic process and observe the sharing of the spoils of the aggregate economic process. These traditions of redistribution, perhaps sensitive to whether the economy is growing or shrinking, may reveal social preferences. In this section, I outline the minimum required structure, followed by the implications I use in relating inequality preferences to economic growth trends. I then outline an estimation strategy and suggest how to compare this strategy and resulting estimates to other estimates of $\eta$.

### *4.2.1 Structural Assumptions*

A common statistical assumption provides much of the needed structure: incomes or consumption, $c_i$, are lognormal (*LN*) (McAlister, 1879) distributed in society in each $t$ period of observation, $c_i(t) \sim LN(\mu(t), \sigma^2(t))$. It is perhaps worth noting that any distribution can be approximated by the lognormal or any other distribution, and various tests beyond the purpose of this paper to discuss suggest whether the approximation is appropriate. A limitation of this estimation method, however, is that the population of interest must be sufficiently large to be described well by the *LN* distribution. Assuming *LN* distributed $c$ is powerful. Analytical estimates for mean, median, and modal values of the distribution are $\bar{c} = exp(\mu + \frac{1}{2}\sigma^2)$, $\tilde{c} = exp(\mu)$, and $c^{Mo} = exp(\mu - \sigma^2)$, respectively. Extending from Emmerling, Groom, and Wettingfeld (2017), respective growth rates are $\bar{g}_t = \frac{1}{t}\left(\Delta\mu_{0,t} + \frac{1}{2}\Delta\sigma^2_{0,t}\right)$, $\tilde{g}_t = \frac{1}{t}\Delta\mu_{0,t}$, and $g^{Mo}_t = \frac{1}{t}\left(\Delta\mu_{0,t} - \Delta\sigma^2_{0,t}\right)$ where $\Delta\mu_{0,t} = \mu_t - \mu_0$ and $\Delta\sigma^2_{0,t} = \sigma^2_t - \sigma^2_0$. While $\tilde{g}_t$ is driven by changes in $\mu$ – so I also refer to it as $g_\mu$ – I refer to changes in the residual of the mean, $\bar{g}_t - \tilde{g}_t = \frac{1}{t}\frac{1}{2}\Delta\sigma^2_{0,t}$ as inequality-driven growth and by $g_{\sigma^2}$.

From the lognormal analytical values, it follows that $\bar{g}_t = \tilde{g}_t = g^{Mo}_t$ when $\bar{g}_t - \tilde{g}_t = 0$. The implication is that no inequality-driven growth occurs if the economy is structured to maximise median growth, $max\, \tilde{g}_t$, resulting from a distribution-independent wealth-generating process.[84] I also note that a vector in $\left(\frac{1}{2}\sigma^2, \mu\right)$ space that maximises mean growth, $max\, \bar{g}_t$, maximises the rate of ascent through $\bar{c}$ level sets.[85] The vector orthogonal to $\bar{c}$ from any initial $\left(\frac{1}{2}\sigma^2_0, \mu_0\right)$ point, then, suggests the optimal growth path and exhibits the relation $\Delta\mu_{0,t} = \frac{1}{2}\Delta\sigma^2_{0,t}$ (see Note 7). There can also be vectors in this space based on preferences over distribution during degrowth. $\Delta\mu_{0,t} = 0$ follows from median degrowth minimization, and I assume the preference for wealth preservation similarly mirrors wealth creation. Importantly, I do not suppose that we actually choose $\sigma^2$ and $\mu$ as a society, but rather these are salient proxies for society's choice of institutional structures responsible for distribution and growth.

[84] For instance, growth may be driven by total, or per-worker savings as generally assumed in aggregate growth models. It may also occur stochastically. The growth process itself is outside the scope of this paper just as many models of economic growth instead ignore distribution.

[85] Or through level sets of a preference-preserving transformation such as $ln(\bar{c})$. We might also use $(\sigma^2, \mu)$, and in either case derive comparable estimates from consumption data as $(ln(\bar{c}) - ln(\tilde{c}), ln(\tilde{c}))$.

**Note 7 – The Optimal Growth Path in $\left(\frac{1}{2}\boldsymbol{\sigma^2}, \boldsymbol{\mu}\right)$ Space**

Suppose instead of treating the distribution of growth $\left(\frac{\partial\sigma^2}{\partial t}, \frac{\partial\mu}{\partial t} \geq 0\right)$ from the perspective of annual national statistics, we treat it more accurately as a continuous process. The unconstrainted (over distribution) proxy objective is $\max_{\mu(t),\sigma^2(t)} \bar{g}_t = \frac{\dot{\bar{c}}}{\bar{c}}$ resulting in optimality condition $\frac{\partial\mu(t)}{\partial t} = \frac{1}{2}\frac{\partial\sigma^2(t)}{\partial t}$ which is the continuous equivalent of $\Delta\mu_{0,t} = \frac{1}{2}\Delta\sigma^2_{0,t}$. As a perhaps unnecessary proof, in the continuous case $\frac{\partial\bar{c}}{\partial t} = \bar{c}\left(\dot{\mu} + \frac{1}{2}\dot{\sigma}^2\right)$, so $\frac{\dot{\bar{c}}}{\bar{c}} = \dot{\mu} + \frac{1}{2}\dot{\sigma}^2 = \bar{g}$. Then, $\bar{g}$ are level sets in $\left(\frac{1}{2}\dot{\sigma}^2, \dot{\mu}\right) = \left((\bar{g} - \tilde{g}), \tilde{g}\right)$ space with form $\dot{\mu} = \bar{g} - \frac{1}{2}\dot{\sigma}^2$. Choose the distribution of $\dot{\mu}$ and $\frac{1}{2}\dot{\sigma}^2$ to maximize growth in $\bar{g}$ which is the transition through $\bar{g}$ level sets. From the right triangle altitude theorem, also known as the geometric mean theorem (Euclid, Elements book four, 360-280 BC), define $f = \frac{\dot{\mu}\frac{1}{2}\dot{\sigma}^2}{c}$ for length of a level set, *c*, and its altitude, *f*, in $\left(\frac{1}{2}\dot{\sigma}^2, \dot{\mu}\right)$ space. So, $\max_{\{\dot{\mu}, \dot{\sigma}^2\}} \frac{\dot{\mu}\frac{1}{2}\dot{\sigma}^2}{c}$ subject to $\bar{g} = \dot{\mu} + \frac{1}{2}\dot{\sigma}^2$. From the resulting Lagrangian with shadow price, $\lambda$, interior solutions satisfy system $\frac{\frac{1}{2}\dot{\sigma}^2}{c} - \lambda = 0$, $\frac{\dot{\mu}\frac{1}{2}}{c} - \lambda\frac{1}{2} = 0$, and $\bar{g} = \dot{\mu} + \frac{1}{2}\dot{\sigma}^2$. So, $\frac{1}{2}\dot{\sigma}^2 = \dot{\mu}$ subject to a $\bar{g}$ (and that $\dot{\mu} = \frac{1}{2}\dot{\sigma}^2 = \frac{1}{2}\bar{g}$).

### *4.2.2 Deriving Critical Points*

Values of $\eta$ take on special significance relating to mean and median growth, at least in a certain class of utility functions. In particular, $\eta = 2$ implies a median maximising growth preference, and $\eta = 1$ a preference for consumption potential or mean maximising growth. I derive these implications structurally using the constant relative risk aversion (CRRA) representative utility form (Atkinson, 1970) which is usually defined, as in Dasgupta (2008), as

$$u(c_i(t)) = \begin{Bmatrix} c_i(t)^{1-\eta}(1-\eta)^{-1}, \; if\ \eta > 0, \eta \neq 1 \\ \ln(c_i(t)), \quad if\ \eta = 1 \end{Bmatrix} \tag{33}$$

and a discounting-free, more than one-period welfare maximisation objective such as

$$max\, W = \sum_{t=0}^{T} \sum_{c=0}^{C_{max}} u\left(c_i(t)\right) \tag{34}$$

where $c_i(t) \sim LN(\mu(t), \sigma^2(t))$. A more comprehensive model is presented in the next chapter, *Mean-Spirited Growth*, that includes discounting, a fully specified objective, and additional constraints. The primary outcome does not change, but rather results in growth pathways centred on the vectors I describe here. I assert that the specific controls by which we distribute the result of our distribution-independent growth process are not important at present and so I do not state them in the objective. As will be seen, the implied-$\eta$ estimates that result can exceed the bounds of $\eta > 0$ required in Equation (33). This occurs as economies may distribute the returns to economic growth in ways that do not match any rational CRRA-based preference.

From the CRRA utility form, $\eta = 1$ implies proportional changes in $c_i$ are of equal utility value across the distribution as also noted by Dasgupta (2008) (Note 8). It follows that mean growth maximisation – plan the economy to grow mean consumption potential – maximises welfare growth. As additional evidence on the relationship between $\eta = 1$ and mean growth maximisation, we might observe that $\ln(c_i(t))$ is a monotonic transformation of $c_i(t)$ and so represent the same ordinal preference.

Note 8 – Proportional Changes are of Equal Utility Value when $\boldsymbol{\eta = 1}$

Dispensing with *t* in the discounting-free model, let a proportional change be $c_1 = ac_0$. When $\eta = 1$, $\Delta u_{1-0} = \ln(c_1) - \ln(c_0) = \ln(ac_0) - \ln(c_0) = \ln(a) + \ln(c_0) - \ln(c_0) = \ln(a)$.

$\eta = 2$ instead implies that percentage changes are of equal utility – that same percentage increases and decreases in consumption are of equal value (Note 9). Such a society then prefers to maximise the mean of growth rates, which Euclid shows would be the geometric mean across the economy in such an application. Critically, under the *LN* assumption the geometric mean is equal to the median, implying a preference for maximising median growth. Supporting the $\eta = 2$ connection, Buchholz and Schumacher (2010) note that when agents in society emphasise their relative position – their distributional status – that $\eta = 2$ is implied as meeting a no-envy condition (in the relative sense). They define this as a preference where no generation of an $i^{th}$ person envies another generation because their distributional status is consistent. I note this is equally applicable to any $i^{\text{th}}$ and $j^{\text{th}}$ persons in a consumption distribution and is most consistent with a median growth objective (also in Note 9). Under the *LN* assumption this is where $\bar{g}_t = \tilde{g}_t = g_t^{Mo}$.

Note 9 – Percentage Changes are of Equal Utility Value when $\boldsymbol{\eta} = \mathbf{2}$

A necessary but not sufficient proof for the connection between median growth emphasis and $\eta = 2$ is that, when $\eta \neq 1$, $\%\Delta u_{1-0} = \frac{\frac{c_1{}^{1-\eta}}{1-\eta} - \frac{c_0{}^{1-\eta}}{1-\eta}}{\frac{c_0{}^{1-\eta}}{1-\eta}} = \frac{c_1{}^{1-\eta} - c_0{}^{1-\eta}}{c_0{}^{1-\eta}} = \left(\frac{c_1}{c_0}\right)^{1-\eta} - 1$. Then for $\eta = 2$ as well as other values $\eta \neq 1$, $\%\Delta u_{1-0} = \frac{c_0}{c_1} - 1$. So, if we suppose that $c_1 = ac_0$, then $\frac{c_0}{ac_0} - 1 = \frac{1}{a} - 1$. The sufficient proof for why $\eta = 2$ should be singled out then follows from Buchholz and Schumacher's (2010) no-envy condition specification. To show this and that the no-envy condition is not dependent on intertemporal framing, first suppose that the welfare objective is over two "people", $i, j$. Then $W = \frac{c_i{}^{1-\eta}}{1-\eta} + \frac{c_j{}^{1-\eta}}{1-\eta}$ if $\eta > 0, \eta \neq 1$, subject to the constraint $c_i + mc_j = y$. The constraint reports the distribution of the returns from production where $0 < m < 1$ is a production and transfer efficiency parameter. Applying the Lagrangian method with shadow price $\lambda$, interior solutions – which are necessary for the two-person society to exist – satisfy $c_i{}^{-\eta} - \lambda = 0$, $c_j{}^{-\eta} - \lambda m = 0$, and $c_i + mc_j = y$. These imply optimal distributions $c_i{}^* = \frac{y}{1+m^{\frac{\eta-1}{\eta}}}$ and $c_j{}^* = \frac{y}{m^{\frac{1}{\eta}}+m}$. In comparison, Buchholz and Schumacher (2010) define no-envy in the relative sense as that person *i* does not envy $j$ such that $\frac{c_i}{c_j} \geq \frac{mc_j}{c_i}$ and conversely so. Minimally, this is $\frac{c_i}{c_j} = \frac{mc_j}{c_i}$, so $c_i{}^2 = mc_j{}^2$. Subject to the output constraint $c_i + mc_j = y$, the no-envy condition implies $c_i = \frac{y}{1+m^{\frac{1}{2}}}$ and $c_j = \frac{y}{m^{\frac{1}{2}}+m}$ which are equal to the optimal distributions $c_i{}^*, c_j{}^*$ when $\eta = 2$. Buchholz and Schumacher provide a similar defence of $\eta = 1$ when absolute changes, such as mean growth, matter.

For values off critical points $\eta = 1, 2$, as $\eta$ increases above $\eta = 2$, losses are increasingly considered worse than symmetric consumption gains around the median, and conversely so as preferences decrease below 2. I also suppose that $\eta = 0$, which is outside the values generally defined for CRRA utility because it does not imply a concave preference over consumption, implies that making anyone better off through mean consumption growth is valued equally, even if people in the rightmost tail receive all the gains. I now present an estimation strategy based on these observations.

### *4.2.3 Estimation Strategy*

To review, $\eta = 1 \implies max\, \bar{g}_t \Leftrightarrow \Delta\mu_{0,t} = \frac{1}{2}\Delta\sigma_{0,t}^2 \Leftrightarrow \tilde{g}_t = \bar{g}_t - \tilde{g}_t$, and $\eta = 2 \Rightarrow max\, \tilde{g}_t \Leftrightarrow \frac{1}{2}\Delta\sigma_{0,t}^2 = 0 \Leftrightarrow \bar{g}_t = \tilde{g}_t$ when $c_i(t) \sim LN(\mu(t), \sigma^2(t))$. I then conjecture that preferences are revealed (Samuelson, 1938, 1948) by growth policies such that $\Leftrightarrow$ replaces $\Rightarrow$ at the start of each logical chain. To use these logical chains in the measurement of $\eta$, suppose $f(a)$ approximates $\eta$ for any real value $a$ and satisfies the two critical point distributional conditions of $\eta = f(\tilde{g}_t = a,\ \bar{g}_t - \tilde{g}_t = 0) = 2$ and $\eta = f(\tilde{g}_t = a,\ \bar{g}_t - \tilde{g}_t = a) = 1$. Also observe that growth and degrowth preferences are a reflection about the negative diagonal in $\left(\frac{1}{2}\sigma^2, \mu\right)$ space. This diagonal necessarily reports stagnant growth and so bisects the space into growth and degrowth half-spaces.

I use the expanded arctangent argument ATAN2 (Organick, 1966) to write a relation that satisfies these conditions and transitions smoothly and continuously through and around $\eta = 1, 2$ to estimate implied-$\eta$ as:

$$\eta \approx \arctan_\eta(\bar{g}, \tilde{g}) \begin{cases} \frac{4}{\pi} ATAN2(\bar{g}_t - \tilde{g}_t, \tilde{g}_t) & if\ \bar{g}_t \geq 0 \\ \frac{4}{\pi} ATAN2\big( -\tilde{g}_t, -(\bar{g}_t - \tilde{g}_t)\big) & if\ \bar{g}_t < 0 \end{cases} \tag{35}$$

where I preserve the sign in the negative growth scenario as the off-the-shelf ATAN2 function uses it to determine which quadrant the user has in mind.[86] This method of estimation also results in $\eta = f(\tilde{g}_t = 0,\ \bar{g}_t - \tilde{g}_t = a) = 0$ and performs as expected on $\eta = [-1, 3]$ during both economic growth and contractions. Such a range encompasses the $\eta$ values expected in preference-based, economy-wide estimates which are generally in $\eta = [1, 2]$, as well as implied preferences for extreme poverty increasing and decreasing redistributions. From the *LN* assumption, estimates outside of $\eta = [1, 3]$ are likely long run socially unsustainable anyway. Values of $\eta < 1$ imply modal degrowth – robbing from the poor to benefit the rich, while values of $\eta > 3$ imply mean degrowth – reducing total future consumption potential in society. Interestingly, Buchholz and Schumacher (2010) also bound reasonable estimates from above by $\eta = 3$ and below by $\eta = 1$ based on "circumstance solidarity." This concept seems fitting based on the implications derived here – any economy based on satisfying the preferences of much of the population must fall within the bounds of neither robbing from the poor nor future generations.

To summarise the estimation strategy, I recognise the decomposition of national growth – using mean and median values or sufficient distributional data – as a vector normalised

[86] Note $ATAN2(\bar{g}_t - \tilde{g}_t, \tilde{g}_t) = ATAN2(x, y) = \begin{cases} \arctan\left(\frac{y}{x}\right)\ if\ x > 0 \\ \arctan\left(\frac{y}{x}\right) + \pi\ \ if\ x < 0\ \&\ y \geq 0 \\ \arctan\left(\frac{y}{x}\right) - \pi\ \ if\ x < 0\ \&\ y < 0 \\ +\frac{\pi}{2}\ \ if\ x = 0\ \&\ y > 0 \\ -\frac{\pi}{2}\ \ if\ x = 0\ \&\ y < 0 \\ \text{undefined if } x = 0\ \&\ y = 0 \end{cases}$. Then for $\eta = 1,2$, $ATAN2(0) = \frac{\pi}{2}$ and $ATAN2(1) = \frac{\pi}{4}$, respectively. Without an increase in intuition, we can clearly also write the argument directly in terms of growth estimates $\bar{g}$ and $\tilde{g}$ or $\Delta\mu_{0,t}$ and $\frac{1}{2}\Delta\sigma^2_{0,t}$ in the arctangent argument.

to the origin in $(g_{\sigma^2}, g_\mu)$ space. The approximation of $\eta$ is the angle of that growth vector. More precisely, I express the angle of the vector in $\eta$ units and assume distributional preferences consistently imply inequality preferences. Figure 30 depicts the transformation of growth direction without consideration of its magnitude into implied-$\eta$ values during both growth and degrowth by using the polar coordinate system.

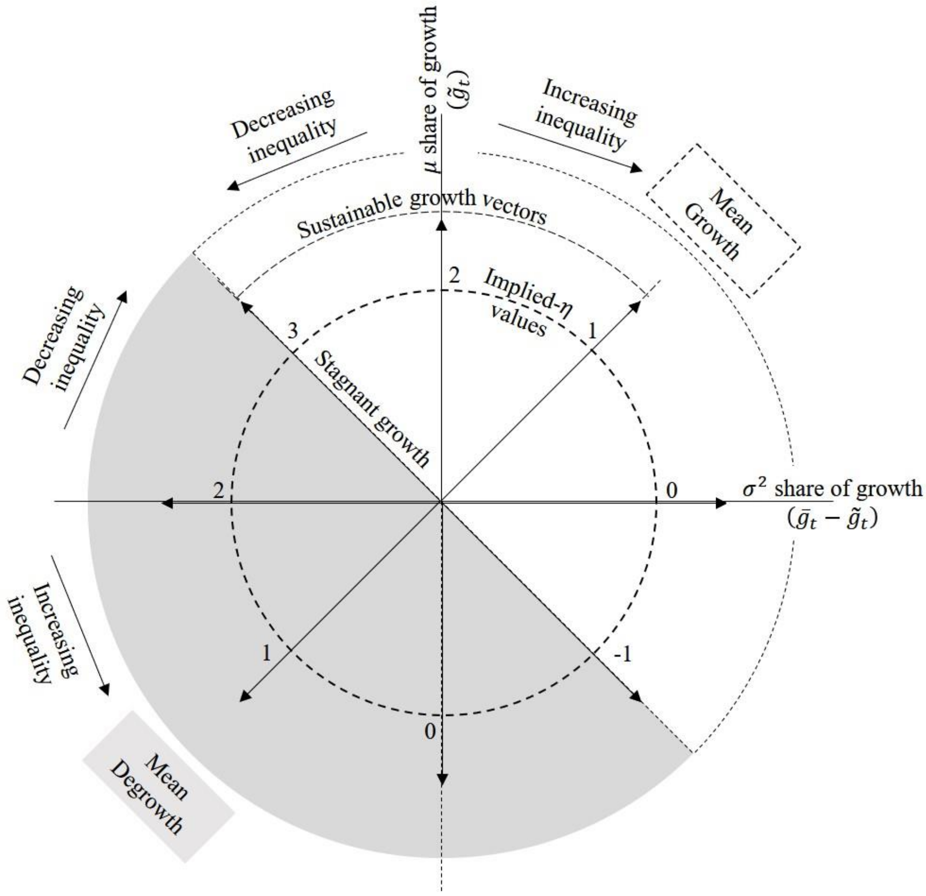


FIGURE 30. IMPLIED-$\eta$ VALUES (ANGLES) ON THE MEAN GROWTH VECTOR CIRCLE

Notes: Angles in $\eta$ units as the implied-$\eta$ measure are derived from estimates of $\tilde{g}$ and $\bar{g} - \tilde{g}$. The relative growth rates of these estimates during growth and degrowth – the distribution of the aggregate returns to economic growth – implies social preference over inequality. Values outside of $\eta = [1, 3]$ are likely long-run socially unsustainable: Values $\eta < 1$ imply modal degrowth – robbing from the poor to benefit the rich, and values $\eta > 3$ imply mean degrowth – reducing total consumption potential of future generations.

#### *4.2.4 Theoretical Sensitivity to Measurement Error*

Suppose Equation (35) is written as a vector-valued function, $f$, and let $D = 2\tilde{g}^2 + \bar{g}^2 - 2\bar{g}\tilde{g}$. When $\bar{g} \neq \tilde{g}$, the resulting Jacobian matrix is[87]

(36) $$\boldsymbol{J}_f(\bar{g},\tilde{g}) = \begin{bmatrix} \bar{g} & -\tilde{g} \\ -\bar{g} & \tilde{g} \end{bmatrix} \frac{1}{D}$$

$\frac{\partial f}{\partial \bar{g}}$ and $\frac{\partial f}{\partial \tilde{g}}$ are otherwise zero when $\bar{g} = \tilde{g} \neq 0$ and undefined when $\bar{g} = \tilde{g} = 0$. The resulting vector Laplacian for nonzero and defined values is

(37) $$\Delta f = \begin{bmatrix} \tilde{g}(\bar{g} - \tilde{g}) - \bar{g}(2\tilde{g} - \bar{g}) \\ \bar{g}(2\tilde{g} - \bar{g}) - \tilde{g}(\bar{g} - \tilde{g}) \end{bmatrix} \frac{2}{D^2}$$

The Laplacian is particularly useful in this application as growth vectors – based on averages of growth measures over time – are themselves averages and so subsequent estimates of $\eta$ are average based. A Laplacian informs on how much a measure will diverge from the average change over time. It is immediate from comparing $\frac{2}{D^2}$ to the values within the vector component of the Laplacian that measurement error is greater for small growth values, particularly those summing to less than two-per cent. So, the growth vector approach appears limited in its ability to measure well the $\eta$ implied by small levels of growth unless they are measured very well. However, the relationship noted in $\Delta f$ suggests more nuance is in order. Setting $\Delta f = 0$ reveals by the quadratic formula that divergence from the mean approaches zero at $\tilde{g}^* = -\frac{1-\sqrt{5}}{2}\bar{g}$ and similarly for degrowth. This relates to a relation $\frac{\tilde{g}}{\bar{g}-\tilde{g}} = \frac{1+\sqrt{5}}{2}$ after simplifying and rationalising. Subsequently, this results in a growth vector implying $\eta \approx 1.30$. However, the benefit is that the general impact of measurement errors is predictable. For long run estimates that are typically greater than 1.30, we should expect short-run estimates of mean and median growth to underestimate $\eta$ in response to measurement error.[88] Conversely, for long run estimates of $\eta$ less than 1.30, corresponding short run estimates will more likely overestimate $\eta$. In summary, short-run

[87] Note $\frac{d\arctan(\alpha)}{d\alpha} = \left(\frac{1}{1+\alpha^2}\right)\alpha'$.

[88] If we believe that, for a given $\bar{g}$, a value $\tilde{g} > \tilde{g}^*$ will underestimate $\eta$. These are in a relation associated with $\eta$ estimates larger than 1.30, and conversely for greater values of $\tilde{g}$.

estimates will tend to underestimate preferences in an economy for either extreme mean- or median-emphasising growth policies.

### *4.2.5 Methodological Discussion*

The increasing availability of national mean and median income and consumption estimates allows the estimation of implied-$\eta$ for nations where other estimates of $\eta$ are not available. This is because the alternatives rely on more privileged data – some access to the economic system of interest. The assumptions we must accept to do this are at least comparable in abstractness to existing revealed-preference approaches. And while we may question the accuracy of estimates other than when $\eta = 1, 2$ as they are mechanically derived, I note that most existing country-level estimates fall within $\eta = [1,2]$.

Implied-$\eta$ estimates should be considered with caution, however. For one, there is likely measurement error in national statistics, particularly estimates based on small survey sizes or limited market access. If the expectation of such errors is zero, however, then there is no effect on long-run estimates. But as shown in the preceding section, short-run estimates – where realisations of measurement error are unlikely to be zero – are concerning. Second, economic growth, and therefore implied-$\eta$ estimates, include the impact of exogenous factors. If we believe the expectation of the influence of exogenous shocks on economic growth is also zero, we can again be unconcerned with long-run estimates on growth trends. However, short-run estimates contain potentially informative volatility effects as such volatility includes information too. If we suppose actual preferences embodied in $\eta$ are stable, a desirable quality of an economic system is that economic distributions also report a stable implied-$\eta$. Unfortunately, the practitioner cannot easily distinguish between measurement error and volatility effects. While it is also desirable to have accurate estimates, the two matters are of different scales of importance – one is a matter of measurement while the other impacts welfare. So, while we might estimate implied-$\eta$ in the short-run and discuss volatility, measurement error in national statistics means that such estimates must be considered cautiously.

We might also explore what is being measured in comparison to other methods. Implied-$\eta$ provides the summary, net effect of policies and institutions – including social norms and historical power-sharing processes – on the distribution of economic gains in society. Often the estimates to follow are higher than from, say, the tax policy-based approach in, for example, Evans (2005). Policy-based estimates only account for part of the picture and often imply a preference for a more unequal distribution supporting mean growth.

Meanwhile, stated preference estimates sometimes suggest that individuals prefer a more equal distribution. Somewhere in between such estimates, the divergent preferences driving various policies in society reconcile and result in the economic distributions from the aggregate economic process that we observe.[89]

### 4.3 Empirical Applications

I estimate long-run implied-$\eta$ values using four country-level datasets. I do so by OLS where data is sufficient, and two-point (starting versus end period) comparisons otherwise. I then estimate short-run implied-$\eta$ values on a subset of the data for further discussion. If distribution mechanisms are entirely resilient to growth volatility, we should expect short- and long-run estimates to be the same as implied-$\eta$ estimates. I then attempt to quantify implied-$\eta$ short-run instability in an appropriate statistic. Finally, I extend the framework to estimate intra-society implied-$\eta$'s between social groups where one has historically held economic and political power.

I use data based on the Luxembourg Income Survey (LIS), and World Inequality Database (WID) to demonstrate different long run estimation methods in the main body. In the appendix, I report additional long run estimates using datasets from the Organisation for Economic Co-operation and Development (OECD) which has substantial overlap with Eurostat data, and from the World Bank. The LIS provides comparable OECD country-level data based on representative household surveys. The WID dataset includes percentile incomes for a more diverse subset of countries collected in several studies with a shared methodology. For the LIS data, I assume a lognormal relationship describes each country-year observation, and then mean and median estimates are derived. In the WID data, for each country-year I test whether a lognormal distribution, and alternatively a gamma distribution, are sufficiently descriptive of the percentile data available. I then derive median and mean estimates. The estimation procedure for implied-$\eta$ uses a comparison of two years in both short-run and long-run cases. Then when sufficient long-run data is available – when growth estimates are statistically significant at standard levels, I also estimate implied-$\eta$ by OLS.

While $\eta$ is typically defined over consumption changes, I remain vague about whether to use income or consumption data. I use both measures as well as different measurement scales – individuals versus household levels in various forms – in the following

[89] The impact of $\eta$ on economic growth policy is explored further in *Mean-Spirited Growth.*

applications. I think this is a matter best left to practitioners as each sort of data implies something a bit different about society. The form of data should be case specific and carefully chosen based on their interests. Even noting that using consumption data is technically the correct basis, interpreting what, exactly, consumption entails and how we should measure it is up for debate, e.g. what do we include in consumption? The four datasets I use have slightly different bases and this likely influences some estimates. Some include various private and state transfers which likely benefit some income and consumption groups more than others. So too, estimates on a household versus per-capita basis should differ if household size is systematically different along the consumption scale. I do not neglect that different income and consumption measures have different policy applications. Rather, I use the data in the bases it is available in (minimally adjusting for inflation) and the reader can decide which is appropriate for their purpose.

### *4.3.1 Long-Run Estimates of Implied-$\eta$*

I first use Thewissen, Nolan, and Roser's (2016) LIS-based data which is adjusted to measures of equivalized disposable household income. This measure of purchasing power parity on a U.S. dollar basis, (PPP($)), is of adjusted incomes including all household inflows – earnings, capital-based income, and private and state transfers less taxes. The result is then divided by the number of "equivalent" adults to arrive at comparable figures. I report LIS-based estimates in Table 10. Due to less frequent sampling, often seven or eight years of data are available per country and regression-based estimates unobtainable with confidence. For those where both OLS and two point-based estimates can be derived, the results are generally similar. Countries with a reputation for more equal distribution generally have higher eta values than the United States which has a reputation for emphasizing economic growth rather than notions of equitability. The single estimate with negative mean growth across the period, Hungary, has a substantially lower implied-$\eta$ estimate which suggests the poor experienced more of the degrowth burden compared to wealthier citizens from the harsh financial crises that occurred in the period. The mean of the 25 implied-$\eta$ estimates (by two-point method) is 1.78 versus mean estimates of 1.35 and 1.42 from Evans' (2005) list of 20 OECD countries. The higher estimate suggests a preference for higher levels of expenditure on the poor, public goods, and programmes benefitting future generations, e.g. enforcing a higher social cost of carbon and more expenditures on environmental preservation.

TABLE 10—GROWTH RATE AND LONG-RUN IMPLIED-$\eta$ ESTIMATES, LIS DATA

| | Period [a] | Two-point | | OLS | |
|---|---|---|---|---|---|
| | | $\bar{g}$ [b] | $\eta$ | $\bar{g}$ [b] | $\eta$ [c] |
| Australia | 1981-2010 (8) | 1.79 | 1.70 | 1.38** | 1.77 |
| Austria | 1994, 2004 | 0.87 | 2.11 | - | - |
| Belgium | 1985, 2000 | 3.10 | 1.53 | - | - |
| Canada | 1981-2010 (10) | 1.02 | 1.66 | 0.90 | 1.59 |
| Czech Republic | 1992, 2010 | 3.98 | 1.89 | - | - |
| Denmark | 1987-2010 (7) | 1.13 | 1.77 | 1.15 | 1.86 |
| Estonia | 2000, 2010 | 6.94 | 2.19 | - | - |
| Finland | 1987-2010 (7) | 2.03 | 1.72 | 1.75 | 1.74 |
| France | 1978-2010 (7) | 0.90 | 2.11 | 0.79** | 2.12 |
| Germany | 1984-2010 (7) | 0.69 | 1.78 | 0.56** | 1.64 |
| Greece | 1995, 2010 | 2.16 | 2.08 | - | - |
| Hungary | 1991, 2012 | -0.25 | 0.26 | - | - |
| Ireland | 1987-2010 (8) | 4.18 | 2.11 | 3.45 | 2.11 |
| Israel | 1986-2010 (7) | 2.24 | 1.78 | 1.67 | 1.80 |
| Italy | 1986-2010 (11) | 0.90 | 2.01 | 0.63** | - |
| Luxembourg | 1985-2010 (8) | 4.19 | 1.94 | 2.63 | 1.92 |
| Netherlands | 1993, 2010 | 1.77 | 1.77 | - | - |
| Norway | 1979-2010 (8) | 3.58 | 1.96 | 2.28 | - |
| Poland | 1992, 2010 | 2.08 | 1.80 | - | - |
| Slovak Republic | 1992, 2010 | 3.06 | 1.84 | - | - |
| Slovenia | 1997, 2010 | 2.22 | 1.99 | - | - |
| Spain | 1980-2010 (8) | 2.13 | 2.04 | 2.15 | - |
| Sweden | 1981-2005 (6) | 2.62 | 1.86 | 2.04 | 1.84 |
| United Kingdom | 1979-2010 (9) | 2.86 | 1.68 | 2.41 | 1.80 |
| United States | 1979-2013 (10) | 0.70 | 0.97 | 0.77 | 1.33 |

*Notes:* Estimates based on the annual rate of long-run growth in PPP-adjusted equivalised disposable household income. All OLS estimates reported are statistically significant at the 1-per cent level or better using Huber–White standard errors (White, 1980) unless otherwise noted to improve readability (** 5 per cent level; * 10 per cent level).

[a] Number of observations in the period in parentheses. [b] Average annual growth rate, in per cent. [c] Statistical significance of OLS-based implied-$\eta$ estimates is the least from the two estimated values used in its composition.

*Sources:* Author calculations based on data from Thewissen, Nolan, and Roser (2016) which is derived from the Luxembourg Income Study (LIS) database.

Table 11 reports WID data-based estimates. These estimates are on percentile, pre-tax national income at the distribution percentile threshold. The income includes pensions and all other sources of transfer and is at the individual, rather than household level of the population over 20 years of age. These estimates are CPI-adjusted but left in the local currency. The estimation procedure is to first fit a *LN*, alternatively a Gamma ($\Gamma$) distribution to the percentile data. I then use a two-sample Kolmogorov-Smirnov test (Massey, 1951) to check whether the fitted distribution represents the data well. The analytical values of these distributions are the basis for estimating implied-$\eta$.[90] While the estimation methodology relies on an *LN* distributional assumption, I substitute $\Gamma$-based estimates whenever the $\Gamma$ distribution fits the data and the *LN* does not. $\Gamma$-based estimation generally, but not exclusively, results in higher estimates. Since the WID data sometimes covers substantially longer periods, e.g. France from 1900-2014, I also report estimates

[90] Though implied-$\eta$ estimates need not be based on fitted analytical values. One can also estimate directly on the data's mean and median values and currently I see no definitive advantage to either approach.

over important national historical periods. Estimating on subperiods also allows the derivation of some implied-$\eta$ estimates for the USSR/Russian Federation which is otherwise too inconsistent across the entire 1961 through 2015 period to derive statistically significant, OLS-based estimates (see Note 10).

TABLE 11—GROWTH RATE AND LONG-RUN $\eta$ ESTIMATES, WID DATA

| | Period [a] | Lognormal | | Gamma | |
|---|---|---|---|---|---|
| | | $\bar{g}$ [b] | $\eta$ | $\bar{g}$ [b] | $\eta$ |
| Brazil [c] | 2001-2015 (15) | 1.24** | - | 1.25 | 2.16 |
| China [d] | 1978-2015 (38) | 5.63 | 1.63 | 5.36 | 1.83 |
| Transition | 1978-1999 (22) | 4.65 | 1.62 | 4.45 | 1.81 |
| Present | 2000-2015 (16) | 9.29 | 1.78 | 8.69 | 1.92 |
| Côte d'Ivoire [e] | 1988-2014 (6) | -1.48** | -0.36 | -1.84 | - |
| Egypt [f] | 1999-2015 (6) | 1.45 | 2.17 | 1.30 | 2.19 |
| France [g] | 1900-2014 (83/102) | 2.21 | 2.27 | 2.43 | 2.10 |
| LIS comparison | 1978-2010 (26/33) | 2.43 | 0.41 | 1.13 | 1.04 |
| Pre-WWII | 1900-1939 (20/27) | - | 2.83 | 0.68 | 2.49* |
| Post-war | 1946-1969 (22/24) | 4.52 | - | 4.58 | - |
| Transition | 1970-1999 (25/30) | - | 2.87 | 1.40 | - |
| Present | 2000-2014 (11/15) | 2.47 | -0.12 | - | - |
| India [h] | 1951-2013 (54/63) | 1.84 | - | 2.05 | 1.89* |
| Postcolonial | 1951-1964 (12/14) | 2.06 | 2.34 | 2.31 | - |
| Transition | 1965-1999 (32/35) | 1.86 | 1.92 | 1.86 | - |
| Present | 2000-2013 (10/14) | 3.74 | 1.68 | 4.42 | 0.59 |
| Palestine [f] | 1996-2011 (10) | -1.33* | -0.28** | -1.26* | -0.23** |
| Russia [i] | 1961-2015 (25/36) | - | - | - | - |
| USSR | 1961-1989 (3/10) | - | - | 2.38 | 2.06 |
| Transition | 1990-1999 (6/10) | - | - | -5.38 | - |
| Present | 2000-2015 (16) | 2.89 | 2.53 | 4.09 | 2.18 |
| Turkey [f] | 1994-2016 (16) | 2.19 | - | 2.06 | 2.28** |
| United States [j] | 1962-2014 (51) | 1.47 | 1.50 | 1.42 | 1.74 |
| LIS comparison | 1979-2013 (35) | 1.44 | 1.68** | 1.56 | 1.75 |
| Transition | 1962-1980 (17) | 1.62 | - | 1.50 | - |
| Reaganomics | 1981-1992 (12) | 2.53 | 0.60 | 1.77 | 1.44 |
| Present | 1993-2014 (22) | 0.81 | - | 1.28 | 1.90 |

*Notes:* Estimates based on growth and distribution of pre-tax income among individuals age 20 and over. Distributions fit to annual percentile income thresholds by maximum likelihood estimation. All OLS estimates reported are statistically significant at the 1-per cent level or better using Huber–White standard errors unless otherwise noted (** 5 per cent level; * 10 per cent level). Statistical significance of implied-$\eta$ estimates is the least from the two estimated values used in its composition.

[a] Number of observations in the period in parentheses. [b] Average annual growth rate, in per cent.

*Sources:* Author calculations based on data in [c] Morgan (2017); [d] Piketty, Yang, and Zucman (2017); [e] Czajka (2017); [f] Alvaredo, Assouad, and Piketty (2018); [g] Garbinti, Goupille-Lebret, and Piketty (2018); [h] Chancel and Piketty (2017); [i] Novokmet, Piketty, and Zucman (2018); [j] Piketty, Saez, and Zucman (2016), available from World Inequality Database (WID), WID.world.

Note 10 – Estimates for the Russian Federation/USSR by the Two-Point Method

We can also derive estimates for the Russian Federation/USSR by the two-point method as a matter of general interest. The economic collapse during the transition period resulted in increases in inequality – the poor experienced (much) greater losses. The negative values estimated suggest a national experience that does not fit any rational CRRA utility-based preference.

| | Period | $g$ | $\eta$ |
|---|---|---|---|
| Overall | 1961-2015 | 1.78 | 1.87 |
| USSR | 1961-1989 | 3.57 | 2.35 |
| Transition | 1990-1999 | -4.69 | -0.39 |
| Present | 2000-2015 | 4.34 | 2.39 |

Notes: Mean growth rates and implied-$\eta$ values for the Russian Federation/USSR over important historical periods.

Sources: Author calculations based on data from Novokmet, Piketty, and Zucman (2018) available from World Inequality Database (WID), WID.world.

I report additional long-run estimates based on OECD and World Bank data – both income and consumption-based in the latter – in appendix *A4.2 Long-Run Estimates of Implied-$\eta$ Based on OECD and World Bank Data*. These estimates are sometimes on substantially shorter periods and show greater variability across the diverse sets of countries involved. The mean of the estimates are 1.89, 1.73, and 1.70 for the OECD, World Bank income, and World Bank consumption data, respectively. These, like the means of the LIS and WID-based estimates, suggest a general social preference for greater public goods expenditure and environmental protection than preceding revealed preference estimates have suggested. I now produce some short-run estimates which perhaps have greater implications for the study of social unrest.

### *4.3.2 Short-Run Estimates – Measuring Economic Volatility*

Long-run estimates imply social preferences embodied in $\eta$. Or more accurately the mean value of $\eta$ implied by the national economic process. Short-run estimates, however, include both measurement error – noted as a substantial concern in the methodology section – and the effect of economic "booms" and "busts" on distribution. We can often estimate implied-$\eta$ annually, and if we believe our data of sufficient quality, find a summary statistic on distributional volatility. Short-run estimates do not imply annually changing social preferences because exogenously driven volatility influences distribution. But what we do measure is the impact of exogenous pressures on economic growth – whether economic distributions are relatively stable or add to inequality by benefitting income groups differently during shocks, booms, and busts. Suppose preferences are stable, or at least slowly changing. A stable implied-$\eta$ is preferable and a volatile one supports beliefs of economic injustice which may increase susceptibility to nationalistic and other self-preservation rhetoric. To explore this concept, I first compare short-run implied-$\eta$ estimates to the tax-based, equal sacrifice approach estimates in Evans (2005) which use a panel of OECD countries in 2002. I then propose some measures to quantify implied-$\eta$ volatility.

Evans (2005) uses Stern's (1977) equal sacrifice approach which relies on data from income tax schedules, the CRRA utility functional form explicitly for structure, and "equality of taxation" traceable to Mills (1848) for legitimisation.[91] If we assume that tax structure is a result of political discourse, generally stable, and representative of social preferences over the tax component of distribution – perhaps a fair assumption in the OECD – then Evans provides a useful comparison. The LIS-based data I use – chosen for quality and consistency – is not annual, but instead available at varying intervals. I choose the shortest available intervals that include the year 2002 and estimate implied-$\eta$. Presented in Table 12, I often arrive at divergent estimates from Evans as well as the preceding long-run implied-$\eta$ estimates. This may suggest instability in the distribution of the returns to economic activity. Instability implies that something akin to "booms" and "busts" also occur in the economic distribution.

TABLE 12—COMPARING EVANS (2005) AND LIS-BASED SHORT-RUN IMPLIED-$\eta$ ESTIMATES

| | Period [a] | $\bar{g}$ [b] | $\eta$ from Evans (2005) [c] | Long-run $\eta$ [d] | Short-run $\eta$ |
|---|---|---|---|---|---|
| Australia | 2001, 2003 | 0.93 | 1.67 | 1.77 | 1.21 |
| Austria | 2000, 2004 | 2.29 | 1.74 | 2.11 | 1.46 |
| Belgium | 1997, 2000 | 5.51 | 1.33 | 1.53 | 0.88 |
| Canada | 2000, 2004 | 1.67 | 1.28 | 1.59 | 2.01 |
| Czech Republic | 2002, 2004 | 4.66 | 1.29 | 1.89 | 1.83 |
| France | 2000, 2005 | 0.61 | 1.37 | 2.12 | 2.36 |
| Germany | 2000, 2004 | 0.42 | 1.36 | 1.64 | -0.05 |
| Hungary | 1999, 2005 | 3.80 | 1.36 | 0.26 | 1.95 |
| Ireland | 2000, 2004 | 4.22 | 1.24 | 2.11 | 1.66 |
| Italy | 2000, 2004 | 1.33 | 1.37 | 2.01 | 1.32 |
| Norway | 2000, 2004 | 2.78 | 1.32 | 1.96 | 1.55 |
| Poland | 1999, 2004 | 0.33 | 1.48 | 1.80 | -0.61 |
| Slovak Republic | 1996, 2004 | 1.19 | 1.52 | 1.84 | 1.39 |
| Spain | 2000, 2004 | -1.49 | 1.18 | 2.04 | 1.26 |
| United Kingdom | 1999, 2004 | 4.12 | 1.24 | 1.80 | 1.87 |
| United States | 2000, 2004 | 0.28 | 1.30 | 1.33 | 1.67 |

*Notes:* Comparison of Evans' (2005) equal sacrifice approached-based $\eta$ estimates to long-run and short-run implied-$\eta$ estimates based on growth in PPP-adjusted equivalised disposable household income.

[a] Start and end years for the two-point, or slope formula-based estimates. [b] Average annual growth rate, in per cent. [c] Average of Evans' "high" and "low" estimates. [d] Best available estimates from Table 10.

*Sources:* Author calculations based on Evans (2005), and Thewissen, Nolan, and Roser (2016) which is derived from the Luxembourg Income Study (LIS) Database.

91 The result is an estimation strategy that takes advantage of differences between marginal and average tax rates. Equality of taxation is related to the concept of equality of sacrifice – that no one is more or less inconvenienced than others.

I next report in Table 13 three estimates of implied-$\eta$ volatility based on the LIS dataset which has some overlap with the countries in Table 12. These volatility measures are based on the span of the LIS-based data available. The measures I compare are the spread as $Max - Min$ of implied-$\eta$ estimates in the period, the coefficient of variation (CV) quotient, $\frac{CV_\eta}{CV_{\bar{g}}}$ which is the normal CV of implied-$\eta$ over the normal CV of mean growth, and the variance of implied-$\eta$ estimates, $Var(\eta)$ (see Note 11 for its derivation). The period of these estimates only includes limited data following the 2008 financial crisis where the extent of distributional resistant to economic volatility would be further tested.

Note 11 – Deriving $\boldsymbol{Var(\eta)}$ in the Polar Coordinates Space

As estimates of $\eta$ are based on converting growth vectors into a polar coordinates-type space, deriving $Var(\eta)$ requires some steps. Working backwards from equation (35), multiply $\frac{\pi}{4}\eta$ to convert $\eta$ estimates into radians. Then note $\cos\theta = \frac{x}{r}$ and $\sin\theta = \frac{y}{r}$ for angle $\theta$ in an $(x, y)$ space with a hypotenuse (or circle with radius) *r*. Assume that the distribution of the returns to growth are independent of the scale of growth and then conveniently set $r = 1$. So, $\cos\left(\frac{\pi}{4}\eta\right) = \bar{g} - \tilde{g}$ and $\sin\left(\frac{\pi}{4}\eta\right) = \tilde{g}$. Use these values for each observation to find their averages and subsequently the average of $\bar{\eta}$ in the *ATAN2* argument as $\bar{\eta} = ATAN2\left(\frac{1}{N}\sum_{i=1}^{N} cos\left(\frac{\pi}{4}\eta_i\right), \frac{1}{N}\sum_{i=1}^{N} sin\left(\frac{\pi}{4}\eta_i\right)\right)$. Then compute the variance in the standard way, here as $Var(\eta) = \frac{1}{N}\sum_{i=1}^{N}(\eta_i - \bar{\eta})^2$.

Lower $Max - Min$ spread and $Var(\eta)$ values are associated with countries that have reputations for pursing greater economic equality and higher standards of living. Estimates of $\frac{CV_\eta}{CV_{\bar{g}}}$, however, are less consistent – countries can have high values of $CV_\eta$ obscured by high values of $CV_{\bar{g}}$, or one measure can be higher without consistency. In comparison to implied-$\eta$ estimates which are independent of growth magnitude, the $CV$ quotient confounds any estimate by reintroducing magnitude. Perhaps a more useful statistic will be developed in future research. But from the statistics I derive, $Var(\eta)$ for the Nordic countries versus other estimates is perhaps indicative of the statistic's value. We might then, for instance, more efficiently counter international extremist recruiting or environmental degradation by directing international efforts and funding to strengthening social safety nets in nations with higher $Var(\eta)$ estimates.

TABLE 13—IMPLIED-$\eta$ VOLATILITY ESTIMATES BASED ON A SUBSAMPLE OF LIS DATA

| | Period [a] | $Max - Min$ | $CV_\eta/CV_{\bar{g}}$ | $Var(\eta)$ [b] |
|---|---|---|---|---|
| Australia | 1981-2010 (8) | 2.87 | 1.13 | 1.47 |
| Canada | 1981-2010 (10) | 3.36 | 0.84 | 1.50 |
| Denmark | 1987-2010 (7) | 1.93 | 0.65 | 0.66 |
| Finland | 1987-2010 (7) | 2.35 | 0.45 | 0.80 |
| France | 1978-2010 (7) | 3.35 | 0.38 | 1.50 |
| Ireland | 1987-2010 (8) | 2.35 | 0.36 | 0.68 |
| Israel | 1986-2010 (7) | 2.64 | 0.86 | 0.97 |
| Italy | 1986-2010 (11) | 3.57 | 0.29 | 1.56 |
| Luxembourg | 1985-2010 (8) | 2.38 | 0.71 | 1.16 |
| Norway | 1979-2010 (8) | 2.20 | 0.52 | 0.47 |
| Spain | 1980-2010 (8) | 2.61 | 0.43 | 1.25 |
| Sweden | 1981-2005 (6) | 2.16 | 0.40 | 0.77 |
| United Kingdom | 1979-2010 (9) | 2.78 | 0.39 | 0.87 |
| United States | 1979-2013 (10) | 2.73 | 0.83 | 0.96 |

*Notes:* Estimates based on the annual rate of long-run growth in PPP-adjusted equivalised disposable household income. Max-Min reports the spread in implied-$\eta$ estimates over the period, $CV_\eta/CV_{\bar{g}}$ the normal coefficient of variation of implied-$\eta$ to that of mean growth, and $Var(\eta)$ the variance of implied-$\eta$ estimates.

[a] Number of observations in the period in parentheses. [b] Uses the mean of the angle for the sample mean calculated as $\bar{\eta} = ATAN2\left(\frac{1}{N}\sum_{i=1}^{N} cos\left(\frac{\pi}{4}\eta_i\right), \frac{1}{N}\sum_{i=1}^{N} sin\left(\frac{\pi}{4}\eta_i\right)\right)$, for the $i = 1{:}\,N$ implied-$\eta$ estimates of a country.

*Source:* Author calculations based on data from Thewissen, Nolan, and Roser (2016) which is derived from the Luxembourg Income Study (LIS) Database.

Having outlined a process to use the implied-$\eta$ estimation methodology to explore short-run volatility, it seems entirely appropriate to consider how accurate estimates may be. The empirical section already cautioned that some estimates taken over small changes in economic growth will be particularly prone to the impact of measurement error. These are in economies with limited economic growth which are also reporting long-run estimates toward the extremes of $\eta$ plausibility. One source of error can be the basis used for deriving estimates of $\bar{g}$ and $\tilde{g}$ as it is entirely possible that the estimates derived by fitting a *LN* or $\Gamma$ distribution will differ from nonparametric estimates. This is generally more of a concern for the estimation of $\tilde{g}$ in the presence of substantial income inequality which results in more income flows to those represented in the rightmost tail of a distribution. Because of this issue, I next explore the impact of the basis for $\bar{g}$ and $\tilde{g}$ estimates.

### *4.3.3 Exploring the Impact of the Parametric Assumption*

It is the intention of this method of estimation that practitioners use high quality and preferably readily available national statistics to estimate implied-$\eta$. The use of the *LN* distribution in this paper allows us to arrive at a convenient method for estimating $\eta$. But as in most of the preceding $\eta$ estimates, the intention is not to imply that the lognormal distribution must be fit to national datasets. To so assert such a requirement would substantially constrain the framework's applicability to those with access to privileged data and a small subset of countries where distributional data is publicly available. However, in the methodology section, the sensitivity of results to measurement error suggests that, particularly for small levels of growth, deviations in $\bar{g}$ and $\tilde{g}$ can bias results. This is found to be truer for estimates that move away from a growth vector implying an $\eta$ of 1.29.

One source of deviation is how we measure $\bar{g}$ and $\tilde{g}$. The theory relies on the *LN* assumption, while in practice it is expected that nonparametric values will normally be used. As one of several other alternatives, a gamma distribution is also possible as used in parts of Table 11. To compare the sensitivity of estimates to such assumptions, I again use the WID distributional data and estimate $\eta$ from the nonparametric mean and median values as well as based on values from fitting the lognormal and gamma distributions. To minimise confounding errors from short-run volatility, I estimate based on the long-run WID data and do not divide it into national subperiods of geopolitical interest. However, for many periods – particularly when growth is substantial – the difference in resulting $\eta$ values in subperiods is often similar to the differences found in the summary period-based estimates.

The comparison $\eta$ estimates and their differences are reported in Table 14. We can observe that difference between *LN*, Γ, and nonparametric-based estimates of $\eta$ are fairly trivial. In many cases, the difference in resulting $\eta$ is small – less than 10-per cent. There are, however, exceptions. Countries with higher existing income inequality based on the Gini coefficient such as Côte d'Ivoire and the United States have a higher divergence between parametric and nonparametric estimates. This is fitting with a scenario where *LN* and Γ distributions do not fit well around the median value due to more substantial wealth flows to households in the rightmost tail of the income distribution. However, analysing the *LN*, Γ, and nonparametric estimates themselves is also informative. We might, for example, observe that the difference between the *LN* and nonparametric-based estimates for the Republic of Côte d'Ivoire are the largest in the sample. However, comparing the *LN* and nonparametric estimates suggests a similar policy – that the powers in control of the economy prefer an unequal distribution. This is also the case for the United States which had the second largest deviation in estimates – between the Γ and nonparametric estimates. In this case, all three $\eta$ estimates point to a similar conclusion – that the economy is designed to increase inequality over time.

TABLE 14—THE IMPACT OF THE LOGNORMAL PARAMETRIC ASSUMPTION

| | Estimate basis | | | Differences | | |
|---|---|---|---|---|---|---|
| | LN | Γ | Non | LN − Γ | LN − Non | Γ − Non |
| Brazil [a] | - | 2.16 | 2.44 | - | - | -0.28 |
| China [b] | 1.63 | 1.83 | 1.79 | -0.20 | -0.16 | 0.04 |
| Côte d'Ivoire [c] | -0.36 | - | 0.21 | - | -0.57 | - |
| Egypt [d] | 2.17 | 2.19 | 2.36 | -0.02 | -0.19 | -0.17 |
| France [e] | 2.27 | 2.10 | 2.13 | 0.17 | 0.14 | -0.03 |
| India [f] | - | 1.89 | 1.79 | - | - | 0.10 |
| Palestine [d] | -0.28 | -0.23 | -0.39 | -0.05 | 0.11 | 0.16 |
| USSR [g] | - | 2.06 | 2.23 | - | - | -0.17 |
| Russia - present [g] | 2.53 | 2.18 | 2.20 | 0.35 | 0.33 | -0.02 |
| Turkey [d] | - | 2.28 | 2.37 | - | - | -0.09 |
| United States [h] | 1.50 | 1.74 | 1.35 | -0.24 | 0.15 | 0.39 |

*Notes:* Comparison of $\eta$ estimates based on Lognormal (LN), Gamma (Γ), and Nonparametric (Non) bases. Based on growth and distribution of pre-tax income among individuals aged 20 and over. Data includes the summary periods in the WID dataset where distributional data is available to avoid confounding short-run volatility issues. The exception is the USSR and present Russian Federation data where each trend is individually stable but in comparison different. Distributions fit to annual percentile income thresholds by maximum likelihood estimation. Only OLS estimates that are statistically significant at the 10-per cent level or better using Huber–White standard errors are used in the analysis. Statistical significance of implied-$\eta$ estimates is the least from the two estimated values used in its composition.

*Sources:* Author calculations based on data in [a] Morgan (2017); [b] Piketty, Yang, and Zucman (2017); [c] Czajka (2017); [d] Alvaredo, Assouad, and Piketty (2018); [e] Garbinti, Goupille-Lebret, and Piketty (2018); [f] Chancel and Piketty (2017); [g] Novokmet, Piketty, and Zucman (2018); [h] Piketty, Saez, and Zucman (2016), available from World Inequality Database (WID), WID.world.

Before concluding the empirical section, I turn to presenting an intra-society measure equivalent to implied-$\eta$ estimates. In societies with clearly defined subgroups and a history of income inequality, an implied-$\eta$ estimate is perhaps informative on how inequality between and within groups is changing over time. This suggests inequality preferences within those groups given the constraints of society as well as the preferences of society in general.

#### *4.3.4 An Extension into Intra-Society Measures of Implied Inequality*

Without the benefit of comparable supporting structure, I devise $\eta$-scaled estimates to describe between-group dynamics. That is, I compare the economic result implied between, say, gender and racial groups in society driven by historical social structures. This synthetic $\eta$ estimate suggests whether the net effect of a policy is to increase ($\eta > 2$), maintain ($\eta = 2$), or decrease ($\eta < 2$) equality between groups. These estimates are on less firm theoretical ground as no distributional assumption seems appropriate, nor does the vector associated with $\eta = 1$ have relevance. Instead, I rely on the vector implying $\eta = 2$ is of equality of growth between the social group of interest which experiences economic growth rate $g_i$, and a dominant baseline group which experiences economic growth rate $g_{\neq i}$. For estimates to make sense, the historically socially dominant group's statistic must be $g_{\neq i}$ and I estimate deviations from it. The growth values resulting in $\eta = 0$ does gain an interesting interpretation – the structure of the economy is such that it caters only to the dominant group.[92] This structure is equivalent in the prior framework to an economy designed only to benefit the wealthy in the right-tail. The between-group synthetic-$\eta$ estimate can be derived as

$$(38)\quad \eta \approx arctan_{groups}(g_{\neq i}, g_i) \begin{cases} \frac{4}{\pi} ATAN2(g_{\neq i} - g_i, g_i) & if\ g_{\neq i} \geq 0 \\ \frac{4}{\pi} ATAN2\big(-g_i, -(g_{\neq i} - g_i)\big) & if\ \bar{g}_t < 0 \end{cases}$$

where values of $g_i$ and $g_{\neq i}$ must be specified at comparable points in their respective distributions – say both are either mean or median growth values – for the estimates to make intuitive sense.

Table 15 provides estimates based on U.S. Census Bureau mean and median household and personal income data from 1967-2017. The overall, implied-$\eta$ estimate on the data is near unity over the 51 years – close to the preceding LIS-based estimate and the general belief that the U.S. economic system is organised to maximise long-run mean (roughly gross domestic product per-capita) economic growth. Within-group estimates use median and mean growth within each population subgroup and are comparable in interpretation to prior estimates. Between-group estimates instead compare the growth vector of, say, households that identify as Black versus White non-Hispanic households. To compare vectors between genders, I instead use personal income estimates. White non-Hispanic households, and male where applicable, are taken as the baseline group because U.S. national-scale policymakers – the legislative and executive branches of the U.S. government – remain overwhelmingly White non-Hispanic and Male.[93] Race and gender-based comparisons are estimated between both median and mean values.

[92] We might also want to consider subtracting 1 or 2 from the estimate if we want to recenter it on values of 1 or 0 instead of on an $\eta$ equivalent scale. But then we sacrifice the interpretation of $\eta = 0$.

[93] In 1967, 2.6-per cent of the U.S. Congress was non-Hispanic White, and 2-per cent female. In 2017, these shares have grown to 20.0-per cent and 19.5-per cent, respectively out of 535 national representatives (The Brookings Institute, 2019).

TABLE 15—LONG-RUN $\eta$ BETWEEN GROUP ESTIMATES, U.S. CENSUS DATA

| | Within-group (median to mean) | versus White, non-Hispanic | |
|---|---|---|---|
| | | at medians | at means |
| Overall (51) | 0.90 | - | - |
| Black (46) | 1.40 | 2.28 | 2.04 |
| Asian (30) | 1.47 | 2.24 | 1.93 |
| Hispanic (46) | 1.08 | 1.80 | 1.81 |
| | | versus Male | |
| Female (51) | 1.97 | 2.35 | 2.23 |
| Male (51) | 1.49 | - | - |

*Notes:* Comparable-scale estimates of $\eta$ values within and between major racial and gender groups in the United States, 1967-2017. All underlying growth estimates are derived by OLS and statistically significant at the 1-per cent level or better using Huber–White standard errors on the number of observations noted in parentheses.

*Sources:* Author calculations based on data from the United States Census Bureau (2018, 2019).

Table 15 suggests that inequality is increasing overall and within each ethnic group, but at a faster rate for White non-Hispanic and Hispanic groups. At median points of comparison, inequality is decreasing between both Black and Asian households and their White non-Hispanic counterparts, but inequality is increasing for Hispanic households. Black and non-Hispanic White convergence around median values is also found in, for instance, Couch and Daly (2004), Jones, Schmitt, and Wilson (2018), Sakano (2002), and Smith (1978). Inequality is also slightly decreasing – or at least stable – between Black and non-Hispanic White households at mean values. Within-gender estimates suggest nearly stable inequality between women, but growing inequality for men. Between-gender estimates suggest decreasing inequality for women versus men. These sort of estimates without a strong theoretical basis should be taken with caution. However, the general implication – decreasing relative economic status of roughly median income white American males is hardly surprising as it is a frequent complaint of that group (Anderson, 2016; Kimmel, 2013).

### *4.3.5 Empirical Discussion*

This research presents a simple method to estimate something comparable to the elasticity of marginal utility of consumption – the $\eta$ value as a measure of inequality preferences, based on publicly available economic growth data. Estimating on such data allows the generation of comparable estimates for countries where data availability is insufficient to estimate $\eta$ using revealed and stated preference methodologies. Implied-$\eta$ estimates are akin to measures of the net effect in society as they include preferences over tax and other institutional structures – traditionally used separately to measure $\eta$ – and other social preferences over the distribution of the returns to economic growth. The long-run estimates are robust to mean-zero errors in the measurement of economic statistics but require sufficient time-series data to ensure this occurs. Short-run estimates are susceptible to measurement error but also include inseparable, desirable information on the volatility of implied-$\eta$. This volatility in economic growth returns may be another source of social unrest even when the long-run trends of economic growth and inequality trends are desirable. I also extend beyond the bounds of the theory developed and provide a method to estimate comparable-scaled values for inequality growth between different groups in a population. These compare growth trends at, say, the median value of two groups, and suggest whether the median consumers of each group are converging in consumption potential. These estimates should be considered with caution. However, the results fit with the general picture of rising non-Hispanic White male discontent with their changing social position, at least in the United States.

Of particular theoretical interest is the result that implied-$\eta$ estimates are very often higher than those estimated by other revealed preference approaches. The mean of estimates on the 20 OECD countries in Evans (2005) are 1.35 and 1.42 (two estimation methods), on the 17 Latin American countries in Moore, Boardman, and Vining (2020) is 1.33, on the six transitioning Eastern European countries in Seçilmiş and Akbulut (2019) are 1.52 and 1.19 in 2000 and 2015, respectively, and the mean of 20 mostly separate estimates on the U.K. alone is 1.58.[94] In comparison, the mean of implied-$\eta$ estimates based on LIS, WID, OECD, World Bank income, and World Bank consumption data are 1.78, 1.58, 1.89, 1.73, and 1.70, respectively.[95] Additionally, the implied-$\eta$ outlier estimates are generally much larger or smaller than in the revealed preference approach sets. Such higher estimates have implications in, for example, setting the social cost of carbon (SCC) and expenditures on environmental protection and public goods (suggesting all should be higher).

I do not find the disparity between implied-$\eta$ and traditional $\eta$ estimates damning, but rather encouraging. There has been a simmering debate in the literature on this matter – descriptive as simmering as it has reached a point of intractability. Repeated estimates in the range of $\eta = [1, 1.5]$ have resulted in some experts such as Stern (2007) recommending these sorts of values be used in policy development, with a preference toward unity. As a

[94] Mean of U.K. estimates is based on estimates in Banks, Blundell, and Lewbel (1997), Blundell (1988), Blundell, Pashardes, and Weber (1993), Cowell and Gardiner (1999), Evans (2004a), Evans (2005), Evans and Sezer (2002), Evans and Sezer (2005), Evans, Kula, and Sezer, (2005), Groom and Maddison (2019), and Stern (1977).

[95] Based on the two-point formula estimates for the LIS-based, OECD, and World Bank estimates; and the best available full period estimates (*LN* when available, alternatively *Γ* based) from the WID data.

result, such values have been integrated into government calculations affecting a variety of planning. Yet, Weitzman (2007) suggests that $\eta = 1$ is the absolute lowest value that a reasonable economist would consider and is certainly not an average. Buchholz and Schumacher (2010), too, put $\eta = 1$ as the lowest bounds of societal economic solidarity and values of $\eta$ are likely higher. The most substantial and methodical rebuke, however, comes from Dasgupta (2008) who suggests that true values of $\eta$ are almost certainly higher – in the range of $\eta = [1.5, 3]$ or even higher, as lower values imply "absurdly high" societal savings ratios and other behaviour. The natural reply to such criticisms has been to observe that rigorous revealed preference estimates suggest low values of $\eta$. Here instead mean implied-$\eta$ estimates are in line with Dasgupta's expectations.

### 4.4 Concluding Remarks

One of the net effects of competing social preferences is the distribution of the returns from economic growth. I estimate these social preferences as the summary statistic implied-$\eta$. Inequality in the distribution of the returns to economic growth has been a matter of substantial debate going back to at least Veblen (1899). More recently, Atkinson (1970) set in motion the debate on how to measure inequality preferences well. I avoid much of the debate altogether by measuring the theoretical indicator for inequality preferences directly from economic growth trends, rather than advocating for some more indirect indicator. The inequality growth vector approach to estimating implied-$\eta$ is admittedly ad hoc. But it is also simple to execute, relies on generally publicly available data, is applicable to a variety of circumstances, and I think the process is fairly intuitive. It also sets a difficult policy challenge – the design of economies aligned with social preferences during both economic booms and busts, not just on the average. At a minimum, I present a measure of inequality that bridges the theory and practice of studying inequality which allows new international and within-country comparisons.

# 5. Mean-Spirited Growth

*We contribute to the Solow-Swan strain of economic growth literature by integrating national distributional aspects into a dynamic model of economic growth. We show that country specific optimal growth policies balance distributional "preferences" against savings propensities of different percentiles of the emerging income distribution. We then present international comparisons of growth decomposition into distributional statistics and propose new measures of economic performance. Historical economic performance reveals heterogeneous patterns of inequality increasing (e.g. US, UK, Germany, China) and decreasing (e.g. Ireland, France, Netherlands, Vietnam) economic growth around the world. These comparisons reveal the implicit efficiency-equity trade-offs in each country's set of national economic growth and social policies.*[*]

Keywords: Inequality, economic growth, distribution, the elasticity of marginal utility, preferences.

JEL codes: D63, O43, P17, P27

[*] This chapter was developed with support from Ben Groom at The London School of Economics and Political Science, and Eli Fenichel at Yale School of the Environment as discussed in the Statement of Inclusion of Conjoint Work. Their support has taken the form of substantial discussions and feedback.

## 5.1 Introduction

Over the last 80 years, per-capita gross domestic product (GDP) growth has become the go-to measure of economic "success" at the national level. As a result, National Accounts dutifully report this measure of economic performance, we compare countries on this basis, and allocate development assistance by this measure. Whether evaluating countries within the purview of the European Bank for Reconstruction and Development (EBRD), Heavily Indebted Poor Countries (HIPC), or New Partnership for Africa's Development (NEPAD) initiatives, one finds per-capita GDP growth an important metric of success. It serves as such explicitly but also signals to non-governmental organisations where to direct limited resources. Sufficient growth to nudge a country from developing, to in-transition, to developed-economy status may result in the loss of substantial aid.

Per-capita GDP growth, $\dot{G}c$, however, is increasingly viewed as a poor measure of growth in wellbeing. For one, Rawl's (1971) conception – institutions, and concept – the outcome of justice, suggest different places have different distributive institutions and hence outcomes. Critically, for non-symmetric distributions – as income and consumption tend to be – mean, median and modal measures of centre differ. $\dot{G}c$ is, roughly, a measure of change in the mean of the income distribution and does not report on changes in the median and mode of a non-symmetric distribution. It provides no information about the distribution nor does any representative agent sort of conception based on it or any other single datapoint. It is even possible that measures of $\dot{G}c$ can be positive without a change in consumption for much of the population – statistical prosperity without real correspondence. The possibility of a disconnect between growth statistics and shared prosperity has, naturally, resulted in increasing calls to move beyond $\dot{G}c$ (e.g., Stiglitz, Sen, and Fitoussi, 2010).

Growth theory has, however, focussed on the mean because it is indispensable – it makes complex questions about society and welfare tractable. In terms of growth and wellbeing, it is then generally assumed that improvements for the representative, generally mean agent imply improvements for the population it represents. In national statistics, this has always been proxied for by per-capita mean income. Within the realm of theoretical exploration where this assumption is explicit, or the limitations imposed by it so well known that it is practically explicit, the limited representativeness of $\dot{G}c$ is clear. However, the practice of placing great importance on $\dot{G}c$ when its representative agent origin is not explicit, for instance in national accounts and development policy, has led to discontent with and debate about the usefulness of $\dot{G}c$.

Because the representative agent assumption is explicit, rising inequality does not contradict the Solow-Swan factor decomposition strain of growth literature (Solow, 1956) as this literature does not say anything at all about the matter. In effect though, the representative agent is too simple – it represents a uniform population.[96] One is tempted to appeal to the Kaldor-Hicks compensation criteria to suggest the representative agent method is sufficient. For instance, Mankiw, Romer, and Weil (1992) suggest their form of factor decomposition is descriptive of "international variation in the standard of living." Yet the non-normal, non-symmetric nature of income implies it is only descriptive of variation in the mean agent's standard of living and not that of the other $N-1$ members of a potentially diverse, $N$ size population. Another way to view this matter is that the factor decomposition strain of economic growth makes no claim about the ownership of factors of production, yet policy based on it presumably assumes a desirable dispersal of returns.

The distributional dimension means that a level of $\dot{G}c$ can be achieved in different ways. The lognormal distribution is particularly useful in visualising this. It has tractable analytical properties and generally fits income and consumption data well. It is well known that the mean, median, and mode of it take forms $\bar{c} = exp(\mu + \frac{1}{2}\sigma^2)$, $\tilde{c} = exp(\mu)$, and $c^{Mo} = exp(\mu - \sigma^2)$. Extending from Emmerling, Groom, and Wettingfeld (2017), the respective growth rates are $\bar{g}_t = \frac{1}{t}\Delta\mu_{0,t} + \frac{1}{t}\frac{1}{2}\Delta\sigma_{0,t}^2$, $\tilde{g}_t = \frac{1}{t}\Delta\mu_{0,t}$ and $g_t^{Mo} = \frac{1}{t}\Delta\mu_{0,t} - \frac{1}{t}\Delta\sigma_{0,t}^2$ where $\Delta\mu_{0,t} = \mu_t - \mu_0$ and $\Delta\sigma_{0,t}^2 = \sigma_t^2 - \sigma_0^2$. Here, we observe it is possible that increases in $\sigma^2$ can drive growth in mean income, increasing inequality. But inequality-driven growth also implies that for many workers – those earning and consuming around the mode – the economic outlook worsens. The most common, modal, experience would then conflict with national statistics that indicate "economic growth."[97] Next, an empirical exercise suggests the importance of the distributional complication.

[96] Alternately, it can be represented by a degenerate distribution (all have the same value).

[97] The mean-modal spread also exceeds the mean-median gap sometimes proposed as a measure of inequality suggesting the latter is insufficient in capturing the issue.

### 5.2 An Empirical Exploration

The factor decomposition literature suggests that levels of $\dot{G}c$ can be achieved in different ways – combinations of capital, labour, productivity, etc. Using the lognormal assumption, levels of $\dot{G}c$ can also be achieved through combinations of growth in $\mu$ and $\sigma^2$ which indicate how distribution is structured in an economy. As with factors of production, policies and institutions can facilitate growth in either one or both distributional dimensions. Growth in $\mu$ and $\sigma^2$ highlight the trade-off between growth in the median and the right tail of the distribution, respectively, for overall per-capita growth. We perform growth distributional decomposition on some internationally comparable datasets containing mean and median income data which suggest how different distributional policies impact growth.

We utilise Luxembourg Income Survey (LIS), Organisation for Economic Co-operation and Development (OECD), World Bank, and World Inequality Database (WID) datasets to derive growth paths that countries have taken. The LIS and OECD provide comparable estimates based on extensive, representative household surveys. The World Bank dataset – also survey-based – includes data on income and consumption for a more diverse set of countries. The WID data instead includes percentile incomes across the distribution for a few countries using a shared methodology. For each country, we assume a lognormal distribution relationship, and then decompose the median to estimate $\mu$ followed by the mean to estimate residual $\sigma^2$. Figure 31 presents the LIS and OECD data decomposition into the share of economic growth contributable to median ($\mu$-driven) and inequality ($\sigma^2$-driven) growth. We can observe that despite using similar household-scale measures, the OECD data generally suggests less unequal growth. More importantly, differences within each dataset are apparent. Nations with similar mean growth rates achieve it by different distributional processes. Notable inequality-increasing outliers are the United States, Germany, and Canada over the last 30 to 40 years. In contrast, Ireland, France, the Netherlands, and Hungary experienced growth in combination with reductions in inequality.[98] Of course, growth and distributional statistics cannot identify all sources of unrest – discontent over immigration and essential commodity and fuel prices (in Hungary, France) can lead to turmoil even when national statistics suggest improvements in general wellbeing.

[98] Also noted, for instance, in Garbinti, Goupille-Lebret, and Piketty (2018) in the case of France.

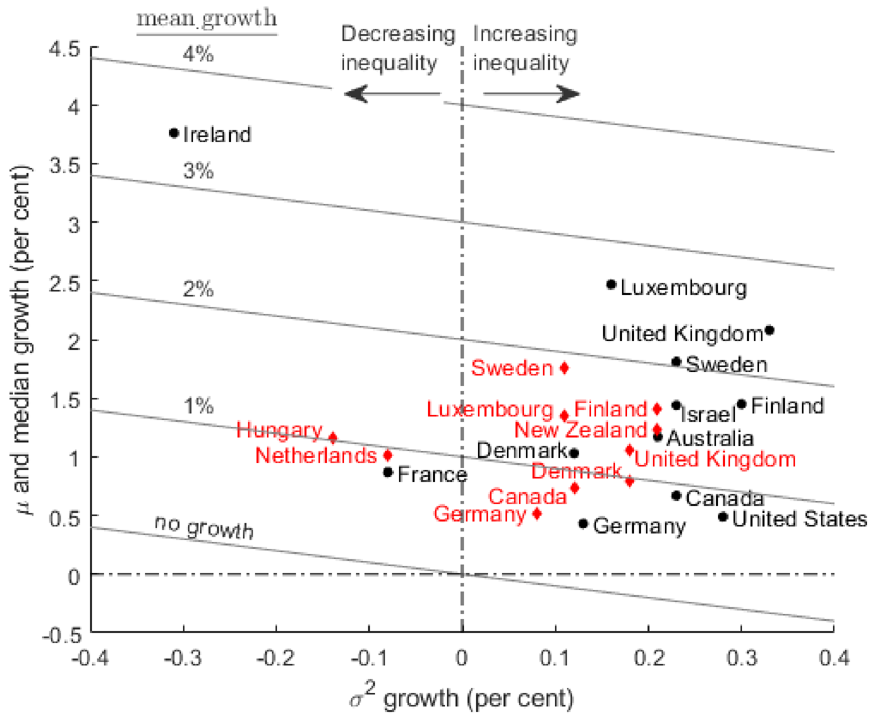


FIGURE 31. LIS AND OECD DATA ANNUAL GROWTH RATE DECOMPOSITION

*Notes:* Luxembourg Income Study (LIS) estimates (black dot markers) based on PPP-adjusted equivalised disposable household income from surveys conducted between 1979 and 2013. OECD estimates (red diamond markers) based on CPI-adjusted household income in national currencies from surveys conducted between 1975 and 2017. OLS estimates of annual growth rates in μ, $\sigma^2$, and mean growth for countries listed in Table 32 of appendix *A5.1 Data Tables Supporting Figure 31 and Figure 32*.

*Source:* Author calculations based on data from OECD (2019), and Thewissen, Nolan, and Roser (2016) which is derived from the LIS Database.

Within the preceding LIS and OECD datasets, all countries experienced $\dot{G}c$ on average along various distributional trajectories. Decomposition of the World Bank dataset in Figure 32 provides a more varied picture as it describes nations with more varied institutions, endowments, and states of development. We might qualitatively make some observations. Along the rightmost edge of the data mass signifying greater inequality-driven growth, we find the United States, Germany, Bangladesh, urban India, and South Africa. These countries are known for increasing concern over inequality, but generally stable political regimes over the period. On the other extreme edge signifying decreasing inequality-driven growth, we find Mexico and several Central and South American countries. These are known for stronger communal traditions but also political instability over the period. We might suggest that while distribution is but one factor in economic "success", too unequal growth in either direction leads to social disquiet. We observe that, at least mechanically, there is an equity-efficiency trade-off that occurs. We can also observe that negative growth is not experienced equally as in the case of Croatia – that growth by itself is not a driver of inequality.

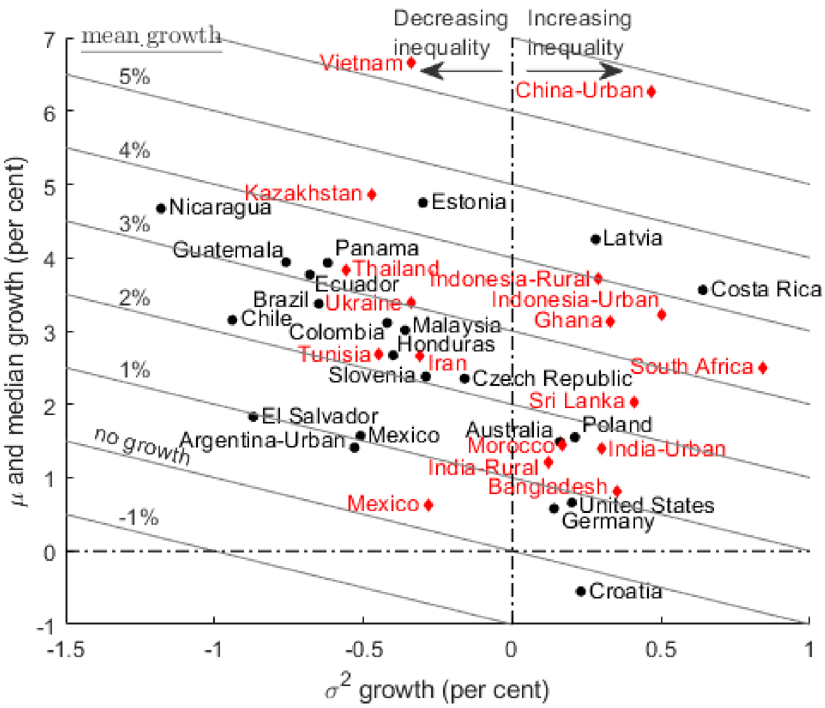


FIGURE 32. WORLD BANK DATA ANNUAL GROWTH RATE DECOMPOSITION

*Notes:* World Bank income-based survey data (black dot markers) and consumption-based data (red diamond markers). OLS estimates of annual growth rates in μ, $\sigma^2$, and mean growth for countries listed in Table 33 of appendix *A5.1 Data Tables Supporting Figure 31 and Figure 32*. Estimates based on PPP-adjusted household per-capita income and consumption expenditures.

*Sources:* Author calculations based on data from World Bank (2019).

National policies are not necessarily designed to share the returns to production in any socially preferred way. It is difficult to make the case that socially destructive levels of inequality are actually in the public's interest. However, when national policies do embody "preferences", different trajectories in $\mu$ and $\sigma^2$ also result. Observed growth paths embody the political costs of growth, distribution and historical ownership of policy. Rich histories have, perhaps, led to traditions of sharing the returns to economic growth in each country that we would generally lump together as "preferences." Growth in $\mu$ versus $\sigma^2$ is one salient result. However, how does an economy move from preferences over sharing to embarking on a growth path? A review of the literature on growth, distribution, and measuring inequality precedes a dynamic growth model that incorporates distribution.

### 5.3 Literature Review

An entire academic industry discusses inequality and growth and produces supplemental statistics to $\dot{G}c$. These generally do not acknowledge the representative agent source of the issue but do tend to explore the non-symmetric distributional departure of measures of centre. Often this is by jumping to conclusions on the matter – lamenting inequality and developing statistics without the nuance of what preferences are in a society.

One introductory statistic for inequality is the spread between per-capita (mean) income and the median, with $median < mean$ indicating inequality (Barro 2000). Measuring the spread of means and medians is important where inequality is concerned because an economic policy that focusses on one or the other – the mean versus median debate, still neglects the distributional issue.[99] Yet the spread, or growth in the spread, is merely descriptive and ignores notions of fairness, institutional constraints, or other "social preferences" over distribution.

Another popular distributional measure is the Gini coefficient (Gini, 1912; Dalton, 1920) in the form summed over the Lorenz curve (Lorenz, 1905). Despite its popularity, Atkinson (1970) discusses why it is insufficient for the task. Among the criticisms, it too neglects preferences, and we append that it only provides a static snapshot. In the context of economic growth, a dynamic measure is more relevant.

Another approach has been to consider the distributional implications of economic growth, sometimes quite formally, without arriving at a statistic. The implication is that either more or less inequality is beneficial, not that a certain amount of inequality is preferred. Early endeavours of this variety include Kuznets (1955, 1973), Stiglitz (1969), Solow (1974), and Leontief (1983). Among other conclusions, they suggest an important role of different savings rates along the income distribution. However, by not taking the inequality problem "beyond the drawing board", we did not inherit a functional statistic from these endeavours.

Advancements in the study of growth and inequality have also progressed alongside general methodological improvements. Van der Ploeg (1983) provides an optimal control approach and fascinating implementation of a predator-prey model to employment and growth. Benhabib and Rustichini (1996) offer a game-theoretic, political economy model

[99] Note, however, the median has an important distributional quality that is robust to increases in the tail.

of appropriation among interest groups. However, as with their predecessors, these remained academic rather than practical endeavours.

Others have instead focused on poverty reduction resulting in more actionable poverty indices. The most introductory of these is a measure of the share of the population living below some poverty line – the headcount ratio. The poverty gap index, Watts index (Watts, 1964; Zheng, 1993), Sen-Shorrocks-Thon index (Sen, 1976; Shorrocks, 1995), and Foster-Greer-Thorbecke indices (Foster, Greer, and Thorbecke, 1984) all improve on the headcount notion by incorporating the intensity of poverty. Other measures such as the Multidimensional Poverty Index (MPI) take a more general perspective on wellbeing as advocated for by Sen (1999). The MPI – for better or worse – summarises progress across the Human Development Index through a subjective weighting scheme to arrive at a measure of welfare (UNDP, 2010; Alkire and Jahan, 2018). One cannot seriously argue that mass poverty is ever socially optimal, and these indices share the merit of focussing the discussion on the most severe of distributional shortcomings. This study in no way seeks to supplant such measures. However, what about cases where the issue is not extreme, pervasive poverty, but rather general discontent with the distribution of growth?

Interest in differences in inequality and growth have resulted in several empirical explorations in recent decades. International comparisons have been particularly fashionable such as by Barro (1991, 2000). These generally link economic growth to measures of inequality such as the Gini coefficient to arrive at some conclusion on whether inequality supports or hinders growth. See overviews in, for example, Shin (2012), Cingano (2014), and Grigoli and Robles (2017). This body of research has been less informative in total, however, as some find inequality positively, and others negatively, related to economic growth.

Persson and Tabellini (1994), like others, find that income inequality is harmful to growth. However, they attempt a plausible, non-savings-based explanation linking income inequality to policies that do not protect property rights and thus do not protect the private returns to investment. Halter, Oechslin, and Zweimüller (2014) add a time dimension – inequality increases economic growth in the short run but reduces it in the long term, perhaps linked to Persson and Tabellini’s institutional argument. Banerjee and Duflo (2003) instead suggest a more complicated relationship – that changes in inequality in either direction reduce growth. Then in a manner not unlike the argument that higher earners save more, Jones (2019) suggests a model where those with the highest incomes drive innovation. Finally, Jorgenson (2018) reiterates the many arguments to date that we should measure welfare rather than income because it captures notions of inequality, poverty, and distribution that income cannot.

Yellen (2016), Chetty et al. (2017), and Fried, Novan, and Peterman (2018) have also weighed in on the enduring challenge of inequality and economic policy. These have generally lamented the state of inequality rather than chart a path forward. Perhaps this is because attempts to simplify the distributional discussion have not always been well-received (Piketty 2015). The level of criticism levelled at incorporating, or even discussing inequality and economic growth are striking given the discussed explicit limitations of the factor decomposition growth literature.

After reviewing the wealth of work, lack of consensus, and intensity of criticism, one may want to just jump to the punchline as in Stiglitz (2016) and prescribe a broad, generally common sense, but expensive set of remedies. Perhaps a more practical approach begins with an observation that factors at both the top and bottom of the income distribution impact economic growth. As such, no single statistic captures whether economic growth benefits a society (Voitchovsky, 2005, 2009). Instead, practitioners should seek to complement $\dot{G}c$ with appropriate context on distribution and preferences in a policy-relevant manner.

Our goal is to arrive at a useful complementary statistic to $\dot{G}c$. We begin by building on the Ramsey-Cass-Koopmans (RCK) model, which fuses the Ramsey rule for how much a society should save, with modern growth theory (Ramsey, 1928; Cass, 1965; Koopmans, 1963). We also note recent empirical work on the relationship between inequality and savings that supports the premise that the wealthy save a larger share of income. U.S. tax data covering a century puts firm numbers in support of the belief that there are differences in savings rates (Saez and Zucman, 2016). As a general matter of allocating scarce personal resources, we expect the trade-off between current consumption and savings to hold beyond the U.S. Closely related, research using a global poll of over 1.7 million people suggests there is some relatively consistent income satiation point where savings rates go from zero for much of the population, to positive for high earners (Jebb, Tay, Diener, and Oishi, 2018).[100]

[100] The global average satiation point is $60,000 to $95,000/year depending on estimation method. The satiation point appears to differ both geographically and demographically.

### 5.4 Incorporating Consumption Distribution into Growth Theory

We have made claims on how modern growth theory and indicator statistics fall short of policy need. We now put foundations from the literature to work by, in effect, making a single change to the Solow-Swan paradigm. While a more sophisticated approach to studying economic growth and inequality than using a representative agent is needed, it should transparently integrate economic distribution yet result in clear, concise policy recommendations. We build on recent advancements in Emmerling, Groom, and Wettingfeld (2017) and incorporate a representative distribution, rather than a representative agent, into an efficient capital market assumption. The resulting model suggests how growth contributes to mean and all other consumption points through competing influences.

We assume the constant relative risk aversion (CRRA) utility form (Atkinson, 1970) which has particularly desirable analytical properties, and the lognormal form (McAlister, 1879) of representative consumption (income minus savings) distribution (Arrow et al. 2014; Pinkovskiy and Sala-i-Martin 2009; Battistin, Blundell, and Lewbel 2009).[101] We then implement the consumption side of the RCK model in a simplified form as objective:[102, 103]

(39) $$\max_{\{\mu(t),\sigma^2(t),t>0\}} W_0 = \int_{t=0}^{\infty}\left(\int_0^{C_{max}} u\big(c_i(t)\big)\,dc(t)\right)e^{-\rho t}dt,$$

where we abstract from uncertainty, note that $c_i(t)\sim LN(\mu(t),\sigma^2(t))$, and that for any agent, $i$:

(40) $$u(c_i(t)) = \begin{Bmatrix} c_i(t)^{1-\eta}(1-\eta)^{-1}, & if\ \eta>0, \eta\neq 1 \\ \ln(c_i(t)), & if\ \eta = 1 \end{Bmatrix}$$

which incorporates the elasticity of marginal utility of consumption, $\eta$. That is, rather than concerning ourselves with the welfare function of a representative agent, we use the objective as in Emmerling, Groom, and Wettingfeld (2017) as the welfare over the

101 Limitations to the representative consumer approach beyond what have already been discussed can be found in Caselli and Ventura (2000).

102 It may be helpful to note $\nabla W_0(\mu_0,\dots\mu_T,\sigma_0^2,\dots,\sigma_T^2) = \left(\frac{\partial W_0}{\partial\mu(0)},\dots,\frac{\partial W_0}{\partial\mu(T)},\frac{\partial W_0}{\partial\sigma^2(0)},\dots,\frac{\partial W_0}{\partial\sigma^2(T)}\right)^{\mathsf{T}}$ where $\frac{\partial W_0}{\partial\mu(t)} = e^{-\rho t}\int_0^{C_{max}} u\big(c_i(t)\big)\left(\frac{ln(c_i(t))-\mu(t)}{\sigma^2(t)}\right)dc(t)$ and $\frac{\partial W_0}{\partial\sigma^2(t)} = e^{-\rho t}\int_0^{C_{max}} u\big(c_i(t)\big)\left(\frac{\big(ln(c_i(t))-\mu(t)\big)^2}{2\sigma^4(t)} - \frac{1}{2\sigma^2(t)}\right)dc(t)$ for $t=(0,\dots,T)$.

103 We have abstracted from population growth in this application which is traditionally included in the RCK model.

distribution as a whole while incorporating preferences over distribution represented in $\eta$. We then want to choose the "institutional structure" to maximise our objective which is subject to $\eta$ and the initial distribution. However, it is impossible to write a satisfactory function for something as ambiguous and far-reaching as the institutional structures of society that distributes the returns from production. Instead, we proxy for it by jumping to the structure's convenient distributional representations - values of $\mu$ and $\sigma^2$.

The objective is subject to a system of differential equations describing developments in capital and consumption possibilities (see Note 12 for their derivation):

(41) $$\dot{k}(t) = \left[\int_0^{C_{max}} s\big(c_i(t)\big)dc(t)\right] - \delta k(t),$$

(42) $$\dot{\bar{c}}(t) = -\frac{u\prime(\bar{c}(t))}{\bar{c}(t)u\prime\prime(\bar{c}(t))}\bar{c}(t)\big(f'\big(k(t)\big) - \delta - \rho\big),$$

where $k(0) > 0$, the production function $f\big(k(t)\big)$ is of simple Cobb-Douglas form, and the savings rate can be dependent on position in the distribution.[104]

Note 12 – Deriving the Mean Consumption State Equation

Derivation of the RCK state equations is well known. Equation (41) is a natural extension of the Solow-Swan and RCK model capital accumulation equations. Equation (42) is derived based on a present value Hamiltonian of equations (39) and (41): For tractability, choose the generally well known value $\bar{c}(t)$ and define $\mathcal{H}_{pv} = e^{-\rho t}u(\bar{c}(t)) + \lambda(t)\dot{k}(t)$ with costate variable $\lambda(t)$ and use the simpler capital accumulation form $\dot{k}(t) = f\big(k(t)\big) - \bar{c}(t) - \delta k(t)$. As an optimal control problem, the first-order necessary conditions for a maximum are $\frac{\partial\mathcal{H}}{\partial\bar{c}} = 0$ and $\frac{\partial\mathcal{H}}{\partial k} = -\dot{\lambda}(t)$. So, $\frac{\partial\mathcal{H}}{\partial\bar{c}} = e^{-\rho t}u'(\bar{c}(t)) - \lambda(t) = 0$ and $\frac{\partial\mathcal{H}}{\partial k} = \lambda(t)\big(f'\big(k(t)\big) - \delta\big) = -\dot{\lambda}(t)$. Then, to find how the state changes over time, $\frac{\partial\dot{\mathcal{H}}}{\partial\bar{c}} = -\rho e^{-\rho t}u'\big(\bar{c}(t)\big) + e^{-\rho t}u''\big(\bar{c}(t)\big)\dot{c}(t) - \dot{\lambda}(t) = e^{-\rho t}u'\big(\bar{c}(t)\big)\Big[-\rho + \frac{u''(\bar{c}(t))\dot{c}(t)}{u'(\bar{c}(t))} - \frac{\dot{\lambda}(t)}{e^{-\rho t}u'(\bar{c}(t))}\Big] = -\rho + \frac{u''(\bar{c}(t))\dot{c}(t)}{u'(\bar{c}(t))} - \frac{\dot{\lambda}(t)}{\lambda(t)} = 0$. Substituting the result of $\frac{\partial\dot{\mathcal{H}}}{\partial\bar{c}}$ into that of $\frac{\partial\mathcal{H}}{\partial k}$ for $-\frac{\dot{\lambda}(t)}{\lambda(t)}$ and rearranging the result as $\dot{c}(t) = -\frac{u'(\bar{c}(t))}{u''(\bar{c}(t))}\big(f'\big(k(t)\big) - \delta - \rho\big) = \dot{c}(t)$. Multiplying $\dot{c}(t)$ by $\frac{\bar{c}(t)}{\bar{c}(t)}$ then allows the substitution of $\frac{1}{\eta} = -\frac{u\prime(\bar{c}(t))}{\bar{c}(t)u\prime\prime(\bar{c}(t))}$ where appropriate.

[104] The per-capita Cobb-Douglas production function adopted is of form $f(k(t)) = Ak(t)^{\alpha}$ with total factor productivity multiplier $A$, and output elasticity of capital $\alpha$. In practice it is fitted to initial period economic parameters.

Equation (42) is a modified Keynes-Ramsey Rule (Ramsey, 1928) which we can restate as $\dot{\bar{c}}(t) = \frac{\bar{c}(t)}{\eta}\left(f'\big(k(t)\big) - \delta - \rho\right)$ where the elasticity of marginal utility with respect to consumption at the mean is $\eta = -\frac{\bar{c}(t)u''(\bar{c}(t))}{u'(\bar{c}(t))}$. The implication is that we weight changes in consumption by an assumed common social preference containing the inverse marginal utility of consumption. This links the literature on inequality preferences to growth and can be contrasted against Negishi (1960). A convenient distributional assumption also links savings and growth as we specify the savings function to include the recent literature on unequal savings rates and satiation points as

(43) $$s(c_i(t), b) = \begin{Bmatrix} 0, & if\ c_i(t) < b \\ \psi(c_i(t)), & if\ c_i(t) \geq b \end{Bmatrix}$$

where $b$ is some point of satiation – an ancillary condition on utility signifying whether necessary consumption is satisfied in each period before the agent saves for later consumption (e.g. continued consumption in retirement). We derive $b$ from observations in Saez and Zucman (2016) and Jebb, Tay, Diener, and Oishi (2018) which can be either relative to the distribution or absolute. The embedded function $\psi(\cdot)$ conceptually satisfies $\psi'(\cdot) > 0$ and at least initially $\psi''(\cdot) > 0$, but in practice requires less sophistication to get the expected results. In appendix *A5.2 Implications of Some Alternative Model* , we explore different interpretations – the implications of choosing a uniform savings rate as well as of setting $b$ as absolute rather than relative. A dystopian, or revolt constraint can also be specified – for instance that median consumption in any period cannot be lower than the initial poverty level. The rational is that if much of the population is pushed into abject poverty, one may expect a collapse of the sort of institutions that make economic activity predictable and modellable.

The operation and design of the model is purposefully simple. An "economy" arrives at an efficient outcome by balancing the influence of the concavity of the distribution-weighted utility function from consumption for all agents, against savings-driven growth supported by the upper tail.[105] The state equations $\dot{\bar{c}}(t)$, $\dot{k}(t)$, and the impact of $\eta$ are in

[105] It can be shown that for concave, symmetric utility forms – exhibiting decreasing marginal utility – that minimizing variance maximizes utility.

turn dependent on the distribution through savings and initial distributional conditions $\mu(0)$ and $\sigma^2(0) > 0$.[106]

### 5.5 A Simulation

In the model, "social preference" over income distribution has a pervasive influence on distribution and growth. Some have made ethical arguments why there might be a social preference at all, while others just assume that such preferences exist (Stern, 2007; Dasgupta, 2008; Tol, 2010). An alternative is to suppose that institutions supporting a set of economic growth paths are constrained by popular mandate. A distribution process then emerges that appears as if society has preferences over the degree of inequality. We can, at a minimum, interpret $\eta$ as a summary parameter of complex institutional arrangements that underpin distribution in society – it implies an $\eta$ as in the preceding paper.

The appropriate value of the social preference indicator $\eta$ is a matter of ongoing debate and methodological development. See, for example, Evans (2005), and Groom and Maddison (2019).[107] Emmerling, Groom, and Wettingfeld (2017) note that estimates of $\eta$ vary from 0.4 to 4 depending on the context, and the preceding chapter finds long-run, national estimates rarely fall outside $\eta = [1, 2.5]$. Notably, some argue for $\eta = 1, 2$ as holding special significance (Buchholz and Schumacher, 2010; Dasgupta, 2008; Groom and Maddison, 2019; Stern, 2007; and Tol, 2010). $\eta = 1$ suggests a social preference for consumption maximisation – an emphasis on developing mean consumption potential, while larger values suggest greater inequality aversion. Some instead specifically advocate for $\eta = 2$ on the premise of emphasising balanced economic growth – an emphasis on median growth. We remain detached from the normative debate and rather explore the implications of inequality preferences on optimal economic distribution.

To form a concrete example of the impact of $\eta$, we start from initial conditions $\mu(0), \sigma^2(0)$ derived from U.S. national statistics on median and mean household income (U.S. Census Bureau, 2019) and calibrate parameters in the preceding model from popular sources.[108] We then perform repeated simulations to obtain preferences for 30-year growth paths in a $\mu$ and $\sigma^2$ space for select parameter values in $\eta = [0.5, 4]$. By the lognormal

[106] These constraints can be operationalized as the 2T vector $(\mu(0), \dots, \mu(T), \sigma^2(0), \dots, \sigma^2(T))^\intercal$ where $\mu(t) = \ln(\bar{c}(t)) - \frac{1}{2}\sigma^2(t)$ and $\sigma^2(t) = 2(\ln(\bar{c}(t)) - \mu(t))$ for $t = (0, \dots, T)$ derived from the mean of the lognormal distribution.

[107] $\eta$, the elasticity of marginal utility with respect to consumption, is interpretable as social inequality aversion among other uses.

[108] Penn World Table 9.0 (Feenstra, Inklaar, and Timmer, 2015) is used to calibrate the production function and other parameters as well as Saez and Zucman (2016) and Jebb, Tay, Diener, and Oishi (2018).

distributional assumption, we can represent results in a grid of average annual growth rates of mean, median, and modal incomes as in Figure 33. Optimised economic progress after 30-years falls along the dashed line resulting from the mean under the lognormal assumption. The direction and extent of pathways in $(\sigma^2, \mu)$ space differ by preference over $\eta$. After an initial correction from the real-world trend because present levels of inequality are not preferred, growth pathways generally fall along what we may approximate as a vector as discussed in the preceding chapter. We also include a projected growth path based on the 1967-2017 U.S. trend. Naturally, the real trend includes all sorts of shocks not included in the $\eta$-based simulation.

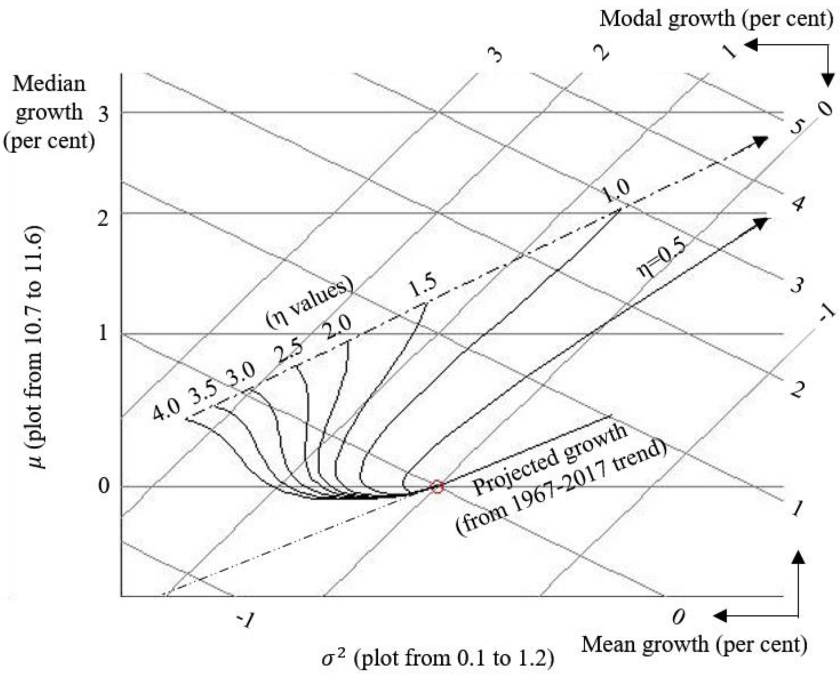


FIGURE 33. OPTIMAL $\mu$ AND $\sigma^2$ PARAMETERS OVER 30 YEARS CONDITIONAL ON $\eta$

*Notes:* Optimal growth paths in $(\mu, \sigma^2)$ space for the U.S. depending on the social preference for inequality represented by η (higher value interpretable as a preference for less inequality) and projected U.S. growth based on the 1967-2017 trend. These are overlaid on a contour map of the implied annualised mean, median, and modal growth rates.

*Sources*: Author calculations based on repeated simulations and data from the U.S. Census Bureau (2019).

We must remain aware of the lognormal distributional assumption when interpreting Figure 33 and the results in general. One issue is that the lognormal approximates the bulk of any income or consumption distribution well, but not ultrawealthy outliers. We could use the gamma distribution or more sophisticated alternatives, but the capacity to represent growth in a space where mean, median, and modal implications can be represented simultaneously would be lost. Instead, the point is to have an intuition for when critical economic and social parameters, like inequality preference, drive the results, and not the distributional assumption.

It is also important to discuss the implications of our welfare function. Over the 30-year planning horizon, the decisionmaker weighs each agent $i$'s benefit in each period $t$ against every other agent's benefit in every period (see discussion in, for instance, Gollier, 2011). The planning horizon is critical because the $\eta$ parameter summarises inequality preferences between every $i$ in every $t$. Preference for less inequality (higher $\eta$) implies not just a preference for more equal distribution in each period, but also greater intertemporal redistribution to earlier periods to balance later gains from economic growth, regardless of the discount rate. Thus, a forward-thinking, multiyear planning society with inequality aversion may behave as if discounting even if $\rho = 0$.

Some observations: After the initial correction of distributional policy to match preferences, planning horizon, and lack of real world instability, we see the $\eta = 2$ path settles into distribution-preserving growth ($\bar{g}_t = \tilde{g}_t = g_t^{Mo}$). The $\eta = 1$ path also settles into what would be mean-maximising growth in a short run, noniterative model. However, a preference for greater inequality, ($\eta < 1$), results in higher average mean growth through higher capital accumulation from savings. This higher annualised growth rate, however, is driven by gains enjoyed in later years at the expense of lower consumption in earlier ones.

Similarly, a preference for economic policy focused on median rather than mean income, advocated for instance by Aghion et al. (2013), European Commission (2014), and Stiglitz (2012) based on inequality aversion, might maximise short-run but not long-run median growth. Policy emphasising greater inequality can also result in higher median growth (eventually) through savings. The issue is whether we can first tolerate lower median growth and then greater long-run inequality. Under the lognormal assumption, another trade-off is a reduction in, or possibly negative, modal growth.

Instead of choosing to monitor median income as our metric, suppose instead a society acts to maximise median income growth. That is, suppose that society chooses policies suggesting an $\eta$ (recall, however, this is a parameter) that maximizes growth in $\tilde{c} = exp(\mu)$. From a sufficiently long-run perspective, we might then choose to maximise growth in the mean, $\bar{c} = exp(\mu + \frac{1}{2}\sigma^2)$ too, since growth in $\sigma^2$ has a positive effect on growth in $\mu$ through savings. But this would require ignoring intergenerational differences and instead supposes that growth in the annualised average is sufficiently rewarding. But this conflicts with what $\eta$ is indicating which is a preference over inequality between all $i$ and $t$. The point is also not to choose $\eta$, a parameter indicating social preferences. The optimal distributional policy instead follows from (and the previous chapter suggests) an $\eta$ parameter representing preferences.

More generally, emphasising either mean or median growth does not necessarily result in increased total wellbeing based on the CRRA utility form. For many social preference values where $\eta \neq 1, 2$, planning for growth in either mean or median statistic is socially suboptimal. Instead, for any social preference over distribution and inequality an entirely different optimal growth path is preferred. A national statistic emphasising economic growth in any form alone is not informative on whether national preferences over distribution are being addressed.

A static statistic is also insufficient to describe progress. Along the optimal growth paths in $(\mu, \sigma^2)$ space, a mean-median relationship or other such comparisons, one value tends to grow comparatively faster in most preference scenarios. Current statistics on distribution do not adequately reflect that even stable social preferences result in changing levels of inequality with changing incomes.

We also note that projected U.S. growth continuing from the 1967-2017 trend – generally predictable in the long-run as a linear estimate fits the 1967-2017 data well – is not generally short-run optimal under plausible inequality preferences which would be indicated by an $\eta$ closer to 1.5 or 2. Instead, the current policy suggests an as-if social preference more unequal. The result is a sacrifice of growth potential by any measure of centre – mean, median, or mode.[109] The result is that any policy "revolution" to a more equitable preference-based growth policy begins with economic upheaval. We find support for this in Banerjee and Duflo (2003) where any preference change incurs a temporary loss in $\dot{G}c$. A policy conflict then occurs – a representative government, knowing $\eta$, cannot justify a growth trend suggesting an inequitable social preference, yet policymakers would be hesitant to embark on a radical policy realignment if $\dot{G}c$ is the metric of economic success.

Finally, we can put comparisons on perhaps more familiar ground as a set of comparative distributions in Figure 34. These emphasise how different values of $\eta$ result in different consumption distributions over time.[110] One might appeal to the Keynes-Ramsey Rule

[109] Graphically, a trend maximizing short run growth in one measure of center, for instance the mean, would run perpendicular to the mean growth contour lines – traversing them by least distance.

[110] Assuming national policy impacting distribution is based on preferences represented by $\eta$.

rearranged as $\frac{\dot{\bar{c}}(t)}{\bar{c}(t)} = \bar{g}_t = \frac{1}{\eta}\left(f'\left(k(t)\right) - \delta - \rho\right)$ to understand their relationship.[111] A preference for less equality, or greater inequality, (smaller $\eta$) results in a preference for more income growth in the right tail. Under the lognormal assumption, this necessarily results in a reduction in modal income growth and a fattening (higher frequency) in at least some upper-intermediary (middle) incomes. There is a greater dispersal across incomes too – less common experience across households which might impact national cohesion. Unfortunately, this outcome again highlights the inadequacy of both mean and median measures. Both measures may register growth, while a subset of agents around the mode are pushed into poverty.

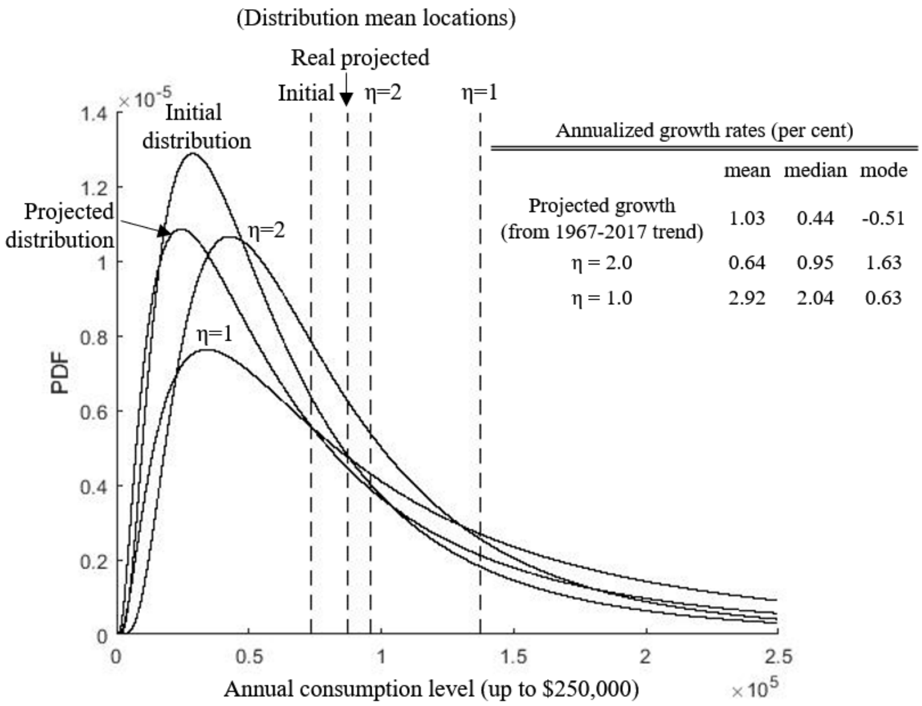


FIGURE 34. CONSUMPTION DISTRIBUTION AT 30 YEARS, BY $\eta$ VALUE

*Notes:* Consumption distributions and measures of centre for select values of η. Initial distribution based on recent U.S. national statistics and projected distribution based on 1967-2017 U.S. growth trend. A smaller η value, representing a greater preference for inequality, results in higher growth in inequality and the right tail. Und the lognormal assumption, the modal statistic may decrease even if the median and mean increase as projected based on current U.S. policy – a particularly adverse effect for much of society.

*Sources*: Author calculations based on simulations and data from the U.S. Census Bureau (2019).

Whatever national preferences over distribution and growth are, a complementary national statistic to $\dot{G}c$ appears warranted. It can inform whether growth policy is placing the economy on a preferred or at least familiar path. Policymakers and the public can then

[111] One may also want to appeal to this in attempting to estimating values of $\eta$ empirically. That is, suppose $\eta$ implied in any period of economic activity can be estimated as a ratio of net marginal product and mean consumption growth. This is not explored here, however.

judge whether the path is desirable in part based on whether they take a normative or positive perspective. We conclude by presenting candidates for the purpose.

### 5.6 Economic Performance in Terms of Distribution

The emphasis of this paper is on the importance of incorporating social preferences over distribution into economic growth policy. At a minimum, policies over distribution result in growth paths as if there is a preference, even if it is one that does not benefit most of society. A statistic indicating when we have departed from a distributional growth path, preference, etc. complements $\dot{G}c$ because it says something about whether growth policy will be regarded as equitable or at least results in outcomes that match expectations. Policy fitness, in turn, informs on whether policy supports social stability. It can also say something about $\dot{G}c$ – whether it will exceed or fall short of what the factor decomposition growth literature predicts because of the savings-distribution relationship.

Whether a growth path matches social preferences is the important characteristic for monitoring distribution, not which path we are on. One way to develop a statistic that informs on this is to incorporate $\eta$ as a benchmark and contrast it against implied-$\eta$ measures as in the preceding paper or other comparisons. Incorporation results in an internationally comparable statistic while retaining national context because we ask how closely growth policies match each society's preference in the aggregate. We then compare how on-target each society's policies are, rather than comparing growth when preferences over how growth is achieved may be radically different.

To incorporate the benchmark, consider a growth statistic over the distribution of form

$$\nu_{(\cdot)} = D(t) - D(\eta) \tag{44}$$

where $D(t)$ reports as the realisation a measure of economic performance in *t*, $D(\eta)$ identifies what the society would prefer in the same units based on social preferences, and a positive (negative) value of $\nu_{(\cdot)}$ suggests greater (lesser) inequality growth than preferred.

An approach in somewhat familiar units readily emerge. An equivalent measure to an $\eta$ growth path is the explicit rate of divergence of two points on the income distribution. Mean and median statistics are often available. Suppose $D(\eta)$ is the growth rate of divergence in mean and median implied by social preference value $\eta$, and $D(t)$ the observed rate of divergence in the same period. The new statistic follows as

(45) $$\nu_{mean/median} = g(t)_{mean/median} - g(\eta)_{mean/median}$$

If the lognormal assumption fits the distribution, we might estimate $\nu_{mean/median}$ from its mean and median analytical values as $\nu(\mu, \sigma^2, \eta)$ where $g(t)_{mean/median} = \frac{1}{t}\frac{1}{2}\Delta\sigma^2_{0,t}$ which is growth attributable to a spread in the distribution. If $g(\eta)_{mean/median}$ is based on $\eta = 2$, signifying a preference for equal growth such that $\bar{g}_t = \tilde{g}_t = g_t^{Mo}$, this implies $g(\eta)_{mean/median} = 0$ and $\nu_{mean/median} = \frac{1}{t}\frac{1}{2}\Delta\sigma^2_{0,t}$ but this is an exception.

We may also want to revisit the normal coefficient of variation (CV) based on its familiarity in the field and general usefulness in discussing distribution. The appropriate statistic, however, is the divergence from preferences over the growth rate of the coefficient of variation, $g_{CV}$:[112]

(46) $$\nu_{(CV)} = g(t)_{CV} - g(\eta)_{CV}$$

The Gini coefficient is a popular and intuitive measure of inequality, and we would be remiss to leave it out. As in the preceding forms, a comparison of growth in the observed Gini coefficient, $g(t)_{Gini}$, to the implied socially optimal trend in a society, $g_{Gini}$, is most informative:

(47) $$\nu_{(Gini)} = g(t)_{Gini} - g(\eta)_{Gini}.$$

Finally, one approach is to state $D(t)$ as the value of $\eta$ implied by distributional growth in each period, $\eta(t)$, e.g. implied-$\eta$. We can then compare this to $D(\eta) = \eta$ as society's generally constant social preference. The statistic on growth and distribution is then[113]

(48) $$\nu_{(\eta)} = \eta - \eta(t)$$

An advantage of benchmark $D(\eta) = \eta$ is that each $\eta$ value identifies a growth path with an approximate rate of divergence, and so has a dynamic perspective "baked in". An issue is that we must measure the abstract quality $\eta(t)$ consistently and frequently. Another issue

[112] From the normal coefficient of variation $CV = \frac{\sigma}{\mu}$, growth is $g_{CV} = \frac{\dot{CV}}{CV} = \frac{\mu\dot{\sigma}-\dot{\mu}\sigma}{\mu\sigma}$.

[113] In this representation of $\nu_{(\cdot)}$, the realisation and pure preferences switch places. This is so that the sign of $\nu_{(\eta)}$ has the same interpretation as in the comparison measures.

is the interpretation of the units of measure as $\nu_\eta$ is in units of divergence from preferences. However, this would hardly be our first economic measure in abstract units.

As an empirical starting point, Table 16 through Table 18 report average annual values for the four proposed measures of $D(t)$ for countries based on five datasets. The values for $g_{mean/median}$ are also equivalent to $\nu_{(mean/median)}(\eta = 2)$ – the values of $\nu_{(mean/median)}$ if we assume the social preference is an equal growth rate across the distribution. Except for the directly $\eta$-based statistic, these values are normalised by mean growth to report the rate of divergence, or inequality growth, per-per cent growth. The $\eta$ measure does not require normalisation because the statistic informs on the $\eta$ path of the economy without regard to the magnitude of growth. When sufficiently long time-series are available as in the WID dataset, we also report values for significant periods of development. Because the measures are composed from OLS estimates on different qualities of the data – annual Gini coefficients, mean, median, and residual values – not all $D(t)_{(\cdot)}$ estimates are statistically significant and reported for all countries. Consequently, the global picture is not representative – only nations with sufficiently stable growth and data collection initiatives are included – those where at least one distributional measure can be calculated at generally acceptable confidence levels.

TABLE 16—GROWTH AND DISTRIBUTION TREND ESTIMATES, LIS AND OECD DATA

| | Period [a] | $\bar{g}$ [b] | $\frac{g_{mean/median}}{\bar{g}}$ | $\frac{g_{CV}}{\bar{g}}$ | $\frac{g_{Gini}}{\bar{g}}$ | $\eta$ |
|---|---|---|---|---|---|---|
| <u>LIS data</u> | | | | | | |
| Australia | 1981-2010(8) | 1.38** | 0.16 | 0.52 | 0.40** | 1.77** |
| Canada | 1981-2010(10) | 0.90 | 0.25 | 1.03 | 0.59 | 1.59 |
| Denmark | 1987-2010(7) | 1.15 | 0.10** | 0.81** | - | 1.86 |
| Finland | 1987-2010(7) | 1.75 | 0.17 | 1.14 | 0.75 | 1.74 |
| France | 1978-2010(7) | 0.79** | -0.10* | -0.46** | -0.63* | 2.12** |
| Germany | 1984-2010 (7) | 0.56** | 0.23** | 0.88** | 0.79** | 1.64** |
| Ireland | 1987-2010(8) | 3.45 | -0.09** | -0.40 | -0.17** | 2.11 |
| Israel | 1986-2010(7) | 1.67 | 0.14 | 0.29 | 0.62 | 1.80 |
| Italy | 1986-2010 (11) | 0.63** | - | - | 0.60* | - |
| Luxembourg | 1985-2010(8) | 2.63 | 0.06** | - | 0.26 | 1.92** |
| Norway | 1979-2010 (8) | 2.28 | - | - | 0.21** | - |
| Spain | 1980-2010 (8) | 2.15 | - | -0.31** | - | - |
| Sweden | 1981-2005(6) | 2.04 | 0.11** | 0.97 | 0.44** | 1.84 |
| United Kingdom | 1979-2010(9) | 2.41 | 0.14 | 0.37 | 0.29** | 1.80 |
| United States | 1979-2013(10) | 0.77 | 0.37 | 1.12 | 0.70 | 1.33 |
| <u>OECD data</u> | | | | | | |
| Canada | 1976-2017 (42) | 0.85 | 0.14 | 0.54 | 0.31 | 1.80 |
| Denmark | 1985-2016 (16) | 0.99 | 0.19 | 1.36 | 0.57 | 1.71 |
| Finland | 1986-2017 (32) | 1.63 | 0.13 | 0.82 | 0.55 | 1.81 |
| Germany | 1985-2016 (14) | 0.60 | 0.13 | 0.43 | 0.97 | 1.82 |
| Hungary | 1991-2016 (15) | 1.02** | -0.14** | -0.60** | -0.30** | 2.15** |
| Israel | 1990-2017 (14) | 1.88 | - | - | 0.17 | - |
| Italy | 1984-2016 (17) | 0.33* | - | - | 1.18* | - |
| Luxembourg | 1986-2016 (15) | 1.46 | 0.07 | 0.23 | 0.46 | 1.90 |
| Netherlands | 1977-2016 (17) | 0.92 | -0.09 | -0.44 | 0.17* | 2.11 |
| New Zealand | 1985-2014 (10) | 1.45 | 0.15** | - | 0.32* | 1.78 |
| Norway | 1986-2017 (14) | 2.19 | - | - | 0.17** | - |
| Sweden | 1975-2017 (15) | 1.86 | 0.06 | 0.29 | 0.49 | 1.92 |
| United Kingdom | 1975-2017 (23) | 1.23 | 0.14 | 0.30 | 0.42 | 1.79 |
| United States | 1995-2017 (13) | 0.54 | - | 1.26 | 0.76 | - |

*Notes:* $g_{mean/median}$ is the growth rate of divergence of mean and median income as an annual percentage change that is approximated by estimating the change in $\ln(mean/median)$. $g_{CV}$ is the growth rate in the coefficient of variation approximated by estimating $ln(CV)$. $g_{Gini}$ is the growth rate of the Gini coefficient. We approximate $\eta$ by the growth vector approach in the preceding paper. A limitation of the CRRA utility function is that only $\eta > 0$ are considered valid and so lower values imply policies diverging from CRRA utility-based preferences. All estimates reported are statistically significant at the 1-per cent level or better using Huber–White standard errors unless otherwise noted (** 5 per cent level; * 10 per cent level). Statistical significance of all inequality estimates is the least of the two estimated values used in their composition.

[a] Number of observations, in parentheses. [b] Average annual growth rate, in per cent.

*Sources:* Author calculations based on data from Thewissen, Nolan, and Roser (2016) which is derived from the Luxembourg Income Study (LIS) Database; and OECD (2019).

TABLE 17—GROWTH AND DISTRIBUTION TREND ESTIMATES, WORLD BANK DATA

| | Period [a] | $\bar{g}$ [b] | $\frac{g_{mean}/_{median}}{\bar{g}}$ | $\frac{g_{CV}}{\bar{g}}$ | $\frac{g_{Gini}}{\bar{g}}$ | $\eta$ |
|---|---|---|---|---|---|---|
| <u>Income basis</u> | | | | | | |
| Argentina--Urban | 1987-2017 (27) | 0.88* | -0.60 | -1.07 | -0.59* | 2.46 |
| Australia | 1981-2014 (10) | 1.65 | 0.10 | 0.14 | 0.20 | 1.87 |
| Bolivia | 1990-2017 (19) | 2.83 | - | -0.58* | - | - |
| Brazil | 1981-2017 (33) | 2.72 | -0.24 | -0.36 | -0.14 | 2.24 |
| Canada | 1981-2013 (11) | 1.09 | - | - | 0.25** | - |
| Chile | 1987-2017 (14) | 2.20 | -0.43 | -0.58 | -0.35 | 2.37 |
| Colombia | 1992-2017 (19) | 2.69 | -0.16* | -0.29** | -0.15* | 2.17 |
| Costa Rica | 1981-2017 (31) | 4.20 | 0.15 | - | 0.09** | 1.77 |
| Croatia | 1988-2015 (8) | -0.32** | -0.73 | -3.90 | -4.16** | -0.51 |
| Czech Republic | 1993-2015 (14) | 2.19 | -0.08 | -0.40 | - | 2.09 |
| Ecuador | 1987-2017 (20) | 3.09 | -0.22* | -0.40** | -0.24** | 2.23 |
| El Salvador | 1991-2017 (23) | 0.96 | -0.90 | -1.47 | -1.42 | 2.56 |
| Estonia | 1993-2015 (14) | 4.45 | -0.07* | -0.27 | -0.16** | 2.08 |
| Germany | 1991-2015 (18) | 0.72 | 0.19 | 0.54** | 0.54 | 1.71 |
| Guatemala | 1986-2014 (5) | - | - | - | - | 2.24 |
| Honduras | 1989-2017 (28) | 2.27 | -0.18 | -0.34 | -0.11** | 2.19 |
| Israel | 1986-2016 (10) | 1.94 | - | - | 0.23** | - |
| Latvia | 1993-2015 (15) | 4.53 | 0.06** | - | 0.19** | 1.92 |
| Lithuania | 1993-2015 (13) | 6.54 | - | -0.13** | - | - |
| Malaysia | 1984-2015 (12) | 2.65 | -0.14 | -0.32 | -0.17 | 2.15 |
| Mexico | 1989-2016 (15) | 1.05* | -0.49 | -0.68 | -0.53* | 2.40 |
| Nicaragua | 1993-2014 (6) | 3.49 | -0.34 | -0.55 | -0.36 | 2.32 |
| Panama | 1989-2017 (24) | 3.31 | -0.19 | -0.33 | -0.21 | 2.20 |
| Poland | 1985-2015 (15) | 1.76 | 0.12 | 0.35 | 0.54 | 1.83 |
| Slovenia | 1993-2015 (13) | 2.09 | -0.14** | -0.66** | - | 2.15 |
| United States | 1986-2016 (10) | 0.86 | 0.24 | 0.43 | 0.31 | 1.62 |
| Uruguay | 1981-2017 (14) | 0.95 | - | - | -0.24* | - |
| <u>Consumption basis</u> | | | | | | |
| Bangladesh | 1983-2016 (9) | 1.17 | 0.30** | 0.75** | 0.61 | 1.48** |
| China--Rural | 1990-2015 (13) | 6.09 | - | - | 0.09 | - |
| China--Urban | 1990-2015 (13) | 6.74 | 0.07 | 0.08** | 0.19 | 1.90 |
| Côte d'Ivoire | 1985-2015 (10) | -2.36 | | -0.19** | - | - |
| Georgia | 1996-2017 (22) | 1.67** | - | - | -0.13* | - |
| Ghana | 1987-2016 (7) | 3.46 | 0.10** | - | 0.20 | 1.87 |
| India--Rural | 1983-2011 (6) | 1.33 | 0.09** | - | - | 1.88 |
| India--Urban | 1983-2011 (6) | 1.69 | 0.18 | 0.21** | 0.33 | 1.73 |
| Indonesia--Rural | 1984-2017 (25) | 4.00 | 0.07 | - | 0.20 | 1.90 |
| Indonesia--Urban | 1984-2017 (25) | 3.73 | 0.13 | 0.11 | 0.23 | 1.81 |
| Iran, Islamic Republic of | 1986-2016 (11) | 2.35 | -0.13 | -0.34 | -0.23 | 2.15 |
| Kazakhstan | 1996-2017 (18) | 4.39 | -0.11 | -0.43 | -0.40 | 2.12 |
| Mauritania | 1987-2014 (7) | 2.24 | - | -0.74** | -0.54** | - |
| Mexico | 1984-2016 (15) | - | - | - | - | 2.53 |
| Morocco | 1984-2013 (6) | 1.61** | 0.11* | - | - | 1.85* |
| Pakistan | 1987-2015 (12) | 2.90 | - | -0.12* | - | - |
| South Africa | 1993-2014 (7) | 3.34 | 0.25** | - | 0.11** | 1.59** |
| Sri Lanka | 1985-2016 (8) | 2.45 | 0.17 | 0.20* | 0.29 | 1.75 |
| Thailand | 1981-2017 (23) | 3.28 | -0.17 | -0.40 | -0.22 | 2.19 |
| Tunisia | 1985-2015 (7) | 2.23 | -0.20 | -0.55 | -0.38 | 2.21 |
| Turkey | 1987-2016 (17) | 2.95 | - | -0.19** | - | - |
| Ukraine | 1992-2016 (19) | 3.03 | -0.11** | -0.50 | -0.47 | 2.13 |
| Vietnam | 1992-2016 (10) | 6.30 | -0.05** | -0.26 | - | 2.07 |

*Notes:* $g_{mean/median}$ is the growth rate of divergence of mean and median income as an annual percentage change, approximated by estimating the change in $\ln(mean/median)$. $g_{CV}$ is the growth rate in the coefficient of variation approximated by estimating $ln(CV)$. $g_{Gini}$ is the growth rate of the Gini coefficient. We approximate $\eta$ by the growth vector approach in the preceding paper. A limitation of the CRRA utility function is that only $\eta > 0$ are considered valid and so lower values imply policies diverging from CRRA utility-based preferences. All estimates reported are statistically significant at the 1-per cent level or better using Huber–White standard errors unless otherwise noted (** 5 per cent level; * 10 per cent level). Statistical significance of all inequality estimates is the least of the two estimated values used in their composition.

[a] Number of observations, in parentheses. [b] Average annual growth rate, in per cent.

*Sources:* Author calculations based on data from World Bank (2019).

TABLE 18—GROWTH AND DISTRIBUTION TREND ESTIMATES, WID DATA

| | Period [a] | $\bar{g}$ [b] | $\frac{g_{mean/median}}{\bar{g}}$ | $\frac{g_{CV}}{\bar{g}}$ | $\frac{g_{Gini}}{\bar{g}}$ | $\eta$ |
|---|---|---|---|---|---|---|
| Brazil [c] | 2001-2015 (15) | 1.25 Γ | -0.14 Γ | -0.20** | -0.23 Γ | 2.16 Γ |
| China [d] | 1978-2015 (38) | 5.63 | 0.23 | 0.05 | 0.24 | 1.63 |
| Transition | 1978-1999 (22) | 4.65 | 0.24 | 0.08 | 0.34 | 1.62 |
| Present | 2000-2015 (16) | 9.29 | 0.15 | -0.02 | 0.06 | 1.78 |
| Côte d'Ivoire [e] | 1988-2014 (6) | -1.48** | -0.41 | -0.31 | - | -0.36 |
| Egypt [f] | 1999-2015 (6) | 1.45 | -0.15 | -0.25 | -0.33 | 2.17 |
| France [g] | 1900-2014 (83/102) | 2.21 | -0.27 | -0.24 | -0.14 | 2.27 |
| LIS comparison | 1978-2010 (26/33) | 2.43 | 0.75 | 0.33 | 0.05 | 0.41 |
| Pre-WWII | 1900-1939 (20/27) | 0.68 Γ | -0.67 Γ | - | -0.40 Γ | 2.83 |
| Post-war | 1946-1969 (22/24) | 4.52 | - | -0.10 | 0.08 | - |
| Transition | 1970-1999 (25/30) | 1.40 Γ | -0.06 Γ | - | - | 2.87 |
| Present | 2000-2014 (11/15) | 2.47 | 1.11 | 0.37 | -0.06 | -0.12 |
| India [h] | 1951-2013 (54/63) | 1.84 | - | -0.08 | 0.15 | 1.89* Γ |
| Postcolonial | 1951-1964 (12/14) | 2.06 | -0.38 | -0.48 | -0.31 | 2.34 |
| Transition | 1965-1999 (32/35) | 1.86 | 0.06 | -0.03 | 0.12 | 1.92 |
| Present | 2000-2013 (10/14) | 3.74 | 0.20 | 0.10 | 0.53 | 1.68 |
| Palestine [f] | 1996-2011 (10) | -1.33* | -0.29 | -0.28 | -0.28* | -0.28** |
| Russia [i] | 1961-2015 (25/36) | - | - | - | - | - |
| USSR | 1961-1989 (3/10) | 2.38 Γ | -0.05 Γ | - | -0.27 Γ | 2.06 Γ |
| Transition | 1990-1999 (6/10) | -5.38 Γ | - | - | -1.44 Γ | - |
| Present | 2000-2015 (16) | 2.89 | -0.79 | -0.36 | -0.22 | 2.53 |
| Turkey [f] | 1994-2016 (16) | 2.06 Γ | - | -0.32* | -0.38 Γ | 2.28** Γ |
| United States [j] | 1962-2014 (51) | 1.47 | 0.29 | 0.08 | 0.37 | 1.50 |
| LIS comparison | 1979-2013 (35) | 1.44 | 0.21** | 0.03 | 0.42 | 1.68** |
| Transition | 1962-1980 (17) | 1.62 | - | -0.06 | - | - |
| Reaganomics | 1981-1992 (12) | 2.53 | 0.66 | 0.27 | 0.33 | 0.60 |
| Present | 1993-2014 (22) | 1.28 Γ | 0.07 Γ | -0.29** | 0.35 Γ | 1.90 Γ |

*Notes:* Annual estimates used as the basis of these measures are derived from percentile distributional data. $g_{mean/median}$ is the growth rate of divergence of mean and median income as an annual percentage change, approximated by estimating the change in $\ln(mean/median)$. $g_{CV}$ is the growth rate in the coefficient of variation approximated by estimating $ln(CV)$. $g_{Gini}$ is the growth rate of the Gini coefficient. We approximate $\eta$ by the growth vector approach in the preceding paper. A limitation is the CRRA utility function is that only $\eta > 0$ are considered valid and so lower values imply policies diverging from CRRA utility-based preferences. All estimates reported are statistically significant at the 1-per cent level or better using Huber–White standard errors unless otherwise noted (** 5 per cent level; * 10 per cent level). Statistical significance of all inequality estimates is the least of the two estimated values used in their composition.

[a] Number of observations, in parentheses, where a lognormal/gamma distribution fits the data sufficiently well based on the two-sample Kolmogorov-Smirnov test (Massey, 1951). [b] Average annual growth rate, in per cent. [Γ] Denotes estimates based on the gamma distribution when lognormal-based estimates do not fit the data sufficiently well – these estimates are explored in appendix *A5.3 The Gamma Distribution Alternative*

*Sources:* Author calculations based on World Inequality Database (WID) data discussed in: [c] Morgan (2017); [d] Piketty, Yang, and Zucman (2017); [e] Czajka (2017); [f] Alvaredo, Assouad, and Piketty (2018); [g] Garbinti, Goupille-Lebret, and Piketty (2018); [h] Chancel and Piketty (2017); [i] Novokmet, Piketty, and Zucman (2018); and [j] Piketty, Saez, and Zucman (2016).

Whatever the form of $D(t)$, a couple approaches can be taken to setting the benchmark $D(\eta)$. One may be called a proactive or normative approach. That is, set a measure of performance based on careful societal introspection (whom do we want to be as a society?) and define the measure of performance as a comparison to it. In a sense, we decide what is "fair" and then evaluate growth in those terms. It is suggested we may refer to this as the European approach.

One may also take a positivist approach. That is, look at where countries are and impute the social preference associated with their status. From a policy perspective, what a practitioner does is make apparent the extent of the trade-off between $\mu$ and $\sigma^2$, or mean and median growth, evident in a country's existing trajectory. We may then ask whether something different – a policy intervention, ought to be done when current and long-run distributional paths differ. It is suggested that we may refer to this as the U.S. approach, though these are generalisations.

In the U.S. approach, historical measures of $D(t)$ replace $D(\eta)$ in future periods without requiring the exogenous discovery of underlying social preferences. In the European approach, one may take these values as a starting point, but need not take the current state of the economy indicated, revealed preference estimates of $\eta$, or long-run or short-run implied-$\eta$ estimates, or other historical indicators of distribution as the benchmark of performance. $D(\eta)$ may in fact differ substantially from that implied by existing policies. In any event, even among experts one method cannot be agreed on (Drupp, Freeman, Groom, and Nesje, 2018), and both have some merit.

### 5.7 Concluding Remarks

If a population has no preference over distribution, GDP growth (not even requiring it in per-capita form) is a sufficient statistic as implicit in it is that more consumption potential without regard to distribution is preferred. A standard argument stemming from the Kaldor-Hicks compensation criteria is to make the pie as large as possible and then figure out how to distribute it later. If the public is confident in the state's omniscience and the elite's generosity, then perhaps GDP growth remains sufficient, whatever distributional preferences are. However, Sen (2000) argues that Kaldor-Hicks advocacy for potential Pareto improvements to be sufficient is not a defensible criterion if there is no mechanism and no intent for gainers to compensate losers. Suppose, given the existence of extreme wealth, poverty, and discontent among the poor and marginalised, that we side with Sen and cannot trust in unmonitored distribution.

A statistic that includes an agnostic representation of preferences is useful in monitoring economic progress. Not taking a stance on what the social preference value ought to be is an advantage. A preference for more inequality may, for instance, be representative of how rapidly a country prefers to modernise through capital accumulation, i.e. China's capital hungry modernisation initiatives. It may also be the product of longstanding religious or cultural beliefs that one would be on questionable ethical grounds to reject outright, i.e. Saudi Arabia's theocratic-supported monarchy. In comparison, measures like the static Gini coefficient imply that a decrease in inequality is preferable. The proposition of new statistic $\nu_{(\cdot)}$ makes no such claim on optimal distribution and instead represents the deviation from any distribution-growth path.

$\nu_{(\cdot)}$ also offers the advantage of representing the dynamic nature of growth better as it uses the rate of divergence in measures of centre. Static alternatives such as the mean-median spread, mean/median ratio, coefficient of variation, Gini, and measures of poverty are only appropriate if the mean and median grow in proportion – which we have shown is unlikely. Static measures cannot represent the changing nature of distribution with economic growth. At best we end up comparing static values in two periods without guidance on whether the change is beneficial.

$\nu_{(\cdot)}$ also challenges that the method of conducting international comparisons changes. Formulating $\nu_{(\cdot)}$ as relative to national preferences has the advantage of informing on how well national policies fit. It is a measure of deviation rather than placing international values side-by-side without context. This suggestion is not to say that comparing components $D(t)$ or $D(\eta)$ between countries does not make for good discussion. But keeping the statistic as the composite $\nu_{(\cdot)}$ is more informative when comparing, say, an OECD member against an LDC with different institutional arrangements, production potential, and social traditions.

In the model, we have treated $\eta$ as a constant parameter, but like target $\dot{G}c$, the unemployment rate, or inflation, it is almost certainly subject to change over time. For instance, societies may be willing to sacrifice equality in favour of rapid modernisation over some period, and then transition to preferring a more equal dispersion once some quality-of-life potential is attainable through redistributions or economic reorganisation.[114] In operation, consider if economic growth is distributed more unequally than preferred because society's preferences have shifted to a more equitable distribution ($\eta$ has increased). Then $\nu_{(\cdot)} > 0$ and an interpretation is that future growth is bought at a higher price – greater sacrifice – than desired in the present given the remaining needs of society today. If instead society changes to favour individual success – a chance at attaining personal wealth and status become paramount ($\eta$ decreases), then $\nu_{(\cdot)} < 0$ implies a more equal distribution is occurring than fitting with a society that now prefers to gamble. Estimating $\eta$ and $D(\eta)$ well is essential work and exploring how the $\eta$ summary of preferences changes over time is an interesting avenue for further research. These are some of the many questions this research leads to, rather than answers.

To conclude, we cannot discard measures of the magnitude of growth such as $\dot{G}c$ because they say something vitally important. They tell us how the per-capita consumption potential

[114] Or conversely so. Society may exhibit a preference for improving the conditions of the poorest first, but then prefer to gamble and allow unequal growth after most of the population's basic needs are met.

of society is changing – whether the capacity for better lives is increasing. But by summarising the welfare distribution into a preference-free measure of centre, they do not inform on how consumption is operationalised. Moving from a representative agent to a representative distribution basis when modelling economic growth reveals what we give up in using the representative agent assumption. Clearly, at least one more statistic is needed. This statistic(s) should portray the dynamic nature of distribution and growth and embody social preferences over trade-offs inherent in national distributional policies. As a practical matter, a companion to $\dot{G}c$ should be simple to formulate and interpret. The proposals on $\nu_{(\cdot)}$ generally meet these conditions and are based on our distributional-growth model. The model itself and the statistics based on it perhaps make sense of some inconsistencies in economic growth and social outcomes. For instance, improvements in measures of economic growth but not measures of national satisfaction, or vice versa, may result from how poorly or well economic distributions match preferences.

# Conclusion – Contributions of Non-Cooperative Environmental Policy Examples and Vector-Based Inequality Analyses

As a thesis must make a distinct contribution to our body of knowledge, I outline here what I think are some qualifying contributions from the chapters of this thesis.

The Introduction lays out a simple argument for non-cooperative, even unilateral strategy for addressing pressing environmental concerns. Using basic game-theoretic language, I identify the bounds when a NEP approach is more likely to be effective than seeking cooperation from all parties. The NEP perspective has substantial implications for environmental protection and redress when limited resources are available. Applying the NEP criteria identifies policies that we might pursue to try and achieve a minimal redress of an environmental problem when first-best and perhaps even second-best and satisficing options do not have sufficient public support to be successful (though a NEP might also be first-best, second-best, or satisficing). It is an alternative decision-making approach like Lipsey and Lancaster (1956) and Simon (1956) applied to environmental policies.

I then study environmental decisions on a local scale that sum to a substantial concern. Among the contributions is the specification of a damages function that is more appropriate for thinking about local-scale environmental problems because it allows for a site's total environmental destruction. The change in the specification is deceptively simple as the impact on analyses and optimal policy are substantial. For one, the damages from any sort of local damage-generating industrial activity in a region can be minimised by clustering at a reduced number of sites (sometimes one) once some scale of total production is reached. It is a redress to the persistent problem of the tragedy of the commons (Stavins, 2011) by way of taking some sites out of production altogether. I observe that the limiting factor is transportation costs, which generally decline over time, and thus the potential to reduce the impact of production on the environment is increasing. For some activities, such as the storage of nuclear wastes, the implication is that policymakers should seek to accumulate waste to as few global sites as possible, and so this research will remain permanently relevant. I have also applied the framework to several settings within and outside environmental policy and find behavioural adaptation, when it can occur, must be carefully considered in local environmental policy design.

Next, I explore policies to address the plastic recycling crisis which now impacts every species and environment on the planet. At the core of the issue is that the producers of the goods that become waste have little responsibility for that waste resulting in pushes for more expansive EPR programs in recent years. I propose a perhaps odd framework that falls under the EPR and DfE umbrellas. It is to optimally impose a micro-tax on producers based on the number of different materials in a product and jointly subsidise recyclers based on the number of materials within that product that they instead process. The result is that any product so regulated becomes profitable to fully recycle through producers reducing its material complexity and ensuring that product's remaining complexity is tied to the profit motive of recyclers. An iterative product improvement process then emerges to make newer versions of products more recyclable. Unfortunately, such a proposal is novel in the environmental, sustainability, and recycling and waste spheres so there is a lack of evidence on their effectiveness. I instead explore the theory and efficiency of the proposal and advocate for its experimental usage.

Then, I explore a large application of NEP – policies to influence the clean consumption through abatement technologies of otherwise dirty inputs to production. To do so, I explore coal consumption in the U.S. energy sector and how power plants that install FGD systems are able to use high sulphur coal while meeting sulphur dioxide emissions restrictions, while their non-FGD counterparts must use cleaner coal which is more expensive per heat unit. I explore this disparity from multiple angles and with diverse empirical techniques. In the process of modelling the power plant abatement and fuel choice decision I also contribute a nonparametric method to identify market structure and use it to test whether the U.S. energy sector exhibits perfect substitutability between high sulphur coal and natural gas or low sulphur coal inputs to production. I also provide estimates of market own- and cross-price elasticities of demand for coal and natural gas which are separated by whether power plants have FGD systems installed to show that the resulting markets behave differently. Finally, after identifying four different types of abatement costs involved with FGD installation and operation, I estimate the two types with limited coverage in the literature. Ultimately, the goal is a general framework and empirical approaches to make policy decisions about energy and similar matters. The results are meant to inform the current and future energy transitions.

While the three applications I present of NEP are not exhaustive, I think they support the case that there are NEP type alternatives to traditional environmental policy approaches. The NEP approach suggests that even when a majority, through the median voter or another decision mechanism, opposes decisive environmental action, regulators might nonetheless design and implement policies that improve environmental quality.

With some apprehension, as the subject is generally outside the bounds of environmental economics, I then report the results from my study of social preferences over inequality and the distribution of economic growth. I develop a theoretical framework for estimating the elasticity of marginal utility of consumption, $\eta$, implied by the distribution of the returns to economic growth in society. The method relies on publicly available mean and median national income or consumption statistics, and so allows estimates that are not possible by other methods. Particularly useful is that this new implied-$\eta$ estimate bridges the practice and theory of inequality by measuring inequality in the same units as used in models of inequality. I provide implied-$\eta$ estimates for several countries and periods based on four datasets. I then include a follow-on measure – the volatility of implied-$\eta$, as a measure of how economic shocks, booms, and busts impact the comparatively wealthy and poor in societies. The framework is then extended to study inequality between sex, racial, and social groups in the same $\eta$ terms.

I then conclude with a second chapter on inequality and economic growth, co-authored with Ben Groom of the London School of Economics and Political Science and Eli Fenichel of Yale University, where we present a more comprehensive model of inequality, economic growth, and distribution in society. Following international comparisons of inequality increasing and decreasing growth trends, an optimal control model is presented where the impact of $\eta$ as a parameter representing social preferences over inequality and economic growth is explored. This chapter concludes with an argument for adopting a companion statistic to GDP per capita and similar growth statistics which reports on the deviation of economic distributions from societal preferences over inequality.

In this thesis I have studied two linked social problems. I first researched cases where presumably scarcity and consumption concerns result in opposition to environmental protection. But then, we should ask whether the distribution of scarcity, alternatively existing consumption patterns, is optimal. So, I then undertook the measurement of social preferences implied by inequality to find a method to measure when preferences and realisations diverge. Environmental protection and economic inequality are two issues locked in a brutal cycle. Suboptimal inequality results in greater scarcity and then opposition to policies that would limit immediate consumption. But then overconsumption of the environment reduces public goods as well as future consumption, resulting in more conflict over resources and inequality. We require approaches to environmental and social policy development that make progress on both problems without limiting consumption. I think I have made some progress toward so addressing both issues in this thesis.

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# Appendices

Some areas of this dissertation warrant further discussion, data sharing, or the presentation of additional results. Such results may be particularly interesting to some readers but are generally not strictly within the environmental scope of the main body of this thesis. Each appendix referenced in the main body is presented in separate sections in the order initially referenced.

## A1.1 Damage Functional Forms

This appendix supports the four marginal damage functional forms in Figure 2 by illustrating their corresponding total damage forms (Figure 35).

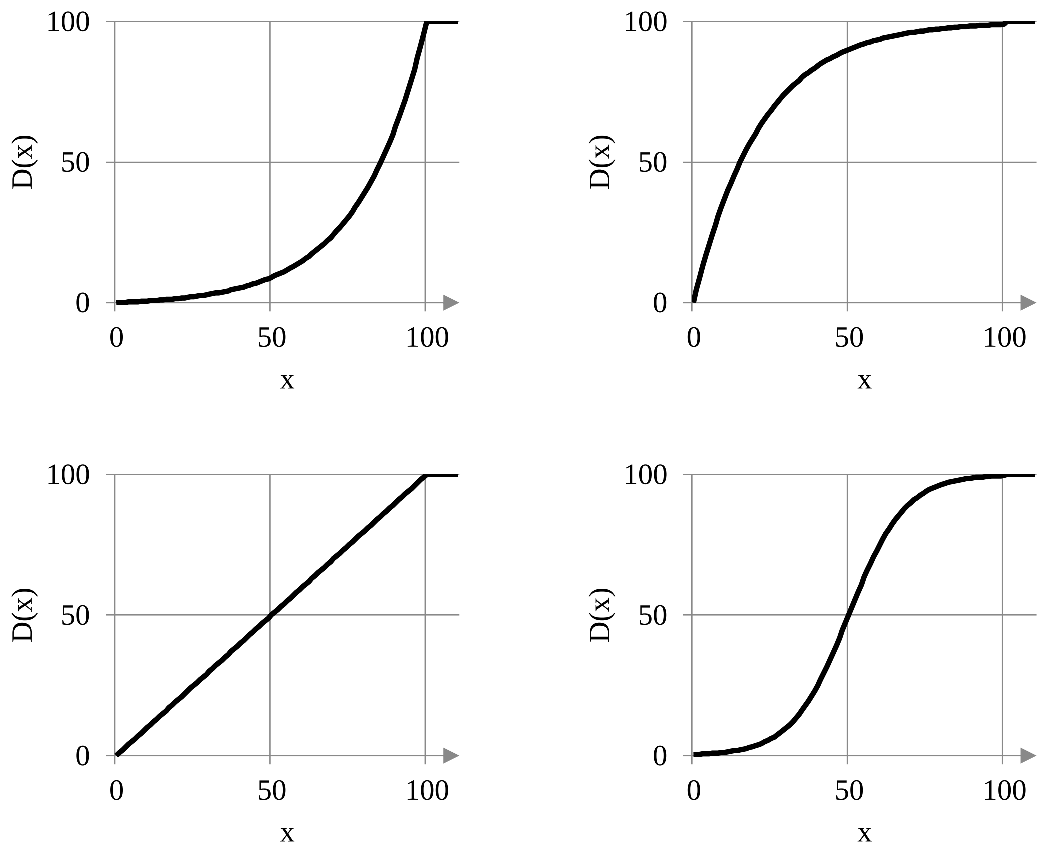


FIGURE 35 GENERAL REPRESENTATIONS OF DAMAGE FUNCTIONAL FORMS

Notes: Four general damage functional forms resulting in the marginal damage forms in Figure 2.

(top-left) Case for dispersion: $d'(x) > 0, d''(x) > 0$.

(top-right) Case for clustering: $d'(x) > 0, d''(x) < 0$.

(bottom-left) Local indifference case: $d'(x) > 0, d''(x) = 0$

(bottom-right) Biological (logistic) case: $d'(x) > 0, d''(x) \geq 0$ $on\ x \leq 50$ and $d'(x) > 0, d''(x) \leq 0$ $on\ x > 50$

## A1.2 Siting Framework Applied Beyond the Environment

The results of this paper are not intended to imply that the 100-per cent damages functional addition, nor aggregating and dispersing policies, are uniquely applicable to matters of environmental policy. Rather, it is a simple adjustment which may be widely applicable. In this section, three additional applications are provided. Two are presented empirically while the third pertains to a subject of greater secrecy where empirics would be difficult but the story is clear.

### *A1.2.1 Prostitution, Illicit Activities, and Safe Sites*

Matters of prostitution, illicit drug use and sales, homelessness, and levels of general law and regulatory enforcement come to mind when searching for social policy-related applications of an aggregating doctrine. Whether in the Holbeck neighbourhood of Leeds in the U.K., or Amsterdam's Red-Light District, city planners sometimes seek to cluster solicitation and prostitution provision into a designated area. If one expects social damages – either as a direct consequence or indirect result of associated behaviours to be substantial or fit the case for clustering, this organisation might be warranted. Solicitation, biowaste, illicit drug use, noise and late hours, and unruly behaviour among patrons might best be avoided entirely in residential neighbourhoods. However, policymakers must remain cognizant of the potential for leakage, general equilibrium versus partial equilibrium effects, and long-run adjustments to supply and demand.

I explore the implementation of the Holbeck Managed Zone (HMZ) in Leeds using a combination RD (Thistlethwaite and Campbell, 1960) and RK design. While prostitution is legal in Great Britain, soliciting in public and several related behaviours remain crimes. The HMZ, officially piloted in October 2014 and later made permanent, is an area where the authorities tolerate solicitation and related activities, and additional social services are sometimes offered to prostitutes. The HMZ itself is small, falling within a roughly two square kilometre area where it is allowed on twenty partial streets in inner Leeds. Reportedly 12-15 women work within the zone each night (BBC, 2018). It has, however, faced strong opposition from some residents of surrounding neighbourhoods in recent years (Bindel, 2019), and in addition to requiring a greater police presence drawn from existing resources, additional officers have become necessary (Leeds City Council, 2018). We should then expect an initial drop in all sorts of criminal activity around the HMZ with implementation, followed by increases in criminal activity over time as leakage occurs and possibly greater supply and demand move into the area.

I use Lower layer Super Output Area (LSOA)-level data on crime reports and population levels for years 2011-2018 from the UK Police (2020) and Office for National Statistics (2019, 2020) to explore the impact of the HMZ implementation on surrounding communities. The UK divides into roughly 35,000 LSOA's, each with a mean population of around 1,600; LSOA's are the smallest statistical scale in the U.K. where population figures are consistently available. Another advantage of LSOA's is that they tend to follow roads and other geographic features. In this case, the HMZ falls primarily within one LSOA, with a few permissible roads extending into a functionally similar LSOA. In total, the two LSOA's encompass about three square kilometres with a population of roughly 1200 each in 2011, and a density of 819 people per square kilometre. But the HMZ is concentrated into two-thirds or less of this area. In all functional aspects, the HMZ area is also less residential and more industrial than neighbouring LSOA's.

The primary interest is the impact on the HMZ's surrounding neighbourhoods, and I compose data series of inner and outer rings of LSOA's around the HMZ. The inner ring is composed of seven LSOA's that border the HMZ's two LSOA's. The seven LSOA's have a combined area of 13.53 square kilometres and a population density of 756 per square kilometre in 2011. An outer ring of 27 LSOA's then borders the inner ring and has a combined area of 37.86 square kilometres and population density of 1,205 per square kilometre in 2011.[115] It appears that the HMZ area is selected to avoid more densely populated neighbourhoods. However, it is a short stroll from such places and the lack of distance is perhaps not sufficient to prevent leakage.

Using crime report data, I compose monthly crime summaries in the form of the number of crimes reported in each zone – HMZ, inner ring, outer ring, and greater Leeds – without any sort of weighting by crime severity. I then divide these by annual LSOA population data from ONS which reports midyear population estimates. The resulting data are the monthly crime report incidences per capita and does not include unreported crimes. I have included all crime data for three reasons. First, the categories used by the U.K. Police changed over this period. Second, officers likely take some discretion in deciding charges

[115] HMZ LSOA's: E01033013 and E01033032. Inner ring LSOA's: E01011366, E01011362, E01011363, E01033015, E01033016, E01011364, and E01011368. Outer ring LSOA's: E01011284, E01011292, E01011293, E01011294, E01011318, E01011369, E01011370, E01011371, E01011372, E01011373, E01011375, E01011482, E01011483, E01011467, E01011678, E01011729, E01011731, E01011732, E01011733, E01011734, E01011736, E01033008, E01033010, E01033011, E01033018, E01033019, and E01033034.

and so a crime that might fall into one category in one period, may later fall into a lesser or more severe category after a change in enforcement policy. Third, I am after the net effect on the neighbourhoods under review.

Unfortunately, the analysis must be more complicated than comparing the HMZ to greater Leeds. A feature of the HMZ is that prostitutes and punters should be able to report other activities they observe to police and social workers. The nature of crime reporting in the HMZ then changes. Any decrease in criminal activity – or what is considered criminal – may be offset by a greater likelihood of reporting. Another issue is that the HMZ and surrounding areas are under greater scrutiny – more intensively policed than greater Leeds.

I summarise the monthly data into yearly observations for scrutiny in Table 19. This includes the reported crime incidence per-capita in the Holbeck Managed Zone (HMZ), two surrounding areas – an inner ring and outer ring encompassing the HMZ, and greater Leeds excluding the HMZ and rings.

TABLE 19 —SUMMARY OF CRIME INCIDENCE AROUND THE HOLBECK MANAGED ZONE

| | HMZ | Inner Ring | Outer Ring | Rest of Leeds | |
|---|---|---|---|---|---|
| 2011 | 0.3623 | 0.4195 | 0.4109 | 0.1197 | |
| 2012 | 0.2723 | 0.4319 | 0.3779 | 0.1000 | |
| 2013 | 0.2406 | 0.4131 | 0.3672 | 0.0921 | |
| 2014 | 0.2177 | 0.3784 | 0.3067 | 0.0838 | HMZ implemented |
| 2015 | 0.2203 | 0.3505 | 0.3052 | 0.0923 | |
| 2016 | 0.2310 | 0.3539 | 0.3523 | 0.1028 | |
| 2017 | 0.2233 | 0.3908 | 0.3730 | 0.1096 | |
| 2018 | 0.2439 | 0.4072 | 0.4032 | 0.1192 | |

*Notes:* Annual summary of the number of crime reports of all types reported in the Holbeck managed zone (HMZ), the immediate surrounding area (inner ring), the area surrounding the inner ring (outer ring), and the rest of Leeds, without any sort of weighting scheme based on severity, etc. The nature of reporting changes in the HMZ after the zone is piloted because prostitutes and those soliciting can now, for example, report other crimes observed. The emphasis in the analysis is on the impact in the inner and outer rings from increased police presence, a shift of activities out of these areas into the HMZ or other areas, and then growth in general supply and demand across the area from insufficient containment within the HMZ.

*Sources:* Author calculations based on data from U.K. Police (2020), Office for National Statistics (2019, 2020).

We should find two changes – first, a decrease driven by changes in the manner of policing, and second, an offsetting increase driven by a greater concentration of activity (supply and demand) into the HMZ and the surrounding area. I illustrate the HMZ case in Figure 36. I plot monthly reported crime incidence for the HMZ (red hollow triangles), the immediate surrounding inner ring (black circles), outer ring (hollow blue circles), and greater Leeds excluding these areas (green hollow circles). Because what is reported within the HMZ has changed, it is only present for non-causal comparison. However, what occurs in the inner and outer rings is informative. We first observe drops in incidence where we cannot separate the effect of changes in police presence from the effect of a change in activity location. These effects are particularly pronounced in the inner ring as we would expect. We then observe an increase in incidence over time in both rings as more criminal and unruly activity concentrates into the area. In a few years, we will also be able to observe the impact of additional police resources since assigned to the area. At present, however, it appears that attempts to aggregate one activity into the HMZ results in substantial spillovers impacting the entire area.

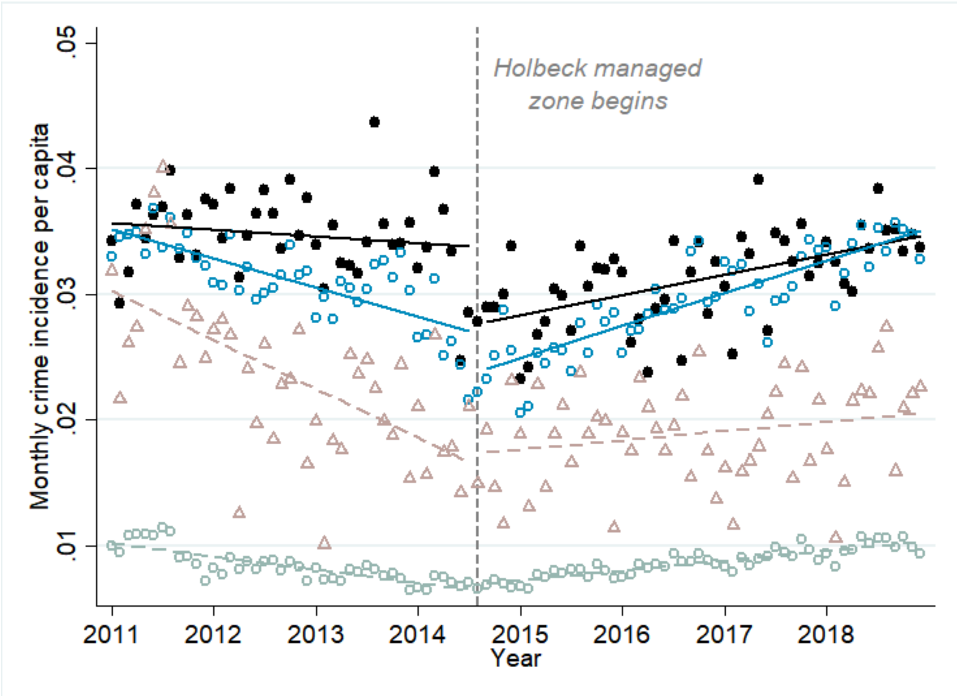


FIGURE 36 PER-CAPITA CRIME REPORTS IN AND AROUND THE HOLBECK MANAGED ZONE

*Notes*: Black solid markers: Crime reports per-capita (incidence) in an inner ring – the area immediately surrounding the Holbeck Managed Zone (HMZ). Blue hollow markers: Incidence in outer ring – the area immediately surrounding the inner ring. Red hollow triangles: Incidence within the HMZ. Green hollow markers: Incidence in greater Leeds excluding the Holbeck managed zone and surrounding rings. We see an initial impact from changes in policy, policing activity, and concentration into the HMZ, followed by increases in criminal activity across the three zones of interest from adaptation.

Sources: Author calculations based on data from U.K. Police (2020) and Office for National Statistics (2019, 2020).

An RD/RK design following the framework of section *1.5.5 Regression Discontinuity and Kink Methodologies and Estimates* results in the empirical estimates reported in Table 20. The estimate of the RK value in the outer ring is of particular interest as it supports the visual presentation that the crime incidence grew more rapidly in the area surrounding the HMZ – evidence of a concentrating of activity into the area. So, clearly caution is in order with clustering policies when adaptation is possible. Site selection is also clearly important – the HMZ and surrounding rings have higher reported crime incidence than greater Leeds to begin with. The moment of implementation, however, does not appear to involve any other particularly causal officially known event. But I do observe that drops in crime incidence in the rings precede official implementation of the zone by a month or two. At the point of discontinuity, what we are attempting to measure is not the effect of allowing solicitation in the HMZ, but the impact of a change in policy, movement of prostitution siting, and greater policing efforts in the surrounding area which might all precede official implementation. Nevertheless, police reports hardly capture an accurate picture. Outreach programmes such as the Basis Sex Work Project, the Leeds City Council, and West Yorkshire Police advocate for the HMZ and similar programmes because of the positive impact on those engaged in prostitution (BBC, 2018; Leeds City Council, 2020).

TABLE 20 — RK AND RD DESIGN ESTIMATES ON HMZ IMPLEMENTATION

| | Inner ring | Outer ring |
|---|---|---|
| Date (*actual* – *k*) | -0.000076 | -.00022*** |
| | (.000046) | (0.000028 |
| RD: D=1(*actual* ≥ *k*) | -0.0051*** | -.0019* |
| | (0.0015) | (0.0011 |
| RK: Year*D | 0.00022*** | .00043*** |
| | (0.000053) | (0.000035 |
| Constant | 0.033*** | .026*** |
| | (0.0013) | (0.00085) |
| Observations | 96 | 96 |
| R-squared | 0.34 | 0.67 |

*Notes:* Holbeck Managed Zone implementation with k=October 2014. Dependent variable is the monthly number of crimes reports per-capita in the area immediately surrounding the HMZ (inner ring), and the area surrounding the inner ring (outer ring), 2011-2018 (see annual summary data reported in Table 19. Greater police presence in the area and a drawing of activity into the zone when the HMZ pilot is implemented decreases crime initially, but then the rate of incidence increases over time as the zone plausibly "spills over" due to close proximity to inner and outer ring neighbourhoods.

Robust standard errors in parentheses. *** Significant at the 1 per cent level, ** Significant at the 5 per cent level, * Significant at the 10 per cent level.

*Sources:* Author calculations based on data from U.K. Police (2020) and Office for National Statistics (2019, 2020).

We can find similar clustering programmes in place when the target audience is particularly hard to reach. Illicit drug users, for instance, are being served by at least 78 drug consumption facilities – colloquially safe injection sites – across Western Europe as of 2018 (EMCDDA, 2018). These provide benefits within drug user communities by lowering disease transmission rates, overdose deaths, and providing access to counselling. However, they also provide community benefits as clustering sites. Similarly, provision of homeless shelters may face opposition from conservative factions in society based on the provision of a public benefit to what they might see as a private problem. Arguments on ethical grounds are unlikely to be persuasive, but arguments based on damage avoidance to personal property might be compelling.

This framework also suggests a basis for nonenforcement zones for other laws. Examples include the designated nonenforcement zone for drug dealers unilaterally implemented by park management to address a particularly intractable drug dealing problem in central Berlin's Görlitzer Park (Connolly, 2019). This is certainly in the spirt of non-cooperative environmental policies. Such applications receive substantial public criticism which should be weighed against any benefits sought. I also note that remuneration for the provision of a public service to impacted residents is missing in each of these examples. I now discuss a policy leading to clustering on a global scale.

#### *A1.2.2 Fighting ISIS and International Terrorism*

The Islamic State (IS) inherited from al-Qaeda a brutal legacy of armed conflict with the U.S. and allies across Western Asia.[116] Following the September 11th, 2001 terrorist attacks, the U.S. and western leaders committed to fighting against generally Salafi-based jihadist organisations in foreign countries rather than at "home". This is generally referred to as the Global War on Terrorism (GWOT). President George W. Bush and Bush administration officials made this case on several occasions. For instance, first publicly in November 2002: "The best way to keep America safe from terrorism is to go after terrorists where they plan and hide." (Bush, 2002), and later even more pertinently: "We are fighting these terrorists with our military in Afghanistan and Iraq and beyond so we do not have to face them in the streets of our own cities." (Bush, 2004). Even before the establishment of IS, foreign fighters were drawn to Afghanistan and Iraq to fight against coalition occupation forces.[117] These occupations drew in fighters who might have otherwise engaged in terrorist attacks in the West. The implication is that allowing a soft border through Turkey, Pakistan, and other routes into such areas reduces the threat to Western Europe and North America. It can also allow identification of nationals who have travelled to the region and attempt to return.

I use data on the number of terrorist attacks from 1990-2018 in Europe, Afghanistan and Iraq, and the rest of the world from the Global Terrorism Database (2020). Yearly summary data is reported in Table 21 for reference. I then use an RD/RK design – without any sort of weighting scheme for severity of attacks or impact – to explore the relief from terrorist

116 To be clear, the Islamic State is also known as the Islamic State of Iraq and the Levant (ISIL), the Islamic State of Iraq and Syria (ISIS), and by the Romanized Arabic acronym "Daesh".

117 The author was part of these forces in 2005 and has firsthand knowledge of efforts to stem the flow of insurgents into Iraq. However, also see Byman (2007).

activities in Europe which occurred as potential terrorists instead concentrated into Afghanistan and Iraq.

TABLE 21 —SUMMARY OF TERRORIST ATTACKS

| | Iraq & Afghanistan | Europe | Rest of the World | |
|---|---|---|---|---|
| 1990 | 2 | 448 | 3,437 | |
| 1991 | 33 | 780 | 3,870 | |
| 1992 | 71 | 860 | 4,140 | |
| 1993 | 6 | 234 | 508 | |
| 1994 | 27 | 679 | 2,750 | |
| 1995 | 23 | 405 | 2,653 | |
| 1996 | 16 | 651 | 2,391 | |
| 1997 | 22 | 597 | 2,580 | |
| 1998 | 8 | 246 | 680 | |
| 1999 | 21 | 374 | 1,000 | |
| 2000 | 24 | 487 | 1,313 | |
| 2001 | 17 | 486 | 1,410 | Invasion of Afghanistan |
| 2002 | 44 | 232 | 1,057 | |
| 2003 | 202 | 221 | 855 | Invasion of Iraq |
| 2004 | 411 | 105 | 650 | |
| 2005 | 772 | 180 | 1,065 | |
| 2006 | 1,120 | 168 | 1,470 | |
| 2007 | 1,388 | 134 | 1,720 | |
| 2008 | 1,520 | 372 | 2,913 | |
| 2009 | 1,640 | 347 | 2,735 | |
| 2010 | 1,721 | 394 | 2,711 | |
| 2011 | 1,729 | 293 | 3,054 | |
| 2012 | 2,906 | 367 | 5,256 | |
| 2013 | 4,295 | 427 | 7,319 | |
| 2014 | 5,758 | 1,177 | 9,973 | Worldwide caliphate declared |
| 2015 | 4,679 | 1,021 | 9,277 | |
| 2016 | 5,000 | 406 | 8,220 | |
| 2017 | 3,917 | 404 | 6,659 | |
| 2018 | 3,138 | 271 | 6,198 | |

*Notes:* Number of terrorist attacks of all types reported in Iraq and Afghanistan, Western and Eastern Europe, and the rest of the world without any sort of weighting scheme based on severity, etc. The nature of reporting changes in Iraq and Afghanistan after invasion and occupation, and we cannot be sure of the quality and consistency of reporting from the rest of the world. In Europe, reporting suggests subsequent decreases after the invasions of Afghanistan and Iraq, and an increase when Islamic State compels attacks globally.

*Sources:* Author calculations based on data from the Global Terrorism Database (2020).

Figure 37 then illustrates the results. I focus on terrorist attacks in Europe (black markers) because data from the rest of the world (hollow red markers) is almost certainly not comprehensive and may be inconsistently reported. I also illustrate data from Iraq and Afghanistan (hollow blue triangles), but this data is undoubtedly inconsistent as the nature of reporting and attacks changes with an occupation. The key takeaway is that the invasion and occupations of Afghanistan, and in particular more accessible Iraq, led to substantial concentrations of extremists into the theatre from abroad including from Europe. Unfortunately, we then see an increase in reported attacks in Europe and the rest of the world over time as the Islamic State has a substantial, digitally savvy recruiting activity. Attacks in all areas are also attenuated by the declaration of a “worldwide caliphate” in

2014 in response to increased pressure in Iraq. Concentrating an unwanted activity does not result in a "free lunch" elsewhere when adaptation is possible.

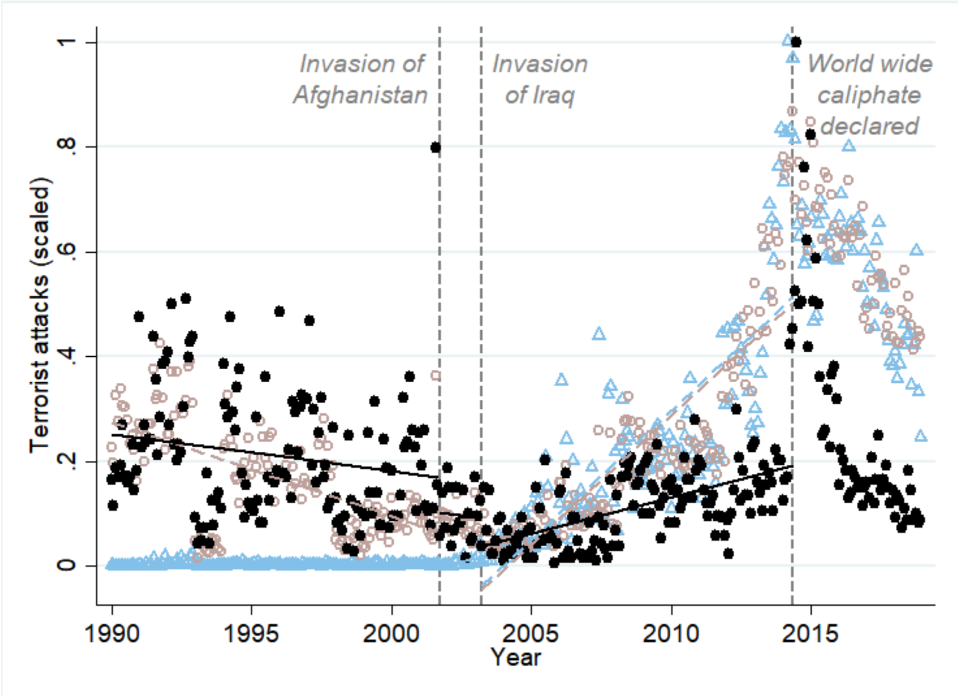


FIGURE 37 TERRORIST ATTACKS IN EUROPE AND THE REST OF THE WORLD

*Notes*: Number of terrorist attacks scaled to their highest values in the period. Solid black markers: Terrorist attacks in Western and Eastern Europe. Blue hollow triangles: Reported terrorist attacks in Iraq and Afghanistan – the Global War on Terror (GWOT) primary theatres of operation during the period of interest. Red hollow markers: Reported terrorist attacks in the rest of the world. We observe drops in terrorist attacks in Europe and the rest of the world with the invasions of Afghanistan and Iraq. These invasions were countered by international anti-occupation forces often using terrorist attacks and recruiting based on extremist ideologies.

Sources: Author calculations based on data from the Global Terrorism Database (2020).

In Table 22 I also provide estimates of the observed effect using the RD/RK design. In this example, the decision to invade Afghanistan and in particular Iraq by the U.S. and coalition forces could coincide with greater vigilance in the European Union. Yet we have also observed that the declaration of a worldwide caliphate in 2014 – a call for dispersed global attacks – could not be prevented by vigilance. We might also at least take the actual timing of the invasion of Iraq as not connected to any individual terrorist's activities. It is clear that both an RD effect occurs – reducing the incidence of terrorist attacks in Europe, followed by an RK effect of increasing activity perhaps driven by greater recruiting and radicalisation activities.

TABLE 22 — RK AND RD DESIGN ESTIMATES ON THE GWOT

| | Afghanistan (k=Oct 2001) | Iraq (k=Apr 2003) |
|---|---|---|
| Date (*actual* – *k*) | -0.46*** | -1.59** |
| | (0.11) | (0.68) |
| RD: D=1(*actual* ≥ *k*) | -24.29* | -155.80*** |
| | (13.22) | (15.63) |
| RK: Year*D | -1.12 | 4.26*** |
| | (0.84) | (0.71) |
| Constant | 60.36*** | 64.63*** |
| | (9.51) | (7.27) |
| Observations | 158 | 151 |
| R-squared | 0.18 | 0.72 |

*Notes:* Invasions of Afghanistan and Iraq on *k* dates as components of the Global War on terrorism (GWOT). Dependent variable is the monthly number of terrorist attacks reported across Western and Eastern Europe from 1990 through early 2020 (see annual summary data reported in Table 21). The estimation period for the Afghanistan effect is January 1990-March 2003, and for the Iraq effect is November 2001-June 2014. There is a substantial drop in attacks in Europe coinciding with the invasions of in particular Iraq by coalition forces, but then an increase in the rate of attacks thereafter.

Robust standard errors in parentheses. *** Significant at the 1 per cent level, ** Significant at the 5 per cent level, * Significant at the 10 per cent level.

*Sources:* Author calculations based on data from the Global Terrorism Database (2020).

I now qualitatively present a historical example where we would not expect adaptation by "polluters" over time.

### *A1.2.3 Development of Nuclear Arsenals*

In a rush to build the first nuclear weapons, the possibility of incidental exposure and damages during research, development, and manufacturing were at first a distant secondary concern. Weapons programmes used a vast network of sites in production – several dozen locations across the U.S., U.K., and Canada – in what would eventually be called the Manhattan Project (Atomic Heritage Foundation, 2019). Several sites were contaminated in the course of mining, processing, and testing materials as well as by accidents. At first, the justification for contamination resulted from a combination of ignorance and the necessity of ensuring victory in World War II. But the scale of the costs involved became impossible to ignore. Current estimates put the liability from the sum of the U.S. nuclear weapons programme at around $377 billion, a considerable share of which accumulated in the early years (U.S. GAO, 2019). The dangers of both radioactive exposure and nuclear proliferation soon changed weapons development. To minimise the threat to population centres, by 1943 much of the most dangerous activities – the production and preparation of plutonium – was transferred to an isolated site near Hanford, Washington which is still used for waste storage.[118] This sort of consolidation occurred in other nuclear programmes as well – at Windscale/Sellafield in the U.K., and until the fall of the U.S.S.R. at Mayak (Pearce, 2019; Diakov, 2011).[119]

### A1.3 Plastic Production Volume, 1939-2016

Reproduction of global plastic production estimates from Geyer, Jambeck, and Law (2017) for years 1950-2015, supplemented by 1939 and 1945 estimates from Freinkel (2011) converted to millions of metric tons (MMT).

TABLE 23 — ESTIMATES OF GLOBAL PLASTIC PRODUCTION BY YEAR

| | Plastic Production (millions of metric tons) | | Plastic Production (millions of metric tons) |
|---|---|---|---|
| 1939 | 0.097 | 1982 | 73 |
| 1945 | 0.371 | 1983 | 80 |
| 1950 | 2 | 1984 | 86 |
| 1951 | 2 | 1985 | 90 |
| 1952 | 2 | 1986 | 96 |
| 1953 | 3 | 1987 | 104 |
| 1954 | 3 | 1988 | 110 |
| 1955 | 4 | 1989 | 114 |
| 1956 | 5 | 1990 | 120 |
| 1957 | 5 | 1991 | 124 |
| 1958 | 6 | 1992 | 132 |
| 1959 | 7 | 1993 | 137 |
| 1960 | 8 | 1994 | 151 |
| 1961 | 9 | 1995 | 156 |
| 1962 | 11 | 1996 | 168 |
| 1963 | 13 | 1997 | 180 |
| 1964 | 15 | 1998 | 188 |
| 1965 | 17 | 1999 | 202 |
| 1966 | 20 | 2000 | 213 |
| 1967 | 23 | 2001 | 218 |
| 1968 | 27 | 2002 | 231 |
| 1969 | 32 | 2003 | 241 |
| 1970 | 35 | 2004 | 256 |
| 1971 | 38 | 2005 | 263 |
| 1972 | 44 | 2006 | 280 |
| 1973 | 51 | 2007 | 295 |
| 1974 | 52 | 2008 | 281 |
| 1975 | 46 | 2009 | 288 |
| 1976 | 54 | 2010 | 313 |
| 1977 | 59 | 2011 | 325 |
| 1978 | 64 | 2012 | 338 |
| 1979 | 71 | 2013 | 352 |
| 1980 | 70 | 2014 | 367 |
| 1981 | 72 | 2015 | 381 |

*Notes:* Estimates of global plastic production in millions of metric tons (MMT). A substantial increase in global plastic production coincides with waste management firms becoming common.

Sources: 1950-2015 data reproduced from Geyer, Jambeck, and Law (2017). 1939 and 1945 estimates converted to MMT from millions of pounds reported in Freinkel (2011).

118 An inexcusable footnote to the Hanford legacy is the forced displacement of Native Americans in the area (Atomic Heritage Foundation, 2017), and possible legacy effects on the Columbia River Gorge from poor containment of early waste storage.

119 Nuclear processing appears to have expanded to other sites in the Russian Federation in recent years. However, Russian or any other domestic policy – particularly secretive ones – does not necessarily match social preferences.

### A1.4 Description of SIC Code 4953: Refuse Systems and Number of New Firms

This appendix reports the definition of refuse systems (waste management firms) and the number of new firms reported in Orbis. Verbatim from the U.S. Department of Labor, Occupational Safety and Health Administration (https://www.osha.gov/pls/imis/sic_manual.display?id=955&tab=description):

> *4953 Refuse Systems*
>
> *Establishments primarily engaged in the collection and disposal of refuse by processing or destruction or in the operation of incinerators, waste treatment plants, landfills, or other sites for disposal of such materials. Establishments primarily engaged in collecting and transporting refuse without such disposal are classified in Transportation, Industry 4212.*
>
> - *Acid waste, collection and disposal of*
> - *Ashes, collection and disposal of*
> - *Dumps, operation of*
> - *Garbage: collecting, destroying, and processing*
> - *Hazardous waste material disposal sites*
> - *Incinerator operation*
> - *Landfill, sanitary: operation of*
> - *Radioactive waste materials, disposal of*
> - *Refuse systems*
> - *Rubbish collection and disposal*
> - *Sludge disposal sites*
> - *Street refuse systems*
> - *Waste materials disposal at sea*

TABLE 24 — ESTIMATES OF NEW WASTE MANAGEMENT FIRMS IN THE EU BY YEAR

| | New Firms | | New Firms | | New Firms | | New Firms |
|---|---|---|---|---|---|---|---|
| 1799 | 1 | 1916 | 3 | 1950 | 26 | 1984 | 346 |
| 1827 | 1 | 1917 | 2 | 1951 | 21 | 1985 | 396 |
| 1832 | 1 | 1918 | 1 | 1952 | 20 | 1986 | 388 |
| 1848 | 1 | 1919 | 5 | 1953 | 19 | 1987 | 439 |
| 1856 | 1 | 1920 | 4 | 1954 | 66 | 1988 | 577 |
| 1866 | 1 | 1921 | 5 | 1955 | 46 | 1989 | 647 |
| 1867 | 1 | 1922 | 6 | 1956 | 55 | 1990 | 956 |
| 1868 | 2 | 1923 | 4 | 1957 | 58 | 1991 | 1257 |
| 1875 | 1 | 1924 | 2 | 1958 | 46 | 1992 | 1308 |
| 1878 | 1 | 1925 | 4 | 1959 | 37 | 1993 | 1406 |
| 1880 | 1 | 1926 | 4 | 1960 | 46 | 1994 | 1406 |
| 1887 | 1 | 1927 | 4 | 1961 | 37 | 1995 | 1508 |
| 1888 | 1 | 1928 | 10 | 1962 | 52 | 1996 | 1510 |
| 1890 | 4 | 1929 | 22 | 1963 | 39 | 1997 | 1621 |
| 1891 | 1 | 1930 | 5 | 1964 | 102 | 1998 | 1672 |
| 1893 | 1 | 1931 | 2 | 1965 | 75 | 1999 | 1708 |
| 1895 | 1 | 1932 | 3 | 1966 | 75 | 2000 | 1995 |
| 1896 | 2 | 1933 | 9 | 1967 | 65 | 2001 | 2015 |
| 1897 | 1 | 1934 | 10 | 1968 | 161 | 2002 | 2371 |
| 1898 | 1 | 1935 | 8 | 1969 | 101 | 2003 | 2584 |
| 1899 | 6 | 1936 | 4 | 1970 | 124 | 2004 | 2831 |
| 1900 | 4 | 1937 | 13 | 1971 | 106 | 2005 | 3080 |
| 1902 | 2 | 1938 | 2 | 1972 | 118 | 2006 | 3319 |
| 1904 | 2 | 1939 | 6 | 1973 | 184 | 2007 | 3688 |
| 1906 | 1 | 1940 | 12 | 1974 | 180 | 2008 | 3760 |
| 1907 | 2 | 1941 | 4 | 1975 | 131 | 2009 | 3861 |
| 1908 | 1 | 1942 | 6 | 1976 | 160 | 2010 | 4352 |
| 1909 | 1 | 1943 | 1 | 1977 | 169 | 2011 | 4476 |
| 1910 | 2 | 1944 | 5 | 1978 | 185 | 2012 | 4721 |
| 1911 | 2 | 1945 | 11 | 1979 | 250 | 2013 | 4987 |
| 1912 | 1 | 1946 | 15 | 1980 | 233 | 2014 | 4660 |
| 1913 | 2 | 1947 | 15 | 1981 | 217 | 2015 | 4426 |
| 1914 | 2 | 1948 | 17 | 1982 | 251 | 2016 | 4255 |
| 1915 | 26 | 1949 | 23 | 1983 | 371 | 2017 | 4171 |
| | | | | | | 2018 | 3692 |

*Notes:* Number of new *SIC code 4953: Refuse Systems* firms in Europe from the first record in 1799 through 2018. These are summed by year of the date of incorporation for firms within the countries that would eventually comprise the 28 member European Union. A substantial increase in global plastic production coincides with the growth in waste management firms post-World War II. The second phase of even more rapid growth begins in the 1970s with the recycling revolution.

*Sources:* Author calculations based on data from Orbis accessed on March 3rd, 2019.

### A1.5 Nuclear Power Plant Construction Permit Approvals, 1955-2018

This appendix supports Figure 9 by reporting the share of commercial nuclear reactor construction permits approved and the nuclear share of U.S. energy sector output spanning the full history of civilian nuclear power. 1955-2011 data are from the U.S. Energy Information Administration, supplemented by 2012-2018 data from the United States Nuclear Regulatory Commission. Substantial public outcry follows the Three-Mile Island accident in 1979 and subsequently increased regulatory measures were put in place. Zero new construction permits are then taken to the approval stage for over three decades.

TABLE 25— U.S. NUCLEAR REACTOR CONSTRUCTION PERMITS AND GENERATION SHARE

| | Construction Permits Issued | Operating Licenses Issued | Operable units | Permanently shut down | Share of energy production (per cent) | |
|---|---|---|---|---|---|---|
| 1955 | 1 | 0 | 0 | 0 | 0 | |
| 1956 | 3 | 0 | 0 | 0 | 0 | |
| 1957 | 1 | 1 | 1 | 0 | 0 | |
| 1958 | 0 | 0 | 1 | 0 | 0 | |
| 1959 | 3 | 1 | 2 | 0 | 0 | |
| 1960 | 7 | 1 | 3 | 0 | 0.1 | |
| 1961 | 0 | 0 | 3 | 0 | 0.2 | |
| 1962 | 1 | 6 | 9 | 0 | 0.3 | |
| 1963 | 1 | 2 | 11 | 1 | 0.3 | |
| 1964 | 3 | 3 | 13 | 1 | 0.3 | |
| 1965 | 1 | 0 | 13 | 0 | 0.3 | |
| 1966 | 5 | 2 | 14 | 1 | 0.5 | |
| 1967 | 14 | 3 | 15 | 2 | 0.6 | |
| 1968 | 23 | 0 | 13 | 1 | 0.9 | |
| 1969 | 7 | 4 | 17 | 0 | 1 | |
| 1970 | 10 | 3 | 20 | 1 | 1.4 | |
| 1971 | 4 | 2 | 22 | 0 | 2.4 | |
| 1972 | 8 | 6 | 27 | 2 | 3.1 | |
| 1973 | 14 | 15 | 42 | 0 | 4.5 | |
| 1974 | 23 | 15 | 55 | 2 | 6.1 | |
| 1975 | 9 | 2 | 57 | 0 | 9 | |
| 1976 | 9 | 7 | 63 | 1 | 9.4 | |
| 1977 | 15 | 4 | 67 | 0 | 11.8 | |
| 1978 | 13 | 4 | 70 | 1 | 12.5 | |
| 1979 | 2 | 0 | 69 | 1 | 11.3 | Three-Mile Island |
| 1980 | 0 | 2 | 71 | 0 | 11 | incident |
| 1981 | 0 | 4 | 75 | 0 | 11.9 | (March 28th) |
| 1982 | 0 | 4 | 78 | 1 | 12.6 | |
| 1983 | 0 | 3 | 81 | 0 | 12.7 | |
| 1984 | 0 | 6 | 87 | 0 | 13.5 | |
| 1985 | 0 | 9 | 96 | 0 | 15.5 | |
| 1986 | 0 | 5 | 101 | 0 | 16.6 | |
| 1987 | 0 | 8 | 107 | 1 | 17.7 | |
| 1988 | 0 | 2 | 109 | 0 | 19.5 | |
| 1989 | 0 | 4 | 111 | 3 | 17.8 | |
| 1990 | 0 | 2 | 112 | 0 | 19 | |
| 1991 | 0 | 0 | 111 | 1 | 19.9 | |
| 1992 | 0 | 0 | 109 | 2 | 20.1 | |
| 1993 | 0 | 1 | 110 | 0 | 19.1 | |
| 1994 | 0 | 0 | 109 | 0 | 19.7 | |
| 1995 | 0 | 0 | 109 | 0 | 20.1 | |
| 1996 | 0 | 1 | 109 | 3 | 19.6 | |
| 1997 | 0 | 0 | 107 | 2 | 18 | |
| 1998 | 0 | 0 | 104 | 1 | 18.6 | |
| 1999 | 0 | 0 | 104 | 0 | 19.7 | |
| 2000 | 0 | 0 | 104 | 0 | 19.8 | |
| 2001 | 0 | 0 | 104 | 0 | 20.6 | |
| 2002 | 0 | 0 | 104 | 0 | 20.2 | |
| 2003 | 0 | 0 | 104 | 0 | 19.7 | |
| 2004 | 0 | 0 | 104 | 0 | 19.9 | |
| 2005 | 0 | 0 | 104 | 0 | 19.3 | |
| 2006 | 0 | 0 | 104 | 0 | 19.4 | |
| 2007 | 0 | 0 | 104 | 0 | 19.4 | |
| 2008 | 0 | 0 | 104 | 0 | 19.6 | |
| 2009 | 0 | 0 | 104 | 0 | 20.2 | |
| 2010 | 0 | 0 | 104 | 0 | 19.6 | |
| 2011 | 0 | 0 | 104 | 0 | 19.3 | |
| 2012 | 2 | 0 | 104 | 0 | 19 | |
| 2013 | 0 | 0 | 100 | 4 | 19.4 | |
| 2014 | 0 | 0 | 99 | 1 | 19.5 | |
| 2015 | 1 | 0 | 99 | 0 | 19.6 | |
| 2016 | 3 | 0 | 99 | 0 | 19.8 | |
| 2017 | 1 | 0 | 99 | 0 | 20 | |
| 2018 | 1 | 0 | 98 | 1 | 19.3 | |

*Notes:* U.S. nuclear power plant construction and operating permits, functional power plants, and share of national energy production. Building permits issued exceeds the number of power plants because not all are completed. Following the 1979 Three-Mile Island accident, several were completed but no new building permits issued for three decades. The number of sites has since declined, and those nearing retirements are unlikely to be replaced.

*Sources:* Data reproduced from Energy Information Administration (1955-2011) and United States Nuclear Regulatory Commission (2012-2018).

## A2.1 Improving Recyclability Through Standardisation Auctions

In this section I outline a second approach to the recycling problem. This method is more applicable when the interest is instead to standardise an entire waste stream. This approach, while possibly very interesting to consider, is outlined in the appendix rather than the main body because the applicable policy instrument that I arrive at is fairly well-known. I show that one efficient approach is to institute a Vickrey-style auction over the right to an industry's product or packaging design, as applicable. So, a complex – even intractable – problem may find a solution in an existing and thoroughly studied instrument.

In some cases, standardisation across firms in an industry might prove more effective in reducing recycling costs. Suppose the issue is not the number of materials in a product and their separability. Rather, the stream of materials is indistinguishable at an acceptable cost. As an example, the most common plastic bottle material, PET, is easily distinguished from HDPE and PP caps and lids.[120] PET sinks in water (density versus water of 1.33-1.39) while HDPE (0.94-0.97) and PP (0.93-0.94) are buoyant (Aguado and Serrano, 1999). HDPE and PP, however, are difficult to distinguish from each other in processing rates that are economical despite having substantially higher value when separated than mixed. Even differences within a plastic class, for instance between different PP bottle compositions, can hinder recyclability (Eriksen, Christiansen, Daugaard, and Astrup, 2019).

We might attempt to modify the prior framework – instead of *N* materials within a product, consider *N* materials across industries producing similar items. However, the issue at hand is markedly different as contamination by just a few firms who disagree with the policy cancels out the efforts of all others. Instead, industry requires a framework to reach universal agreement on a standard. This calls for efficient industry standardisation through a redistribution auction.

Unfortunately, the conditions where industry standardisation occurs spontaneously are stringent. Ostrom (1990) lists conditions common in communities reaching agreement without state intervention based on hundreds of case studies, and Coase (1960) outlines the difficult set of conditions for an agreement. The most evident takeaways from Ostrom and Coase are that spontaneous order is rare and takes substantial effort to maintain. Furthermore, even if industry recognises an innovation as unequivocally beneficial, the rate of technological change might be unacceptably slow and unpredictable (see a within-firm case study by Carlson, slow industry adoption over seven-decades by Snowden, and a case

[120] And occasionally PET and polyethene (polyethylene) (PE) caps and lids.

of industry failure to adopt altogether by Calomiris, in *Coordination and Information*, 1995). The state might instead require a more immediate industry change yet seek to preserve the industry's expertise in devising an appropriate standard and then require its universal adoption.

It might be optimal for producers to adopt a single standard, e.g. HDPE or PP for all bottle caps, more efficient to raise a tax for development of efficient sorting means, or more efficient for firms to opt-out of the problem altogether by removing their product from the market or make them markedly distinguishable, e.g. HDPE and PP caps in different shapes. Here I contribute to the first option – efficiently choosing an industry standard, and address how to choose one by pairing auction and redistribution mechanisms.

### *A2.1.1 Finding an Efficient Industry Standard and Defining Evaluation Criteria*

Suppose the goal is to standardise an industry, and on the advice of Ostrom, Coase, and the observation that standardisation has not already occurred, the regulator does not expect an industry to standardise without intervention. Note that many forms of potentially recyclable refuse, e.g. bottles, labels, and bottle caps, are differentiated for marketing purposes because firms receive an advantage from customer recognition. Despite the firm's private benefit from product differentiation through marketing, due to the severity of the environmental impact of nonrecycled waste, the regulator decides to set a standard.[121] Deciding to set a standard does not inform on what the standard should be, however. Moreover, unlike standards in the sense of quantity limitations, product standardisation is multidimensional, and a regulator is less appropriately informed than industry to decide. Society would benefit from a mechanism by which industry itself decides a standard meeting basic recyclability and efficiency criteria.

Standardisation of materials entering the recycling stream is important because even small shares of impurities markedly reduce the value of recycled material. To have a familiar language, let us continue to suppose we want to improve the recyclability of bottle caps. The goal is to ensure all bottle caps are the same, but leave their design, within reason, to the industry through a decision process.[122] The example of bottle caps is particularly

[121] Weitzman (1974) discusses when a price versus quantity-based regulation is more efficient. Here I do not intend that we reduce quantity at all, but rather standardize an industry's product. There is an all-or-nothing quality to success in the current scenario, and so neither a tax nor quantity regulation is applicable. Rather, I propose a framework for industry to choose a specification-based standard.

[122] "Within reason" means that industry be allowed to design the product through some industry decision process, but the result is constrained to meeting some minimum requirements on recyclability.

fitting because they are entirely recyclable, but one of the barriers to profitable recycling is the cost of sorting their variety into differentiated lots.

I present a simple case: Suppose we only require a uniform (standardised) product – recyclability concerns beyond homogeneity are not within the scope of this analysis. Further, there is a strict all or nothing nature to the problem – any heterogeneity makes all units of the waste unprofitable to recycle. This condition makes the problem different from the chapters of this thesis and many environmental problems where marginal changes in pollution levels have value. Here, heterogeneity results in entire lots of waste not being recycled and instead released into the environment with uncertain, long-term, and negative consequences.

For structure, volumes of the material to be regulated are $a_i > 0$ for all firms in the industry, $i = 1{:}N$, such that $\sum_{i=1}^{N} a_i = a$ and I scale to firm market shares, $\frac{a_i}{a} = s_i, \cdots, \frac{a_N}{a} = s_N$ such that $\sum_{i=1}^{N} s_i = s = 1$. Each share $s_i$ is associated with a product type $\mathcal{s}_i$ differing by firm in a fully differentiated market. Alternatively, we may view $s_i$'s as groups of firms within a market that share a product design and coordinate to influence regulation. Suppose firms and market shares already exist – that the regulator seeks to apply a standard to an existing market. As with many environmental problems, the regulator seeks to correct an environmental problem occurring from a perhaps longstanding production tradition. That is, we do not realize there is going to be an environmental problem until it occurs. As a matter of expositional efficiency, I rank market shares from the largest, $s_1$, to smallest, $s_N$. Initially also suppose perfect information about and between firms – everyone knows each other's market shares and costs of changing product type. Also suppose that $s_i$'s are stable and that small changes in cost do not lead to substantial changes in market share. Finally, assume initially that there are no fixed costs to changing product types and the relevant marginal cost, in the sense of being dependent on $s_i$, of changing to another type is a constant, shorthanded $c(\mathcal{s}_i \rightarrow \mathcal{s}_j) = c$ for all $i \neq j$, and comparisons of $s_i$ to $s_j$, $cs_i$ to $cs_j$, and $acs_i$ to $acs_j$ are equivalent for any scalars $c$, $a$. An implication is that if standard monitoring and enforcement costs are per-unit and uniform, we can disregard them in this calculation.

One depiction of industry and the costs of switching product types is presented in Figure 38. If all firms switch to an entirely different standard, $\mathcal{s}_o$, $o \notin 1{:}N$, then the total cost of industry compliance is $\sum_{i=1}^{N} s_i c = c \sum_{i=1}^{N} s_i = c$. But industry can achieve a uniform, ceteris paribus more recyclable product at a lower cost than enforcing a wholly foreign standard that does not match any existing firm's product type. Since setting the industry standard as any firm $i$'s type allows that firm to avoid cost $cs_i$, choosing any $\mathscr{s}_i$ achieves homogeneity at a lower cost than by choosing $\mathscr{s}_o$. Further,

(49) $$c(1-s_1) \leq c(1-s_2) \leq \cdots \leq c(1-s_N) \leq c$$

and with strict inequalities whenever market shares are not equal. So, for an industry where firm $i = 1$ has at least as much market share as any other, a least cost, efficient solution is to set the industry standard to an industry modal product type such as $\mathscr{s} = \mathscr{s}_1$. The cost to the industry of change is then $c^* = c(1 - s_1)$.

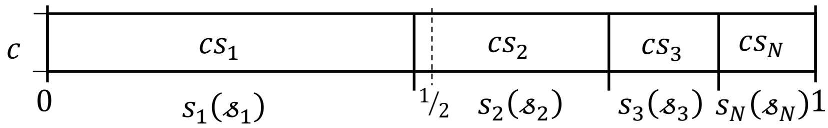


FIGURE 38. INDUSTRY MODEL WITH UNIFORM MARGINAL COST OF CHANGE

*Notes:* Representation of the entire industry as market shares $s_i$ of different product types $\mathscr{s}_i$ for all $i = 1{:}N$. The marginal cost of changing product type is c such that the cost for the entire industry to change to an outside option, $\mathscr{s}_0$, is c. If a standard is set to $\mathscr{s} = \mathscr{s}_i$, then $c_i = 0$, else $\mathscr{s} \neq \mathscr{s}_j$, then $c_j = cs_j(\mathscr{s}_j)$. I order shares from largest to smallest which has no corresponding relevance in the real world so choosing the standard at the ½ mark is arbitrary. It is immediate that choosing $\mathscr{s}_1$ results in the least industry cost, $c^* = c(1 - s_1)$ which is an efficient solution.

However, all firms $i \neq 1$ have reason to oppose standardisation to $\mathscr{s}_1$ as each incurs cost $cs_i$ that the industry leader avoids. Further, unless $s_1 \geq \frac{1}{2}$, the savings to firm $i = 1$ cannot be sufficient to offset the cost to all other firms, not that the industry leader has a unilateral, unprompted incentive to offer recompense anyway. By just choosing $\mathscr{s}_1$, the regulator picks policy winners and losers and may attempt to do so against prohibitive levels of opposition. So, at a minimum, the regulator should consider policy mechanisms that usually allow industry to arrive at the efficient solution on their own. We should also compare policy alternatives using some rational criteria that capture the competing interests of firms and the public. I emphasise three often-used criteria to judge competing policies: efficiency, political feasibility (shorthand: feasibility), and equitability.

An advantage is that we begin with the knowledge that an efficient policy is one that results in $\mathscr{s} = \mathscr{s}_1$ which is the standard set to a modal product type (here Firm 1's type). In comparison, the most feasible policy is the solution that minimises opposition to setting an $\mathscr{s} = \mathscr{s}_i$. We might set opposition in terms of cost – a rational perspective – as $cs_i$'s and then discuss in terms of shares.[123] In this sense, too, setting $\mathscr{s} = \mathscr{s}_1$ minimises total industry opposition, but by satisfying only one firm – the largest. We might also measure opposition as the number of firms that must pay at all to convert. In this all-or-nothing perspective, setting $\mathscr{s} = \mathscr{s}_i$ for any $i$ without compensation to firms $j \neq i$ are equally worse than any alternative except setting $\mathscr{s} = \mathscr{s}_o$. The appropriateness of these feasibility measures, which I call type I and II, is specific to industry lobbying behaviour. The most equitable solution, in comparison, is one where the post-policy marginal cost paid is equal between firms, that $\alpha_1 c = \alpha_2 c = \cdots = \alpha_N c$ for policy prescription $\alpha_1, \alpha_2, \dots, \alpha_N$. This may also be referred to as the envy-free state – the prescription where no firm prefers the per-unit cost of converting that any of their competitors must pay. Weaker measures are sometimes appropriate. We might consider whether a policy is revenue neutral – whether the revenue collected from firms stays within the industry or is captured or supplemented by the state. Setting $\mathscr{s} = \mathscr{s}_1$ is efficient and maximises type I feasibility, but alone does nothing to address equitability nor type II feasibility. It is also revenue neutral as no money has changed hands. I now compare some alternative mechanisms using these criteria against the regulator picking a winner.

Two immediate alternatives are to choose the mean or median product type. Suppose we randomly assign the standard – equivalently hold a lottery. One method is an unweighted lottery where each firm receives one ticket of equal probability of winning, then $Pr(\mathscr{s} = \mathscr{s}_i) = \frac{1}{N}, \sum_{i=1}^{N} \frac{1}{N} = 1$, and $Pr(\mathscr{s} \neq \mathscr{s}_i) = \left(1 - \frac{1}{N}\right)$. For the unweight (u) lottery, the expected industry cost is, $E\left[c^U | Pr(\mathscr{s} = \mathscr{s}_i) = \frac{1}{N}\right] = c\left(1 - \frac{1}{N}\right) > c(1 - s_1) = c^*$ unless all firms have equal market share, then this is met with equality as the median would equal an industry mode as well as the mean.[124] Alternatively, suppose the lottery is weighted (w) by share – the value of each firm's ticket is in proportion to their market share. Then, $Pr(\mathscr{s} = \mathscr{s}_i) = s_i$, $Pr(\mathscr{s} \neq \mathscr{s}_i) = 1 - s_i$, and recall $\sum_{i=1}^{N} s_i = s = 1$. Then, $E[c^w | Pr(\mathscr{s} = \mathscr{s}_i) = s_i] = c(1 - \sum_{i=1}^{N} {s_i}^2) > c(1 - s_1) = c^*$.[125] Again, if all firms hold equal market share, this relation is instead met with equality as the mean would equal an industry mode and the median. In both cases the expected total conversion cost of outright setting standard $\mathscr{s} = \mathscr{s}_1$ is less than if we set by a lottery. In addition to not addressing feasibility and equity

123 This is a simplification, there are likely differences between the cost of switching and willingness to pay to avoid switching in practice (e.g. funding lobbying efforts).

124 Note $E\left[c^U | Pr(\mathscr{s} = \mathscr{s}_i) = \frac{1}{N}\right] = \sum_{i=1}^{N} \frac{1}{N} c(s - s_i) = \frac{1}{N} cs \sum_{i=1}^{N} 1 - \frac{1}{N} c \sum_{i=1}^{N} s_i = c - \frac{1}{N} c = c\left(1 - \frac{1}{N}\right) > c(1 - s_1) = c^*$.

125 Note $E[c^w | Pr(\mathscr{s} = \mathscr{s}_i) = s_i] = \sum_{i=1}^{N} s_i c(s - s_i) = \sum_{i=1}^{N} s_i cs - \sum_{i=1}^{N} s_i c s_i = cs \sum_{i=1}^{N} s_i - c \sum_{i=1}^{N} {s_i}^2 = c(s^2 - \sum_{i=1}^{N} {s_i}^2) = c(1 - \sum_{i=1}^{N} {s_i}^2) > c(1 - s_1) = c^*$.

concerns, we find these lotteries are unlikely to be efficient. Small firms – those where $s_i < \frac{1}{N}$, however, are in particular more likely to support the unweighted lottery because it provides a larger probability that their type is chosen and thus, their expected cost is lower. Nevertheless, the odds are stacked against any one firm, and in addition to being inefficient, total expected type I feasibility decreases while equitability is unchanged.

We might also suppose a process akin to visually selecting the median and choose the product type at the ½ share mark. One issue is that there is no natural reason why firms are ranked in this manner. But by whatever sorting method, if a firm has $s_1 \geq \frac{1}{2}$ guaranteeing they are the market leader, and if we suppose market share is a naturally continuous concept as depicted for convenience – which it is not – then choosing the median results in the efficient solution. We should otherwise accept the ordering and depiction is arbitrary, though convenient, and that choosing the firm at the ½ mark is another form of lottery by market share.

#### *A2.1.2 Regulating Industry with Paired Auction and Redistribution Mechanisms*

What the regulator requires is a mechanism that allows industry itself to choose a standard, preferably efficiently. The lack of standardisation without intervention suggests there are barriers to this process. One issue is that standardisation requires all but one firm to accept an expense to standardise. Another is that firms are in competition and would not want to give away competitive advantage, either in knowledge, advertising potential, or market position to competitors. The mechanism needs to aggregate firm preferences without incurring large transaction costs while protecting the privacy of firm valuations. Open bargaining between several firms clearly does not meet these requirements. However, a variety of auction mechanisms exist that would allow industry to choose a product type endogenously. Some of these preserve bidder privacy – through sealed bids – and can be conducted with low transaction costs. Surely some forms of auction are more desirable than a regulator picking winners and losers or using chance. Collecting auction revenue, for instance, provides an opportunity to explore to what extent both feasibility and equitability measures can be improved.

To be clear, I explore an auction to choose the industry standard. That is, the winning firm specifies aspects of the product that eventually enters the recycling stream, and all producers in the industry then use this product design.[126] This process clearly requires non-excludability – that no competitor can be excluded from using the design. Allowing excludability, instead, assigns monopoly rights to a firm, and as such, every firm would have the same bid amount – the monopolist's value of the entire industry. With these conditions in place, I provide a very brief review of the auction literature, explore bidding mechanisms that should result in the efficient solution $\mathcal{s} = \mathcal{s}_1$, and then discuss how the revenue can be used to address the other desirability criteria.

#### *A2.1.2.1 A Briefest Review of the Auction Literature*

It is fashionable to observe that the earliest surviving, Western, record of auctions occurs in the 5th century BC (Herodotus, 1824 trans.) which is also around the time of the first surviving recorded instance of random allocation by drawing lots (Leviticus 16:8, King James Version). It is also fashionable to note that auctions have occasionally found use at points of great historical importance, e.g. the auctioning of the Roman empire in 193 AD (Hekster and Zair, 2008). However, the study of allocation by chance, within the study of probability theory, received earlier substantive theoretical interest beginning in the 16th century (Hacking, 2013). Auctions, in comparison, did not receive substantive theoretical treatment until the 1960's (Cassady, 1967). We might trace modern interest in auction theory specifically to Vickrey's (1961) study of second-price sealed-bid auctions which I will show are, here too, half of an efficient solution to optimal standard selection that also addresses other criteria.

Many auction mechanisms have been explored since 1961 from Vickrey's game-theoretic perspective. See Klemperer (1999) for a survey, Clarke (1971) and Groves (1973) for important theoretical contributions, and Milgrom (2012) for an analysis of several examples. The auction literature includes propositions to address environmental problems specifically too. The auction of tradeable emissions permits receives substantial interest. See an early proposal to use a Vickrey auction in Dasgupta, Hammond, and Maskin (1980) to get firms to reveal their true costs of abatement, or a recent discussion of alternatives in He and Chen (2014). But proposals and applications have been found in many other environmental areas, e.g. optimal allocation of the commons (Montero, 2008), maintaining renewable natural capital quality (Teytelboym, 2019), and conservation auctions to preserve biodiversity (de Vries and Hanley, 2016). The theory supporting the combination of auctions and paired revenue redistributions has been explored at length in, for example,

[126] So, we must continue to require some minimum recyclability aspect of the product.

Green and Laffont (1979) as an early yet comprehensive work. Porter, Shoham, and Tennenholtz (2004) also provide an intuitive introduction to auction redistributions with similar criteria to the ones I propose. The auction I propose is, albeit, roughly backwards of Porter, Shoham, and Tennenholtz's mechanism. The other agents (firms) in their setting pay (uniformly) for the most efficient firm to complete some task. Here instead, I propose the firm that would incur the greatest loss from performing a task bids to win such that it pays all other firms in the industry through an auction revenue redistribution to complete it instead. This task might be viewed as a "bad" by industry members as their participation in the auction takes the form of bids to avoid a greater loss. Moulin (2010) provides a proof that "goods" and "bads" can be handled equivalently in a Vickrey auction – rather than the firm with the highest valuation receiving the good, we search among agents by comparing disutility's.

Close parallels also occur in the field of artificial intelligence (AI) and applied computing. Cavallo (2006) initiated a wave of interest in the field by proposing a Vickrey auction paired with a redistribution to all participants of a form we might trace back to Bailey (1997). See, for example, Guo and Conitzer (2008) on the dominance of Vickrey auction-Bailey redistributions versus other auction forms, Guo and Conitzer (2010) on the efficiency of Bailey type redistribution mechanisms versus other forms, and efforts to overcome the budget balance issue without losing efficiency or making other compromises in Mishra and Sharma (2018). AI, applied computing, and similar applications occur in settings where enriching an external party through auction revenue – generally the seller of a good or the state – has no useful equivalent. Rather, the goal is to allocate efficiently and redistribute auction revenue within the system (here industry) while retaining the auction's strategy-proof quality – the concept that players (here firms) report their true value of the item being auctioned to the auction mechanism.

I add to the auction-redistribution literature by first outlining a paired auction and distribution mechanism resulting in an efficient allocation under the conditions of the preceding section. I then find the industry conditions necessary for efficient allocation without more intrusive intervention. I then relate the framework to auctions with more realistic information and industry structural assumptions – the result is proposing a Vickrey-type auction to overcome, foremost, informational constraints, followed by a redistribution of the winning bid to all other firms to offset conversion costs.

### *A2.1.2.2 A Paired Loss Avoidance Auction and Redistribution*

I continue with the preceding industry and perfect information structure. Suppose a first-price auction is held for the right to industry's standard. Each firm has a private valuation of the loss they incur if the industry does not standardise to their type, $v_i(\mathcal{s}_i)|_{i\ loses} = cs_i(\mathcal{s}_i)$, such that, dispensing with the $\mathcal{s}_i$ type notation, $v_1 \geq v_2 \geq \cdots \geq v_N$, which are related by strict inequalities whenever market shares are not equal. Alternatively, the value to a firm of retaining their type as the industry standard is zero, $v_i(\mathcal{s}_i)|_{i\ wins} = 0$. Tautologically the cost of changing their type to the industry standard when their type is already the industry standard is zero. The potential payoffs to *i* with bid $b_i$ in this loss avoidance auction are then[127]

$$(50) \qquad p_i = \begin{cases} -b_i & if\ b_i > b_j\ for\ all\ i \neq j\ (i\ wins) \\ -cs_i & if\ b_i < b_j\ for\ at\ least\ one\ i \neq j\ (i\ loses) \end{cases}$$

Firm *i* receives a higher payoff from winning when $-b_i > -cs_i$ and so bids up to $cs_i \geq b_i$. With knowledge of the payoff functions of all other firms, we expect bid $b_i \geq cs_j$ for $i < j$ as I rank firms by market share, and expect Firm 1 to place the winning bid of $b_1 = cs_2$ (plus some nearly zero tiebreaking amount, $\varepsilon$). If firms bid above their cost of transitioning to the standard, they risk incurring a larger loss. But also, firms recognise that their competitors, including their next closest smaller competitor lose more if they bid more than their own transition costs as well. Each firm, including Firm 1, then chooses the least bid from the interval $cs_i \geq b_i \geq cs_{i+1}$, and so chooses $b_i = cs_{i+1}$ (plus $\varepsilon$).

Even without a redistribution mechanism, the process remains efficient because Firm 1 has the highest valuation and bids to win such that $\mathcal{s} = \mathcal{s}_1$ results – the type adopted across the industry is again the modal type. It remains politically infeasible and inequitable, however, and all firms $i \neq 1$ oppose it on the grounds of conversion costs and loss of advantage to Firm 1. Winning Firm 1 might also oppose it because they are paying for the right to keep using their design, which is already theirs by trademark, copyright, patent, or tradition. However, Firm 1 might also support the framework when they gain an advantage from industry transition. Their per-unit cost of industry transformation – in the form of their

[127] That is, the resulting conditional or dependent valuation and payment (bid) pairs, $(v_i, b_i)$, in the function $p_i = v_i - b_i$ are $(0, b_i)$ if *i* wins, and $(-cs_i, 0)$ if *i* loses. The tiebreaking case can be specified as a lottery between parties in a tie, has an expectation of $\frac{-(b_i+cs_i)}{number\ of\ ties}$, and is unimportant to the greater discussion.

bid paid – is less than or equal to that of any other firm in the form of the incurred cost of changing to the standard product type.

To address equity and feasibility – but sometimes risking efficiency in the process – consider a market-share weighted redistribution of the winning bid to the rest of the industry. Since $(1 - s_1) = (s_2 + s_3 + \cdots + s_N)$, the market shares, excluding Firm 1 as the winning bidder, are of the form $\frac{s_{i \neq 1}}{1 - s_1}$ and share-weighted distributions are

$$\lambda_i = cs_2 \left(\frac{s_i}{1 - s_1}\right), i \neq 1. \tag{51}$$

Knowing (or being able to conjecture) on the market shares of firms is a substantial advantage in determining fair redistributions. In other settings, the practitioner cannot determine any equivalent of this measure. In addition to causing mathematical complexity during analysis, distribution weights must often be suggested of symmetric size, $\lambda_i = \lambda = \frac{1}{N}$ for all *N* firms involved in the redistribution as a best guess of sorts. See, for example, Bailey (1997), Cavallo (2006), Green and Laffont (1979), and Porter, Shoham, and Tennenholtz (2004). In comparison to an $\frac{s_{i \neq 1}}{1 - s_1}$ distribution, one of size $\frac{1}{N}$ is undesirable for large, but not small firms which might even receive a profit from converting.

But then, in this setting I must ask whether it is ever the case that Firm 1 does not prefer to win and instead have their conversion costs partially offset by a competitor. Suppose Firm 1 follows a min-max strategy – minimise the maximum cost another firm's actions can impose on them. Then the choice of payoffs are $-cs_2$ when Firm 1 bids successfully, and $b_{w \neq 1}\left(\frac{s_1}{1 - s_{w \neq 1}}\right) - cs_1$ when another firm, *w*, wins and Firm 1 receives a distribution of *w*'s bid. Consistent with the min-max assumption, Firm 1 believes Firm 2 will bid to win and Firm 1 expects the payoff to be

$$p_1 = \begin{cases} -cs_2, & Firm\ 1\ wins \\ b_2\left(\frac{s_1}{1 - s_2}\right) - cs_1, & Firm\ 1\ loses\ and\ Firm\ 2\ wins \end{cases} \tag{52}$$

I compare these payoffs to determine when the market structure – the shares of firms – results in Firm 1 bidding to win. From $-cs_2 > b_2\left(\frac{s_1}{1 - s_2}\right) - cs_1$,

$$s_1 > s_2 \left(\frac{\mathrm{c}}{\mathrm{c} - \frac{b_2}{1 - s_2}}\right) \tag{53}$$

is required for Firm 1 to bid $b_1 > b_2$. That is, $s_1$ must be at least large enough that Firm 1, with such beliefs over $b_2$, prefers to win rather than take a share-weighted distribution of Firm 2's bid but pay the rest of their own conversion cost. Also consistent with a min-max strategy, Firm 1 bids as though they believe Firm 2 will asymmetrically bid $b_2 = cs_2$. Then, the two largest market shares must satisfy $s_1 > s_2\left(\frac{1-s_2}{1-2s_2}\right)$. On the other end of the spectrum, Firm 1 believes Firm 2 to follow a symmetric strategy and bid $b_2 = cs_3$, then

(54) $$s_1 > s_2\left(\frac{1-s_2}{1-s_2-s_3}\right)$$

which requires information on the size of the three largest firms. Notice that these functions are increasing in $s_2$, and $s_2$ and $s_3$, respectively – the alternative values of taking a pay-out increases for Firm 1. Suppose $s_3 = 1 - s_1 - s_2$, a triopoly, then $s_1 > \sqrt{s_2(1-s_2)}$ is required. A plot of the market structure required for the efficient solution to emerge is illustrated in Figure 39. It illustrates, as a set of level curves, the minimum required $s_1$ size for different conjectures on Firm 2's bidding behaviour and Firm 2's market size – the minimum conditions for an efficient solution to result without more complex state intervention. Focussing on the min-max strategy, I note along the blue line the minimum necessary $s_1$ shares for the conjecture that Firm 2 bids $b_2 = cs_2$. Then, Firm 2 share sizes $s_2 \leq \frac{1}{3}$ can result in an efficient solution as long as Firm 1 is sufficiently large – in the case when $s_2 = \frac{1}{3}$, $s_1 = \frac{2}{3}$ is required. As an alternative for comparison, suppose that Firm 1 expects Firm 2 to act symmetrically by bidding $b_2 = cs_3$ and observes that $s_3 = \frac{1}{2}s_2$ such that $b_2 = \frac{1}{2}cs_2$. Then for $s_2 = \frac{1}{3}$, $s_1 \geq \frac{5}{12}$ is sufficient to result in an efficient solution (if Firm 1 conjectures correctly).

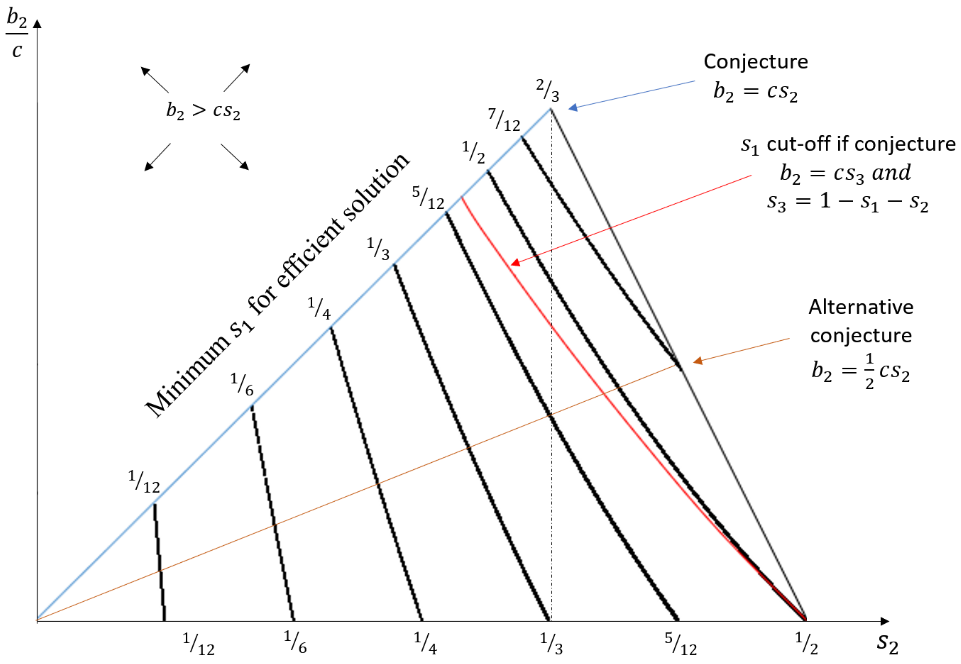


FIGURE 39. MINIMAL MARKET STRUCTURE FOR AN EFFICIENT BID STRATEGY

*Notes:* Level sets of the minimum size of the largest firm, Firm 1, based on the second largest, Firm 2, in the industry that is required for Firm 1 to bid to win. If Firm 1 uses a min-max strategy (minimise the maximum loss that other firms can force them to incur), Firm 1 must be at least as large as the values on the blue line for any Firm 2 market share on the horizontal axis. If Firm 1 instead supposes that Firm 2 has a symmetric strategy to their own, the alternative payoff to Firm 1 – the share they receive of Firm 2's bid revenue – decreases. In the case of a triopoly, the additional maximum constraint as the red line occurs. I also illustrate an alternative conjecture that Firm 2 bids half their value (orange line) which is also the relevant conjecture if Firm 1 believes Firm 2 to follow a symmetric strategy and $s_3 = \frac{1}{2}s_2$.

Figure 40 re-illustrates the market structure where the efficient solution results when Firm 1 follows a min-max strategy. These are the minimum market shares for Firm 1, given Firm 2, under which Firm 1 intends to win. We observe that as the size of Firm 2 increases, a decreasing variety of industry arrangements results in the efficient solution. As industry approaches the duopoly extreme, Firm 2 is also fully compensated for transitioning to Firm 1's type.

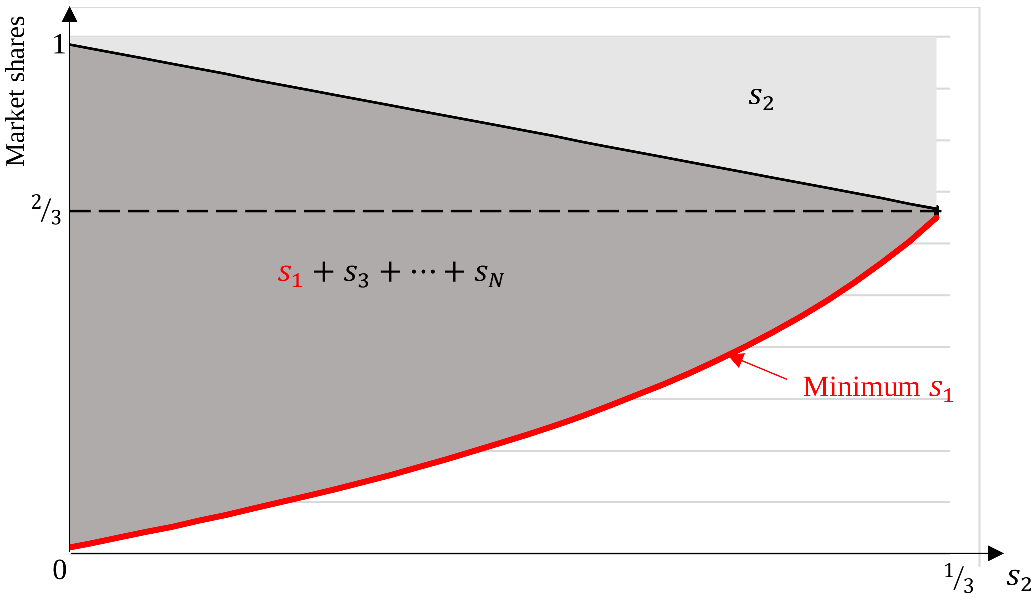


FIGURE 40. MARKET SHARES RESULTING IN EFFICIENT BIDS UNDER MIN-MAX STRATEGY

*Notes:* Market structure leading to an efficient solution ($\mathscr{s} = \mathscr{s}_1$) if Firm 1 assumes a min-max strategy. The minimum Firm 1 market share necessary (red line) increases as the alternative value – a redistribution share of another firm's (Firm 2's) winning bid increases. In the duopolistic case, $s_2 \leq \frac{1}{3}$ is required for the efficient solution to emerge without further bid and distribution structure.

Additional intervention can overcome market structural bounds. I have previously noted that researchers have proposed a redistribution weight of $\lambda = \frac{1}{N}$ when no equivalent of market shares can be determined. Montero (2008) also proposes a framework that includes the winning bidder in a more complex redistribution scheme. Such modifications, in effect, result in different effective market shares in the distribution and lose either revenue neutrality or progress on other criteria in the process. It is also a minor point to extend the framework to idiosyncratic marginal costs. Suppose $c > 0$ is the share of marginal costs common to all firms in the industry, let $c_i > 0$ be firm *i* specific marginal costs, and $\overline{s_i} > 0$ be firm *i*'s unweighted market share. Let $s_i = \left(1 + \frac{c_i}{c}\right)\overline{s_i}$ and the preceding framework holds.

Maintaining an unmodified structure, when $\mathscr{s} = \mathscr{s}_1$, industry retains maximum type I feasibility when the auction redistributes all of Firm 1's bid to Firms $2{:}N$. But, with a redistribution, we instead have an industry where perhaps all firms oppose the standard and type II feasibility is zero because everyone pays, albeit less than with no redistribution mechanism at all.[128] One possible outcome is that a coalition or cartel of firms organises to disrupt any efforts to decide on or enforce an industry standard.

[128] With the exception of the preceding duopoly case.

The impact on equitability is more promising as it must improve with redistribution. Importantly, Cavallo (2006) emphasises the separability of bid and redistribution mechanisms – if the bid is efficient and Nash, distributional concerns can be considered independently. Observe when the regulator assigns the standard to any firm *i* without redistribution, $\alpha_i = 0$ and all $\alpha_{j \neq i} = 1$. But in the auction mechanism, suppose the market structure is such that Firm 1 wins. For Firm 1, $\alpha_1 c = -\frac{cs_2}{s_1}$, and for all other firms $i \neq 1$, $\alpha_i c = \frac{cs_2\left(\frac{s_i}{1-s_1}\right) - cs_i}{s_i}$ from their bid and share weighted distribution and adjustment costs, respectively. Then redistribution weights are $\alpha_1 = -\frac{s_2}{s_1}$ and $\alpha_{i \neq 1} = s_1 + s_2 - 1$. Setting $\alpha_1 = \alpha_{i \neq 1}$, full equitability occurs with industry structure such that $s_2 = \frac{s_1(1-s_1)}{1+s_1}$ which can only be satisfied when Firm 2 is small, at most $(s_1, s_2) \approx \left(\frac{5}{12}, \frac{1}{6}\right)$. Discovering the market structure resulting in equitability is more academic than practical, however. In practice, if a regulator prefers equitability and is willing to risk efficiency, they might prefer to fractionally redistribute – subsidise conversion or impose fees – and instead distribute $\frac{\alpha_1}{\alpha_{i \neq 1}} cs_i = \frac{s_2}{s_1}\left(\frac{1}{1-s_1-s_2}\right) cs_i$ to losing firms. It is shown in Green and Laffont (1979) that equivalent measures to efficiency, feasibility, and equitability are traded off in the pursuit of a balanced budget. This has also been explored in, for instance, AI and applied computing in Guo and Conitzer (2008).

Unfortunately, the perfect information assumption is not entirely realistic either. I now observe parallels between the outlined process and a Vickrey type auction with imperfect information.

#### *A2.1.2.3 Using a Vickrey Auction-Redistribution with Imperfect Information*

Experience tells us that when we move to the practice of environmental policy, the perfect information assumption, alternatively assuming truthful reporting, must be discarded. We instead prefer mechanisms that compel truthful reporting by making it in each relevant participant's interest to do so and which still results in an efficient solution. While there are several auction types, the second price sealed bid (SPSB) Vickrey auction can be designed as a loss avoidance auction which generally arrives at the same solution as the preceding framework despite imperfect information. In this section, I explore how a Vickrey auction followed by a redistribution compares in terms of the three criteria of efficiency, feasibility, and equitability

Tailored to this application, in the Vickrey auction we expect each firm to bid $b_i = cs_i$, expect to pay $cs_{i+1}$ if they win, and the winner to be Firm 1. But with the loss of perfect information, I must assess whether firms are likely to have accurate estimates of competitor's bids to bid accordingly, or whether they must resort to another strategy. This difference is vital because we have observed market structures in the perfect information version where Firm 1 is better off bidding to lose and receive a redistribution share instead. As firms in an industry requiring regulation of this sort are potentially large and established, I find it unlikely that the largest – those ranking first or second in an industry – would be unaware of their market share and that of their largest competitors. With cost, too, I believe it misguided to assume leading firms do not have a decent idea of their closest competitor's costs, and at a minimum, would conjecture their competitor's costs are not unlike their own. When we believe firms to be confident in their estimates of their competitor's position, the possibility of Firm 1 bidding to lose remains. When instead there is substantial uncertainty, we might expect operators to always bid $b_i = cs_i$ – the standard Vickrey result – and an element of certainty in how firms act emerges from market uncertainty.

In feasibility, we should not expect type I to change when firms expect the efficient solution to emerge from the Vickrey auction, as it is also the type I most feasible solution.[129] I again suppose that small firms are informed on large firm market shares, and perhaps costs, as benchmarks to compare their operations against. Then, their expectation over the redistribution share to expect is similar to the perfect information case. When we instead cannot suppose they have access to useful information, we should expect type II feasibility to decrease because firms cannot calculate an expected redistribution to compare to their cost of conversion. Risk averse firms would then rather not risk converting.

In equitability, we observe that firm expectations on $\alpha_i$ depend on the certainty of knowledge about the leading two or three firms in the market. This expectation, however, is not relevant to the bid decision due to the separability of bidding and redistribution mechanisms. However, the state must have accurate information on each firm's $s_i$ and $c_i$, or $c$ when constant, to distribute fairly. Without this information, some firms have an incentive to overstate their $s_i$ and $c_i$ measures to increase the compensation they would receive. Compounding the matter, observe that $c_i s_i$ is suggested by a firm's bid as we expect their bid to be $b_i = c_i s_i$. Then, many firms – any not expecting to inadvertently win – have an incentive to overbid if the regulator knows all bid amounts. This is at least

[129] Noting too the prior discussion of increased certainty in bids when firms are uncertain about competitors.

applicable to firms not expecting to win – a firm expecting to win can bid truthfully because they do not expect a distribution. The incentive to overbid results in uncertainty about the bidding behaviour of firms that are nearly, but not quite, the market leader, and the strategy-proof property of standard SPSB Vickrey auctions with a redistribution is in jeopardy, in contrast to Cavallo (2006). However, a simple solution is to ensure that the agent in charge of redistributions is unaware of the nonwinning bid amounts and that the bidders are aware of this naivety – in essence the auctioneer 'burns' their information after determining the auction winner. This correction does not address the private information problem when determining pay-outs, but this problem is hardly unique to this setting nor should we expect it to be solved here. As in Bailey (1997), Cavallo (2006), Green and Laffont (1979), and Porter, Shoham, and Tennenholtz (2004), the worst case is that the regulator falls back on distribution weights $\lambda = \frac{1}{N}$. But surely even weighting by publicly observable market shares is preferred by larger – and perhaps more influential – firms.

The SPSB Vickrey auction in a setting of uncertainty appears as though it will perform comparable to the perfect information auction when the regulator takes certain precautions. I do observe, however, that the quality of information available to bidders about the largest firms in their industry impacts bidding behaviour. Uncertainty increases the likelihood that Firm 1 bids their value, but also increases opposition among smaller risk-averse firms. I also observe that concerns about equitability and conversion costs can lead to strategic bidding and loss of efficiency, but this is addressed with the compartmentalisation of bidding and distribution processes. The cost is that we forgo knowledge that would, in a nonstrategic setting, allow us to redistribute fairly (according to $s_i$'s). We could explore alternatives to Vickrey, but at this stage I do not find sufficient value in doing so: In the Vickrey auction, we have an instrument with well-known properties which is efficient and no less feasible or equitable than the perfect information case. However, the proposed application remains unproven, and I defer judgement until it is seen in practice.

## A3.1 Data Preparation

Sulphur content is reported as per cent by weight by the EIA which lists a one-per cent sulphur content as 1.00, and mandates reporting the average sulphur content of the purchased load or the monthly consumption, as applicable, to the nearest 0.01-per cent or 1/100th of a per cent. Ash content follows the same format but to a required reporting accuracy of 0.1-per cent or 1/10th of a per cent. The conversion to per-mmBTU for a fuel order or consumption data using sulphur as the example simplifies down conveniently. Note from

(55) $$(sulphur\ percent\ of\ a\ ton\ of\ coal) * 10^{-2} = sulphur \frac{tons}{ton\ of\ coal}$$

then,

(56) $$\left(sulphur \frac{tons}{ton\ of\ coal}\right) * \left(20 * 10^{2} \frac{pounds}{ton}\right) = sulphur \frac{pounds}{ton\ of\ coal}$$

Dividing by the mmBTU per ton of the fuel purchased or consumption data,

(57) $$\frac{sulphur \frac{pounds}{ton\ of\ coal}}{\frac{mmBTU}{ton\ of\ coal}} = sulphur \frac{pounds}{mmBTU}$$

Putting this together in one step using the sulphur per cent and mmBTU per ton reported by the EIA,

(58) $$\frac{(sulphur\ percent) * (20)}{(mmBTU)} = sulphur \frac{pounds}{mmBTU}$$

I also note that the data contains several forms of petroleum-based fuel oils. I include a conversion for these sources as a matter of general interest because there is some nuance involved that is sometimes overlooked. Each of these has a different weight per unit of measure. The conversion from per cent by weight of a barrel to pounds per mmBTU follows:

(59) $$(sulphur\ percent\ of\ a\ barrel\ of\ fuel) * 10^{-2} = sulphur \frac{barrels}{barrel\ of\ fuel}$$

then,

(60) $$\left(sulphur\,\frac{barrels}{barrel\;of\;fuel}\right) * \left(42\,\frac{gallons}{barrel}\right) = sulphur\,\frac{gallons}{barrel\;of\;fuel}$$

The weight of fuel oil per gallon differs by the density, so each fuel oil grade has a different pound per gallon conversion factor. For instance, the weight of residual fuel oil is 7.88 pounds per gallon on average. Using this as the conversion factor would be a mistake, however. What needs to be converted is the gallons of sulphur per barrel of fuel oil into pounds of sulphur per barrel. Using the fuel oil conversion, i.e. 7.88 pounds/gallon implicitly assumes the sulphur-to-fuel oil weight ratio is the same as the volume ratio, which it is not – the sulphur constituent is substantially heavier than fuel oil on average. I instead construct the correct conversion factor from sulphur's weight per volume measure of 2.07 grams/millilitre at room temperature,

(61) $$\left(2.07\,\frac{grams}{milliliter}\right) * \frac{\left(1000\frac{milliliters}{liter}\right)*\left(3.78541176\frac{liters}{gallon}\right)}{\left(28.3495231\frac{grams}{ounce}\right)*\left(16\frac{ounces}{pound}\right)} \approx 17.2750\,\frac{pounds}{gallon}$$

Note the 17.275 pounds/gallon conversion factor differs from the 15.2 pounds/gallon factor found in some industrial sources. The smaller published number is based on a density of 1.819 grams/millilitre – the density of sulphur at its boiling point of around 240 degrees Fahrenheit. As power plants are probably reluctant to accept fuel oil at the power plant gate at such a temperature, 17.275 pounds/gallon is more appropriate. Using it in the sulphur per cent-to-pounds conversion,

(62) $$\left(sulphur\,\frac{gallons}{barrel\;of\;fuel}\right) * \left(17.275\,\frac{pounds}{gallon}\right) = sulphur\,\frac{pounds}{barrel\;of\;fuel}$$

Dividing by the mmBTU per ton for the fuel purchases or consumption data,

(63) $$\frac{sulphur\frac{pounds}{barrels\;of\;fuel}}{\frac{mmBTU}{barrels\;of\;fuel}} = sulphur\,\frac{pounds}{mmBTU}$$

Putting it all together to use sulphur per cent and mmBTU per barrel reported,

(64) $$\frac{(sulphur\;percent)*(7.2555)}{(mmBTU)} = sulphur\,\frac{pounds}{mmBTU}$$

Natural gas does not contain sulphur or ash content, and so no conversion is necessary.

Finally, I reconcile the price of emissions in terms of permit costs with the model's data format. Using the efficiency of the scrubbing technology, conversion factors for sulphur-to-sulphur dioxide ($S \rightarrow SO_2$), weight, and price of permits, I note that

$$(65) \qquad P_{emissions} = \frac{\left(Sulphur \frac{pounds}{mmBTU}\right) * (1 - efficiency) * (S \rightarrow SO_2) * \left(P_{permits}/ton\right)}{\left(2 * 10^3 \frac{pounds}{ton}\right)}$$

results in permit costs required per mmBTU, which can also be transformed into the average price per ton of coal by multiplying it by the mean mmBTU heat content per short ton. The sulphur-to-sulphur dioxide conversion factor, derived from their molecular weights, is approximately 1.9979. In practice, permit prices per ton of sulphur dioxide emissions have entered this research rather roughly. However, I provide for context the annual $SO_2$ allowance auction weighted average spot prices from winning bids from the U.S. Environmental Protection Agency (EPA) in Table 26. Allowance prices have plummeted in recent years (see a discussion of this in Schmalensee and Stavins (2013)). According to the EPA, emissions are well below the emissions budget (goal) of the Acid Rain Program (ARP) and related schema, resulting in downward pressure on allowance prices. Regardless of the cause, these changes impact the budget share of an emitting dirty versus clean alternative.

TABLE 26— ANNUAL EPA $SO_2$ AUCTION AVERAGE SPOT PRICE

| | Spot price ($) | 7-year advance ($) |
|---|---|---|
| 2005 | 702.51 | 297.49 |
| 2006 | 883.1 | 275.13 |
| 2007 | 444.39 | 193.35 |
| 2008 | 389.91 | 136.14 |
| 2009 | 69.74 | 6.65 |
| 2010 | 37.71 | 2.07 |
| 2011 | 2.81 | 0.17 |
| 2012 | 0.67 | 0.13 |
| 2013 | 0.28 | 0.04 |
| 2014 | 0.45 | 0.04 |
| 2015 | 0.11 | 0.03 |
| 2016 | 0.06 | 0.02 |

*Notes:* Spot price and 7-year advance price of winning bids (weighted average).

*Source:* Compiled from U.S. EPA (2015) annual auction data.

### A3.2 Glossary of Energy Sector Terminology and U.S. Organizational Chart

The U.S. energy sector includes different organisational levels of energy producers, and a brief taxonomy is in order. It should be just sufficient for the use at hand.

*NERC region*: The North American Electric Reliability Corporation (NERC) system has existed in some form since 1968. Beginning as a voluntary association, the modern NERC has the authority to enforce energy sector standards impacting North America. The NERC divides the U.S. energy sector into eight regional, regulatory entities.

*Firm*: This organisational level may operate one or several separate generating locations. The data I use is at a finer resolution, and so the firm is mentioned little in the specific application but likely has an important role in the underling power plant strategic decision making.

*Power plant:* Synonymously a facility or power station. A power plant is a single location receiving fuel and then producing electricity. Production itself may occur in one or more subunits within the power plant operating on the same or different fuels and using some of the same or separate infrastructure. The binding connection is geographic as well as managerial – the power plant supplies generation to the same outgoing transmission lines and intra-power plant decision making comes from a single authority. Much of the data is at this level or finer. Fuel purchasing decisions are recorded at the power plant level.

*Boiler:* This is the smallest relevant division involved in fuel consumption and is synonymous for my purposes with a furnace. Others have referred to this scale as the generator level, but the generator is what creates electrical output, not what consumes fuels. The boiler is the scale where the transformation of fuels into heat energy occurs and thus is the finest scale of fuel consumption data possible. These systems may be single-fuel or mixed, depending on design and operation. A boiler feeds steam into a turbine driving a generator or similar system, and waste gases pass through any abatement system on their way to the waste stack. These connections need not be one-to-one but often are.

This organisational description is presented in Figure 41 where the specific NERC abbreviations are not important to the task at hand.

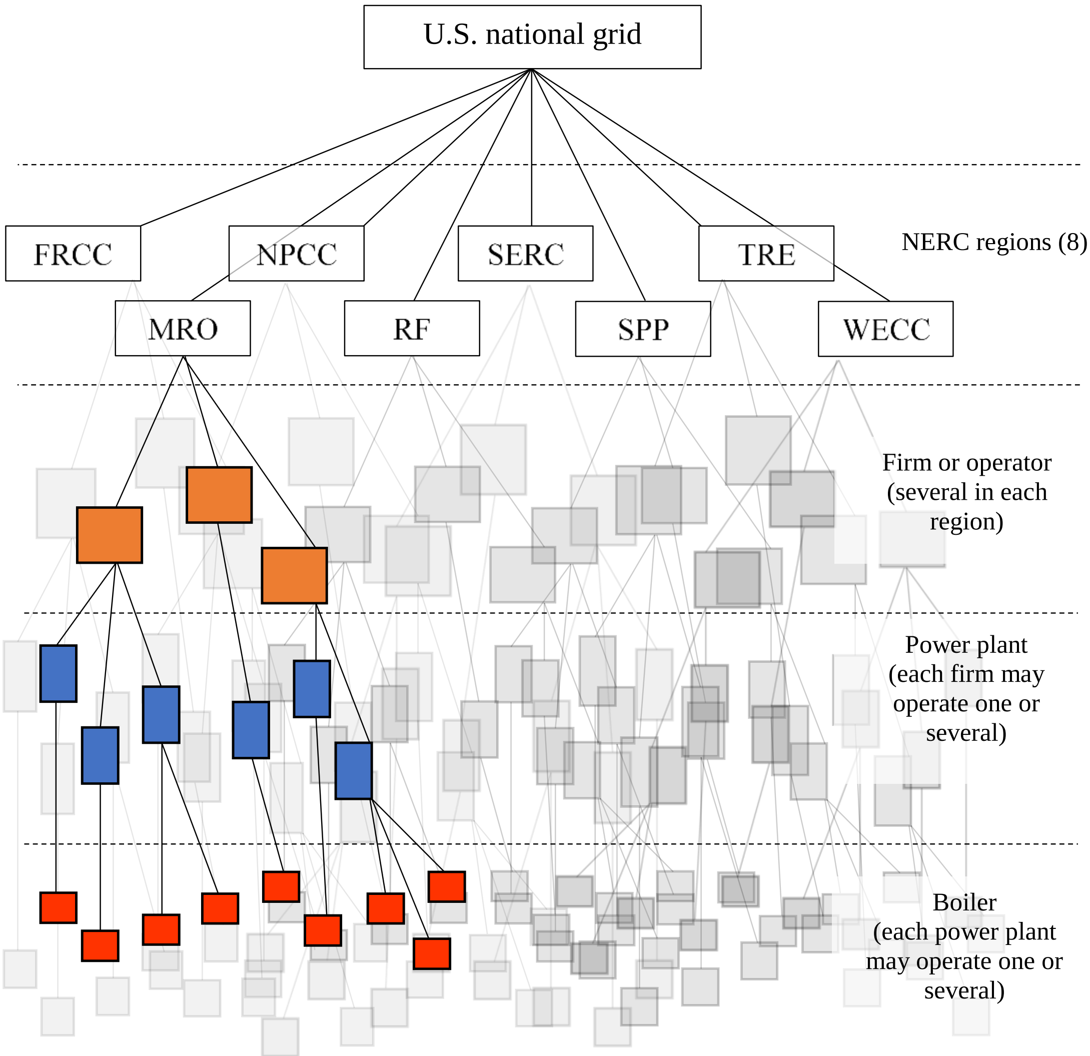


FIGURE 41 HIERARCHY OF THE U.S. ENERGY SECTOR.

*Notes*: One region, Midwest Reliability Organization (MRO), is arbitrarily chosen to highlight.

### A3.3 Returns to Scale in Electricity Generation

In this section I caution on the claim that $\beta_D = \beta_C = 1$, that electricity generation at power plants exhibits constant returns to scale as in, for instance, Christensen and Greene (1976) or Bernstein and Parmeter (2019). I present an argument on why this may occur in the data but does not inform on the underlying production relationship.

To summarise, power plants can adjust their output by adjusting the fraction of time they or their subunits are active in the period of measure instead of by increasing their instantaneous fuel consumption. Power plants, representative of a whole class of industry, rely on some exogenous and expensive technology where $\ell$ suggests its operational status which fixes the level of all other inputs within a period, $t$. This technology is an engineering constraint resulting in some operating range or specification and operating outside of it "voids the warranty". As this $\ell$ is fixed, and in turn fixes all other inputs by engineering relation in *t*, the firm can only choose whether to operate or not. In the case at hand, power plants either produce $E_t = f(D, C, \ell = 1) = D + C$, shut down $E_t = f(D, C, \ell = 0) = 0$, or perhaps only operate some subunit generators if so organised. At power plants, there may be many such constraints – boilers, generators, feed systems, cooling systems, and emissions controls. A second factor is that in the data, a period *T* may be composed of many *t*. An example is monthly data when the firm makes hourly production decisions. In a simpler case, consider a period of measure where *T=2t*. Then measured output in *T* is either 0, $E = f_t(D, C)$, or $2E = 2f_t(D, C)$ for 0, 1, or 2 periods *t* where the power plant is in operation. Within *T* then, the power plant will exhibit constant returns to scale as a doubling of inputs doubles output. However, this does not inform on the relationships between *D*, *C*, and *E* – only the share of *T* when the power plant is active changes. The aggregate in *T* may exhibit constant, decreasing, or increasing returns to scale or be entirely constant – the point is we cannot tell from most data. Adding yet another dimension, even if a power plant operates continuously in *T*, it may be composed of several sub-plant boilers only operating part-time and the issue stands. Even if a counterfactual were to exist that allows comparisons under two separate scale relationships, this might not be informative either. Consider that $E_{CRTS} = aE_{DRTS}$ for a given $D$ and $C$. But the difference in production, *a*, may be inseparable from other matters of power plant efficiency. Whenever any of these complicating dimensions exist, we cannot then estimate returns to scale in the sort of data we generally have available and interpret the result as informative on the underlying production technology. We might go about suggesting that a power plant operator reserves sufficient emergency capacity – say additional generators – such that they can behave in such a way that is equivalent in result to a CRTS or other relationship.

### A3.4 Exploring the Interior Result of the Input Selection Model

The behaviour of the single interior solution of case (ii) is explored. Assume $P_C > P_D$ remains in effect even with compliance costs added into $P_D$ and that output *E* is determined first, or fuel purchased based on an expectation. The baseline production framework without regulation is

(66) $$\pi = P_E E - P_D D - P_C C$$

$$subject\ to\ E = f(D, C) = f(D) + f(C)$$

As $E$ is set, and with an emissions standard, the firm selects $D$ and $C$ to minimise cost,

(67)
$$\min_{D,C,\lambda} P_D D + P_C C$$
$$subject\ to\ E = f(D,C) = f(D) + f(C)$$
$$and\ \mathcal{E} \geq f(\alpha, D) = \alpha D$$

The simplified Lagrangian and the Karush-Kuhn-Tucker (KKT) conditions are then

(68)
$$\mathcal{L}_{D,C,\lambda_1,\lambda_2} = P_D D + P_C C + \lambda_1[E - f(D,C)] + \lambda_2[\mathcal{E} - \alpha D]$$

(69)
$$\mathcal{L}_D = P_D - \lambda f_D = 0$$

(70)
$$\mathcal{L}_C = P_C - \lambda f_C = 0$$

(71)
$$\lambda_1 \mathcal{L}_{\lambda_1} = \lambda_1[E - f(D,C)] = 0$$

(72)
$$\lambda_2 \mathcal{L}_{\lambda_2} = \lambda_2[\mathcal{E} - \alpha D] = 0$$

where $P_D, P_C, E, \mathcal{E}, \alpha \geq 0$, $D, C, \lambda_1, \lambda_2 \geq 0$, and $f_D$ is the first derivative of the production function with respect to the dirty input: $\frac{\partial f(D,C)}{\partial D} = f_D(D,C) = f_D(D)$ and similarly so for $f_C$. Second derivatives then follow as $f_{DD}$ and $f_{CC}$. The variables $D, C, \lambda_1, \lambda_2$ are endogenous, and $P_D, P_C, E, \mathcal{E}, \alpha$ exogenous. The resulting Jacobian is

(73)
$$\begin{array}{c c} & \begin{array}{cccc} \underline{C} & \underline{D} & \underline{\lambda_1} & \underline{\lambda_2} \end{array} \\ \begin{array}{c} \mathcal{L}_D \\ \mathcal{L}_C \\ \mathcal{L}_{\lambda_1} \\ \mathcal{L}_{\lambda_2} \end{array} & \begin{bmatrix} 0 & -\lambda_1 f_{DD} & -f_D & -\alpha \\ -\lambda f_{CC} & 0 & -f_C & 0 \\ -f_C & -f_D & 0 & 0 \\ 0 & -\alpha & 0 & 0 \end{bmatrix} = |J| \end{array}$$

with transposed vectors of exogenous variables,

$$\begin{array}{ccccc} \underline{P_D} & \underline{P_C} & \underline{E} & \underline{\mathcal{E}} & \underline{\alpha} \\ \begin{bmatrix} -1 \\ 0 \\ 0 \\ 0 \end{bmatrix} & \begin{bmatrix} 0 \\ -1 \\ 0 \\ 0 \end{bmatrix} & \begin{bmatrix} 0 \\ 0 \\ -1 \\ 0 \end{bmatrix} & \begin{bmatrix} 0 \\ 0 \\ 0 \\ -1 \end{bmatrix} & \begin{bmatrix} \lambda_2 \\ 0 \\ 0 \\ D \end{bmatrix} \end{array}$$

$\frac{\partial \mathcal{L}}{\partial E} = \lambda_1$ is the change in the optimal value of the cost minimisation objective function with a change in $E$ – assumed positive, and $\frac{\partial \mathcal{L}}{\partial \varepsilon} = \lambda_2$ is the change with a relaxation of the emissions cap. The signs of values and derivatives for the rest of the endogenous variables are assumed as

$$\frac{D, C}{(+)} \quad \frac{f_D, f_C}{(+)} \quad \frac{f_{DD}, f_{CC}}{(-)\ or\ 0}$$

Under the assumption of the $E = D + C$ form for the production function, $f_{DD} = f_{CC} = 0$, and is negative otherwise. Then by the implicit function theorem,

(74) $$\frac{\partial D}{\partial \varepsilon} = \frac{\begin{bmatrix} 0 & 0 & -f_D & -\alpha \\ -\lambda f_{CC} & 0 & -f_C & 0 \\ -f_C & 0 & 0 & 0 \\ 0 & -1 & 0 & 0 \end{bmatrix}}{|J|} = \frac{1}{\alpha}$$

(75) $$\frac{\partial D}{\partial \alpha} = \frac{\begin{bmatrix} 0 & \lambda_2 & -f_D & -\alpha \\ -\lambda f_{CC} & 0 & -f_C & 0 \\ -f_C & 0 & 0 & 0 \\ 0 & D & 0 & 0 \end{bmatrix}}{|J|} = \frac{-D}{\alpha}$$

and subsequently

(76) $$\frac{\partial C}{\partial \varepsilon} = \frac{\begin{bmatrix} 0 & -\lambda_1 f_{DD} & -f_D & -\alpha \\ 0 & 0 & -f_C & 0 \\ 0 & -f_D & 0 & 0 \\ -1 & -\alpha & 0 & 0 \end{bmatrix}}{|J|} = \frac{-1}{\alpha}\frac{f_D}{f_C}$$

(77) $$\frac{\partial C}{\partial \alpha} = \frac{\begin{bmatrix} \lambda_2 & -\lambda_1 f_{DD} & -f_D & -\alpha \\ 0 & 0 & -f_C & 0 \\ 0 & -f_D & 0 & 0 \\ D & -\alpha & 0 & 0 \end{bmatrix}}{|J|} = \frac{D}{\alpha}\frac{f_D}{f_C}$$

When the condition $P_C > P_D$ holds,

(78) $$\frac{\partial D}{\partial P_D} = \frac{\partial D}{\partial P_C} = \frac{\partial C}{\partial P_D} = \frac{\partial C}{\partial P_C} = 0$$

and if $E = \bar{E}$ is relaxed but the emissions cap remains,

(79) $$\frac{\partial D}{\partial E} = 0 \; and \; \frac{\partial C}{\partial E} = \frac{1}{f_C}$$

The mathematical formulation with the emissions cap in place behaves as suggested in Figure 25 and Figure 26 of the main text. Related to $\frac{\partial D}{\partial \alpha} = \frac{-D}{\alpha}$, we should also expect that an increase in any abatement technology efficiency that is subsequently installed will increase demand for *D*.

### A3.5 Coefficients from Linear Logit Estimation

I present the coefficient estimates from the dynamic linear logit model group result for the year 2010 and later – the modern market. These are used to calculate the elasticities in the main body.

TABLE 27— COEFFICIENTS FROM FITTING GROUPED DYNAMIC LINEAR LOGIT SYSTEM

| | Coefficient | Standard error | | Coefficient | Standard error |
|---|---|---|---|---|---|
| /a1 | -0.001 | . | /a2 | 0.006 | . |
| /phi12 | -0.304 | (0.129) | /g2 | 0.001 | (0.005) |
| /phi13 | -0.647 | (0.082) | /d2 | -2.493 | (0.052) |
| /phi23 | -0.271 | (0.080) | /d22 | -0.013 | (0.046) |
| /g1 | -0.004 | (0.005) | /nerc21 | -2.238 | (0.090) |
| /gamma1 | 0.009 | (0.011) | /nerc22 | 2.353 | (0.062) |
| /gamma2 | 0.004 | (0.007) | /nerc23 | -2.299 | (0.085) |
| /d1 | -2.222 | (0.061) | /nerc24 | 1.206 | (0.069) |
| /d11 | 0.058 | (0.063) | /nerc25 | 0.117 | (0.064) |
| /nerc11 | -1.060 | (0.064) | /nerc26 | 0.635 | (0.054) |
| /nerc12 | 1.442 | (0.072) | /nerc27 | 0.311 | (0.056) |
| /nerc13 | -1.661 | (0.077) | /yy11 | 0.037 | (0.048) |
| /nerc14 | 2.561 | (0.066) | /yy12 | -0.061 | (0.057) |
| /nerc15 | 0.907 | (0.064) | /yy13 | -0.175 | (0.051) |
| /nerc16 | -1.051 | (0.063) | /yy14 | -0.207 | (0.047) |
| /nerc17 | -1.390 | (0.062) | /yy15 | -0.274 | (0.058) |
| /y11 | -0.026 | (0.049) | /m22 | -0.010 | (0.065) |
| /y12 | -0.222 | (0.058) | /m23 | 0.067 | (0.066) |
| /y13 | -0.208 | (0.052) | /m24 | 0.075 | (0.067) |
| /y14 | -0.333 | (0.049) | /m25 | -0.023 | (0.067) |
| /y15 | -0.345 | (0.060) | /m26 | -0.204 | (0.066) |
| /m12 | 0.028 | (0.068) | /m27 | -0.363 | (0.067) |
| /m13 | 0.083 | (0.069) | /m28 | -0.309 | (0.066) |
| /m14 | 0.139 | (0.070) | /m29 | -0.002 | (0.067) |
| /m15 | -0.013 | (0.070) | /m210 | 0.090 | (0.067) |
| /m16 | -0.238 | (0.069) | /m211 | 0.159 | (0.067) |
| /m17 | -0.428 | (0.070) | /m212 | -0.015 | (0.066) |
| /m18 | -0.357 | (0.070) | | | |
| /m19 | -0.050 | (0.070) | | | |
| /m110 | -0.025 | (0.070) | | | |
| /m111 | 0.045 | (0.070) | | | |
| /m112 | -0.043 | (0.069) | | | |
| R-squared | model 1 (lnS13) | 0.9456 | | | |
| R-squared | model 2 (lnS23) | 0.8997 | | | |

*Notes:* 576 observations. Estimated by fitting a system of equations by iterative feasible generalized nonlinear least squares (FGNLS).

*Source:* Author calculations based on data from EIA (2016a, 2016c).

### A3.6 Discussion of Why Domestic Policy May Not Impact Trade Partners

While the focus of this paper is the domestic energy sector, it is certainly worth discussing the potential for transmission of domestic abatement policy effects onto the international market. A concept explored in the main body is that targeted and limited intervention in the price of abatement technologies can lead to cleaner consumption of dirtier fuels domestically. Here, I explore the extension that dirtier fuels may be imported – and cleaner ones exported. This concept of “negative leakage” – in this case to leverage technology to increase domestic sulphur demand and draw polluting fuels to their cleanest use – has received some attention. More generally the result is a shifting of higher polluting production or consumption back into a more regulated market. Winchester and Rausch (2013) present a model where emissions regulations in one region may decrease emissions elsewhere. However, in their accompanying computable general equilibrium (CGE) model, fossil fuel supply elasticities would have to be impractically large – approaching infinity – for negative leakage to actually occur. Yet Baylis, Fullerton, and Karney (2014) develop an alternative framework and CGE model where they establish the possibility of negative leakage. Such a framework is then explored further in Elliott and Fullerton (2014). Others, such as Fowlie, Reguant, and Ryan (2016), identify negative leakage effects in real domestic policy, but in a downstream market (cement production). It is perhaps also important to note that Baylis, Fullerton, and Karney (2013) find that negative leakage situations always lead to net welfare losses – from decreasing consumption in the second economy as well as the first.

In this appendix I outline an example partial equilibrium model which suggests that one of the second-order effects of abatement technology adoption in a regulated economy is the reduction in emissions in an unregulated trade partner through price. I then discuss why such negative leakage is unlikely and link to existing literature on the matter. I present the discussion in terms of a set of nations in agreement deciding how to influence an obstinate trade partner and note that the non-adopter’s response must be voluntary. Consider a two-period, sequential decision where the international “Consensus” leads. As the majority demander of inputs and able to coordinate through the accord, Consensus primarily determines the price in the open market. The authority that rejects shared regulation, “Defector”, is small and their influence on open market price is minimal – simplified to zero.[130, 131] Without these comparative size assumptions, an international environmental accord is unlikely to be effective anyway. Whether the motive of Defector is to free ride (discussed at length in Barrett, 2003), or the policy would result in a net loss for Defector, is outside the model. At a minimum, assume Defector's industry benefits at least initially from non-adoption, else they would advocate joining the accord.[132] Both Defector and Consensus have an objective of providing a generally homogenous good, such as consumer energy, given their regulatory environment or possible lack thereof in Defector.[133]

The setup of a small defector versus global consensus raises the critique of whether Defector's actions are worth addressing at all. I suggest there are many cases where such a defection is still relevant. In the case of global greenhouse gas emissions, atmospheric concentrations have reached a level where a net decrease is required to maintain current temperatures (Pfeiffer, Millar, Hepburn, and Beinhocker, 2016). That is, all additional sources of greenhouse gas emissions contribute to the crisis. There are many other pollutants where even small doses are harmful, e.g. dioxins that increase cancer risk and that of a variety of other ailments (Sany et al., 2015), certain CFC's resulting in ozone layer depletion (Barrett, 2003; Lickley et al. 2020), and exposure to fissile waste (Pearce, 2019). As an example, consider a nation intent on nuclear energy or arms production, but without sufficient regulatory empowerment to safeguard operations or store spent fuels safely. Even a small risk of radiation mishap can be of a regional, if not global, concern, e.g. the Windscale nuclear disaster of 1957 contaminating much of Northern Europe (Nelson, Kitchen, and Maryon, 2006; Pearce, 2019).

When cases fitting the outlined scenario occur, the efforts of Consensus would be strengthened by policy tools that compel Defector to behave as though they abide by the Consensus' international agreement. Another way to view this is to have policy tools that make the returns to defection insufficient. However, complex international relationships make traditional policy responses undesirable. What if Defector controls a vital resource that they can restrict in reply? Examples include total control of non-Soviet chromium,

[130] That is, $\frac{\partial P_i}{\partial Q_i^{def}} = o$ for input i. As $o \to 0$, $P_i\left(Q_i^{cons}, Q_i^{def}\right) \to P_i(Q_i^{cons})$. This scenario is representative of an economy which has never invested in clean energy, and, due to a lack of relevant natural resources, is wholly dependent on international markets for energy, which is not implausible. I also impose that Defector is sufficiently near Consensus-controlled sources that Defector does not face prohibitively greater transportation costs.

[131] Espínola-Arredondo and Muñoz-García, (2011) note that under certain conditions a defector might invest in clean technology anyway – that defection serves other geopolitical purposes such as signalling.

[132] Additionally, if the optimal selection for Defector were clean fuels, there would not be opposition to the accord save for the prior footnote. Presumably, national policymakers, like many of us, are unlikely to turn down a free lunch.

[133] Enforcement and compliance are assumed perfect in Consensus – a sufficiently empowered regulator is needed for the policy to be effective anyway.

manganese, platinum, and vanadium supplies by apartheid South Africa (Thomson, 2008), a variety of rare earth minerals currently almost exclusively sourced in China (Tse, 2011; NBR, 2019), and threats against free movement through the Strait of Hormuz by Iran (Talmadge, 2008; Ratner, 2018). A less direct, less confrontational approach may be in order when seeking to influence regimes. Choosing which technologies receive research and development support, for instance, is hardly controversial in comparison to trade tariffs, export subsidies, or threats of retaliation.[134] To develop an expectation of whether such policies will be useful, we require a model that transforms domestic, uncoordinated site-level behaviour into international effects.

To arrive at the international effects first requires aggregating the demand of a set of perfect substitute power plants into market demand that is at least downward sloping in price. This aggregation results from differences in the price at which firms switch between clean and dirty inputs due to differences in transportation, mining, and transaction costs at power plants and sources. Of the two abatement variable costs I have discussed, I use efficiency in this analysis. At Consensus power plants, let $\frac{\partial Q_D}{\partial eff} > 0 \; and \; \frac{\partial Q_C}{\partial eff} < 0$ including from zero to some efficiency (abatement technology adoption).[135] Let aggregation result in a same sign change in market demand. Then, from individual firm increases in abatement technology efficiency, market demand for dirty (clean) inputs increases (decreases) and the dirty fuel share of the market increases.

### *A3.6.1 Partial Equilibrium Effects of Abatement Technology in Consensus*

I use market demand notation, $\mathcal{D}$, and supply, $\mathcal{S}$, for dirty, $_D$ (sub-D), and clean, $_C$ (sub-C), inputs. From the preceding sections, emissions cap, $\bar{\mathcal{E}}$, and abatement efficiency, $eff$ influence the dirty input share, $\mathscr{s}_D$. Recall efficiency enters the regulated market participant's decision as $\delta = 1 - eff$ which here I repurpose as a mean, resulting in effective emissions cap $\bar{D} = \frac{\bar{\mathcal{E}}}{\delta\alpha}$. For simplicity, I work from changes in efficiency. Prices of dirty and clean inputs remain $P_D$ and $P_C$, and I denote excess demand by $\Theta$. Consensus market in equilibrium (implicitly including exports) is described by the system

[134] Cirone and Urpelainen (2013) suggest that trade sanctions may even increase free riding as they reduce the cost of further unilateral actions.

[135] The capacity of the majority to impact the market requires that consumers are not rigidly contracted – that demand and supply are not perfectly inelastic, and that Consensus can exert market power through international policy coordination.

(80) $$\mathcal{D}_D - \mathcal{S}_D = \Theta_D(\mathcal{D}_D, \mathcal{S}_D) = 0$$

(81) $$\mathcal{D}_C - \mathcal{S}_C = \Theta_C(\mathcal{D}_C, \mathcal{S}_C) = 0$$

(82) $$\mathcal{D}_D(\mathscr{s}_D, P_D),\ \frac{\partial \mathcal{D}_D}{\partial P_D} \leq 0\ and\ \frac{\partial \mathcal{D}_D}{\partial \mathscr{s}_D} > 0$$

(83) $$\mathscr{s}_D(\bar{\mathcal{E}}, eff),\ \frac{\partial \mathscr{s}_D}{\partial \bar{\mathcal{E}}} < 0\ and\ \frac{\partial \mathscr{s}_D}{\partial eff} > 0$$

(84) $$\mathcal{S}_D(P_D),\ \frac{\partial \mathcal{S}_D}{\partial P_D} > 0$$

(85) $$\mathcal{D}_C\big((1 - \mathscr{s}_D), P_C\big),\ \frac{\partial \mathcal{D}_C}{\partial P_C} \leq 0\ and\ \frac{\partial \mathcal{D}_C}{\partial \mathscr{s}_D} < 0$$

(86) $$\mathcal{S}_C(P_C),\ \frac{\partial \mathcal{S}_C}{\partial P_C} > 0$$

From the first condition, $\mathcal{D}_D - \mathcal{S}_D = \Theta_D(\mathcal{D}_D, \mathcal{S}_D) = 0$, the sign of the partial derivative $\frac{\partial \mathcal{D}_D}{\partial \mathscr{s}_D} > 0$ implies that an increase in supply, $\mathcal{S}_D$, occurs to maintain equilibrium. Rewriting excess demand in terms of the relevant supply and demand primitives, $\Theta_D\big(\mathscr{s}_D, P_D(\mathscr{s}_D)\big)$ where the implicit effect of dirty input share $\mathscr{s}_D$ on $P_D$ is explicitly acknowledged. The effect in market equilibrium is then $\frac{\partial P_D}{\partial \mathscr{s}_D} > 0$. Then, from the assumption of aggregation from power plant level activity, $\frac{\partial \mathscr{s}_D}{\partial eff} > 0$, and thus $\frac{\partial P_D}{\partial \mathscr{s}_D}\frac{\partial \mathscr{s}_D}{\partial eff} > 0$. Abbreviating, $\frac{\partial P_D}{\partial eff} > 0$ and similarly the second market-clearing condition finds $\frac{\partial P_C}{\partial eff} < 0$.[136]

### *A3.6.2 Resulting Impact on Defector*

Defector is unable to influence price and demands the cheapest and generally dirty input when pursuing cost minimisation subject to world prices. I abstract from geographical, transportation cost, and market barriers – Defector is close to Consensus and on generally friendly terms. I assume that Defector is responsive to price and so their demand curve must be downward sloping. Through the Consensus influence on price, Defector's quantity demanded must then decrease when Consensus adopts abatement technologies because the price of the dirty fuel increases. Being otherwise indifferent between the dirty and more expensive clean alternative places a limit on the effect possible, but one which

[136] Defining a more complex market where $\mathcal{D}_D$ has meaningful cross price elasticities such that $\mathcal{D}_D(\mathscr{s}_D, P_D, P_C)$ and $\mathcal{D}_C\big((1 - \mathscr{s}_D), P_C, P_D\big)$ complicates the result without adding value to this deliberately simple discussion.

accomplishes the Consensus goal as well. I use the notation for Defector, $^{def}$ (superscript-def) and suppress notation for Consensus on the factor primitives. As $\frac{\partial \mathcal{D}_D^{def}}{\partial P_D} < 0$ when $P_D < P_C$ (else equal to zero), and $\frac{\partial P_D}{\partial eff} > 0$, it follows that $\frac{\partial \mathcal{D}_D^{def}}{\partial P_D}\frac{\partial P_D}{\partial eff} < 0$, abbreviated $\frac{\partial \mathcal{D}_D^{def}}{\partial eff} < 0$.

As a specific functional form, I use constant elasticities of demand and supply. In Consensus, for dirt fuel share, $\mathcal{s}_D(\bar{\mathcal{E}}, eff) > 0$, efficiency multipliers satisfying $\zeta_D, \zeta_C > 0$ and $\xi_D, \xi_C > 0$, elasticities of demand $\epsilon_D, \epsilon_C < 0$, and elasticities of supply $\eta_D, \eta_C > 0$. Then, determining the demand for dirty and clean inputs jointly,[137]

$$\mathcal{D} = \mathcal{D}_D + \mathcal{D}_C = \mathcal{s}_D \zeta_D {P_D}^{\epsilon_D} + (1 - \mathcal{s}_D)\zeta_C P_C^{\epsilon_C}, \tag{87}$$

$$\mathcal{S} = \mathcal{S}_D + \mathcal{S}_C = \xi_D {P_D}^{\eta_D} + \xi_C {P_D}^{\eta_C} \tag{88}$$

resulting in separate market-clearing prices for a given level of $\mathcal{s}_D$,

$$P_D = \left[\mathcal{s}_D\left(\frac{\zeta_D}{\xi_D}\right)\right]^{\frac{1}{\eta_D - \epsilon_D}} \text{ and } P_C = \left[(1-\mathcal{s}_D)\left(\frac{\zeta_C}{\xi_C}\right)\right]^{\frac{1}{\eta_C - \epsilon_C}}. \tag{89}$$

For Defector with efficiency multiplier $\zeta^{def} > 0$ and elasticity of demand $\epsilon^{def} < 0$,

$$\mathcal{D}^{def} = \zeta^{def}[\min(P_D, P_C)]^{\epsilon^{def}}. \tag{90}$$

It follows that when $P_D < P_C$,

$$\frac{\partial \mathcal{D}^{def}}{\partial eff} = \zeta^{def}\left(\frac{\zeta_D}{\xi_D}\right)^{\frac{\epsilon^{def}}{\eta_D - \epsilon_D}}\left(\frac{\epsilon^{def}}{\eta_D - \epsilon_D}\right)\mathcal{s}_D^{\frac{\epsilon^{def}}{\eta_D - \epsilon_D} - 1}\frac{\partial \mathcal{s}_D}{\partial eff} < 0 \tag{91}$$

recalling that $\frac{\partial \mathcal{s}_D}{\partial eff} > 0$, and noting $\left(\frac{\epsilon_D^{def}}{\eta_D - \epsilon_D}\right)$ is negative and all other components of the derivative are positive. Notionally, Figure 42 partially illustrates when abatement efficiency improvements in Consensus result in reduced emissions in Defector. However, as $\eta_D$ is generally very elastic and $\epsilon_D$ and $\epsilon^{def}$ inelastic, we should expect that drastic action – large changes in efficiency in Consensus – would be required to affect meaningful interior solution changes in Defector. However, when $P_D$ is close to $P_C$, Consensus action may push Defector over a tipping point that results in Defector undertaking a clean energy transition

[137] We might also suppose $\zeta = \zeta_D = \zeta_C$ and $\xi = \xi_D = \xi_C$ suggesting indifference between fuels in the aggregate as well.

(see Farmer et al., 2019 and Beinhocker, Farmer, and Hepburn, 2018 on the importance of tipping points in pursuing climate goals). Reaching a tipping point is aided by the implication of the second market-clearing condition in Consensus, that $\frac{\partial P_C}{\partial eff} < 0$.

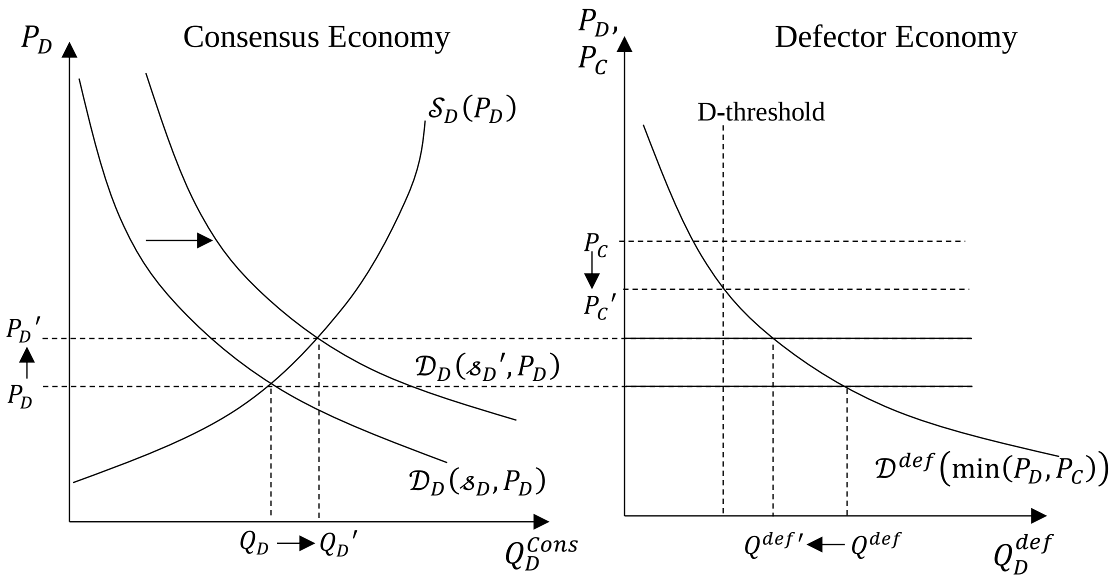


FIGURE 42 CONSENSUS AND DEFECTOR AGGREGATE SUPPLY AND DEMAND

*Notes:* Abatement adoption and efficiency improvements result in increased dirty fuel share, from $\mathscr{s}_D$ to $\mathscr{s}_D'$ in Consensus and subsequently an increase in the shared dirty fuel price from $P_D$ to $P_D'$. Defector then decreases emissions due to decreased quantity demanded, while Consensus emissions do not increase due to abatement technology adoption. However, due to relative supply and demand elasticities in Consensus and the demand elasticity in Defector, changes in Defector quantity demanded and emissions may be small. We can similarly illustrate a simultaneous decrease in $P_C$ to $P_C'$. These changes lower D-threshold, the tipping point where Defector instead chooses the clean alternative.

### *A3.6.3 Domestic Postscript: Lack of Transmission to International Markets*

The primary purpose of this appendix has been to suggest that, through a coordinated effort, a consensus of nations can compel an environmental accord defector to reduce emissions voluntarily. To make the argument, I have explored changes in domestic price resulting from technological changes that allow nations to consume dirty inputs to production cleanly. The model is tested on the U.S. domestic energy market where coal power plants use FGD systems that allow them to consume high sulphur coal without emitting the full sulphur content as sulphur dioxide. I now discuss why the U.S. domestic process has not led to the "export of cleanliness" abroad.

First, when we move to the international trade in energy, the effects of changes in U.S. abatement technology have been overwhelmed by greater market forces. The central theme of this research has been that domestic decisions about energy policy and other environmental and trade concerns can result in negative leakage. In doing so, I have assumed no radical changes in supply or technology impacting the clean alternative. But ceteris paribus has not been the case in the U.S. as natural gas supply has increased with the shale gas boom resulting in unprecedented price decreases. The result is that demand for every quality of coal has plummeted domestically, while U.S. natural gas has remained landlocked until very recently (EIA, 2020c; EIA, 2020d).[138] Figure 43 illustrates the replacement of coal-based electricity generation by natural gas starting in the mid-2000's. A result is that more U.S. coal of all qualities is available to sell abroad.

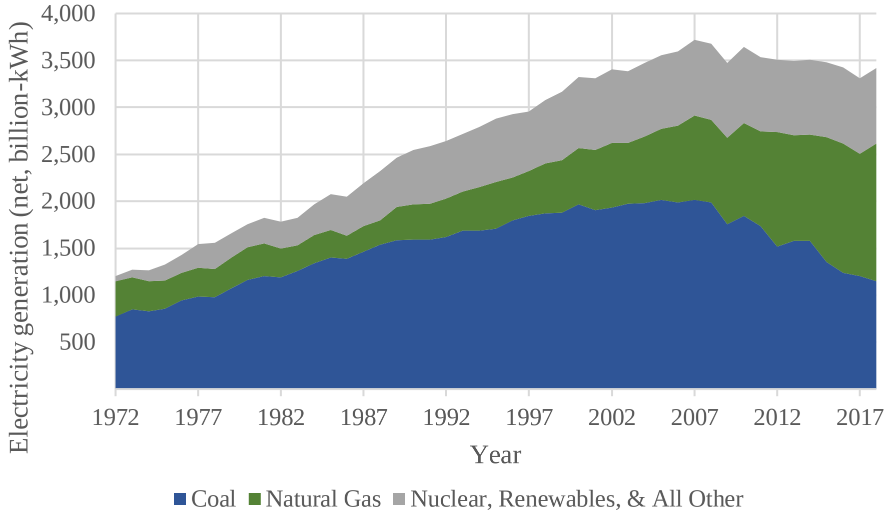


FIGURE 43 U.S. ELECTRICITY GENERATION BY FUEL TYPE

*Notes:* Trend in U.S. net electricity generation by fuel type – coal, natural gas, and all other types including nuclear and renewable sources. Following the shale gas boom in the mid-2000's we observe cheaper natural gas displacing coal-based energy production.

*Sources*: Author calculations based on data from EIA (2020a).

We can also observe in Figure 44 that coal exports were in decline following the implementation of the Acid Rain Program and up until the shale gas boom but have increased since. Interpreting in terms of the domestic market, we first observe a reduction in domestic slack, follow by oversupply as power plants transition away from coal.

[138] In late 2017 the U.S. became a net exporter of natural gas, primarily to Canada and Mexico. Exports of U.S. LNG have also recently increased rapidly: by 560% in 2015-206, 280% in 2016-2017, 53% 2017-2018, and 69% in 2018-2019 from small initial levels.

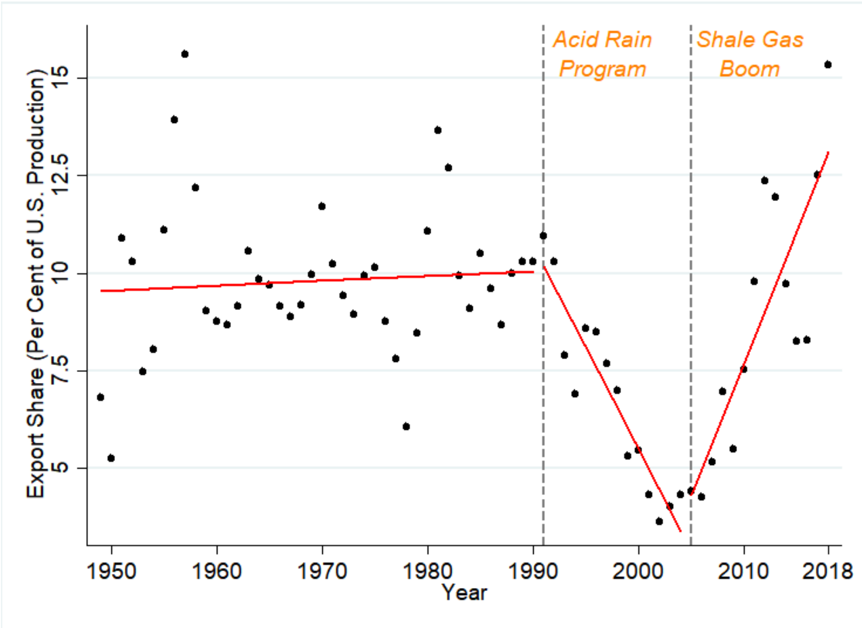


FIGURE 44 SHARE OF U.S. COAL PRODUCTION EXPORTED

*Notes*: After first declining in response to the Acid Rain Program, tradable permits, and FGD usage, the share of total U.S. coal production exported increases substantially following the shale gas boom in the mid-2000s which resulted in power plant fuel switching and coal supply slack.

*Source*: Author calculations based on data from EIA (2019).

Another departure from this paper's theory is that the U.S. alone does not fit the definition of a consensus. That is, it does not alone determine world price. Responding to rather than setting world prices, U.S. coal competes against other sources, and only when at a price advantage can mines export (IEA, 2019). When sufficient details allow estimates of export coal sulphur content – since 1989 – we see both domestic-bound and export coal follow a similar trend in sulphur content until the shale gas boom as illustrated in Figure 45. However, slack in coal supply since 2005 eliminates any opportunity to test the negative leakage effect from domestic abatement technology. Instead, we observe a marked increase in the share of dirty fuel in exports – emissions leakage – while the U.S. transitions to cleaner natural gas domestically. Domestic policies that reduce emissions without accounting for where dirty fuels end up can instead cause leakage to less regulated spheres.

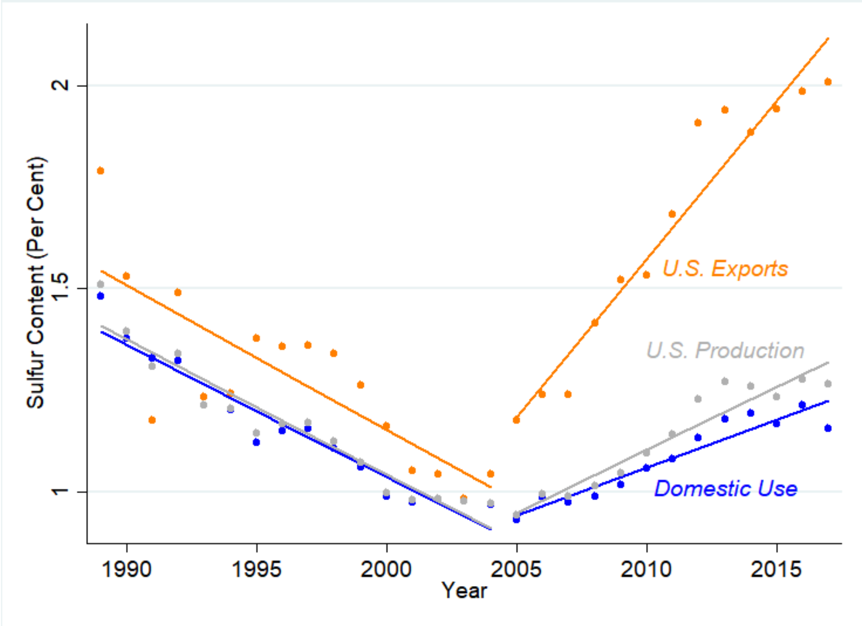


FIGURE 45 COAL SULPHUR CONTENT OF EXPORTS VERSUS DOMESTIC POWER PLANT USE

*Notes*: Trend in the coal sulphur content of U.S. production, energy sector domestic consumption, and coal exports. Shale gas expansion since the mid-2000's lead to unprecedented slack in the U.S. coal supply of all qualities. Subsequently, sufficient high sulphur coal remains on the market to be priced cheaper and purchased by firms in unregulated markets.

*Source*: Author calculations based on data from EIA (1993, 1994, 1995, 2000, 2018a).

So then, what has been the impact on global emissions and the environment? We can observe the destination for U.S. coal since 2000, illustrated in Figure 46. There is first an increase in coal exports to the E.U., U.K., and Ireland, then Japan and South Korea – areas of less concern as they are regulated. But then, we observe increases in exports to nearly unregulated India and China.

However, when we move away from the Consensus-Defector framework, there may be many suppliers and demanders of goods. In the case of coal, globally there are many sources of high sulphur coal at different costs. It is quite possible, then, that U.S. high sulphur coal exports displace more expensive, roughly equally high sulphur sources rather than displacing cleaner energy sources. On this proposition I encounter other emerging research: Wolack (2016) and Knittel, Metaxoglou, Soderbery, and Trindade (2019) model the international trade in energy and find the global displacement of dirty coal by similarly dirty U.S. coal. The U.S. alone, it seems, cannot disrupt the global status quo. So then, the Consensus requirement noted in this research appears essential.

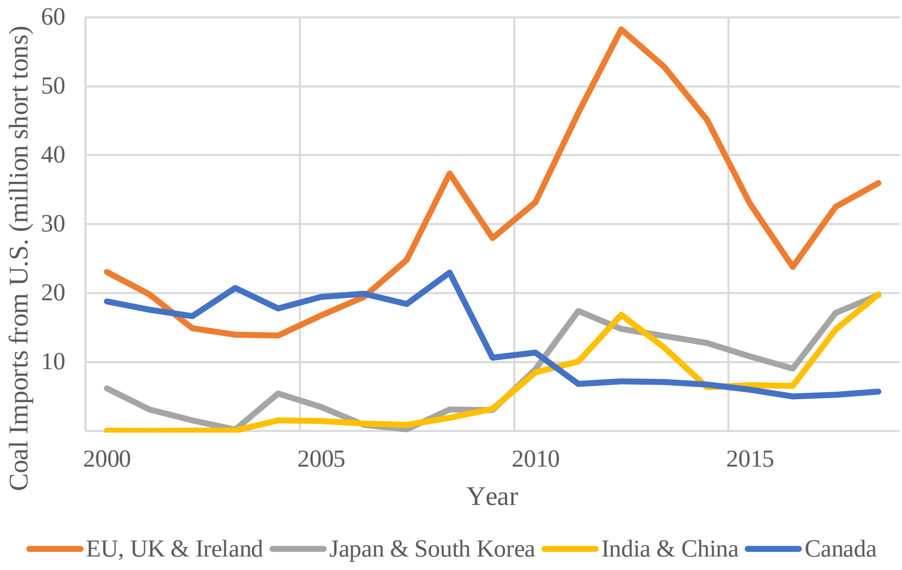


FIGURE 46 IMPORTERS OF U.S. COAL, 2000-2018

*Notes:* Trends in the import of U.S. coal. Increases in imports (ranked by country or body) to the #1 EU, and #10 UK and Ireland, followed by #3 Japan and #4 South Korea dominate increases in U.S. exports. These are countries with relatively clean consumption (sulphur removal) capacities. Increases then follow in #2 India and #13 China. #6 Canada decreased imports substantially from 2008 onward during the shale gas boom. #2 India, #3 Japan, & #4 South Korea had modest gains, and #5 Brazil, #8 Ukraine, #9 Morocco, #11 Egypt, #14 Chile, & #15 Argentina had flat demand over the period and there were modest increases in #7 Mexico, #12 Turkey (not pictured but adds context).

*Sources*: Author calculations based on EIA data (EIA, 2020b).

### A4.1 Comparison of η and Implied-η Estimates

In this section, several $\eta$ estimates from the literature are compared to implied-$\eta$ estimates derived in this paper. As discussed, implied estimates are usually, but not exclusively higher and represent the net effect of tax and all other economic and social policies on the distribution of economic growth. The basis and period of observation for the implied-$\eta$ estimates differ by source.

TABLE 28—COMPARISON OF $\eta$ ESTIMATES TO IMPLIED-$\eta$ ESTIMATES

| | $\eta$ estimates from the literature | Implied-$\eta$ estimates | | | | |
|---|---|---|---|---|---|---|
| | | LIS [t] | WID [u] | OECD [v] | WB-Income [w] | WB-Consumption [w] |
| Argentina (1) [a] | 1.23 | - | - | - | 2.46 | - |
| Australia (2) [b] | 1.67 | 1.77 | - | 1.85 | 1.87 | - |
| Austria (2) [b] | 1.74 | 2.11 | - | 2.21 | 2.18 | - |
| Belgium (3) [b, c] | 1.42 | 1.53 | - | 2.43 | 2.16 | - |
| Bolivia (1) [a] | 1.06 | - | - | - | 2.09 | - |
| Brazil (1) [a] | 2.09 | - | 2.16 | 2.41 | 2.24 | - |
| Canada (3) [b, d] | 1.37 | 1.59 | - | 1.80 | 2.01 | - |
| Chile (1) [a] | 1.22 | - | - | 2.44 | 2.37 | - |
| Colombia (1) [a] | 1.61 | - | - | - | 2.17 | - |
| Costa Rica (1) [a] | 1.11 | - | - | 1.45 | 1.77 | - |
| Czech Republic (4) [b, e] | 1.35 | 1.89 | - | 1.99 | 2.09 | - |
| Ecuador (1) [a] | 1.06 | - | - | - | 2.23 | - |
| El Salvador (1) [a] | 1.13 | - | - | - | 2.56 | - |
| Estonia (2) [e] | 1.20 | - | - | - | 2.08 | 1.63 |
| France (3) [b, f] | 1.36 | 2.11 | 2.27 | 1.90 | 1.98 | - |
| Germany (3) [b, c] | 1.41 | 1.64 | - | 1.82 | 1.71 | - |
| Guatemala (1) [a] | 1.13 | - | - | - | 2.24 | - |
| Honduras (1) [a] | 1.16 | - | - | - | 2.19 | - |
| Hungary (4) [b, e] | 1.59 | 0.26 | - | 2.15 | -0.54 | 1.7 |
| India (1) [g] | 1.64 | - | 1.89 | - | - | 1.81 |
| Ireland (2) [b] | 1.24 | 2.11 | - | 1.65 | 2.22 | - |
| Italy (4) [b, h] | 1.31 | 2.01 | - | 1.84 | 1.58 | - |
| Japan (2) [b] | 1.46 | - | - | -0.84 | - | - |
| Latvia (2) [e] | 1.13 | - | - | 2.09 | 1.92 | 1.96 |
| Mexico (1) [a] | 2.71 | - | - | 1.87 | 2.40 | 2.53 |
| Netherlands (1) [c] | 1.60 | 1.77 | - | 2.11 | 2.12 | - |
| New Zealand (2) [b] | 1.50 | - | - | 1.78 | - | - |
| Nicaragua (1) [a] | 1.14 | - | - | - | 2.32 | 2.04 |
| Norway (2) [b] | 1.32 | 1.96 | - | 1.93 | 2.11 | - |
| Panama (1) [a] | 1.15 | - | - | - | 2.20 | - |
| Paraguay (1) [a] | 1.00 | - | - | - | 0.71 | - |
| Peru (1) [a] | 1.05 | - | - | - | 2.47 | - |
| Poland (4) [b, e] | 1.29 | 1.80 | - | 2.18 | 1.83 | 1.83 |
| Portugal (2) [b] | 1.43 | - | - | 2.68 | 2.88 | - |
| Slovak Republic (4) [b, e] | 1.50 | 1.84 | - | 2.19 | 2.00 | 2.18 |
| Spain (3) [b, c] | 1.32 | 2.04 | - | 0.02 | 1.37 | - |
| Turkey (3) [b, i] | 1.17 | - | 2.28 | 2.02 | - | 2.09 |
| United Kingdom (20) [b, c, j, k, l, m, n, o, p, q, r] | 1.58 | 1.68 | - | 1.79 | 2.55 | - |
| United States (4) [b, d, s] | 1.47 | 1.33 | 1.5 | 1.62 | 1.62 | - |
| Uruguay (1) [a] | 1.71 | - | - | - | 2.19 | - |
| Venezuela (1) [a] | 1.04 | - | - | - | 1.34 | - |

*Notes:* Comparison of $\eta$ estimates from revealed preference studies against implied-$\eta$ estimates using the best available from Table 10, Table 11, Table 29, Table 30, and Table 31. National $\eta$ estimates from the literature are the mean of the reported estimates when more than one is available (number of estimates used in parentheses).

*Sources:* Derived or replicated from [a] Moore, Boardman, and Vining (2020), [b] Evans (2005), [c] Evans and Sezer (2005), [d] Kula (1984), [e] Seçilmiş and Akbulut (2019), f Evans (2004b), [g] Kula (2004), [h] Percoco (2008), [i] Evans, Kula, and Nagase (2014), [j] Stern (1977), [k] Blundell (1988), [l] Blundell, Pashardes, and Weber (1993), [m] Banks, Blundell, and Lewbel (1997), [n] Cowell and Gardiner (1999), [o] Evans and Sezer (2002), [p] Evans (2004a), [q] Evans, Kula, and Sezer (2005), [r] Groom and Maddison (2019), [s] Moore, Boardman, and Vining (2013). Author calculations based on data from [t] Thewissen, Nolan, and Roser (2016) which is derived from the Luxembourg Income Study (LIS) Database, [u] World Inequality Database (WID) set including from Morgan (2017), Alvaredo, Assouad, and Piketty (2018), Garbinti, Goupille-Lebret, and Piketty (2018), Chancel and Piketty (2017), and Piketty, Saez, and Zucman (2016), [v] OECD (2019), and [w] World Bank (2019).

## A4.2 Long-Run Estimates of Implied-η Based on OECD and World Bank Data

This appendix reports additional implied-$\eta$ estimates. OECD (2019) data is from a collection of national household income surveys. For European Union countries, data is based on the Statistics on Income and Living Conditions (EU-SILC) survey, and for other countries are conducted by the respective national governments. As with the LIS-based data, these measures use the equivalised household disposable income concept but define it as "the total income received by the households less the current taxes and transfers they pay, adjusted for household size with an equivalence scale." The OECD data is then CPI adjusted (2010 basis) for comparison. Some of the estimates cover timespans of less than a decade and should be taken with caution. When the estimates of median and residual growth are statistically significant, OLS-based values are used to derived implied-$\eta$. Otherwise estimates for each country are derived using growth between the first and last year of data available in the two-point method.

TABLE 29—LONG-RUN IMPLIED-$\eta$ ESTIMATES, OECD DATA

| | Period [a] | Two-point | | OLS | |
|---|---|---|---|---|---|
| | | $\bar{g}$ [b] | $\eta$ | $\bar{g}$ [b] | $\eta$ [c] |
| Australia | 2000, 2016 | 3.14 | 1.85 | - | - |
| Austria | 2007, 2016 | 0.69 | 2.21 | - | - |
| Belgium | 2004, 2016 | 0.73 | 2.43 | - | - |
| Brazil | 2006, 2013 | 3.82 | 2.41 | - | - |
| Canada | 1976-2017 (42) | 0.99 | 1.91 | 0.85 | 1.80 |
| Chile | 2006, 2017 | 2.30 | 2.44 | - | - |
| Costa Rica | 2010, 2017 | 1.15 | 1.45 | - | - |
| Czech Republic | 1992-2016 (16) | 2.86 | 1.99 | 2.00 | - |
| Denmark | 1985-2016 (16) | 1.14 | 1.68 | 0.99 | 1.71 |
| Estonia | 2004, 2016 | 8.77 | 2.14 | - | - |
| Finland | 1986-2017 (32) | 1.83 | 1.71 | 1.63 | 1.81 |
| France | 1996, 2016 | 0.92 | 1.90 | - | - |
| Germany | 1985-2016 (14) | 1.03 | 1.86 | 0.60 | 1.82 |
| Greece | 1974, 2016 | 0.12 | 2.84 | - | - |
| Hungary | 1991-2016 (15) | 0.74 | 2.12 | 1.02** | 2.15** |
| Iceland | 2004, 2015 | 1.22 | 2.05 | - | - |
| Ireland | 2004, 2016 | 0.43 | 1.65 | - | - |
| Israel | 1990-2017 (14) | 2.77 | 2.00 | 1.88 | - |
| Italy | 1984-2016 (17) | 0.48 | 1.84 | 0.33* | - |
| Japan | 1985, 2015 | -0.02 | -0.84 | - | - |
| Korea | 2006, 2017 | 3.43 | 1.62 | - | - |
| Latvia | 2004, 2016 | 8.18 | 2.09 | - | - |
| Lithuania | 2004, 2016 | 8.19 | 1.98 | - | - |
| Luxembourg | 1986-2016 (15) | 2.29 | 1.87 | 1.46 | 1.90 |
| Mexico | 1984-2016 (9) | 1.59 | 1.87 | 0.88** | - |
| Netherlands | 1977-2016 (17) | 0.93 | 2.18 | 0.92 | 2.11 |
| New Zealand | 1985-2014 (10) | 1.59 | 1.47 | 1.45 | 1.78 |
| Norway | 1986-2017 (14) | 2.54 | 1.93 | 2.19 | - |
| Poland | 2005, 2016 | 5.72 | 2.18 | - | - |
| Portugal | 2004, 2016 | -0.33 | 2.68 | - | - |
| Russia | 2008, 2016 | 3.37 | 2.55 | - | - |
| Slovak Republic | 2004, 2016 | 3.75 | 2.19 | - | - |
| Slovenia | 2004, 2016 | 0.69 | 2.09 | - | - |
| Spain | 2007, 2016 | -1.29 | 0.02 | - | - |
| Sweden | 1975-2017 (15) | 2.42 | 1.88 | 1.86 | 1.92 |
| Switzerland | 2006, 2015 | 0.65 | 2.23 | - | - |
| Turkey | 2004, 2015 | 5.96 | 2.02 | - | - |
| United Kingdom | 1975-2017 (23) | 1.62 | 1.75 | 1.23 | 1.79 |
| United States | 1995-2017 (13) | 0.93 | 1.62 | 0.54 | - |

*Notes:* Estimates of annual rate of long-run growth in CPI-adjusted equivalised disposable household income. All OLS estimates reported are statistically significant at the 1-per cent level or better using Huber–White standard errors unless otherwise noted (** 5 per cent level; * 10 per cent level).

[a] Number of observations in the period in parentheses. [b] Average annual growth rate, in per cent. [c] Statistical significance of OLS-based $\eta$'s is the minimum of the significance of median and inequality-driven growth estimates

*Source:* Author calculations based on data from OECD (2019).

The World Bank (2019) provides a more extensive, but varied dataset that I use to estimate implied-$\eta$ values. Table 30 reports income-based, and Table 31 reports consumption-based estimates. The data is derived from a collection of more than 1,500 representative household surveys conducted by each nation or international agencies on their behalf. These, for instance, included more than 2-million total households in 2015. All data is 2011 PPP($) adjusted (with CPI based on IMF data) and based on household per-capita income and consumption expenditure data. In some cases, e.g. China, both urban and rural statistics are available which account for intra-country differences in prices. When sufficient data for statistical significance is available, OLS-based estimates of median and residual growth rates are used as well as two-point estimates of implied-$\eta$.

TABLE 30—LONG-RUN IMPLIED-$\eta$ ESTIMATES, WORLD BANK INCOME DATA

| | Period [a] | Two-point | | OLS | |
|---|---|---|---|---|---|
| | | $\bar{g}$ [b] | $\eta$ | $\bar{g}$ [b] | $\eta$ [c] |
| Argentina--Urban | 1987-2017 (27) | -0.24 | 1.99 | 0.88* | 2.46 |
| Australia | 1981-2014 (10) | 2.10 | 1.80 | 1.65 | 1.87 |
| Austria | 2003, 2015 | 1.07 | 2.18 | - | - |
| Belgium | 2003, 2015 | 1.13 | 2.16 | - | - |
| Belize | 1993, 1999 | -3.54 | 1.52 | - | - |
| Bolivia | 1990-2017 (19) | 2.76 | 2.09 | 2.83 | - |
| Brazil | 1981-2017 (33) | 4.00 | 2.27 | 2.72 | 2.24 |
| Bulgaria | 1992, 2014 | -0.28 | -0.53 | - | - |
| Canada | 1981-2013 (11) | 1.16 | 2.01 | 1.09 | - |
| Chile | 1987-2017 (14) | 4.77 | 2.38 | 2.20 | 2.37 |
| China--Rural | 1981, 1987 | 11.59 | 1.84 | - | - |
| China--Urban | 1981, 1987 | 6.51 | 1.96 | - | - |
| Colombia | 1992-2017 (19) | 1.89 | 2.16 | 2.69 | 2.17 |
| Colombia--Urban | 1988, 1991 | 0.37 | 2.84 | - | - |
| Costa Rica | 1981-2017 (31) | 9.44 | 1.81 | 4.20 | 1.77 |
| Croatia | 1988-2015 (8) | -0.14 | -0.59 | -0.32** | -0.51 |
| Cyprus | 2004, 2015 | 0.02 | -0.98 | - | - |
| Czech Republic | 1993-2015 (14) | 3.08 | 2.13 | 2.19 | 2.09 |
| Denmark | 2003, 2015 | 1.34 | 1.47 | - | - |
| Dominican Republic | 1986, 2016 | 0.52 | 1.99 | - | - |
| Ecuador | 1987-2017 (20) | 2.90 | 2.18 | 3.09 | 2.23 |
| El Salvador | 1991-2017 (23) | 1.68 | 2.56 | 0.96 | 2.56 |
| Estonia | 1993-2015 (14) | 7.40 | 2.15 | 4.45 | 2.08 |
| Finland | 2003, 2015 | 1.70 | 2.06 | - | - |
| France | 2003, 2015 | 2.02 | 1.98 | - | - |
| Germany | 1991-2015 (18) | 0.74 | 1.89 | 0.72 | 1.71 |
| Greece | 2003, 2015 | -2.59 | -0.04 | - | - |
| Guatemala | 1986-2014 (5) | 4.68 | 2.36 | - | 2.24 |
| Honduras | 1989-2017 (28) | 2.53 | 2.50 | 2.27 | 2.19 |
| Hungary | 1987, 2015 | 0.09 | -0.54 | - | - |
| Iceland | 2003, 2014 | 0.94 | 2.02 | - | - |
| Ireland | 2003, 2015 | 0.57 | 2.22 | - | - |
| Israel | 1986-2016 (10) | 2.83 | 2.07 | 1.94 | - |
| Italy | 2003, 2015 | -0.30 | 1.58 | - | - |
| Korea, Republic of | 2006, 2012 | 1.77 | 2.05 | - | - |
| Latvia | 1993-2015 (15) | 8.59 | 1.87 | 4.53 | 1.92 |
| Lithuania | 1993-2015 (13) | 17.53 | 1.97 | 6.54 | - |
| Luxembourg | 2003, 2015 | 0.11 | -0.69 | - | - |
| Malaysia | 1984-2015 (12) | 4.02 | 2.26 | 2.65 | 2.15 |
| Malta | 2006, 2015 | 3.52 | 1.83 | - | - |
| Mexico | 1989-2016 (15) | 0.49 | 2.60 | 1.05* | 2.40 |
| Netherlands | 2004, 2015 | 1.13 | 2.12 | - | - |
| Nicaragua | 1993-2014 (6) | 5.80 | 2.38 | 3.49 | 2.32 |
| North Macedonia | 2009, 2015 | 2.58 | 2.50 | - | - |
| Norway | 2003, 2015 | 3.21 | 2.11 | - | - |
| Panama | 1989-2017 (24) | 5.68 | 2.26 | 3.31 | 2.20 |

| | | | | | |
|---|---|---|---|---|---|
| Paraguay | 1990-2017 (21) | 0.82 | 0.71 | 1.14** | - |
| Peru | 1997, 2017 | 2.90 | 2.47 | - | - |
| Philippines | 2000, 2015 | 1.04 | 2.40 | - | - |
| Poland | 1985-2015 (15) | 3.00 | 1.90 | 1.76 | 1.83 |
| Portugal | 2003, 2015 | 0.13 | 2.88 | - | - |
| Romania | 1989, 2015 | -1.03 | -0.16 | - | - |
| Serbia | 2012, 2015 | -0.61 | 1.21 | - | - |
| Slovak Republic | 1996, 2015 | 2.52 | 2.00 | - | - |
| Slovenia | 1993-2015 (13) | 2.83 | 2.22 | 2.09 | 2.15 |
| Spain | 2003, 2015 | 0.90 | 1.37 | - | - |
| Sweden | 2003, 2015 | 3.18 | 1.77 | - | - |
| Switzerland | 2006, 2015 | 0.85 | 2.19 | - | - |
| United Kingdom | 2004, 2015 | 0.34 | 2.55 | - | - |
| United States | 1986-2016 (10) | 1.07 | 1.52 | 0.86 | 1.62 |
| Uruguay | 1981-2017 (14) | 1.17 | 2.19 | 0.95 | - |
| Uruguay--Urban | 1992, 2005 | -1.71 | -0.28 | - | - |
| Venezuela, Republica | 1981-2006 (13) | -1.31 | 1.34 | -2.94 | - |

*Notes:* Estimates of the annual rate of long-run growth in 2011 PPP($) adjusted household per-capita income. All OLS estimates reported are statistically significant at the 1-per cent level or better using Huber–White standard errors unless otherwise noted (** 5 per cent level; * 10 per cent level).

[a] Number of observations in the period in parentheses. [b] Average annual growth rate, in per cent. [c] Statistical significance of OLS-based $\eta$'s is the minimum of the significance of median and inequality-driven growth estimates

*Source:* Author calculations based on data from the World Bank (2019).

TABLE 31—LONG-RUN IMPLIED-$\eta$ ESTIMATES, WORLD BANK CONSUMPTION DATA

| | Period [a] | Two-point | | OLS | |
|---|---|---|---|---|---|
| | | $\bar{g}$ [c] | $\eta$ | $\bar{g}$ [c] | $\eta$ |
| Albania | 2002, 2012 | 1.73 | 2.31 | - | - |
| Algeria | 1988, 2011 | 1.03 | 2.56 | - | - |
| Armenia | 1999, 2017 | 4.02 | 2.08 | - | - |
| Azerbaijan | 1995, 2005 | 8.37 | 2.05 | - | - |
| Bangladesh | 1983-2016 (9) | 1.37 | 1.40 | 1.17 | 1.48** |
| Belarus | 1998, 2017 | 17.04 | 2.10 | - | - |
| Benin | 2003, 2015 | 0.42 | -0.16 | - | - |
| Bhutan | 2003, 2017 | 5.70 | 2.20 | - | - |
| Bosnia and Herzegovina | 2001, 2011 | 5.70 | 1.92 | - | - |
| Botswana | 1985-2015 (5) | 3.00 | 1.92 | 2.46* | - |
| Bulgaria | 1989, 2007 | -2.62 | -0.03 | - | - |
| Burkina Faso | 1994, 2014 | 5.27 | 2.40 | - | - |
| Burundi | 1992, 2013 | 1.29 | 1.24 | - | - |
| Cameroon | 1996, 2014 | 4.11 | 2.15 | - | - |
| Central African Republic | 1992, 2008 | 6.69 | 2.31 | - | - |
| China--Rural | 1990-2015 (13) | 14.72 | 1.97 | 6.09 | - |
| China--Urban | 1990-2015 (13) | 16.98 | 1.80 | 6.74 | 1.90 |
| Côte d'Ivoire | 1985-2015 (10) | -1.87 | 0.07 | -2.36 | - |
| Croatia | 1998, 2010 | -2.66 | 0.06 | - | - |
| Djibouti | 2002, 2017 | 1.01 | 2.00 | - | - |
| Egypt, Arab Republic of | 1990-2015 (8) | 1.23 | 1.93 | 0.92 | - |
| Estonia | 1995, 2004 | 4.06 | 1.63 | - | - |
| Eswatini | 1994, 2009 | 10.10 | 2.26 | - | - |
| Ethiopia | 1995, 2015 | 2.68 | 2.48 | - | - |
| Fiji | 2002, 2013 | 0.82 | 1.95 | - | - |
| Gambia, The | 1998, 2015 | 9.66 | 2.31 | - | - |
| Georgia | 1996-2017 (22) | -0.03 | -0.86 | 1.67** | - |
| Ghana | 1987-2016 (7) | 5.19 | 1.76 | 3.46 | 1.87 |
| Guinea | 1991, 2012 | 12.67 | 2.19 | - | - |
| Guinea-Bissau | 1993, 2010 | 0.62 | -0.38 | - | - |
| Hungary | 1998, 2007 | 4.88 | 1.70 | - | - |
| India--Rural | 1983-2011 (6) | 2.11 | 1.84 | 1.33 | 1.88 |
| India--Urban | 1983-2011 (6) | 2.46 | 1.66 | 1.69 | 1.73 |
| Indonesia--Rural | 1984-2017 (25) | 7.58 | 1.94 | 4.00 | 1.90 |
| Indonesia--Urban | 1984-2017 (25) | 6.06 | 1.75 | 3.73 | 1.81 |
| Iran, Islamic Republic of | 1986-2016 (11) | 2.28 | 2.28 | 2.35 | 2.15 |
| Jamaica | 1988, 2004 | 2.96 | 1.51 | - | - |
| Jordan | 1986, 2010 | -0.04 | 2.40 | - | - |
| Kazakhstan | 1996-2017 (18) | 3.87 | 2.17 | 4.39 | 2.12 |
| Kenya | 1992, 2015 | -1.73 | 1.16 | - | - |
| Kosovo | 2003, 2017 | 3.71 | 1.98 | - | - |
| Kyrgyz Republic | 1998, 2017 | 1.33 | 2.60 | - | - |
| Lao People's Democratic Republic | 1992, 2012 | 0.95 | 1.92 | - | - |

| | | | | | |
|---|---|---|---|---|---|
| Latvia | 1997, 2009 | 6.63 | 1.96 | - | - |
| Lesotho | 1986, 2010 | -1.18 | 0.20 | - | - |
| Liberia | 2007, 2016 | 5.85 | 1.94 | - | - |
| Lithuania | 1996, 2008 | 6.79 | 1.75 | - | - |
| Madagascar | 1993, 2012 | -1.11 | 0.14 | - | - |
| Malawi | 1997, 2016 | -2.41 | 1.57 | - | - |
| Mali | 1994, 2009 | 6.27 | 2.51 | - | - |
| Mauritania | 1987-2014 (7) | 3.07 | 2.27 | 2.24 | - |
| Mexico | 1984-2016 (15) | 0.13 | 2.76 | - | 2.53 |
| Moldova | 1997, 2017 | 5.41 | 2.21 | - | - |
| Mongolia | 1995-2016 (9) | 3.94 | 1.99 | 5.00 | - |
| Montenegro | 2005, 2014 | 1.90 | 1.71 | - | - |
| Morocco | 1984-2013 (6) | 2.85 | 1.87 | 1.61** | 1.85* |
| Mozambique | 1996, 2014 | 5.15 | 2.01 | - | - |
| Namibia | 2003, 2015 | 5.18 | 2.43 | - | - |
| Nepal | 1995, 2010 | 5.83 | 2.10 | - | - |
| Nicaragua | 1993, 2005 | -1.13 | 2.04 | - | - |
| Niger | 1992-2014 (6) | 3.02 | 2.03 | 2.51** | - |
| Nigeria | 1985, 2009 | 0.32 | -0.07 | - | - |
| North Macedonia | 1998, 2008 | 11.42 | 0.97 | - | - |
| Pakistan | 1987-2015 (12) | 5.06 | 1.95 | 2.90 | - |
| Philippines | 1985-2015 (11) | 1.78 | 2.01 | 1.40 | - |
| Poland | 1993-2016 (21) | 4.69 | 1.83 | 3.13 | - |
| Romania | 1998, 2016 | 2.28 | 2.16 | - | - |
| Russian Federation | 1993-2015 (21) | 2.73 | 2.32 | 4.59 | - |
| Rwanda | 1984-2016 (6) | 1.01 | 0.34 | 1.07** | - |
| Samoa | 2002, 2013 | 0.07 | 2.89 | - | - |
| Senegal | 1991, 2011 | 2.42 | 2.56 | - | - |
| Serbia | 2002, 2015 | -0.14 | 2.52 | - | - |
| Slovak Republic | 2004, 2009 | 4.23 | 2.18 | - | - |
| Slovenia | 1998, 2003 | 0.70 | 1.57 | - | - |
| South Africa | 1993-2014 (7) | 2.81 | 1.38 | 3.34 | 1.59** |
| Sri Lanka | 1985-2016 (8) | 3.62 | 1.64 | 2.45 | 1.75 |
| Tajikistan | 1999, 2015 | 12.89 | 1.90 | - | - |
| Tanzania | 1991, 2011 | 2.73 | 1.70 | - | - |
| Thailand | 1981-2017 (23) | 5.20 | 2.23 | 3.28 | 2.19 |
| Timor-Leste | 2001, 2014 | 0.59 | 2.63 | - | - |
| Togo | 2006, 2015 | 0.92 | 2.32 | - | - |
| Tonga | 2001, 2015 | -0.09 | -0.84 | - | - |
| Tunisia | 1985-2015 (7) | 3.11 | 2.32 | 2.23 | 2.21 |
| Turkey | 1987-2016 (17) | 3.49 | 2.09 | 2.95 | - |
| Uganda | 1989-2016 (9) | 1.42 | 1.87 | 1.84 | - |
| Ukraine | 1992-2016 (19) | 2.72 | 2.05 | 3.03 | 2.13 |
| Uzbekistan | 1998, 2003 | -6.91 | 0.20 | - | - |
| Vietnam | 1992-2016 (10) | 12.16 | 2.11 | 6.30 | 2.07 |
| West Bank and Gaza | 2004, 2016 | 0.63 | 2.37 | - | - |
| Yemen, Republic of | 1998, 2014 | -1.77 | -0.13 | - | - |
| Zambia | 1991, 2015 | -0.11 | -0.50 | - | - |

*Notes:* Estimates of the annual rate of long-run growth in 2011 PPP($) adjusted household per-capita consumption expenditures. All OLS estimates reported are statistically significant at the 1-per cent level or better using Huber–White standard errors unless otherwise noted (** 5 per cent level; * 10 per cent level).

[a] Number of observations in the period in parentheses. [b] Average annual growth rate, in per cent. [c] Statistical significance of OLS-based $\eta$'s is the minimum of the significance of median and inequality-driven growth estimates

*Source:* Author calculations based on data from the World Bank (2019).

## A5.1 Data Tables Supporting Figure 31 and Figure 32

This section reports the data underlying Figure 31 and Figure 32 (Table 32 and Table 33, respectively).

TABLE 32— DECOMPOSITION OF GROWTH INTO DISTRIBUTION PARAMETERS, LIS AND OECD

| | Period [a] | $\bar{g}$ [b] | μ-driven growth [b] | $\sigma^2$-driven growth [b] |
|---|---|---|---|---|
| LIS data | | | | |
| Australia | 1981-2010 (8) | 1.38** | 1.17** | 0.21 |
| Canada | 1981-2010 (10) | 0.90 | 0.67 | 0.23 |
| Denmark | 1987-2010 (7) | 1.15 | 1.03 | 0.12** |
| Finland | 1987-2010 (7) | 1.75 | 1.45 | 0.30 |
| France | 1978-2010 (7) | 0.79** | 0.87 | -0.08* |
| Germany | 1984-2010 (7) | 0.56** | 0.43* | 0.13** |
| Ireland | 1987-2010 (8) | 3.45 | 3.76 | -0.31** |
| Israel | 1986-2010 (7) | 1.67 | 1.44 | 0.23 |
| Luxembourg | 1985-2010 (8) | 2.63 | 2.47** | 0.16** |
| Sweden | 1981-2005 (6) | 2.04 | 1.81 | 0.23** |
| United Kingdom | 1979-2010 (9) | 2.41 | 2.08 | 0.33 |
| United States | 1979-2013 (10) | 0.77 | 0.49 | 0.28 |
| OECD data | | | | |
| Canada | 1976-2017 (42) | 0.85 | 0.73 | 0.12 |
| Denmark | 1985-2016 (16) | 0.99 | 0.80 | 0.18 |
| Finland | 1986-2017 (32) | 1.63 | 1.41 | 0.21 |
| Germany | 1985-2016 (14) | 0.60 | 0.52 | 0.08 |
| Hungary | 1991-2016 (15) | 1.02** | 1.16** | -0.14** |
| Luxembourg | 1986-2016 (15) | 1.46 | 1.35 | 0.11 |
| Netherlands | 1977-2016 (17) | 0.92 | 1.01 | -0.08 |
| New Zealand | 1985-2014 (10) | 1.45 | 1.23 | 0.21** |
| Sweden | 1975-2017 (15) | 1.86 | 1.76 | 0.11 |
| United Kingdom | 1975-2017 (23) | 1.23 | 1.05 | 0.18 |

*Notes:* Luxembourg Income Study (LIS) estimates in PPP-adjusted equivalised disposable household income, and OECD estimates in CPI-adjusted household income, of the annual rate of long-run growth. Statistically significant at the 1-per cent level or better using Huber–White standard errors unless otherwise noted (** 5 per cent level; * 10 per cent level).

[a] Number of observations in the period in parentheses. [b] Average annual growth rate, in per cent.

*Sources:* Author calculations based on data from OECD (2019), and Thewissen, Nolan, and Roser (2016), which is derived from the Luxembourg Income Study (LIS) Database.

TABLE 33— DECOMPOSITION OF GROWTH INTO DISTRIBUTION PARAMETERS, WORLD BANK

| | Period [a] | $\bar{g}$ [b] | μ-driven growth [b] | $\sigma^2$-driven growth [b] |
|---|---|---|---|---|
| Income | | | | |
| Argentina--Urban | 1987-2017 (27) | 0.88* | 1.41** | -0.53 |
| Australia | 1981-2014 (10) | 1.65 | 1.49 | 0.16 |
| Brazil | 1981-2017 (33) | 2.72 | 3.37 | -0.65 |
| Chile | 1987-2017 (14) | 2.20 | 3.15 | -0.94 |
| Colombia | 1992-2017 (19) | 2.69 | 3.11 | -0.42* |
| Costa Rica | 1981-2017 (31) | 4.20 | 3.56 | 0.64 |
| Croatia | 1988-2015 (8) | -0.32** | -0.55 | 0.23 |
| Czech Republic | 1993-2015 (14) | 2.19 | 2.35 | -0.16 |
| Ecuador | 1987-2017 (20) | 3.09 | 3.77 | -0.68* |
| El Salvador | 1991-2017 (23) | 0.96 | 1.83 | -0.87 |
| Estonia | 1993-2015 (14) | 4.45 | 4.75 | -0.30* |
| Germany | 1991-2015 (18) | 0.72 | 0.58 | 0.14 |
| Guatemala | 1986-2014 (5) | - | 3.94* | -0.76 |
| Honduras | 1989-2017 (28) | 2.27 | 2.67 | -0.40 |
| Latvia | 1993-2015 (15) | 4.53 | 4.25 | 0.28** |
| Malaysia | 1984-2015 (12) | 2.65 | 3.01 | -0.36 |
| Mexico | 1989-2016 (15) | 1.05* | 1.57** | -0.51 |
| Nicaragua | 1993-2014 (6) | 3.49 | 4.67 | -1.18 |
| Panama | 1989-2017 (24) | 3.31 | 3.93 | -0.62 |
| Poland | 1985-2015 (15) | 1.76 | 1.55 | 0.21 |
| Slovenia | 1993-2015 (13) | 2.09 | 2.38 | -0.29** |
| United States | 1986-2016 (10) | 0.86 | 0.66 | 0.20 |
| Consumption | | | | |
| Bangladesh | 1983-2016 (9) | 1.17 | 0.82** | 0.35** |
| China--Urban | 1990-2015 (13) | 6.74 | 6.27 | 0.47 |
| Ghana | 1987-2016 (7) | 3.46 | 3.13 | 0.33** |
| India--Rural | 1983-2011 (6) | 1.33 | 1.22 | 0.12** |
| India--Urban | 1983-2011 (6) | 1.69 | 1.39 | 0.30 |
| Indonesia--Rural | 1984-2017 (25) | 4.00 | 3.71 | 0.29 |
| Indonesia--Urban | 1984-2017 (25) | 3.73 | 3.23 | 0.50 |
| Iran, Islamic | 1986-2016 (11) | 2.35 | 2.66 | -0.31 |
| Kazakhstan | 1996-2017 (18) | 4.39 | 4.86 | -0.47 |
| Mexico | 1984-2016 (15) | - | 0.63** | -0.28 |
| Morocco | 1984-2013 (6) | 1.61** | 1.44* | 0.17* |
| South Africa | 1993-2014 (7) | 3.34 | 2.50** | 0.84** |
| Sri Lanka | 1985-2016 (8) | 2.45 | 2.04 | 0.41 |
| Thailand | 1981-2017 (23) | 3.28 | 3.84 | -0.56 |
| Tunisia | 1985-2015 (7) | 2.23 | 2.68 | -0.45 |
| Ukraine | 1992-2016 (19) | 3.03 | 3.38 | -0.34** |
| Vietnam | 1992-2016 (10) | 6.30 | 6.65 | -0.34** |

*Notes:* Estimates of the annual rate of long-run growth in 2011 PPP($) adjusted household per-capita income and consumption expenditures. All OLS estimates reported are statistically significant at the 1-per cent level or better using Huber–White standard errors unless otherwise noted (** 5 per cent level; * 10 per cent level).

[a] Number of observations in the period in parentheses. [b] Average annual growth rate, in per cent.

*Source:* Author calculations based on data from the World Bank (2019).

### A5.2 Implications of Some Alternative Model Assumptions

In this appendix, four adjustments to the modelling assumptions are explored – changing to a constant savings function, the impact of a relative versus absolute satiation point in the original savings function, changes to the point in the distribution where satiation is reached, and changes to the planning horizon in the economy.

The savings functional form employed – a judgement call with surprisingly limited implications – has received enough attention in seminars to warrant an appendix. Our colleague Dr Roger Fouquet observes that our differential savings rate model is akin to supposing a uniform savings rate but that higher-income members of society earn a higher return on savings in the aggregate production process. We might suppose these higher returns result from a more scrutinous selection of investments (the use of investment services) and might decline as we move down the income scale. Satiation point $b$ then would denote the point where abnormally high returns begin to occur on average.

Other colleagues suggest using a constant savings rate akin to setting $b = 0$ and specifying total savings as $s\big(c_i(t)\big) = \bar{s}\sum_{i=1}^{N} c_i(t)$ for $i = 1\colon N$ (all) of the population, and a distributional and time constant savings rate $\bar{s}$ which we can also interpret as the average national savings rate. We compare this to the current, differential savings function which is approximately stepped with both constant lower (below $b$), and upper (above $b$) savings rates of 0 and $\hat{s}$. Suppose total consumption below $b$ is $C_L$, above is $C_U$, and $0 \leq b < 1$ is the population share below $b$ and is relative (constant). Then the differential case savings total is $(1-b)\hat{s}C_U = I_U$ and uniform case savings are $\bar{s}(bC_L + (1-b)C_U) = I_{L+U}$. Setting $I_U = I_{L+U}$ and writing as $\bar{s}$ identifies the rate equivalent to $\hat{s}$:

$$\bar{s} = \hat{s}\left(\frac{b}{(1-b)}\frac{C_L}{C_U} + 1\right)^{-1} \tag{92}$$

Suppose we set $\bar{s}$ according to its parameters in the initial period and then compare the implications for total, savings-driven growth. From the previous paper, we observe for $\eta = 2$ growth paths, $\frac{C_L}{C_U}$, is constant, for $\eta > 2$ is increasing, and for $\eta < 2$ is decreasing. Further, the change in this ratio occurs slowest for $\eta$ values closest to two and increases slowly as we follow $\eta$ further away. For pathways around $\eta = 2$ we can specify a value of $\bar{s}$ leading to similar results, with higher $\eta$ the $\bar{s}$-based model would suggest higher growth and for lower $\eta$ less growth. Since most long-run national estimates are around $\eta = [1.5, 2]$, our savings specification is perhaps more superficial novelty than departure.

Another matter is the implication of setting the satiation point as a relative versus absolute bound. That is, we might choose between modelling households as saving once they have surpassed some percentile in the distribution, e.g., 80$^{th}$, versus some level, e.g., \$100,000 in real take-home income or consumption. One may be predisposed to assume an absolute satiation point is more appropriate as it is not uncommon to say “if I can just make six-figures, I would save for retirement.” However, recent evidence on savings (Saez and Zucman, 2016) suggest the relative based distribution may be more appropriate. Also note that if the absolute condition were true, and always so, then in say the year 1900 it would have also been true. Undoubtedly, most households have become financially better off in the interim 120 or so years, yet most of the population continues to save as though they reside below the satiation point. The results of this paper have assumed a relative satiation point, but we present a comparison in Figure 47 in the form of a 100-year simulation repeated for preferences denoted by $\eta = 1$, and only differing by whether $b$ is relative or absolute and set at the same initially equivalent values. In the short term – within 30-years or so – little difference emerges. However, over the extended period, differences in consumption and distribution become substantial. Under a relative assumption, greater inequality must be emphasised to accommodate growth, while under an absolute assumption, the capital saturation point maximising mean income is reached earlier. The form of satiation point may be a minor point, however, as one cannot generally expect a 100-year projection to carry much weight.

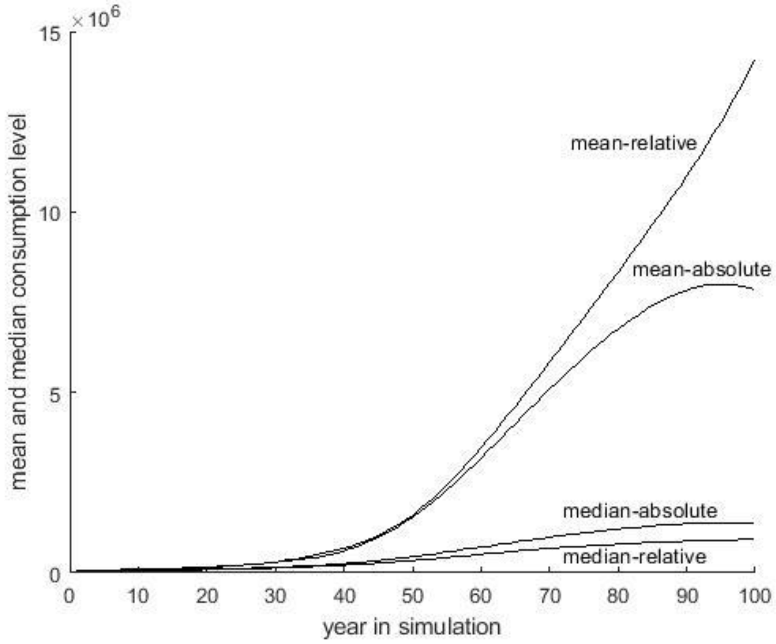


FIGURE 47. MEAN AND MEDIAN TRAJECTORIES OVER 100 YEARS

*Notes:* Differences in growth of mean and median consumption levels (η=1 assumed) by whether the consumption satiation point (point of positive savings) is relative or absolute. The impact of absolute versus relative is small in foreseeable years (the next three or so decades) but has a substantial impact further out. Under a relative satiation assumption, higher inequality is necessary to drive growth resulting in a more extensive spread. Under an absolute assumption, an economy reaches steady-state capital and consumption levels over a shorter time. As a practical matter, it is likely inaccurate to have confidence in economic projections one-hundred years out given the unpredictability of advancements and limitations.

Figure 48 reports the results of changing the relative satiation point with $\eta = \{1, 2, 3\}$ preferences simulated in each case. The 80$^{th}$ percentile satiation point is the default value based on interpretations of Saez and Zucman (2016) and Jebb, Tay, Diener, and Oishi (2018). Alternatives of 60$^{th}$ and 70$^{th}$ percentile relative satiation points are then compared. We observe that as the satiation point is reduced, growth paths pivot toward comparatively more equal growth. One interpretation is that there is less need to rely on the savings of the wealthy to drive economic growth.

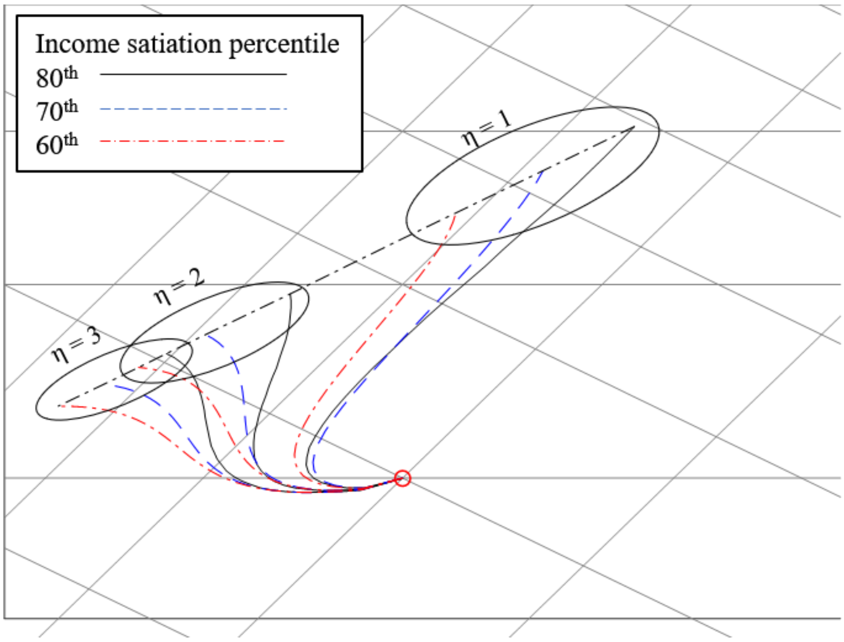


FIGURE 48. SENSITIVITY ANALYSIS OF GROWTH PATHS TO INCOME SATIATION PERCENTILE

*Notes:* Growth paths for $\eta = \{1, 2, 3\}$ preferences are compared under 80th, 70th, and 60th percentile income satiation points in the savings function. Compared to the default 80th percentile assumption, as the satiation point is reduced, there is less need to rely on the savings of the wealthy to drive economic growth.

Finally, I explore whether the planning horizon matters under the model assumptions. Figure 49 reports the results from repeating the simulation with different planning horizons. For each planning horizon, $\eta = \{1, 2, 3\}$ are simulated and subsequently the resulting growth paths are ordered from right to left. Naturally, with a longer planning horizon there is more time for growth to occur. However, the growth paths themselves differ. With a shorter planning period, there is less growth in later years to balance out an emphasis on investment in earlier years. So, with a shorter planning period, the growth shifts toward reducing inequality, or at least not growing inequality as quickly. One interpretation is that with a longer planning period, the population basically grows its way out of comparative poverty. However, to do so much of the population spends their working lives poor.

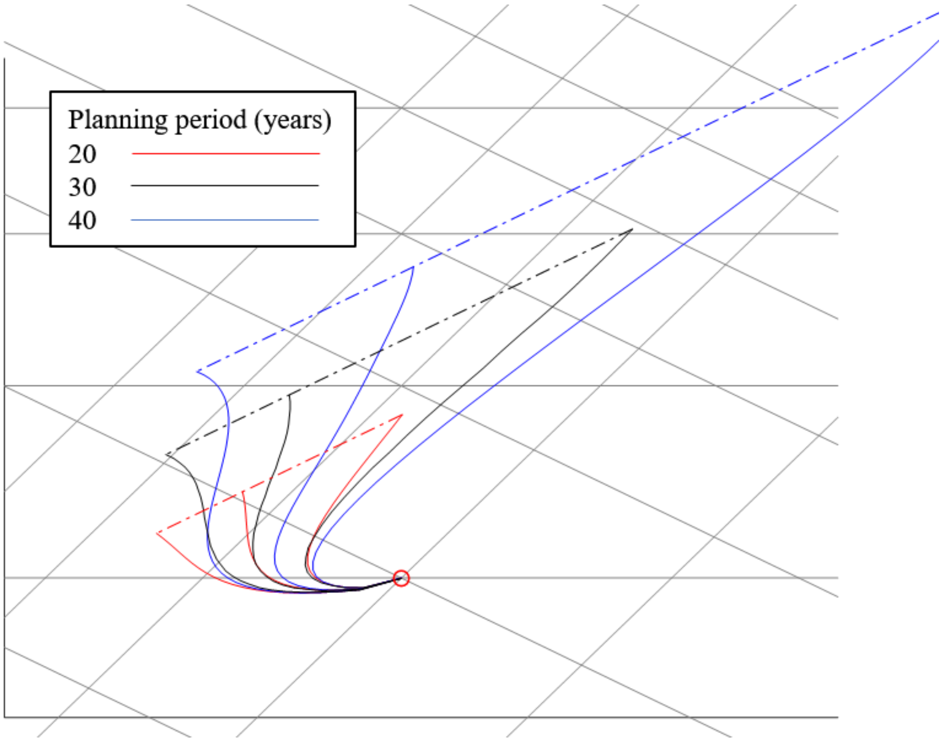


FIGURE 49. SENSITIVITY ANALYSIS OF GROWTH PATHS TO THE PLANNING HORIZON

*Notes:* Alternative planning periods in the simulation of 20, 30, and 40 years with $\eta = \{1, 2, 3\}$ in each planning horizon resulting in growth paths ordered from right to left. In the shorter planning period, where later year growth cannot be relied on, the economic plan must emphasis growth in the lower percentiles of the income distribution to maximize summed welfare. With the longer planning period, the economy can grow its way out of poverty in line with what some macroeconomists advocate for. But with longer planning periods, much of the population spend their working lives comparatively poor in order to contribute directly or indirectly to a wealthier society later.

### A5.3 The Gamma Distribution Alternative

Because we derive $\mu$ and $\sigma^2$ values from fitting a lognormal distribution directly on WID percentile values, we use this opportunity to present decomposition under a competing distributional assumption. The gamma distribution, $c_i(t) \sim \Gamma(k(t), \theta(t))$, provides one such alternative. It follows from a similar decomposition process to the lognormal: from the gamma mean, $k\theta$, growth decomposes as $\bar{g} = g_k + g_\theta$. For context, a decrease in shape parameter (k) increases the skewness of the gamma distribution, $2/\sqrt{k}$. Table 34 provides lognormal and gamma decomposition for the WID percentile distribution data.

TABLE 34— DECOMPOSITION OF GROWTH INTO LN AND Γ PARAMETERS, WID DATA

| | Period [a] | Lognormal distribution | | | Gamma distribution | | |
|---|---|---|---|---|---|---|---|
| | | $\bar{g}$ [b] | μ-driven growth [b] | $\sigma^2$-driven growth [b] | $\bar{g}$ [b] | θ-driven growth [b] | k-driven growth [b] |
| Brazil [c] | 2001-2015 (15) | 1.24** | 1.54 | - | 1.25 | 0.86** | 0.39 |
| China [d] | 1978-2015 (38) | 5.63 | 4.34 | 1.29 | 5.36 | 8.02 | -2.66 |
| Transition | 1978-1999 (22) | 4.65 | 3.54 | 1.10 | 4.45 | 7.45 | -3.00 |
| Present | 2000-2015 (16) | 9.29 | 7.89 | 1.40 | 8.69 | 10.20 | -1.51 |
| Côte d'Ivoire [e] | 1988-2014 (6) | -1.48** | -2.09 | 0.61 | -1.84 | - | - |
| Egypt [f] | 1999-2015 (6) | 1.45 | 1.67 | -0.22 | 1.30 | 0.38 | 0.92 |
| France [g] | 1900-2014 | 2.21 | 2.81 | -0.60 | 2.43 | 1.87 | 0.56 |
| Pre-WWII | 1900-1939 (20/27) | - | 0.91 | -0.70 | 0.68 | - | 0.92** |
| Post-war | 1946-1969 (22/24) | 4.52 | 4.39 | - | 4.58 | 4.56 | - |
| Transition | 1970-1999 (25/30) | - | 1.67 | -1.36 | 1.40 | - | - |
| Present | 2000-2014 (11/15) | 2.47 | -0.26 | 2.73 | 0.18 | - | - |
| India [h] | 1951-2013 (54/63) | 1.84 | 1.82 | - | 2.05 | 2.59 | -0.55** |
| Postcolonial | 1951-1964 (12/14) | 2.06 | 2.84 | -0.78 | 2.31 | - | - |
| Transition | 1965-1999 (32/35) | 1.86 | 1.75 | 0.11 | 1.86 | 1.71 | - |
| Present | 2000-2013 (10/14) | 3.74 | 2.98 | 0.76 | 4.42 | 12.69 | -8.27 |
| Palestine [f] | 1996-2011 (10) | -1.33* | -1.71 | 0.38 | -1.27* | - | -0.77 |
| Russia [i] | 1961-2015 (25/36) | - | - | - | 0.60 | 2.42 | -1.85 |
| USSR | 1961-1989 (3/10) | - | - | - | 2.38 | 1.62 | 0.76 |
| Transition | 1990-1999 (6/10) | - | - | - | -5.38 | - | -9.59 |
| Present | 2000-2015 (16) | 2.89 | 5.17 | -2.29 | 4.09 | 2.37 | 1.72 |
| Turkey [f] | 1994-2016 (16) | 2.19 | 3.03 | - | 2.06 | - | 1.51** |
| United States [j] | 1962-2014 (51) | 1.47 | 1.03 | 0.43 | 1.42 | 2.28 | -0.86 |
| Transition | 1962-1980 (17) | 1.62 | 1.53 | - | 1.50 | 1.43 | 0.07 |
| Reaganomics | 1981-1992 (12) | 2.53 | 0.86 | 1.68 | 1.77 | 3.84 | -2.07 |
| Present | 1993-2014 (22) | 0.81 | 1.13 | - | 1.28 | 1.55 | -0.28 |

*Notes:* Growth and distribution of pre-tax income among individuals age 20 and over. Lognormal and gamma distributions fit to annual percentile income thresholds by maximum likelihood estimation. Under the lognormal distributional assumption, analytical values of $\bar{c} = exp(\mu + \frac{1}{2}\sigma^2)$ and $\tilde{c} = exp(\mu)$ enable decomposition of growth. Decomposition under a gamma distributional assumption follows from the mean, $k\theta$, when $c_i(t) \sim \Gamma(k(t), \theta(t))$ and thus $\bar{g} = g_k + g_\theta$. A decrease in shape parameter (k) suggests an increase in the skewness of the gamma distribution as $2/\sqrt{k}$. Statistically significant at the 1-per cent level or better using Huber–White standard errors unless otherwise noted (** 5 per cent level; * 10 per cent level).

[a] Number of years within the period where the lognormal/gamma distribution fits well (two-sample Kolmogorov-Smirnov test) in parentheses. [b] Average annual growth rate, in per cent.

*Sources:* Author calculations based on data discussed in [c] Morgan (2017); [d] Piketty, Yang, and Zucman (2017); [e] Czajka (2017); [f] Alvaredo, Assouad, and Piketty (2018); [g] Garbinti, Goupille-Lebret, and Piketty (2018); [h] Chancel and Piketty (2017); [i] Novokmet, Piketty, and Zucman (2018); [j] Piketty, Saez, and Zucman (2016), and available from World Inequality Database (WID), WID.world.