Policy Evolution Under The Clean Air Act

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Nearly half a century has elapsed since 1970, when the first Earth Day was celebrated, the U.S. Environmental Protection Agency (EPA) was established, and the U.S. Clean Air Act (CAA) was passed with essentially unanimous bipartisan support.[2] It was not the first Federal law to deal with air pollution – that was the Air Pollution Control Act of 1955 – and it was technically only an amendment to the original Clean Air Act of 1963 (Stern 1982). But it was the first environmental law to give the Federal government a serious regulatory role. The 1970 Act established the basic architecture of the U.S. air pollution control system and became a model for many subsequent environmental laws in the United States and abroad.

In this article, we describe and assess the evolution of air pollution control policy under the Clean Air Act with particular attention to the types of policy instruments used. This evolution was driven at various times by the emergence on the policy agenda of new problems, by innovation and experimentation by EPA, and by changes in the Clean Air Act itself. We begin by outlining the key provisions of the 1970 Act and the main changes Congress made to it over time. We then turn to a generic assessment of the major types of policy instruments that have been employed by EPA.

Finally, we trace and assess the historical evolution of EPA's policy instrument use.[3] Until roughly 2000, EPA made increasing use of market-based instruments, enabled in part by major amendments to the CAA in 1977 and 1990 that passed with overwhelming bipartisan support. In more recent years, however, environmental policy has become a partisan battleground. While EPA's interpretation of the CAA has continued to evolve, it has not been possible to amend it to enable an efficient response to climate change or to address other problems.

The Evolution of the Clean Air Act: 1970-1990

The 1970 Act was a response to increased environmental activism and fears that states would compete by lowering their environmental standards, as well as industry worries about facing a multitude of state-level mandates. This short, 24-page law gave the EPA Administrator considerable discretion and authority to set and change regulations and to enforce compliance.[4]

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[2] There was one negative vote in the House of Representatives, none in the Senate.

[3] A systematic study0 g6 [(Rein tT70.7becam9 Tw.5a8 a)]TJ -2118 0 0 094 Tw aala12.6210.4(a)2atey0 g6 [(u)-2. (ltit4(mi

The law contained four key provisions. First, the Administrator was charged with identifying pollutants that are produced by numerous or diverse sources and have "an adverse effect on public health or welfare" and with promulgating a system of National Ambient Air Quality Standards (NAAQS) for

Beginning in the late 1980s, climate change emerged as a significant issue. Then-candidate George H.W. Bush promised in 1988 to use the "White House Effect" to address the emerging problem of the greenhouse effect, and the Senate ratified the U.N. Framework Convention on Climate Change in October, 1992, without a roll-call vote. By the time legislation to deal with climate change received serious consideration in 2009, however, environmental politics had changed dramatically, with Congressional Republicans almost universally opposed to environmental regulation.

In June, 2009, the U.S. House of Representatives passed legislation – H.R. 2454, the American Clean Energy and Security Act of 2009 or the Waxman-Markey bill – that included an economy-wide emissions trading system to cut carbon dioxide (CO_2

The second panel of Table 1 examines the use of the four types of policy instruments across regulated sectors of the economy: electricity generation, other stationary sources, and mobile sources. The command and control mainstays of the original 1970 Act – technology standards and performance standards – have been used in all domains, while emissions trading has been applied only to stationary sources.

Most economists would agree that economic efficiency — achieved when the difference between benefits and costs is maximized — ought to be one of the fundamental criteria for evaluating environmental protection efforts (Pareto 1896; Kaldor 1939; Hicks 1939).[9] Discussions in the environmental policy realm, however, have more frequently employed a more modest criterion — cost-effectiveness (minimizing the costs of achieving some given objective) — largely because of the challenges of measuring the benefits of environmental protection. Assuming effective enforcement, on which all policy instruments depend for their effectiveness, and the same emissions objective, performance standards are at least as cost-effective as technology standards because they provide greater flexibility to minimize compliance costs.

When emissions from multiple sources are well-mixed, so that emissions from all sources produce the same damages per unit of pollution, cost-effectiveness requires that all sources that exercise some degree of emissions control experience the same marginal abatement cost (Baumol and Oates 1988). In principle, governments could employ non-uniform performance standards to bring about the cost-effective allocation of control responsibility among emissions sources with heterogeneous control costs, but to develop such a set of standards, the government would need to know the marginal abatement cost functions of all sources. Costs are generally heterogeneous, and the government rarely, if ever, knows sources' cost functions. As a consequence, command and control methods are rarely, if ever, cost-effective.

There are two ways the government can achieve the cost-effective allocation of control property in a cost-effective way. Some fifty years ago, Crocker (1966) and Dales (1968) proposed emissions trading systems that could provide such a market solution. Such systems are of two basic types: credit programs and cap-and-trade systems. Under credit programs, credits are assigned (created) when a source reduces emissions below the level set by existing, source-specific limits; these credits can enable the same or another firm to meet its control target.

Under a cap-and-trade system, an allowable overall level of pollution is established and allocated among firms in the form of allowances.[10] Firms that keep their emissions below their allotted level may sell their surplus allowances to other firms or, in many systems, bank them for later use. It is in the interest of each source to carry out abatement up to the point where its marginal control costs are equal to the market-determined price of tradable allowances. Hence, the environmental constraint is satisfied, and marginal abatement costs are equated across sources, satisfying the condition for cost-effectiveness.

Except under unusual conditions, the unique cost-effective equilibrium is achieved independent of the initial allocation of allowances (Montgomery 1972, Hahn and Stavins 2011). This independence property is a key reason why cap-and-trade systems have been employed rather than tax systems in representative democracies. The government can set the overall emissions cap and then allocate the available (and valuable) allowances among regulated sources to maximize support for the initiative without reducing the system's environmental performance or driving up its cost.

Even when the assumption that emissions are well-mixed is only approximately correct, taxes or emissions trading may still be superior to command and control if costs differ substantially across sources. If source-specific damages differ too much, however, command and control may be superior. If sources are relatively isolated, trading may produce "hot spots," areas of unacceptably high concentrations, without further policy protections. In addition, neither taxes nor emissions trading have been used to regulate mobile sources, though tradable performance standards have been employed, as we discuss below..

The Evolution of Policy Instrument Use

Under the original 1970 Act, all Federal air pollution regulation involved either technology or performance standards. At that time, some environmental advocates argued that implementing greater flexibility through tradable rights to emit pollution would inappropriately legitimize environmental degradation, while others questioned the feasibility of such an approach (Mazmanian and Kraft 2009). But, over time, as the Act was amended and EPA's interpretation of its provisions evolved, air pollution regulation evolved from sole reliance on conventional, command-and-control regulations to greater use of emissions trading.[11] This evolution has come to a halt in the last decade.

EPA's First Experiments with Emissions Trading in the 1970s

Beginning in 1974, EPA experimented with emissions trading among stationary sources through four programs – netting, bubbles, offsets, and banking. Under netting or bubbles, firms that reduced

[10] In some cap-and-trade systems most allowances are auctioned off, notably in the Regional Greenhouse Gas Initiative in the northeast United States (Burtraw et al. 2006) and the California cap-and-trade program (California Legislative Analyst's Office, 2017), but auctioning has not played an important role under the Clean Air Act. While abatement is certain under cap-and-trade regimes, allowance prices are not. Weitzman (1974) began a large literature comparing the two approaches under uncertainty.

[11] U.S. Environmental Protection Agency (2001) provides a comprehensive discussion of the use of economic incentives in all U.S. environmental protection programs through 2000, but it must be recognized that command-and-control regulations were still the norm (Hahn 2000).

emissions below the level required by law received credits usable against higher emissions elsewhere within the firm, so long as total, combined emissions did not exceed an aggregate limit (Tietenberg 1985; Hahn 1989; Foster and Hahn 1995). By the mid-1980s, EPA had approved more than 50 bubbles, and states had authorized many more under EPA's framework. Estimated compliance cost savings from these bubble programs exceeded $430 million (Korb 1998).

The offset program, which was explicitly authorized by the 1977 Amendments, allowed trades between firms. Firms wishing to establish new sources in areas that were not in compliance with NAAQS could offset their new emissions by reducing existing emissions through internal sources or through agreements with other firms. Finally, under the banking program, firms could store earned emission credits for future use, allowing for either internal expansion or sale of credits to other firms.

EPA codified all four programs in its Emissions Trading Program in 1986, but the programs were never widely used. States were not required to use the programs, and uncertainties about their future course may have made firms reluctant to participate (Liroff 1986). In addition, individual trades were subject to administrative approval, and trades were required to produce significant net emissions reductions, raising transactions costs. Nevertheless, companies such as Armco, DuPont, USX, and 3M traded emissions credits, and a market for transfers developed. Even this limited degree of participation in EPA's post-1974 trading programs may have saved between $5 billion and $12 billion over the life of the programs (Hahn and Hester 1989).

The Leaded Gasoline Phasedown in the 1980s

Lead in gasoline fouls catalytic converters, which were required in new U.S. cars starting with 1975 models to reduce emissions of carbon monoxide and hydrocarbons. To avoid this problem, the EPA required that only unleaded gasoline be used in cars with catalytic converters. In the late 1970s, there was growing concern about the threat of lead emissions to human health, and EPA began to phasedown gasoline lead beginning in 1979. It initially set different performance standards for refineries of different sizes to account for the higher compliance costs of smaller refineries, but smaller refineries still found it difficult to meet the requirements (Newell and Rogers 2007).

In late 1982, EPA launched a trading program aimed at reducing the burden of the phasedown on smaller refineries. Unlike a textbook cap-and-trade program, in which a fixed quantity of allowances is given or sold to compliance entities, there was no explicit allocation of allowances (Hahn 1989). If a refiner produced gasoline with a total lead content that was lower than the amount allowed, it earned lead "credits" that EPA allowed it to trade. This structure is sometimes referred to as a tradable performance standard. When EPA promulgated an accelerated phaseout of lead in 1985, they added a banking provision that allowed lead credits could also be saved for later use. This created an incentive for refineries to make early reductions in lead content to help them meet the lower limits that took effect over time.

Overall, this program, which was terminated at the end of 1987,[12] was successful in meeting its environmental targets (Anderson, Hofmann, and Rusin 1990; Newell and Rogers 2007), and resulted in leaded gasoline being removed from the market faster than anticipated. In each year of the program, more than 60 percent of the lead added to gasoline was associated with traded lead credits (Hahn and Hester 1989). This high level of trading far surpassed levels observed earlier under EPA's Emissions Trading Program in the 1970s. The level of trading and the rate at which the production of leaded gasoline was reduced suggest that the program was relatively cost-effective (Kerr and Maré 1997; Nichols 1997). EPA estimated that from 1985 through 1987, the program resulted in savings of approximately 20 percent relative to approaches that did not include trading (U.S. Environmental Protection Agency, Office

[12] By 1988, when a uniform performance standard was imposed, very little leaded gasoline was produced in the U.S. The 1990 Amendments banned all lead beginning in 1996.

of Policy Analysis 1985). In addition, the program provided significant incentives for cost-saving technology diffusion (Kerr and Newell 2003).

As the first environmental program in which trading played a central role, the lead phasedown program demonstrated that a trading system could be both environmentally effective and economically cost-effective. In addition, in contrast to the Emissions Trading Program, the lead phasedown program demonstrated that transaction costs in such a system could be low enough to permit substantial trade. The lack of a prior approval requirement was an important factor in the success of lead trading (Hahn and Hester 1989). Also, as in later trading programs, the ability to bank credits enabled significant cost savings and early reductions.

Stratospheric Ozone Protection

Following U.S. ratification of the Montreal Protocol in 1988, Congress imposed an excise tax on chemicals that deplete stratospheric ozone. The tax took effect in 1990 (U.S. Congress 1989). Beginning in 1989, EPA set up an emissions trading system for ozone-depleting chemicals (ODCs) that was expanded after the 1990 Amendments (Hahn and McGartland 1989). Producers were required to have adequate allowances. Limits were placed on both the production and use of ODCs by issuing allowances that limited these activities. Different types of ODCs have different effects on ozone depletion, so each ODC was assigned a different weight on the basis of its depletion potential. Through mid-1991, there were 34 participants in the market and 80 trades, but no studies were conducted to estimate cost savings.

The timetable for the phaseout of ODCs was subsequently accelerated, and the tax on CFCs was raised over time (Reitze 2001). It served as a windfall profits tax, to prevent private industry from retaining scarcity rents created by the quantity restrictions (Merrill and Rousso 1990; U.S. Environmental Protection Agency 2001). The tax may have become the binding instrument, but there was considerable debate regarding which mechanism should be credited with the ultimately successful reduction in the use of these substances, for which U.S. production ceased in 1995 (Cook 1996).

Sulfur Dioxide Allowance Trading

Throughout the 1980s, there was growing concern that acid precipitation – due mainly to emissions of SO_2 from coal-fired power plants – was damaging forests and aquatic ecosystems (Glass, et al. 1982). Because costs of reducing these emissions differed dramatically across sources, however, legislative proposals using command-and-control instruments failed to attract sufficient support. That changed with the 1990 Amendments, which addressed this issue by requiring EPA to launch the SO_2 allowance trading program, eventually covering all non-trivial power plants with a declining cap representing a 50 percent reduction from 1980 levels (Ellerman et al. 2000).

The government freely allocated allowances to power plants to emit specific quantities of SO_2, based primarily on actual fuel use during the 1985-1987 period.[13] If annual emissions at a regulated facility exceeded its allowance allocation, the owner could comply by buying additional allowances or reducing emissions – by installing pollution controls, shifting to a fuel mix with less sulfur, or reducing production. If emissions at a regulated facility were below its allowance allocation, the facility owner could sell the extra allowances or bank them for future use.

Although government auctioning of allowances would have generated revenue that could have been used – in principle – to reduce distortionary taxes, thereby reducing the program's social cost

[13] In addition, the statute required EPA to withhold about 2.8% of all allowance allocations each year, sell them at an annual auction, and return the proceeds in proportion to firms from which allowances had been withheld (Ellerman et al 2000).

(Goulder 1995), this efficiency argument was not advanced at the time. Because the entire investor-owned electric utility industry was subject to cost-of-service regulation in 1990, it was assumed that the value of free allowances would be passed on to consumers and thus not generate windfall profits for utilities. Just as important, the ability to allocate free allowances helped to build significant political support for the program (Joskow and Schmalensee 1998). Because of the independence property associated with cap-and-trade systems, the initial allocation of allowances could be designed to maximize political support without compromising the system's environmental performance or raising its cost.

The program performed well, with SO_2 emissions from electric power plants decreasing 36 percent between 1990 and 2004 (U.S. Environmental Protection Agency 2011), even though electricity generation from coal-fired power plants increased 25 percent over the same period (U.S. Energy Information Administration 2012). The program delivered emissions reductions more quickly than expected, as utilities made substantial use of the ability to bank allowances for future use. With continuous emissions monitoring and a $2,000/ton statutory fine for any excess emissions, enforcement was exceptionally stringent, and compliance was nearly perfect (Burtraw and Szambelan 2010).

Because emissions were not well-mixed and emissions from different power plants had different impacts, some worried that trading might produce "hot spots" of unacceptably high SO_2 concentrations. Computer models had predicted that plants that had the most impact on ecosystems had the lowest costs of reducing emissions, however. Subsequently, the pattern of emissions reductions was found to be broadly consistent with those predictions. No significant hot spots emerged (Ellerman et al. 2000; Swift 2004).[14]

The cost of the program was significantly reduced by the substantial deregulation of railroads in 1980, which caused rail rates to fall and thus reduced the cost of burning low-sulfur Western coal in the East (Keohane 2003; Ellerman and Montero 1998;

beginning of the first compliance period, which provided regulated entities with some degree of certainty, thereby facilitating their planning and limiting allowance price volatility in early years. As with the lead trading program, the absence of requirements for prior approval of trades contributed to low transaction costs and substantial trading (Rico 1995). Banking allowances was again important, accounting for more than half of the program's cost savings (Carlson et al. 2000; Ellerman et al. 2000).

Regional Programs under Clean Air Act Authority

Two other programs that merit attention were not Federal programs per se, but rather regional programs executed under Clean Air Act authority: the Regional Clean Air Incentives Market (RECLAIM) in the Los Angeles area, and NO_x trading in the East.

First, the South Coast Air Quality Management District, which is responsible for controlling emissions in a four-county area of southern California, launched the Regional Clean Air Incentives Market (RECLAIM) in 1993 to reduce emissions of nitrogen oxides (NO_x) and in 1994 to reduce SO_2 emissions from 350 affected sources, including power plants and industrial sources in the Los Angeles area, replacing command-and-control regulations (Ellerman, Joskow, and Harrison 2003). RECLAIM Trading Credits (RTCs) were allocated for free, with the NO_x and SO_2 caps declining annually until 2003, when the market reached its overall goal of a 70% emissions reduction (Ellerman, Joskow, and Harrison 2003). The compliance period was a single year, and banking was not allowed. A unique aspect of this program's design was its zonal nature: trades were not permitted from downwind to upwind sources, reflecting differences in marginal source-specific damages.

The program was predicted to achieve significant cost savings via trade (Johnson and Pekelney 1996; Anderson 1997), and by June 1996, 353 program participants had traded more than 100,000 tons of RTCs, with a value of over $10 million (South Coast Air Quality Management District 2018). Emissions at RECLAIM facilities were some 20 percent lower than at facilities regulated with parallel command-and-control regulations, hotspots did not appear, and substantial cost savings were achieved (Burtraw and Szambelan 2010; Fowlie, Holland, and Mansur 2012).

In the program's early years, allowance prices remained in the expected range of $500 to $1,000 per ton of NO_x. During California's electricity crisis in 2000-2001, however, some sources of electricity were eliminated, which required dramatic increases in generation at some RECLAIM facilities. This caused emissions to exceed permit allocations at those facilities, and, in the absence of a pool of banked allowances, resulted in a dramatic spike in allowance prices -- to more than $60,000/ton in 2001 (Fowlie, Holland, and Mansur 2012). The program was temporarily suspended. Prices returned to normal levels (about $2,000/ton) by 2002, with all sources rejoining the program by 2007. As of July 2018, the twelve-month moving average of NO_x prices was $2,530/ton (South Coast Air Quality Management District 2018).

The other regional program of interest is NO_x trading in the eastern United States. Under EPA guidance, and enabled by the 1990 Amendments, in 1999 eleven northeastern states and the District of Columbia developed and implemented the NO_x Budget Program, a regional NO_x cap-and-trade system. Given the significant adverse health effects of ground-level ozone (smog formed by the interaction of NO_x and volatile organic compounds in the presence of sunlight), the goal of the program was to reduce summertime ground-level ozone by more than 50% relative to 1990 levels (U.S. Environmental Protection Agency 2004). Some 1,000 electric generating and industrial units were required to demonstrate compliance each year during the summer ozone season.

The region covered by the program was divided into upwind and downwind zones, reflecting differences in source-specific damages, and allowances were given to states to distribute to in-state sources. Sources could buy, sell, and bank allowances within limits reflecting the seasonal nature of the ozone problem. Upwind states were given less generous allowance allocations as percentages of 1990 emissions. However, trading across zones was permitted on a one-for-one basis, and the two zones made similar reductions from baseline emissions levels (Ozone Transport Commission 2003).

In 1998, EPA had issued a SIP Call, which required 21 eastern states to submit plans to reduce their NO_x emissions from more than 2,500 sources. The Call created an interstate cap-and-trade program, known as the NO_x Budget Trading Program, which went into effect in 2003, replacing the NO_x Budget Program. In 2005, the NO_x Budget Trading Program was effectively replaced by the Clean Air Interstate Rule (CAIR), which reduced allowance allocations under the acid rain program. In July 2008, however, an Appeals Court ruled that the Clean Air Act did not give EPA authority to amend the acid rain program. Finally, in 2015, CAIR was replaced by the Cross State Air Pollution Rule (CSAPR), which does not allow interstate trading.

At the outset, the NO_x Budget Program market was characterized by uncertainty because some trading rules were not in place when trading commenced. This resulted in high price volatility during the program's first year, although prices stabilized by the program's second year (Farrell 2000). Overall, under the NO_x

EPA's proposal listed specific targets for each state, but gave the states many ways to meet their targets, including: increasing the efficiency of fossil-fuel power plants; switching electricity generation from coal-fired plants to natural gas-fired plants; developing new low-emissions generation (including renewable and nuclear generation); and more efficient end-use of electricity. States were also given flexibility to employ any of a wide variety of policy instruments, including market-based trading systems. Furthermore, states could work together to submit multi-state plans. The regulation was to be finalized in June, 2015 and implemented in 2020.

The state-by-state approach in the CPP did not guarantee cost-effectiveness, because under the formula employed, marginal abatement costs would vary greatly across states. However, encouragement was given to states to employ cap-and-trade systems, and EPA emphasized its willingness to consider multi-state implementation plans. Although EPA was not guaranteeing cost-effectiveness, it was certainly allowing for it, indeed attempting to facilitate it.

Because GHGs are well-mixed globally, climate change is particularly well suited to the use of market-based instruments. But this also means that global damages are unaffected by the location of emissions. Thus any jurisdiction taking action will incur the direct costs of its actions, but the direct climate benefits will be distributed globally. Hence, the direct climate benefits a jurisdiction reaps from its actions will almost certainly be less than the costs it incurs, even if global climate benefits are much greater than global costs. Despite this logic, the central estimate of annual net benefits (benefits minus costs) of the CPP in 2030 in EPA's Regulatory Impact Analysis (RIA) was $67 billion (U.S. Environmental Protection Agency 2014b).[17] How could this be?

Table 2 shows the two answers. First, EPA did not limit its estimate of climate benefits to those received by the United States, but used an estimate of global climate benefits. Second, EPA also quantified and included (the much larger) benefits of human-health impacts associated with reductions in correlated, non-GHG air pollutants.

It would certainly be inappropriate to use a global measure of benefits in analysis of all U.S. regulations (Gayer and Viscusi 2016). Doing so would imply that a labor policy that increased U.S. employment but cut employment in competitor economies would have zero benefits! On the other hand, it can be argued that counting only domestic benefits is inappropriate for a global commons problem (U.S. National Academy of Sciences 2017).

Suppose a domestic U.S climate benefits number were used in the RIA, rather than a global number. EPA estimated global climate benefits of the rule in 2030 using a mid-range 3% discount rate to be $31 billion. According to the Obama administration's Interagency Working Group on the Social Cost of Carbon (2010), U.S. benefits from reducing GHG emissions would be, on average, about 7 to 10 percent of global benefits. If U.S. benefits were thus 8.5% of global benefits, they would amount to about $2.6 billion, considerably less than the RIA's estimated total annual compliance costs of $8.8 billion. This validates the intuition that for virtually any jurisdiction, the direct climate benefits it reaps from reducing GHG emissions will be less than the costs it incurs.[18]

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new traditional coal plants, but since there were no new coal plants planned or likely to be built, due to the relative prices of coal and natural gas, the rule had no real impacts and was not particularly controversial.

[17] See note 3, above, for the role of RIAs in the regulatory process.

[18] There are abundant caveats to this simple analysis. One is that if the proposed U.S. policy increased the probability of other countries taking climate policy actions, then the impacts on U.S. territory of such foreign policy actions would merit inclusion even in a traditional U.S.-only benefit-cost analysis. Trying to quantify this effect would be speculative at best.

Table 1:
Major Categories of Pollutants and Sectors Regulated by the Clean Air Act

		Policy Instrument Used			
		Technology Standard	Performance Standard	Emissions Trading	Taxes
Pollutant Category	Criteria Pollutants	*	*	*	
	Toxic / Hazardous Pollutants	*	*		

Stratospheric
Ozone Depletion Ozone Done Dep

* Greenhw (e Gas TD .005371e Tc ( )TjJ 3.655 -1.15 TD 0 Tc 0 Tw 6.125285 1

Table 2:
Estimated Benefits and Costs of Clean Power Plan Rule in 2030
(EPA's Regulatory Impact Analysis, Mid-Point Estimates, Billions of Dollars)

	Climate Change Impacts from CO_2		Domestic Health Impacts from Correlated Pollutants plus …	
	Domestic	Global	Domestic Climate Impacts	Global Climate Impacts
Benefits				
Climate Change	$3	$ 31	$3	$31
Health Co-Benefits	--	--	$45	$45
Total Benefits	$3	$ 31	$48	$76
Total Compliance Costs	$9	$ 9	$ 9	$ 9
Net Benefits (Benefits – Costs)	- $6	$ 22	$ 39	$ 67

SOURCE: Authors' calculations, based on Table ES-7 (page ES-19) and Table ES-10 (page ES23) of June, 2014, Regulatory Impact Analysis of proposed Clean Power Plan rule (U.S. Environmental Protection Agency 2014b), adopting mid-point estimates, using 3% discount rate, and domestic shares of global climate benefits from the Interagency Working Group on Social Cost of Carbon (2010).

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