



Evaluating the Effectiveness of Environmental Offset Policies

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Robert Hahn and Kenneth Richards

Abstract

In the real world, taxes and cap-and-trade systems are rarely implemented in their pure form. In this paper, we examine a related approach that has been used widely in practice – which we refer to as an “offset.” The idea behind offsets is to encourage firms or entities that may not be a part of the main regulatory system to produce environmental improvements. These improvements can then be used to offset pollution reduction requirements for regulated entities. This paper analyzes how offsets are used in practice, and identifies key economic and political factors that help explain the use of offsets in certain situations. We find that offsets may often fail to take adequate account of environmental or ecosystem damages. We argue that the effectiveness of an offset policy depends on the political and institutional context in which it is developed.

Evaluating the Effectiveness of Environmental Offset Policies

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1. Introduction

One of the great ideas invented by economists was to suggest using “market mechanisms” to achieve a given environmental outcome at least cost (Pigou 1920; Crocker 1966). The two textbook examples most often presented in economics classes are taxes and cap-and-trade. Taxes are an example of a pricing instrument and cap-and-trade is a quantity instrument. Both of these instruments work by putting a price on pollution, thus encouraging those firms with the least expensive pollution control options to make use of them.

In the real world, taxes and cap-and-trade systems are rarely implemented in their pure form. In this paper, we examine a related approach that has been used widely in practice – which we refer to as an “offset.” A key goal of an offset program is to reduce the costs of achieving a particular level of environmental protection. The costs include not only compliance costs, but the additional private transactions costs and public administrative costs associated with the offset program.

The idea behind offsets is to get firms or entities that may not be a part of the core regulatory system to produce environmental improvements, which can then be used to offset pollution reduction requirements in the core regulatory system (Liroff 1980). So, for example, if a developer’s activities take away wetlands in one neighborhood, he may be required to provide an equivalent amount of wetlands, or eco-system services in another area.

In principle, offsets, like other market mechanisms, have the potential to dramatically lower the costs of achieving a given environmental quality target. For example, U.S. Environmental Protection Agency analysis (EPA) of the 2009 Waxman-Markey climate bill indicated that the abatement cost associated with the bill would be 20 percent higher if international offsets had not been incorporated (EPA 2009).

The paper analyzes how offsets are used in practice, and identifies key factors that help explain the use of offsets in certain situations. We analyze the rich array of applications of offsets in environmental and energy regulation. Examples are drawn from offset programs for limiting greenhouse gas emissions, maintaining ecosystem services for wetlands, achieving local air pollution goals, water quality protection, and promoting energy efficiency.

We find that while offsets may be theoretically attractive in some applications, there are many obstacles to their effective implementation. In general, there is a tradeoff between the certainty of achieving additional environmental protection, on the one hand, and the potential to lower the cost of regulation, on the other. We argue that the effectiveness of an offset policy depends on the political and institutional context in which it is developed.

Section 2 of the paper offers a new definition of offsets, and addresses issues in evaluating and designing these programs. Section 3 assesses the performance of offset programs in practice and presents three case studies. The political economy of offsets is examined in section 4. Finally, section 5 presents conclusions and suggests areas for future research.

2. Definition, Evaluation and Design of Offsets

Environmental offsets have been discussed broadly in the literature, but without a particular focus on this instrument (e.g., Stavins, 2003). Some analysts focus on offsets in the context of EPA's air emissions trading program started in the 1970's, where offsets are described as a mechanism to allow new sources of criteria air pollutants in nonattainment areas "provided they acquire sufficient emission reduction credits from existing sources" (Tietenberg and Lewis, 2008).

We suggest that the focus on U.S. EPA's emissions trading program produces too narrow a conception of environment offset programs. We examine several programs that are developed in conjunction with specific environmental regulations, either at the domestic or international level. Purely voluntary offset programs, such as those offered to travelers by airlines to offset their carbon emissions, are not included in our discussion because they are not formally connected to a core regulatory program through a law or regulation.

An environmental offset program is often associated with some form of regulation, which we call a *core regulation*. The core regulation could range from a command-and-control system to a cap-and-trade system. It constrains the activities of the covered parties, inducing them to engage in costly environmental protection activities.

Suppose now that there are other external activities that are not covered by the core regulation that, if implemented, would have the effect of promoting the same, or other, environmental goals. Suppose also that at least some of the activities outside the core regulation are less costly to implement. Clearly, if outside activities can be substituted for some of the more costly core activities, then the overall cost of meeting any environmental protection goal can be reduced. Ideally, the program will encourage private parties to undertake all offset activities that are lower cost than core activities, up to the point where the marginal cost of core activities and offset activities are equal.

Formally, we define an *offset policy instrument* as an addition to a core regulatory program that allows regulated entities to meet a portion of their obligations through environmental improvements that take place outside of the core. The offset activity is supposed to provide an environmental benefit at least as large in value as the environmental damage caused by the relaxation in the core program. Note that under this definition, a pure cap-and-trade system or emission tax would not involve an offset instrument because all sources would be covered by the core regulation. However, offset provisions could be added, as they were in the Waxman-Markey proposed legislation to curb U.S. carbon dioxide emissions.

Under our definition, some environmental programs are actually offset systems, even though they have not traditionally been recognized as such. For example, we would include wetlands mitigation banking programs and point-nonpoint source water pollution trading programs.

A critical question is how to evaluate the performance of an offset program. This involves setting a benchmark for comparison. Two benchmarks appear to be considered in the literature: First, comparing the offset policy to a situation in which there is no offset policy, but there is still a core regulatory program; and second, comparing an offset program to a situation in which there is a well-defined market-based program, such as cap-and-trade or taxes.

Note that using a benchmark of no-offset policy could lead to improvements in environmental quality and/or reductions in costs with the introduction of an offset policy. In contrast, comparing an offset program to a full-blown cap-and-trade program that is functioning well will very likely lead to the result that the offset program is more costly to administer, and quite possibly worse in environmental terms because offset programs generally fail to put clear boundaries around regulated activities.

It is important to consider indirect as well direct costs and benefits of offset programs in evaluating them. Two important indirect costs of offset programs that have received little attention in the literature are the deadweight costs associated with revenue loss and the potential costs of subsidizing an activity. If an offset program decreases revenues going to the government, say because an offset replaces revenues from an emissions tax or the auctioning of an allowance, then the deadweight loss associated with raising the revenue from somewhere else needs to be taken into account. This cost could be on the order of \$.30 for each \$1 of lost revenue (Goulder et al., 1999). In addition, to the extent that offsets represent a subsidy to pollution generating activities, they can encourage inefficient entry into particular industries. To our knowledge, these costs have not been estimated. It is possible, that after accounting for the welfare impacts associated with revenue losses, offsets may fail to pass a benefit-cost test.

Some scholars suggest that there are significant indirect benefits of offsets as well. One possible benefit is that offset policies have served as a stepping stone to the use of more efficient market-based approaches, such as the allowance trading system to reduce sulfur dioxide emissions in the U.S. This is plausible, but difficult to evaluate. For example, industry may have become more supportive of market-based approaches as experience was gained (Haddad and Palmisano, 2001). This argument in support of offsets must be weighed against the possibility that the existence of offset programs in developing countries may make it less likely that they will sign on to binding emission caps without additional payoffs of some kind.

A second indirect benefit is that offsets, by subsidizing certain technologies, may give a boost to the development of these technologies. In general, we believe that offsets represent an inefficient way of providing subsidies for particular kinds of energy sources such as renewables because that is not usually their primary objective.

2.1 Design Challenges and Possible Solutions

The design of an environmental offset program entails several critical elements. To implement an offsets program, it is necessary to (1) determine the basis for identifying equivalency between the environmental impacts of the regulatory relaxation and the offset effort (2) develop an ex ante offset certification process for certifying credits, including the timing of rewards, (3) identify the dimensions and unit of measure for the environmental tradeoff (e.g., metric tons of emissions, acres of land), (4) determine the type(s) of reference cases (baselines) that offsetters will be allowed or required to use, (5) determine the ex post monitoring requirements, (7) choose a trading ratio for the exchange between offset credits and regulatory relief, and (8) choose a form of reward (e.g., additional allowances, relief from a regulation).

A key point is that all these design challenges apply generally to market-based systems, such as taxes or cap-and-trade systems. An important difference, however, is the establishment of a reference case. In a pure cap-and-trade system, for example, an emitter will be required to have a number of allowances that corresponds to his actual emissions. That is, his baseline will frequently be zero. In contrast, the baseline under most offset instruments is less clear.

The basic design challenge for offset systems is to balance the goals of protecting the environment (*i.e.*, assuring the offset program does not compromise the core regulation), increasing the use of low cost offset opportunities, and reducing the transactions costs of implementing the offset program. There are many aspects of an offset system that may introduce uncertainties in the actual level of environmental quality that will be achieved. For example, the environmental property rights for even the core group of regulated entities may not always be

well-defined under the current regulatory system because there may be a difference between a firm's actual activities and what it is permitted to do.

When the offsetting activities are outside the core regulated activities, the baseline level of emissions can be even more difficult to assess because the regulator must guess at the appropriate reference case (Montero 2000). Regulators may want to take into account the fact that the firms most likely to participate in the system are those that get a generous level of property rights relative to their actual baseline level of pollution. This is sometimes referred to as the problem of adverse selection. The baseline problem is further compounded by the fact that some firms involved in generating offsets may be able to affect their baseline allocation by strategically manipulating the regulatory decision maker (sometimes referred to as the problem of moral hazard). The baseline problem raises two distinct issues: first, when a firm or agent reports reducing a ton of emissions, the expected reductions may be less than a ton; and second, the uncertainty around the total level of emissions reductions may be considerable.

In general, there are no simple solutions to the offset design problem. There are three different kinds of solutions that have been proposed. One is to give out enough property rights to all possible offsetting parties so that there will be no issues with adverse selection or moral hazard. The problem with this approach is that it may not be feasible politically because it could involve significant wealth transfers. A second approach is to try to get better information on the nature of the baseline or counterfactual. This approach has serious limitations: it is frequently hard to identify what a firm or industry's emissions might have been in the absence of the offset policy. Bushnell (2011) suggests a randomized trial to ascertain the likely impact of an offset policy. Even if this were doable, we are not sure how much light it would shed on establishing a relevant baseline for particular entities that are eligible to participate in such a program. A third approach is to impose restrictions on the offset program itself. Examples include putting a cap on offsets, taxing offsets relative to other types of transactions, and limiting who can trade offsets. These measures may limit the number of offset trades, but will not likely do much to address problems associated with adverse selection or moral hazard. Precisely because there are fundamental problems with design, we see offsets as a second-best approach that should be employed when other more efficient approaches are not likely to be politically or legally feasible.

3. Offset Programs in Practice

While offset programs are simple in concept, experience suggests that administration of offset programs has been challenging in practice. To explore the potential of the offset approach and the factors that limit their effectiveness, we consider the experience with several different types of programs. First we provide an overview of several offset programs, examine how those programs are structured, and assess the impacts they have had on environmental outcomes and cost. Then we examine three programs in detail: the Clean Development Mechanism for greenhouse gas emissions, point-nonpoint source water pollution trading programs, and wetlands mitigation banking.

While carbon offsets may be the most visible application in the media and academic literature, the concept of offsets substantially predates the focus on climate change. Offsets provisions were not only incorporated into the Clean Air Act in 1977, but have been integrated into a wide range of environmental regulations and programs, covering water quality, habitat protection, and energy as well as greenhouse gases and air quality. Geographically the offsets approach has

been employed in many different countries, including the United States, Canada, the United Kingdom, France, Italy, Brazil, Australia and New Zealand, and at local, state/provincial, national, regional and global scales.

Table 1 provides an overview of a number of offset programs that illustrate the basic nature of this approach as well as the breadth of its application. No attempt was made to provide an exhaustive list of offset programs.

The table illustrates two points. First, offsets have been used in a wide range of applications, including greenhouse gas emissions, air and water quality, energy, and habitat and biodiversity. Second, consistent with the definition of offsets, all programs in Table 1, with one exception, include a combination of a core regulation and an offset program intended to lower the costs of the core program without compromising the environmental goal. The exception is the Montgomery County tradable development right program, which we include to illustrate the boundaries of the offset concept.

Below we present three case studies in offsets: the Clean Development Mechanism, non-point source water pollution, and wetlands. These studies illustrate the wide range of applications of this mechanism, as well as the challenges in design that arise in different contexts.

3.1 The Clean Development Mechanism

The goal of the Kyoto Protocol's first commitment period is to reduce the aggregate developed nations' greenhouse gas emissions by 5.2 percent below 1990 levels between 2008 and 2012, the first commitment period. These emissions include the six greenhouse gases: carbon dioxide, methane, nitrous oxide, hydrofluorocarbons (HFCs), perfluorocarbons, and sulphur hexafluoride.

To reduce the costs of achieving this goal, Kyoto includes mechanisms such as emissions trading among participating countries. The Kyoto Protocol also includes the Clean Development Mechanism (CDM), which allows Annex I nations (developed nations with individual emission budgets) to implement emissions reduction projects in non-Annex I nations that are *not* subject to any caps. These projects generate certified emission reductions (equivalent to one metric ton of CO₂) that can be used by Annex I nations to help achieve their emission reduction requirements.

The Kyoto Protocol defines two goals for the CDM: cost savings in meeting carbon emission budgets of Annex I nations and the promotion of sustainable development. The CDM can meet the first goal by letting Annex I nations implement projects in developing nations, where emission reductions often can be accomplished at a much lower cost than in developed nations.

CDM projects have been significant by some measures. According to the UNFCCC (2010), there are about 2,500 CDM projects registered. As of November 2010 certified emission reductions equivalent to a total of 451 million metric tons of CO₂ emissions had been issued since the start of the CDM. The World Bank estimates that from 2008-2012 the CDM will provide offsets equivalent to 1 billion metric tons CO₂ equivalent. One main driver in the expansion of the CDM has been the European Union Emissions Trading Scheme (EU ETS) that has allowed for certified emission reductions from the CDM to offset domestic emissions (Ellerman and Buchner 2007).

To date, most certified emission reductions have been issued for projects in a small number of countries. China is the largest generator of CERs, accounting for 51%, followed by India with 18%, then South Korea with 13%, and Brazil with 10% (UNFCCC 2010).

When assessing the CDM it is important to note that the objective of the CDM is not to reduce greenhouse gas emissions *per se*, but to reduce the cost of greenhouse gas emission

control and promote sustainable development of non-annex I countries. It may also be a key factor that gets countries to participate in an agreement.

CDM has faced some major problems, and its effectiveness has been debated in the literature (Millard-Ball 2012, Victor 2011, and Grubb et al. 2009). A key concern is whether emissions reductions have been additional, in the sense of truly reducing greenhouse gas emissions on a local or global basis. Under recent rules, project developers can employ one of over 130 approved methodologies for determining baselines and monitoring the project, or they can submit a new approach to an oversight UN board for approval (UNEP 2010). Development of such methodologies is costly and time-consuming. Furthermore, there are strong incentives for host countries and buyers of the certificates to overstate the baseline emissions.

There is evidence that some projects that are not additional have been approved by the executive Board. For example, in China all renewable energy projects were deemed eligible to receive CDM credits, even though China's energy policy already calls for some renewable energy development to meet growing demand (Wara and Victor 2008).

Despite a significant amount of research into the area of whether emission reductions would actually occur (which is sometimes referred to as "additionality" in the CDM literature), there is a paucity of good data on this subject. For example, Schneider (2009) assessed 93 CDM projects that were sampled from the 768 projects registered as of July 2007 to evaluate the methodologies used to demonstrate emission reductions. His analysis concludes that "...that the current tools for demonstrating additionality are in need of substantial improvement. ... In a considerable number of cases it is questionable whether the emission reductions are actually additional." (Schneider, 2009, p. 242). Similarly, Michaelowa and Purohit (2007) evaluate 52 CDM projects in India, finding that only about half of the projects even considered alternative projects in their assessment of emission reductions. The authors assess 19 of the projects in more detail, and find that fewer than half of the large projects provide sufficient information about whether these projects would yield emission reductions in their documentation.

The main objective of the CDM is to reduce the cost of emission abatement for Annex I nations, but some evidence suggests that CDM may actually confound cost-effective pollution reduction in some cases. For example, abating all HFC-23 emissions in the developing world is estimated to cost approximately \$31 million per year. Because of the subsidy implicit in CDM credits, however, industrialized nations spent significantly more (€270-€750 million, or roughly \$400-\$1200 million) to achieve these reductions in 2005 (Wara 2008). According to Hepburn (2007), the prices paid for certified emission reductions are approximately ten times the actual marginal cost of HFC-23 removal. In fact, the removal of HFC-23 for the generation of certified emission reductions is more lucrative than production of the refrigerant from which the HFC-23 is a byproduct (Wara and Victor 2008). In an effort to address this problem, rules were later changed to restrict HFC-23 projects.

Several studies have considered the transactions costs associated with the CDM mechanism. Examples of transactions costs include the costs of applying for credit, monitoring projects and reporting results. Transaction costs are often cited as a serious challenge to the successful deployment of CDM. CDM likely has significantly higher transaction costs than a pure cap-and-trade regime among regulated sources because each project has to be approved.

Some scholars have noted that the CDM market is working reasonably well because of the level of activity that is observed. We would agree that the CDM market is functioning, but the real question is whether its benefits justify its costs. We do not really know the proportion of projects that actually result in incremental emission reductions, for example, and do not have

very good data on costs.

3.2 Nonpoint Source Water Pollution

Most water pollution regulations under the U.S. Clean Water Act focus on point sources of pollution, which are required to obtain permits under the National Pollution Discharge Elimination System (NPDES). However, in many cases, it is much less costly for unregulated nonpoint sources (such as agricultural runoff) to reduce pollution discharges than for point sources to install control technologies. Allowing point sources to offset their own emissions by providing or paying for nonpoint reductions may achieve the same water quality goals but at lower social cost.

The Clean Water Act does not specifically authorize water quality trading, and does not even directly address nonpoint sources of water pollution. This creates a regulatory environment where offset programs face legal constraints and barriers (Woodward et al. 2002). However, there are at least two regulatory mechanisms under which trading programs may be implemented – one that imposes constraints on groups of emitters and one that affects individual firms.

The programs that best fit with the definition of offset programs tend to use clearinghouse structures, which convert nonpoint source reductions to credits that point sources can purchase. For several of these programs, however, no trading has occurred because restrictions on point sources are not binding (Cherry Creek), nonpoint source reductions are considered more costly than point source reductions (Long Island Sound), or the point source driving the program goes out of business before the program is complete.

Measurement creates a particular challenge for the inclusion of nonpoint sources in offset markets. Nonpoint source pollution generally cannot be directly traded because it cannot be accurately measured at a reasonable cost. Compared to air pollution emissions, which are relatively fungible, water pollution exchanges between sources (especially involving nonpoint sources) are less comparable and more expensive to measure (King and Kuch 2003). In most offset programs and agreements, credits are generated based on an *ex ante* basis, with certain activities that are expected to result in pollution reduction—not for actual, measured reductions (Fang, et al. 2005).

Water quality trading programs are implemented at the watershed level, which introduces further complications when compared with air quality trading programs. For example, watersheds are generally much smaller than airsheds, which make for much thinner markets (Woodward, et al. 2002). Also, although nonpoint sources are not federally regulated under the CWA, many states and localities have best management practices regulations. These alternative regulations can complicate identification of the reference case or baseline. For example, programs that expand “green payments” to farmers for practicing good land management change the baseline for nonpoint source runoff.

It is difficult to establish equivalency between point discharges and nonpoint nutrient discharge reductions and in calculating the reductions from these projects. This lends support for case-by-case, regulator-approved trades rather than a “commodity-style” market system, which is likely to increase transaction costs.

In much of the literature, it appears that the mark of a successful program is that actual trades occur. However, in most cases, the stated program objective is for environmental quality goals to be achieved at the lowest economic cost. In some cases, programs were declared successful even in the absence of trading because this larger goal was recognized. In other cases, the

program was deemed a success because it facilitated industrial growth in an area (usually a one-time agreement) without sacrificing environmental quality (Breetz et al. 2004). None of the cases cited in the literature resulted in decreased water quality, which means that there is no evidence that any program so far has sacrificed environmental targets in pursuit of trading.

Some evidence suggests that point/nonpoint trading can result in significant cost savings. For example, a preliminary economic analysis of a program in the Miami Conservancy District in Ohio estimates that nonpoint source reductions that could meet point source demand for credits could be achieved at approximately 8-26 percent of the costs of using point source controls exclusively (Keiser & Associates 2004), and that total savings from the offset program could total \$314 - \$385 million (EPA 2008).

Despite a high degree of interest in water quality trading, especially between point and nonpoint sources, most projects to date have not experienced a high trading volume, and many pilot programs have not been scaled up to full deployment (EPA 2008). Fang et al. (2005) suggest that the two main reasons for the low level of trading are high transaction costs and opposition from environmental groups.

There is a great deal of support in the literature for transaction costs being a major hurdle to the successful deployment of water quality offset programs (Breetz et al. 2004; Fang et al. 2005; Morgan and Wolverton 2005). These include permit negotiation, trade partner searches, administrative expenditures, communications between regulated sources and regulatory agencies, credit verification, post-project site inspection, and routine project management (Fang et al. 2005).

To account for uncertainty, trading ratios are sometimes used. For example, two or more units of nonpoint pollution reductions may be required to allow for a one-unit increase in point source discharges. However, this can reduce the cost savings associated with nonpoint source offsets and also puts an effective tax on exchanges.

The picture that emerges from point-nonpoint source trading is one of little trading because transaction costs are high. A substantial number of trades appear to be one-time offset agreements between individual companies and state regulators. For the trades that have occurred, it appears that environmental quality has been preserved. Furthermore, simulations suggest there is a potential for cost savings.

3.3 Wetlands

Wetlands provide many ecologically and economically valuable services, including, water storage, sediment trapping, and wildlife habitat (Hoeltje and Cole 2007). Section 404 of the Clean Water Act governs the nation's wetlands. It is jointly implemented by the Army Corps of Engineers and the EPA. Anyone wishing to dredge or fill a wetland for development (or any other purpose) is required to obtain a permit from the Corps.

Wetland mitigation is widespread. Every year, about 47,000 acres of new wetlands are required to compensate for 21,000 acres of permitted losses (Kihlslinger 2008).

Mitigation banks began in the late 1980s and early 1990s (Robertson 2006). Instead of constructing mitigation wetlands individually, permit holders can instead purchase credits from a bank.

Mitigation banking has grown rapidly in the last 20 years. There were 219 active banks with 50,000 hectares in 29 states by 2002 (Spieles 2005), and today there are more than 300, with 78 percent of them being for-profit ventures (Robertson 2006; Hough and Robertson 2009). Of the active banks, 75 of them are sold out of credits (Hough and Robertson 2009). Overall,

mitigation banks account for about a third of all mitigation wetland acres in the United States.

Performance standards are a critical element of wetland mitigation banks. Although there are no national standards in place to guide performance goals, the vast majority of cases employ performance standards focused on vegetation (Spieles 2005; Matthews and Endress 2008). These standards are often established somewhat arbitrarily without reference to a particular natural wetland and without measurable targets (Matthews and Endress 2008). Although some performance standards are objective and measurable, others include highly subjective measures of success, such as “good survival” of planted trees.

Enforcement is frequently a problem in wetland mitigation. Monitoring, submission of reports, and long-term maintenance are frequently performed poorly (Kihslinger 2008). Further, the Corps does not generally write penalties into permits, so they have no legal recourse in the case of noncompliance (GAO 2005).

Most studies of ecological equivalency between natural and compensatory wetlands have discouraging results, with only about 17-21 percent of mitigation sites adequately replacing the functions of lost wetlands (Kihslinger 2008). Matthews and Endress (2008) found that only 23 out of 76 evaluated sites achieved all of their stated goals, and 8 failed to achieve any goals. Further, the sites that achieved a higher proportion of goals virtually always had fewer goals and lower thresholds for success.

System-wide impacts are also a major concern (Hoeltje and Cole 2007; Kettlewell et al. 2008). Often, evaluations of individual projects do not consider watershed-wide impacts that accrue when multiple wetlands are replaced via mitigation projects. This is especially important given that many wetlands functions are heavily dependent on their surroundings. Many mitigation wetlands are located far away from or even outside the watershed of the natural wetland they are intended to replace.

The picture that emerges from wetlands mitigation is mixed. There is more trading than in some other programs, such as point-nonpoint source offsets. The net environmental impact is not always positive, though; markets exist, but are thin; and establishing equivalency across different kinds of wetlands is difficult.

4. Why are there Offset Programs?

Based on theory and practice, we believe that offset policies are an imperfect economic instrument that can often serve useful political ends. To see why offsets are likely to be inefficient, consider the following thought experiment. Suppose property rights were well-defined, easily enforced, and governments could agree on an environmental target. Then, it would be more efficient to have a cap-and-trade system than a cap-and-trade system combined with an offset system.

Yet, notwithstanding the fact that offset policies may be inefficient, we observe these policies in many applications. We believe there are several reasons for the use of offsets, the primary ones being political and legal.

Suppose that voters can only imperfectly monitor the actions of politicians and legislators. Then, an offset regime can sometimes offer decision makers an opportunity to deceive voters. For example, politicians may promise a certain amount of environmental quality (say greenhouse gas reductions in the United States). They may then introduce an offset provision that gives the appearance that it will allow equivalent reductions, but because of possible gaming on the part of participants, will not deliver as promised. In a sense, this is a win

for politicians because they can gain politically while shifting the general costs (in terms of less environmental quality than promised) to the poorly informed public.

A related explanation for offsets is that there may not be agreement about the underlying distribution of property rights. For example, a developed country might be willing to agree to a firm goal on a greenhouse gas emissions target, but a developing country might not. In this case, the developing country might find it advantageous to allow an offset program (say in the form of CDM) to proceed without agreeing to a firm target. Indeed an offset program may allow that country and project developers in that country to claim they are helping to solve the problem. At the same time, the developing country is not committing itself to a firm target, but is giving itself the option of extracting rents (in the form of financial transfers from the developed country) in the future.

Some proponents of offsets have argued that encouraging countries like China to participate in an offset system may make it more likely that these countries join a cap-and-trade regulatory regime later on. While it is possible that such regimes may increase a country's monitoring and enforcement capabilities, they also make it *less* attractive for that country to join a cap-and-trade regime because of the benefits that now accrue to the country under the offset regime (Hepburn, 2007).

Similarly, some commentators have argued that the Clean Development Mechanism may make it more likely that developing nations will eventually agree to nationwide targets on carbon dioxide or greenhouse gas emissions. For example, Ellis et al. (2007) argue that a long-term international commitment to a mechanism like CDM may actually encourage more countries to make commitments early on, so that they can take advantage of low-cost reductions at the outset instead of leaving the lowest cost reductions available for CDM investors. But if most of the low-cost reductions are used up, it may make it harder to economically and politically justify stringent developing country targets later. A related argument is that offsets may allow developing nations to develop institutional capabilities, which could increase their demand for reducing climate change emissions. While offsets may affect the demand for environmental quality in a country, the size of the effect is likely to be small.

Rather than encouraging countries to make binding commitments, we believe that the CDM as implemented currently is likely to have the opposite effect. A country will typically weigh the costs and benefits of entering into a climate agreement with binding targets. Consider three possibilities for a developing country: 1) it has no binding target; 2) it has no binding target and uses CDM; or 3) it has a binding target. From the perspective of the developing country, purely on narrow economic grounds, situation 2 is likely to be preferable to 1 because CDM increases revenues coming into the country without imposing any significant costs. At the same time, CDM actually *raises* the opportunity cost for that country of entering into a binding agreement at a given level – that is option 3 becomes relatively less attractive to the developing country. The reason is that CDM serves as an effective subsidy to the developing country. In this sense, CDM may create perverse incentives, since governments in developing countries have incentives *not* to impose emission restrictions if it means that they can set a relaxed baseline and attract lucrative CDM investments.

While offsets may make it less likely that a country not participating in an agreement will join an agreement later on, they may increase the likelihood that some countries will participate in an agreement at the outset. This is because offsets can lower the cost of achieving the stated target for those countries that agree to a binding target.

Another possible explanation for the use of offsets is that they help address potential legal constraints. Consider the case of climate change regulation. There are two basic strategies to reduce net carbon emissions—reduce emissions and increase forest carbon sequestration. One approach to reducing net emissions would be to include both activities under a core regulation that not only required sources to reduce emissions, but mandated the establishment of tree plantations on certain private land holdings. However, the latter law would not likely be constitutional in the United States; in general, the government can prevent harms (regulate emissions) but not require private production of benefits (capturing carbon dioxide). Similarly, the government can prevent the destruction of wetlands, but it cannot force landowners to create new wetlands without compensation. Thus, even though activities such as carbon sequestration and wetlands creation are effective ways to provide environmental quality, they must be induced through the offset system rather than required by regulation.

Finally, offsets may be seen as an evolutionary step in moving away from command-and-control regulation that may be attractive precisely because it is incremental in scope. In that sense, they may serve as a bridge between command-and-control systems and cap-and-trade systems. They can provide some flexibility in reaching a regulatory goal, while at the same time, imposing restrictions that certain interest groups and elected officials view as desirable.

A critical question for policy makers and analysts interested in promoting economic efficiency is how to move from an offset regime to an approach that likely to be more efficient and effective. The answer will differ in specific applications, but the broad outlines of a solution must rely on a careful analysis of the political winners and losers associated with various reforms. Victor (2010), for example, argues that the U.S. could create an offset system under CDM that both encourages greater competition among suppliers and imposes higher quality thresholds for certifying offsets. It remains to be seen whether the U.S. would want to impose such thresholds, but the idea is clever in that it relies on a third party – in this case, the U.S. – to help transform the status quo.

Conclusions and Areas for Further Research

This paper provides a survey and synthesis of the literature on the use offsets. We began by developing a definition for offsets because the term has been used loosely in the literature. Our definition focuses on offsets as a mechanism for reducing the cost of core regulatory activities while still trying to maintain the same, or a comparable, level of environmental quality. We then analyzed theoretical justifications for offsets as well as how they are used in practice.

The economic theory of offsets is fairly well understood. With perfect information, theory requires equating marginal abatement costs across sources. When one allows for the fact that offsets could involve gaming on the part of the regulated entity or a country, the theory is more complicated, and suggests that offsets could easily give rise to inefficiencies and reduce environmental quality. The incentives to overstate emission reductions need to be taken into account in assessing the likely effectiveness of offsets.

There is a need for more realistic empirical modeling of the impact of offsets. Many of the models of offsets that are used to assess costs savings and environmental impacts assume, for simplicity, that they will be implemented effectively. Yet, this does not appear to be the case. Furthermore, the impact of offsets on government revenues and production of pollution-intensive goods also deserves more careful study.

In addition to examining the performance of some offset programs, we discussed the reasons why offsets are used. We believe offsets are probably best understood in terms of a political compromise regarding market-based approaches. Politicians wish to allow some flexibility in achieving an environmental outcome, but may be unwilling or unable to specify property rights completely. For example, in the case of climate change, the U.S. has very imperfect control of sources of greenhouse gas emissions outside of its jurisdiction.

Policy makers and scholars may be interested in identifying conditions under which offsets are likely to “work.” Measuring whether an offset program works depends on the objectives that policy makers have for that program. If, for example, the objective is to subsidize certain “green” investments, then offsets may work, but they should be compared against relevant alternatives, such as direct subsidies or auctions. If the objective is to ensure a particular environmental goal is met in a cost-effective manner, then offsets may be problematic because of problems in specifying the counterfactual. Still, if it is not possible to regulate in a way that includes all sources of emissions, then offsets may be a reasonable alternative. If offsets are used, care should be taken to ensure that the likely results of an offset policy are made clear to the public and interested parties. We recognize, however, that such transparency may not be desired by politicians, who may have incentives to hide some of the less desirable features of an offset regime. Thus, academics can play a useful role in characterizing how these policies actually work.

Our bottom line is that offsets are not a panacea, either in terms of promoting economic efficiency or ensuring a prescribed level of environmental quality. Still, we think in certain cases, they represent a reasonable approach for making environmental and economic progress on difficult challenges. The devil is frequently in the details, and these details need to be studied carefully before making an informed judgment about the likely effectiveness of particular offset programs.

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Table 1: Examples of Offset Provisions

Core Program	Offset Provisions
Greenhouse Gases	
Kyoto Protocol	Clean Development Mechanism (2000-) Joint Implementation (2008-)
UN Framework Convention on Climate Change	Activities Implemented Jointly (1999-)
Alberta Climate Change and Emissions Management Act	Offset Credit System (2007-)
Air Quality	
U.S. Clean Air Act	Offsets Provisions, Clean Air Act Amendments of 1977 (1977 -)
South Coast Air Quality Management District Regulations(California)	1610 Car Scrappage Program ((1993 -)
Water Quality	
Protection of Environmental Operations Act 1997 (New South Wales, Australia)	Green Offsets for Sustainable Development (2003-2005)
U.S. Clean Water Act, NPDES	Point-nonpoint source trading - Cherry Creek, Colorado (1997-)
U.S. Clean Water Act, NPDES	Impact offset provisions - Oregon Revised Statute (2002-)
U.S. Clean Water Act, NPDES	Tar-Pamlico Nutrient Program, North Carolina (1990-)
Energy	
EU Directive of Energy End Use Efficiency and Energy Services	White certificates, Italian National Energy Efficiency Plan 2007(2005-)
Texas Senate Bill 7 (1999): Texas Electric Restructuring Act and Senate Bill 20 (2005)	Renewable Energy Certificates Provisions of Texas Utilities Code (2001-)
European Renewable Energy Directive	UK Renewables Obligation Certificate (2002-)
Habitat and Biodiversity	
Canadian Fisheries Act	Policy for the Management of Fish Habitat (1976-)
U.S. Clean Water Act, Section 404	Joint US EPA –Army Corps of Engineers Memorandum of Agreement (1990) (1975-)
U.S. Endangered Species Act	Conservation Banking (1995-)
New Zealand Resource Management Act	Section 108(2) Resource offsets (1991-)
Brazil National System of Conservation Units	Article 36 conservation units
Australia Environmental Protection and Biodiversity Act	Victoria Vegetative Offsets (1999-)
Montgomery County, Maryland, USA, Master Plan	Transferable Development Rights (1980-)
European Union 1979 Birds Directive 79/409/EEC; 1992 Habitats Directive 92/43/EEC	Compensatory Measures on Natura 2000 Network

Smith School of Enterprise and the Environment
University of Oxford
South Parks Road
Oxford, OX1 3QY
United Kingdom

E enquiries@smithschool.ox.ac.uk
T +44 (0)1865 614942
F +44 (0)1865 614960

www.smithschool.ox.ac.uk/research/stranded-assets/

