TRANSACTION COST IMPLICATIONS OF TWO APPROACHES TO FORESTS IN CLIMATE CHANGE POLICY: NEW ZEALAND AND CALIFORNIA

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Abstract

New Zealand and California present an opportunity to assess how two different designs for incorporating forests in climate policy affect transaction costs for participants in the forest sector. Forests play a prominent role in achieving the greenhouse gas (GHG) emissions reduction goals established by each policy. In New Zealand, the forest sector provides an important option for domestic GHG emissions reductions in an economy where opportunities in other sectors, like agriculture and energy, may be limited. In California, offsets from forests are projected to have the greatest technical potential of any approved offset project type, and will be an important option for reducing the costs of compliance in regulated sectors. This research investigates the different approaches taken by New Zealand and California, the circumstances surrounding each policy, and the transaction cost implications for forest participants under each programme.

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Introduction

The growing concentration of greenhouse gases (GHGs) in the earth's atmosphere threatens the stability of the global climate system (IPCC, 2007). Addressing climate change will require strategies to control anthropogenic GHG emissions to the atmosphere. The policies enacted by New Zealand and California are at the forefront of global efforts to confront climate change, and rely on a market-based mechanism known as "cap-and-trade." By equating the marginal costs of emissions reductions among emitters, cap-and-trade programmes can provide an economically efficient means of reducing GHG emissions. With origins in the United States' trading programme to reduce NOx and SOx pollution under Title IV of the Clean Air Act (§§401-416), cap-and-trade programmes have since been adopted by multiple jurisdictions for the reduction of GHG emissions (Solomon, 1999). This has included countries party to the Kyoto Protocol, such as New Zealand, as well as programmes developed by sub-national entities, including the Regional Greenhouse Gas Initiative in the northeastern United States and California's Global Warming Solutions Act of 2006.

Forests can serve as substantial sinks of carbon, and can offer one of the most cost-effective means of addressing atmospheric GHG concentrations (Dixon et al., 1994). These systems sequester atmospheric carbon dioxide via photosynthesis, and can store this carbon for long periods in organic material. The climate policies developed by New Zealand and California are unique in that they are the only two comprehensive cap-and-trade programmes to include the forest sector. However, while both seek to harness the carbon sequestration potential of forests in reducing net GHG emissions, the policy designs for doing so are divergent. In New Zealand, forests are included as a capped sector under the country's cap-and-trade programme (Boston, 2011). In contrast, California has not included the forest sector under its emissions cap, but rather as an uncapped sector that is eligible to generate offset credits for carbon sequestration (California Air Resources Board, 2008).

The policies of New Zealand and California establish a new environmental market for GHG emissions reductions and carbon sequestration. However, because the environmental "goods" involved are not well defined, these policies must be designed to quantify and assign trading rights to emissions reductions and carbon sequestration before they can be transacted in the marketplace (McCann et al., 2005). Policy design can greatly impact the "transaction" costs involved in bringing an environmental good to market, and, to the extent possible, should be considered in any evaluation of a particular policy approach (Coggan et al., 2013). The indirect nature of transaction costs can make this quite challenging, however, and requires an assessment of costs beyond the direct charges to producers and consumers, such as the resources expended in the establishment of the institutions necessary to define and address the problem (Marshall, 2013; McCann, 2013). Because many transaction costs may be unobservable at the outset, a degree of uncertainty is unavoidable during initial policy design. In reality, policymakers rarely have the ability to empirically analyse the cost performance of one programme against an alternate approach—but the climate policies of New Zealand and California present an opportunity to do just that.

This analysis investigates how the different approaches taken by New Zealand and California affect transaction costs for participants in the forest sector. This focus is selected both for the relative availability of transaction cost data for forest participants, and because levels of participation—and therefore the ultimate success of a carbon market—are contingent upon barriers to entry such as costs. By better understanding these two approaches and the transaction cost implications of each for participants in the forest sector, the experiences of New Zealand and California can provide a useful starting point for determining the most cost-effective way to incorporate forests in future climate policy. Part I of this article provides an overview of ecosystem services and environmental markets, the transaction costs associated with these markets, and the design of the cap-and-trade mechanism. Part II provides background on New Zealand's emissions profile, climate policy, and approach to including forests in cap-and-trade, while Part III does the same for California. Part IV investigates the transaction costs of these two approaches for participants in the forest sector, Part V considers price factors under each market, and Part VI discusses the implications of these findings.

I. THE ESTABLISHMENT OF ENVIRONMENTAL MARKETS

A. Ecosystem Services

Forests play a dual role with respect to climate change. When conserved, forest ecosystems can sequester and store large amounts of atmospheric carbon dioxide (Dixon et al., 1994). However, when forests are harvested, the carbon they store is emitted to the atmosphere as GHGs. If forests are converted to an alternative land use upon harvest, future forest sequestration potential also may be sacrificed (Pregitzer and Euskirchen, 2004; Fischlin et al., 2007). The ability of terrestrial systems like forests to remove and store carbon from the atmosphere is important to regulating the global climate system, and the loss of these systems can have considerable implications for climate stability (Dixon et al., 1994).

The role of forests in regulating the global climate through the sequestration and storage of atmospheric carbon is an example of an ecosystem service. Ecosystem services have been defined as ecosystem processes that directly or indirectly support human well-being (Levy et al., 2005). These services have been categorized broadly by the functions they perform, which include the production of ecosystem goods, and the regulation of biogeochemical and biospheric processes (de Groot et al., 2002). While production functions are important to providing raw materials for human consumption, which range from food to energy resources, regulation functions are critical to maintaining a healthy, functioning biosphere. Consequently, regulation functions are thought of as fundamental to the maintenance of all other ecosystem service functions (de Groot et al., 2002). However, unlike production functions, which generally create discrete, easily commoditized ecosystem goods like timber, the services arising from regulation functions are much more diffuse in nature, and often do not lend themselves to ready quantification and exchange in the traditional marketplace. As a result, markets often fail to produce sufficient quantities of ecosystem services, and instead tend to favour more easily commoditized production functions that may lead to the degradation and destruction of regulation functions (Gatzweiler, 2005).

1. Public and Private Goods

In general, goods and services can be conceived of as occurring somewhere on a spectrum of excludability and subtractability. Excludability denotes the degree to which potential beneficiaries may be excluded from using a good or service, while subtractability refers to the diminishment of a good or service's value one user incurs for other users (Ostrom and Ostrom, 1999; Ostrom et al., 1999). Those goods and services that are perfectly excludable and perfectly subtractable are termed "pure private goods." In contrast, those goods and services that are perfectly inexcludable and perfectly non-subtractable are described as "pure public goods (Ostrom and Ostrom, 1999)." Where a good or service occurs on this spectrum dictates the most effective means for its provision and production. Traditional markets are effective in supplying goods and services that are more characteristic of pure private goods, but begin to fail as goods and services tend more toward pure public goods. The inability to exclude users from public goods means that markets break down for these goods on the demand side, where beneficiaries are unwilling to pay for something they can already receive free of charge (Ostrom and Ostrom, 1999).

To overcome the market's failure to supply public goods, some form of collective action is necessary to induce beneficiaries to provide compensation for the production of a public good. Governments may act to create demand for a public good, then levy compulsory measures such as taxes to ensure that beneficiaries provide their proportionate share in paying for the production of that good (Ostrom and Ostrom, 1999). Once governments have taken steps to create demand for a public good, decisions are required regarding the provision and production of the good itself. Provision choices refer to decisions over whether and how much of the good to supply, while production considerations pertain to the actual processes involved in creating the good itself (Ostrom and Ostrom, 1999).

The amorphous nature of ecosystem services arising from regulation functions renders them challenging to quantify and monetize for inclusion within private markets. The benefits of these services often are neither easily excludable nor subtractable, but rather

accrue freely to society at large. As a result, these ecosystem services often tend to be more characteristic of public goods in nature (Ostrom and Ostrom, 1999). In contrast to ecosystem services arising from production functions, which often are more reminiscent of private goods, the inability to exclude users and extract compensation for ecosystem public goods means that little incentive exists in the market for their production. With respect to forests and climate, this market failure falls at the intersection of two distinct environmental "tragedies:" (1) A failure to internalize the externalized climate *costs* of forest harvest and conversion; and (2) a failure to internalize the externalized climate *benefits* of forest management. These externalized costs lead to inefficiently high levels of GHG emissions to the atmosphere while the externalized benefits result in inefficiently low levels of forest carbon sequestration and storage (Wayburn & Chiono, 2010).

2. Externalized Costs

In 1968, Garret Hardin articulated the notion of the "tragedy of the commons," which described the market failure arising when property rights to moderate resource usage were poorly developed or non-existent in a commons (Hardin, 1968). Hardin's "commons" referred to an open-access resource, or a resource that exhibits non-excludability and subtractability with respect to resource users. The "tragedy" that occurred was the consequent resource degradation that resulted when it was difficult or impossible to hold users accountable for the full costs of their use (Hardin, 1968). These costs were not borne wholly by those responsible for them, but rather were "externalized" to other users.

In the case of climate change, the atmosphere can be thought of as an open-access resource that is "used" both as a disposal medium for GHG emissions as well as a means of regulating the global climate system. Because of the difficulty in assigning rights to the use of the atmosphere as a receptacle for emissions, however, holding users accountable for their emissions can be challenging or impossible. This has consequences for other users due to the subtractability the atmosphere exhibits with respect to emissions: When excessive levels of GHGs are emitted, the resulting disruption of global climate regulation marginalizes those who depend on the atmosphere for a stable climate.

However, the open-access nature of the atmosphere makes limiting emissions via private markets difficult, and emitters generally bear few if any costs for their emissions. As a result, markets fail in dictating an efficient outcome with respect to emissions, and lead to levels of atmospheric GHGs that threaten global climate stability (Rose, 1991). Despite being "externalized" from the cost calculus of individual emitters, these emissions are not costless, however, but rather are borne by society at large in the form of reduced air quality and the disruption of the global climate system.

3. Externalized Benefits

While Hardin's tragedy is one of externalized costs, the failure to include the benefits of an activity within the economic cost calculus can lead to a distinct, if related, type of market failure (Lant et al., 2008). When the benefits of an activity are externalized, beneficiaries may enjoy a service without compensating those responsible for its production. Ecosystem services are a classic example of this. Due to the difficulties inherent in their quantification and the challenges of excluding beneficiaries, the value of these services often is not included in the economic cost calculus of those whose lands produce them. As a result, markets often fail to produce sufficient levels of ecosystem public goods due to the externalization of their economic value. This phenomenon has been termed the "tragedy of ecosystem services (Lant et al., 2008)."

While ecosystem services arising from regulation functions can be thought of as a type of public good, provision and production decisions for these services differ from that of traditional public goods. For instance, the decision over whether or not to supply a particular ecosystem service generally does not need to be made—it already is supplied by the ecosystem. Rather, provision decisions for ecosystem services entail determining whether or not to continue providing a service through the maintenance of the ecosystem that produces it (Gatzweiler, 2005). As a result, choices regarding the provision of ecosystem services often are not made directly by a governmental entity. Instead, the amount and quality of an ecosystem service provided are dictated by the management choices made on the lands that produce that service. Where private lands are involved, decisions regarding the supply of ecosystem public goods often remain in the hands of

private resource managers (Lant et al., 2008). Further, unlike private goods, where the price of a good relays information about its demand, tax-supported expenditures for public goods convey little information about demand for the good itself. As a result, governments must establish alternate means of determining how much of a public good to provide (Ostrom and Ostrom, 1999). With respect to global climate change, this may correspond to emissions and sequestration targets that are designed to maintain atmospheric GHG concentrations at levels that limit the disruption of the global climate system (IPCC, 2007). However, these questions often are just as much about stakeholder values as they are about science, and political factors can exert a strong influence in driving these decisions (Clark, 2002).

4. Correcting Market Failures

Forest management decisions are unique in that they occur at the intersection of these two environmental tragedies with respect to climate. The costs of GHG emissions from harvest and conversion typically are externalized from the economic calculus while landowners generally receive no compensation for the climate benefits their forests provide (Patterson and Coelho, 2009). As a result, private markets dictate levels of harvest and conversion that are inefficient with respect to maintaining a stable global climate while no market demand exists for the public goods forests provide in terms of carbon sequestration and storage.

Overcoming this market failure requires steps to internalize both the externalized climate costs and externalized climate benefits of forest management. This necessitates measures to induce the provision and production of a public good when the factors of production are largely privately controlled, as well as the establishment of disincentives for forest management activities that sacrifice forest carbon storage and sequestration (Wayburn & Chiono, 2010). In some instances, negotiation among affected parties can allow for the internalization of externalized costs or benefits and yield an efficient outcome even in the absence of clearly defined property rights (Lant et al., 2008). However, negotiation among affected parties is not inherently costless, and increases with the number of stakeholders. Additionally, where externalized benefits are involved, beneficiaries free-

riding on a service may have little incentive to enter into negotiations to compensate producers for something they currently enjoy free of charge (Hahn and Stavins, 1992). With respect to global climate change, where the externalities affect a global constituency, the sheer number of stakeholders renders any collective bargaining approach infeasible, and some form of government intervention is necessary to overcome this market failure (Esty, 1996).

In general, governments may intervene to induce desired behaviour and correct market failures through two different approaches: A "command-and-control" regulatory approach; or a more-flexible, incentive-based market mechanism (Ackerman and Stewart, 1988). Under command-and-control approaches, governments specify compliance strategies, and regulated entities are provided with little flexibility in meeting a designated environmental outcome (Stewart, 1988). In contrast, market-based approaches take steps to monetize externalized costs and benefits, and then harness the power of the market to correct behaviour. By creating incentives that internalize the value of desired behaviour and disincentives that internalize the costs of undesirable behaviour, this approach offers regulated entities considerably more flexibility in choosing compliance strategies (Hahn and Stavins, 1992).

Unlike command-and-control regulation, market approaches retain operating decisions in the hands of those best qualified to make them—the entities themselves. This enables regulated parties to select a level of environmentally desirable behaviour that is appropriate given the economic conditions created under the new environmental market (Newell and Stavins, 2003). The advantages of market approaches over command-and-control regulation include greater efficiency, cost effectiveness, and transparency, as well as the ability to promote investment and innovation (Stewart, 1988). However, while cost-effectiveness can be considerably improved through market-based mechanisms, differences in design may greatly impact the relative costs associated with a particular approach. The next section investigates where these costs arise in environmental markets.

B. Transaction Costs

When private markets fail to provide socially optimal levels of environmental goods and services, government intervention may be justified (Willis, 1997; Vatn, 1998; Wayburn & Chiono, 2010). The adoption of market-based policies that assign property rights to environmental goods can facilitate trade in these goods (Coggan et al., 2013). However, because the "good" involved in many environmental problems may not be well defined, resources often must be expended in the quantification and assignment of property rights to the environmental good itself (McCann et al., 2005). Such indirect costs have been termed "transaction costs," and generally are conceived of as those costs beyond the direct expenditures for inputs and production in making a trade (Stavins, 1995; Antinori and Sathaye, 2007). By lowering the effective price a seller receives and raising the effective price a buyer pays, transaction costs reduce welfare by suppressing exchanges that otherwise would have been mutually beneficial (Stavins, 1995). In context to climate policy, transaction costs may include those costs associated with defining and quantifying GHG emissions reductions or sequestration such that these goods can be traded in an environmental market.

Transaction costs can be influenced considerably by the legal, social, and political factors or rules that determine the context within which economic activity takes place (Coggan et al., 2013). As a result, the actual cost of producing an identical good made with the same materials can vary greatly depending on these factors of the "institutional environment (Antinori and Sathaye, 2007; Coggan et al., 2013)." The arrangements within this environment include the governance structures used to organize and contract for the exchange of property rights (Williamson, 1998; Buitelaar, 2004). While considerable attention has been devoted to the role of transaction costs in the arrangement of private firms, much of this analysis treats the institutional environment as a given (Coase, 1937; Coase, 1960; Demsetz, 1969; Williamson, 1985). However, in the context of environmental markets, the institutional environment does not necessarily constitute an immutable factor. Unlike private firms trading within long-established markets governed by long-established rules, the guidelines for environmental markets often are created along with the market itself. As a result, the institutional environment may be regarded as

an object of design in environmental markets rather than a factor to be designed around (Pannell, 2008; Pannell and Wilkinson, 2009; McCann, 2013; Pannell et al., 2013). However, while cost minimisation is frequently a stated goal in policy design, the way in which design factors affect the institutional environment and transaction costs in environmental markets is still not widely recognized (Colby, 2000). As a result, transaction costs often represent a substantial proportion of the overall costs of environmental regulation (Pannell, 2008; Pannell and Wilkinson, 2009; Pannell et al., 2013).

C. Cap-and-Trade

Under New Zealand and California's respective climate laws, the two jurisdictions have enacted policies to limit GHG emissions through what is known as a "cap-and-trade" mechanism (California Air Resources Board, 2008; New Zealand Climate Change Response [Emissions Trading] Amendment Act, 2008). Cap-and-trade programmes are a type of market-based approach that create an environmental market for emissions reductions and sequestration by establishing a limit, or "cap," on GHG emissions from a regulated sector. A cap may be designed to cover only certain sectors, or it may extend to the entire economy. To comply with an emissions cap, entities in regulated sectors must either reduce emissions themselves, or "trade" with other entities to purchase emissions reductions those entities have created. For regulated sources with high marginal costs of emissions abatement, it may be more economical to pay another entity with lower marginal abatement costs to reduce emissions rather than do so in-house. As long as aggregate levels of emissions abatement are sufficient to ensure that overall emissions decline to the level of the cap, it is immaterial to the effectiveness of the policy where in the economy these reductions occur (Esty, 1996). This approach in effect seeks to internalize the externalized costs of GHG emissions in terms of their role in global climate change.

While emissions abatement focuses on forgoing the release of GHGs in capped sectors, the removal, or sequestration, of atmospheric carbon dioxide also can be used to limit the concentration of atmospheric GHGs. In cases where an emissions cap does not extend to the entire economy, sources and sectors outside the cap may still be eligible to receive credit for sequestration that is considered "additional" to what is required by law or would have otherwise occurred under a status quo scenario. A regulated entity may satisfy a portion of its compliance obligations by purchasing credits for this sequestration to "offset" its GHG emissions. This in effect internalizes the economic value of carbon sequestration and storage in terms of its role in mitigating global climate change.

The effectiveness of an offset programme relies on ensuring the credibility of the offset credits generated. In essence, to truly cancel out GHG emissions, sequestration sold as offsets must actually be additional to any sequestration that would have otherwise occurred under a "business-as-usual" scenario. A number of criteria based upon guidance given in the Kyoto Protocol have been developed to ensure the integrity of offset projects, and the requirements for offsets in California's programme are based upon this language (United Nations, 1998). For these credits to be eligible for compliance under California's law, they must be (California Health and Safety Code §38562(d)(1) and (2)):

(1) "real;"

(2) "permanent," or not reversible. If reversible, mechanisms must exist to replace any reversals and ensure all credited reductions endure for at least 100 years;

(3) "quantifiable," or accurately measured and calculated for all GHG sources, sinks, and reservoirs within the offset project boundary;

(4) "verifiable," or well documented and transparent such that all offset project data lend themselves to an objective review by an accredited verification body;

(5)   "enforceable," or under an authority that can hold a party liable and take appropriate action if any provisions of the offset programme are violated; and

(6)   "additional," or representing reductions exceeding any otherwise required by law, regulation, or legally binding mandate, and exceeding any that would otherwise occur in a conservative "business-as-usual" scenario.

If all of these criteria can be satisfied, then the sequestration achieved may be deemed additional, and therefore considered eligible to "offset" a corresponding quantity of GHG emissions released to the atmosphere by a regulated entity. This can provide emitters within capped sectors additional flexibility in their compliance with an emissions cap, and reduce the overall costs of a cap-and-trade programme. By sourcing emissions abatement or sequestration from entities with the lowest marginal costs, cap-and-trade policies theoretically achieve a desired level of emissions reductions at the least possible cost (Esty, 1996).

The decision to include or exclude a particular sector within an emissions cap constitutes an element of policy design and impacts the institutional environment in which a carbon market is established. This choice may be influenced by a variety of factors, including the nature of the sector itself, its relative contribution to the economy's overall emissions, political factors, and the overall goals and objectives of the policy itself. While the unique political, legal, economic, and geographic circumstances of New Zealand and California render any comparison imperfect, these policies nonetheless present an opportunity to gain insight on the transaction cost implications of two different approaches to forests in climate policy. The next two sections provide an overview of these policy designs and the circumstances surrounding them.

II. NEW ZEALAND BACKGROUND

A. National Circumstances

1. New Zealand's Emissions Profile

In 2011, New Zealand's total emissions were 72.8 million tonnes CO_2e (Ministry for the Environment, 2013a). While New Zealand produces barely 0.2% of global CO_2e emissions, its per capita emissions are relatively high compared to other developed nations (Boston, 2011). This is largely attributable to the considerable quantities of methane and nitrous oxide generated from the agricultural sector, which constitute almost 50% of the country's total emissions. However, due to current limitations in reducing emissions from the agricultural sector and an already high reliance on renewable energy, options for reducing emissions in New Zealand are relatively limited when compared to those of many other countries (Boston, 2011). The forest sector presents a considerable opportunity for helping solve the challenges posed by the high contribution of emissions from the agricultural sector, and removed 16.8 million tonnes CO_2e in 2011 (Ministry for the Environment, 2013a). However, this potential is complicated by the highly variable pattern of planting and deforestation, making policies that discourage conversion and promote sequestration critical to meeting New Zealand's climate objectives (Boston, 2011).

2. New Zealand's Forest Sector

New Zealand's forest sector is the country's third largest merchandise export earner, and contributes about 3% to New Zealand's GDP (Forest Owners Association & Ministry for Primary Industries, 2013). Its forests are distinguished into two types: natural forests and plantation forests. Natural forests cover about 30% (8.1 million hectares) of New Zealand's total land area while plantation forests cover about 8% (2 million hectares). No timber is legally harvested from natural forests on public lands, and most harvesting on private natural forests must be undertaken on a sustainable basis. Indeed, it was estimated that only 0.05% of New Zealand's total timber production in 2011 was from the harvest of natural forests (Ministry for the Environment, 2013a). As a result of these sustainable

harvesting requirements, carbon in natural forests is considered to be in a steady state over the long term and is not included in national inventories. In contrast, plantation forests are intensively managed for timber production, and are a prominent part of New Zealand's national emissions profile (Ministry for the Environment, 2013a). Radiata pine (*Pinus radiata*) is the primary commercial species, and is typically grown on even-aged rotations ranging between 25 and 32 years (Karpas & Kerr, 2011).

B. New Zealand Climate Policy

In 2002, the passage of the Climate Change Response Act (CCRA) established New Zealand's emissions reduction obligations under the Kyoto Protocol into domestic law. These obligations required New Zealand to limit its GHG emissions to 1990 levels by 2012, or take responsibility for any emissions over this level (Ministry for the Environment, 2013b). The initial policy guidance for incorporating forests into the CCRA contained several controversial proposals, including the government retention of rights to all sequestration credits on forests planted on or after 1 January 1990, and a deforestation cap on forests established on or before 31 December 1989 (Rive, 2011).

In 2005, a comprehensive review of the government's climate change policies focused particular attention on the proposals surrounding forests. While a December 2005 meeting of the cabinet noted the contentious nature of the proposed government retention of all benefits and liabilities on forests established on or after 1 January 1990, it deferred final decisions on the matter pending further review (Rive, 2011). In 2007, a policy document produced by the Ministry of Agriculture and Forestry outlined the government's overarching principles for forests within a proposed cap-and-trade system (Ministry of Agriculture and Forestry, 2007). This included a revised proposal for forestry that devolved to landowners both credits for sequestration and liabilities for the emissions released from the deforestation of their land (Rive, 2011).

In December 2007, legislation was introduced to amend the CCRA for the provision of a cap-and-trade programme that was consistent with these principles. With the passage of

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the Climate Change Response (Emissions Trading) Amendment Act in 2008, the legislative framework for New Zealand's cap-and-trade programme, known as the Emissions Trading Scheme (ETS), was established. While several complementary grant schemes focus on promoting forest sequestration, the ETS is regarded as the country's main instrument for encouraging afforestation and reducing deforestation (Ministry for the Environment, 2009).

1. Forests in the Emissions Trading Scheme

Following the rules governing forests under article 3.3 of the Kyoto Protocol, the New Zealand ETS draws a distinction between forests according to their date of establishment. For countries party to the Protocol, Article 3.3 limits sequestration credits to forests established on or after 1 January 1990, stating (United Nations, 1998):

> The net changes in GHG emissions by sources and removals by sinks resulting from direct human-induced land-use change and forestry activities, limited to afforestation, reforestation and deforestation since 1990 measured as verifiable changes in carbon stocks in each commitment period, shall be used to meet the commitments under the Article for each Party included in Annex I.

Accordingly, forests planted after 31 December 1989, or "post-1989 forests," are not included under New Zealand's emissions cap, but are eligible to receive credits for sequestered carbon. However, if post-1989 forests do receive credit for sequestered carbon, they are liable for any emissions arising from the reversal of these stocks. In contrast, forests established prior to 1 January 1990, known as "pre-1990 forests," are included under the emissions cap. These forests are liable for any emissions arising from deforestation, and are not entitled to receive any credits for contributions as a carbon sink. At the end of 2011, 2,186 landowners had registered as participants in the post-1989 forests programme, and more than 13.8 million tonnes of sequestration had been reported on participating post-1989 forest lands (Environmental Protection Authority, 2013). Between 1920 and 1990, an estimated 1.4 million hectares of this "pre-1990" plantation forest were established (Forest Owners Association & Ministry for Primary Industries, 2013; Ministry for the Environment, 2013a).

III. CALIFORNIA BACKGROUND

A. State Circumstances

1. California's Emissions Profile

In 2011, California's total GHG emissions were 448.11 million tonnes CO_2e. Transportation accounted for 38% of overall CO_2e emissions, and was the largest single contributing sector with 168.42 million tonnes CO_2e in 2011 (California Air Resources Board, 2013a). The forest sector currently provides a net carbon sink, which preliminary research suggests is around 4.7 million tonnes CO_2e annually, although this estimate is in the process of being revised (California Air Resources Board, 2008; California Air Resources Board, 2014a).

2. California's Forest Sector

Forests in California cover roughly 13.4 million hectares, or nearly a third of the state's total land area. Over half (7.6 million hectares) of these forested lands are managed by the federal government, and most are within the National Forest System (Christensen et al., 2008). Private owners hold about 40%, or 5.3 million hectares, and small state and local holdings compose the remaining approximately 0.4 million hectares. About 7.7 million hectares are softwood forests consisting primarily of mixed conifers, while oak forests are the most common hardwood forest type, occupying about 4 million hectares throughout California (Christensen et al., 2008). Despite being the fourth largest lumber-producing state in 2008, increasing timber management constraints on private lands and a large reduction in harvest levels on public lands have resulted in a continual decline in California's overall timber production (Christensen et al., 2008). Between 2001 and 2006, wood fibre production from federal lands was only about 10% of the state's total production, down from an average of 40% between 1963 and 1987 (California State Board of Equalization, 2006).

B. California Climate Policy

In 2006, the passage of Assembly Bill 32 (AB 32), known as the Global Warming Solutions Act, required California to reduce its greenhouse gas emissions to 1990 levels by 2020 (California Health and Safety Code §38550). Achieving the goals of AB 32 will require California to limit its emissions to 431 million tonnes CO_2e by 2020, a 15% reduction from the projected "business-as-usual" scenario (California Air Resources, Board 2014a).

In 2008, the California Air Resources Board (ARB), the state agency charged with overseeing the law's implementation, released an initial scoping plan outlining recommendations for achieving the GHG emissions reductions required by AB 32. The scoping plan contained a mix of strategies combining direct regulations, market-based approaches, voluntary measures, and other emissions-reduction initiatives, but the centrepiece of California's emissions-reduction strategy was the creation of a statewide cap-and-trade programme (California Air Resources Board, 2008). While the cap-and-trade programme does not extend to every economic sector in the state, it covers the sources responsible for roughly 85% of California's GHG emissions (California Air Resources Board, 2011a).

Under California's policy, forests are not included in the cap, but the forest sector may generate offset credits that are eligible under the cap-and-trade programme. Emitting facilities may use these offset credits and those from certain other uncapped sectors and sources to satisfy up to 8% of their compliance obligations (California Air Resources Board, 2011a). Offsets will be an important component of helping emitting entities meet their compliance obligations under California's cap-and-trade programme, and forecasts suggest that the total demand for offsets could exceed 200 million tonnes CO_2e by the end of 2020 (Stevenson et al., 2012).

1. Forest Offset Projects

To be eligible under California's programme, offset credits must be generated using either a Compliance Offset Protocol (COP) or one of the "early-action" protocols approved by ARB (California Code of Regulations §95970). Under these approved protocols, forest offsets may be created via three project types: Reforestation, Avoided Conversion, and Improved Forest Management (IFM). Reforestation projects involve restoring tree cover on land that is not at its full stocking level; Avoided Conversion projects conserve forests where conversion is imminent; and IFM projects, which are referred to as "Conservation-Based Forest Management" (CBFM) projects in certain early-action protocols, alter forest management to increase carbon stocks relative to a business-as-usual scenario (California Code of Regulations §95970).

At the beginning of 2014, 108 offset projects had been listed under ARB-approved compliance or early-action protocols with the Climate Action Reserve (CAR), the primary offset registry approved for use with AB 32 (Climate Action Reserve, 2014a). Currently, these projects have generated about 6.7 million tonnes of sequestration that are potentially eligible to be issued ARB offset credits for use under California's cap-and-trade programme. While only 29 of these offset projects are in the forest sector, they currently account for more than half (3.5 million) of all the emissions reductions generated using ARB-approved protocols (Climate Action Reserve, 2014a). While forest projects located anywhere in the contiguous United States are eligible under the compliance protocols, most of the reductions created by ARB-approved forest projects (3.2 million) are from the 14 projects located in California, all of which are either IFM or CBFM projects (Climate Action Reserve, 2014a). Forest projects are thought to have the greatest technical potential for providing GHG reductions of any project type eligible under California's programme, and preliminary modelling suggests that forests could technically supply between 500 and 700 million tonnes of offset credits by 2020 (Stevenson et al., 2012).

IV. TRANSACTION COSTS OF FOREST SEQUESTRATION

A. Transaction Cost Data

Transaction costs often are defined as those indirect costs arising beyond expenditures for the simple inputs of production (Stavins, 1995; Antinori and Sathaye, 2007). Despite potentially composing a large part of the overall costs of environmental regulation, the diffuse nature of transaction costs often means that they are not discretely tracked by participants (Pannell, 2008; Pannell and Wilkinson, 2009; Pannell et al., 2013). In environmental markets, these costs frequently arise from the rules and guidelines established for the creation of an environmental good. This may include the expenses involved in satisfying regulatory requirements, or the expenditures necessary in quantifying and defining the environmental good itself to allow for its exchange in the marketplace. While data on permitting and regulatory fees often are readily available, information on the costs involved in creating an environmental good itself may be more difficult to ascertain.

For New Zealand, forestry consultant fees are used as a proxy for the transaction costs involved in each step of defining and quantifying forest carbon for trading in the ETS. The large number of forest participants and relative standardization of forest projects has rendered the use of consultants an important strategy under New Zealand's programme (S. Orme, personal communication, 21 November 2013). Along with an overview of the regulatory permitting costs incurred in establishing a project, this approach gives some sense of the transaction costs involved in bringing forest carbon to market under New Zealand's programme. For California, cost data on regulatory permitting also may be easily obtained, but the considerably less-standardized nature of forest projects under California's programme means that most projects have been undertaken by entities with in-house specialization in forestry and carbon policy rather than via external consultants. As a result, the transaction cost estimates for California's programme rely primarily on financial information provided by forest project participants coupled with data on the necessary regulatory permitting fees. These estimates represent costs on projects ranging from 5,000 to 10,000 hectares for 6 of California's 14 forest projects.

B. Quantifying Forest Carbon in New Zealand's Programme

The guidelines governing New Zealand's forest sector are based on Article 3.3 of the Kyoto Protocol, and in effect create a baseline for forest carbon stocks at 1990 levels (United Nations, 1998) To maintain this stocking level, forests established prior to 1 January 1990 are essentially under an emissions cap, and owners of these lands are liable for any emissions arising from the conversion of their forests. Any forests established after 31 December 1989 are considered additional to this baseline level, and may be eligible to receive credits for sequestration. Due to this policy design, methodologies must be in place to both quantify emissions from the deforestation of pre-1990 forests as well as any sequestration in post-1989 forests participating in the ETS.

1. Post-1989 Forests

In New Zealand, post-1989 forest owners are not required to become participants in the ETS, but may elect to do so if they wish to receive credit for the carbon sequestered on their land. Credits may be claimed for any carbon stocks on post-1989 forest land, but once credit is claimed, landowners become liable for emissions arising from the reduction of these credited stocks (Cameron, 2011). Post-1989 participants must monitor carbon stock changes on their forests and report these changes to determine their liabilities or entitlements. To achieve this, landowners are required to delineate a "carbon accounting area" (CAA) on land that will be enrolled in the ETS. Carbon stock changes are then calculated on each CAA by subtracting the carbon stocks at the beginning of the reporting period from the stocks present at the end (Lough & Cameron, 2008).

Total carbon stocks on post-1989 forests include both the carbon stored above ground in the tree bole, branches, and leaves, as well as that stored below ground in tree roots. For post-1989 forests, two methods are available for quantifying changes in forest carbon stocks. The first approach is based on a series of "look-up" tables provided in the New Zealand Forestry Sector Regulations (Ministry of Agriculture and Forestry, 2011a). These tables provide pre-calculated carbon stock values for a given forest type, age, and location. Forest carbon stocks are determined by looking up an appropriate carbon

stocking factor from the tables, and then multiplying it by the area of a CAA to calculate total carbon stocks. These generic look-up tables are based on national averages, and may be used by those participants with less than 100 hectares enrolled in a project. To improve carbon stock approximations on larger projects, a second quantification approach, known as the field measurement approach (FMA), utilizes customized look-up tables based on field measurements taken from a network of randomly located sample plots on the participant's land (Ministry of Agriculture and Forestry, 2011a).

After carbon stock changes have been quantified, post-1989 forest participants must self-report their entitlements or liabilities though an emissions return. Voluntary returns may be submitted on an on-going, annual or multi-year basis for participants to claim credits for sequestration on their CAAs. Mandatory returns must be submitted within three months of the conclusion of the 5-year crediting period in which a post-1989 forest landowner is participating. This return provides sequestration and emissions information for the entire crediting period, and allows the participant to claim or surrender credits for any outstanding entitlements or liabilities (Cameron, 2011).

2. Pre-1990 Forests

Pre-1990 forest owners may harvest their lands without incurring any liability for emissions as long as the land is replanted or allowed to naturally regenerate to forest. However, if the land is "deforested," or changed from forestry to an alternate, non-forest use, landowners automatically become participants in the ETS, and are liable for the emissions that arise from this deforestation. Emissions from pre-1990 forests also are calculated using look-up tables based on forest age, location, and species, but these tables are distinct from those used for determining sequestration on post-1989 forests (Ministry of Agriculture and Forestry, 2011b). In accordance with New Zealand's obligations under the Kyoto Protocol, no credit is given for carbon in wood products, and all emissions are assumed to occur instantaneously upon deforestation (United Nations, 1998).

C. Transaction Costs for New Zealand Forest Participants

For New Zealand forest landowners, the first step to participating in the ETS is to open a holding account at the New Zealand Emission Unit Register (NZEUR). The NZEUR is New Zealand's only official credit registry, and manages all emissions and credit transactions under the ETS. There are no fees associated with opening an NZEUR account, and landowners may either choose to open an account themselves, or hire a forestry consultant to do so on their behalf for about $500 NZD (in 2013 dollars).

Once an NZEUR account has been opened, post-1989 forest owners must then submit an application to the New Zealand Ministry for Primary Industries (MPI) registering them as a participant. This requires a $562 MPI processing fee, and the application must include a property description as well as a map delineating the CAAs to be enrolled under the ETS (Ministry of Agriculture and Forestry, 2011a). If a forestry consultant is hired to file an ETS application on a participant's behalf, project areas under 30 hectares cost approximately $1,000, and $10 per hectare is assessed for each additional hectare above this level (Woodnet, 2012). If subsequent CAAs are added to an existing project, an MPI fee of $102 is incurred, while reconfiguring CAAs requires $562 in MPI processing fees (Ministry of Agriculture and Forestry, 2011a). Using a consultant to add or reconfigure CAAs starts at around $250 and increases with complexity (S. Orme, personal communication, 29 January 2014).

After a post-1989 forest project landowner has successfully applied to participate in the ETS, the carbon stocks for each CAA must be calculated. For projects under 100 hectares, the generic look-up tables may be used to quantify carbon stocks. If a project exceeds 100 hectares, the participant must quantify their carbon stocks using the FMA. Where the FMA is required, pricing depends on the forest type (exotic or indigenous) and size of the project. For exotic forests, MPI regulations require a minimum of 30 sample plots, and these are priced at between $300 to $350 a plot depending on travel time and logistics. As a result, FMA sampling costs start at $9,000 for exotic forests of 100 hectares, and will increase as larger projects require more plots. For indigenous forests, a minimum of 15 plots is required, and these generally are priced at between $450 and

$600 a plot due to the greater complexity involved in sampling indigenous forests. For these forests, FMA sampling starts at $6,750, and increases in cost according to the number of additional plots required on projects larger than 100 hectares (S. Orme, personal communication, 29 January 2014). Once sample plots have been established, stocking data from the plots are submitted to MPI for the creation of project-specific look-up tables (Ministry for Primary Industries, 2012). These tables are then used by the project participant to quantify the carbon stocks on the post-1989 forest project. No fees are assessed by MPI for administering the FMA.

Once carbon stocks have been quantified, a post-1989 forest participant must submit an emissions return detailing the carbon entitlements or liabilities arising on a project. The MPI fee for filing an emissions return is $102 (Ministry for Primary Industries, 2012). If a consultant is used, filing an emissions return costs $170 for returns on up to 5 CAAs plus $10 per each additional CAA (Woodnet, 2012). If CAAs contain multiple age classes or species, additional fees may apply. If an emissions return shows a net increase in carbon stocks, a corresponding amount of credits is to be transferred into the participant's NZEUR account within two weeks. If the emissions return demonstrates a net decrease in carbon stocks, the participant must surrender credits covering this liability within 20 working days of submitting an emissions return. An overview of transaction costs for forest participants under New Zealand's programme is provided in Table 1.

TABLE 1. NEW ZEALAND FOREST PARTICIPANT TRANSACTION COSTS

Description	MPI Fees Fixed	Consultant Fees Fixed	Variable	Occurrence
Opening NZEUR Account	$0	$500		Initial
ETS Participant Application	$562	$1,000[a]	$10/ha	Initial
FMA Sampling (exotic)	$0	$9,000*	$300 - 350/plot	Initial
FMA Sampling (indigenous)	$0	$6,750*	$450 - 600/plot	Initial
Adding CAAs	$102	$250		On-going
Reconfiguring CAAs	$562	$250		On-going
Emissions Return Filing	$102	$170**	$10/CAA	On-going

[a] For projects up to 30 hectares.
*For projects up to 100 hectares.
**For returns with up to 5 CAAs.

D. Quantifying Forest Carbon in California's Programme

Under California's programme, forests are not included as a capped sector, but individual forest projects may be eligible to receive offset credits for the generation of any "additional" sequestration. Because managed forests in California are largely of mixed age-class, the date of establishment cannot be used as a baseline against which to quantify additional sequestration for offset credit. Rather, a baseline that reflects a "business-as-usual" counterfactual must be modelled for each individual forest project.

Of the three ARB-approved forest project types, IFM projects are anticipated to have the greatest offset generation potential, and could account for up to 61% of all forest offsets generated by 2020, or about 36 million tonnes of carbon sequestration (Stevenson et al., 2012). These projects already have been a substantial contributor to California's overall offset supply, and were responsible for 31% (1.6 million) of the total 5.4 million offset credits issued by ARB at the beginning of 2014 (California Air Resources Board, 2014b). Because of the importance of these projects to future offset supplies as well as their current abundance, this investigation of transaction costs for forests participants in California's programme focuses on IFM projects.

E. Improved Forest Management Projects

Improved Forest Management projects generate emissions reductions by adopting management approaches that increase forest carbon stocks relative to a business-as-usual baseline scenario. This baseline is an estimate of the carbon stocking levels that would have occurred in the absence of a forest project. Carbon sequestered above and beyond this baseline level is eligible to receive credits for emissions reductions.

1. Project Listing and Initial Inventory

The first step in undertaking an IFM project is listing the project with an ARB-approved offset registry. This requires submitting all necessary project documentation, inventory data, and attestations to the registry. Inventory data must include a quantification of

carbon stocks in all required pools as well as any incidental emissions arising as a result of project activity. All plot data upon which forest carbon inventory estimates are based must have been sampled within 12 years of the inventory (California Air Resources Board, 2011b).

2. Baseline Determination

Once stocking levels have been quantified in an initial inventory and a project has been listed, a baseline must be established to calculate emissions reductions. For IFM projects, a baseline is determined by modelling the carbon stocks in standing live trees through a series of growth and harvesting scenarios that reflect all legal and financial constraints on the land. Stocks are modelled out 100 years under these scenarios and then averaged over this timeframe so that the baseline is expressed as a single average value for carbon stocks per hectare per year (California Air Resources Board, 2011b). Once the baseline for standing live carbon is determined, baselines for all other required carbon pools must also be estimated. This includes the carbon in standing dead trees, soils, and harvested wood products. These values also are averaged over 100 years, and then added to the baseline for standing live carbon to produce a final baseline for all carbon pools within a forest project.

3. Monitoring

Offset project participants are required to monitor onsite carbon stocks and submit an Offset Project Data Report (OPDR) each year for the duration of a project's 100-year lifetime. These reports must include an estimate of carbon stocks in all required pools, and incorporate any new forest inventory data obtained during the previous year (California Air Resources Board, 2011b). To calculate the emissions reductions arising from a forest project, both the project's "primary" (intended) effects as well as its "secondary" (unintended) effects must be quantified. For IFM projects, the primary effects are the changes in carbon stocks in standing live trees, standing dead trees, soil, and harvested wood products resulting from the altered management regime under the project. Secondary effects include any emissions arising from vehicles or equipment associated with the project, the shifting of harvesting activities from the project area to

other forestlands (known as "leakage"), or the decomposition of discarded wood products that originated from the project. Each year, the actual change in carbon stocks must be quantified by subtracting the prior year's carbon stocks from the current year's carbon stocks. The difference between this value and the change in baseline carbon stocks over the same period represents the additional carbon sequestered as a result of the project. Subtracting any emissions arising from secondary effects gives the project's net GHG emissions reductions for that year.

4. Verification

To guarantee the authenticity and permanence of the sequestration generated, IFM projects must satisfy rigorous third-party verification requirements. For all ARB-approved forest projects, "permanent" means enduring for 100 years. Projects are required to undergo third-party verification with on-site visits after the submission of the first OPDR, and again at least every six years thereafter for the duration of a project's life. During verification, ARB-accredited third-party verifiers must provide a detailed review of the forest carbon inventory as well as a re-measurement of sample plots and a comparison of these measurements with inventory data. ARB will not issue offset credits for emissions reductions until a project's OPDR has been approved by an accredited third-party verifier (California Air Resources Board, 2012c). At the discretion of the offset project participant, less-intensive verifications may be provided in interim years between full verifications. These "desk verifications" do not require a field visit, and may allow a project participant to be issued offset credits for sequestration in years between full verifications (California Air Resources Board, 2012c).

5. Reversals

In the event of the emission, or "reversal," of credited carbon sequestration on an IFM project, the requirements for replacing the reversed carbon depend on the nature of the reversal itself. If the reversal is deemed intentional, such as through harvest or land conversion, the offset project participant must surrender emissions allowances to compensate for the carbon emitted. To discourage strategic behaviour, allowances must be surrendered on a greater than 1:1 ratio, the magnitude of which decreases with the age

of a project at the time of reversal (*i.e.*, the older the project at the time of intentional reversal, the lesser the replacement penalty). To provide insurance against unintentional reversals, such as from fire or windthrow, a project must contribute a certain percentage of credits to a "Forest Buffer" account each year. The size of this contribution depends on an assessment of the project's risk of reversal (California Air Resources Board, 2011b). If a project undergoes an unintentional reversal, credits from this account are used to replace the reversed credits.

F. Transaction Costs for California Forest Participants

To participate under California's programme, all projects must first open an account with an approved offset registry, such as the Climate Action Reserve (CAR), which is the primary registry in use under California's cap-and-trade programme. Opening an account with CAR requires a one-time setup fee of $570 NZD (in 2013 dollars), and an annual account maintenance fee of $570 each year thereafter (Climate Action Reserve, 2014b).

For IFM projects, the costs for modelling and initial baseline determination range from $11,402 to $17,103 for projects between 5,000 and 10,000 hectares in size. A forest inventory must be undertaken prior to project commencement, and again every 6 to 12 years thereafter, depending on the degree of change in forest stocks. These inventories generally range from around $19 to $29 a hectare, depending on forest stocking levels and stand heterogeneity (personal communication, J. Golinkoff, 29 October 2013; personal communication, L. Wayburn, 10 December 2013; personal communication, P. Swedeen, 10 June 2014). Once all inventory data, project documentation, and appropriate attestations have been compiled, these materials must be submitted to a registry for listing, which incurs a project submittal fee of $798 (Climate Action Reserve, 2014b). Costs involved in the preparation of documentation for listing generally are about $2,000 (personal communication, P. Swedeen, 10 June 2014).

Once a project has commenced, an OPDR must be compiled each year over the project's 100-year lifetime. These reports must include a quantification of carbon stocks and any new inventory data from the year prior. The costs of preparing an OPDR can vary considerably depending on the status of a forest project. If no substantial alterations of forest carbon stocks have occurred over the year prior, such as from harvest or disturbance, then an OPDR may be based primarily on modelling, and the time and expense involved in its preparation relatively minimal. However, if carbon stocks have been altered appreciably, new inventory data may be required, and costs may increase considerably. Depending on the complexity required for an OPDR in a given year, these costs can range from an estimated $4,560 to as much as $21,892 annually (personal communication, J. Golinkoff, 6 February 2014; personal communication, L. Wayburn, 8 August 2013; personal communication, R. Holderman, 01 June 2014; personal communication, P. Swedeen, 10 June 2014).

Verification costs are incurred upon project commencement and again at least every 6 years thereafter. Preparing the initial documentation for project verification costs between $4,000 and $5,000, and the verification itself is estimated to range between $32,838 and $68,413 for projects between 5,000 and 10,000 hectares in size. "Desk verifications" are undertaken between full verifications at the discretion of the offset project participant, and are estimated to cost between $10,946 and $16,419 (personal communication, J. Golinkoff, 6 February 2014; personal communication, L. Wayburn, 8 August 2013; personal communication, R. Holderman, 01 June 2014; personal communication, P. Swedeen, 10 June 2014). After a positive verification, credits issued by the registry incur a $0.25 fee per credit. When these credits are sold, transferring the credits from a project's account to an offset buyer's account requires a registry credit transfer fee of $0.03 per credit (Climate Action Reserve, 2014b). To protect against reversals, projects are required to hold "insurance" in the form of a contribution of a percentage of credits generated by the project to the Forest Buffer account. The magnitude of this contribution varies by project according to the assessed risk of reversal, and the actual cost of these requirements for participants depends on the current market price for credits. An overview of transaction costs for forest participants under California's programme is provided in Table 2.

TABLE 2. CALIFORNIA FOREST PARTICIPANT TRANSACTION COSTS

| Description | Registry Fees | | Participant Costs | | |
	Fixed	Variable	Fixed*	Variable	Occurrence
Opening Registry Account	$570				Initial
Project Submittal Fee	$798				Initial
Baseline Determination			$11,402 - $17,103		Initial
Listing Documentation			$2,280		Initial
Initial Verification Documentation			$4,560		Initial
Inventory				$19 - $29/ha	Initial and On-going (every 6 to 12 years) **
Registry Account Maintenance Fee	$570				On-going (annual)
OPDR Preparation			$4,560 - $21,892		On-going (annual)
Verification			$32,838 - $68,413		On-going (every 6 years)
Desk Verification			$10,946 - $16,419		On-going (discretionary)
Forest Buffer/Leakage				Dependent	On-going (upon credit issuance)
Registry Credit Issuance		$0.25/credit			On-going (at time of credit registration)
Registry Credit Transfer Fee		$0.03/credit			On-going (at time of credit sale)

*For projects between 5,000 and 10,000 hectares.
** All on-going costs are for the 100-year lifetime of the project.

V. PRICE FACTORS

While a consideration of transaction costs is an important factor in the overall performance of an environmental market, the rules governing a market also can have considerable impact on the price of goods transacted in that market. If price-containment mechanisms are included in an environmental market, some form of price-floor or supply control may keep prices high enough to in part compensate for other design features that may lead to higher transaction costs for producers. However, where these mechanisms are lacking, the price signal may be weakened such that participation in a market is compromised despite minimal cost barriers to entry for producers. The following sections consider these factors for the carbon markets of New Zealand and California.

A. Eligible Compliance Instruments in New Zealand

Under New Zealand's ETS, the primary unit of trade is the "New Zealand Unit" (NZU), which is equivalent to one tonne of carbon dioxide-equivalent emissions. The NZU is the unit the New Zealand government creates and distributes for trading under the ETS, and post-1989 forest participants are awarded NZUs for the carbon sequestration created on their lands. However, several types of international credits generated under the Kyoto Protocol also are eligible to be used for compliance under New Zealand's ETS (Ministry for the Environment, 2013c). These include:

(1) Emission Reduction Units (ERUs), which are generated by Joint Implementation projects that reduce emissions or create sequestration in Kyoto Annex B countries;

(2) Removal Units (RMUs), which are awarded to Kyoto Annex B countries for net sequestration generated by land use, land-use change, and the forestry sector; and

(3) Certified Emissions Reductions (CERs), which are generated by Clean Development Mechanism projects that reduce emissions and support sustainable development or create forest sequestration in developing countries.

The lack of restrictions on the use of ERUs, RMUs, and CERs under New Zealand's programme has exerted a considerable downward pressure on the price of carbon overall in the country's trading programme (World Bank, 2012). In 2011, a depressed global carbon market and a strong New Zealand dollar enabled regulated entities in New Zealand to purchase sufficient international credits to satisfy compliance obligations for several years. Indeed, of the 27 million units regulated entities surrendered for compliance in 2012, 95% were international credits (Ministry for the Environment, 2013d).

Barring the emplacement of any restrictions on the use of international credits, regulated entities will continue to be able to use these compliance instruments until 31 May 2015. After this date, New Zealand will no longer have access to these international credits due to the country's decision not to become a signatory to the Kyoto Protocol's second compliance period. Regulated entities will then be required to surrender NZUs or banked New Zealand-originated assigned-amount units from the first Kyoto compliance period, and it is expected that these constraints will begin to strengthen the demand for NZUs as regulated entities plan for 2015 and beyond (World Bank, 2014).

B. Eligible Compliance Instruments in California

Under California's cap-and-trade programme, regulated entities must either surrender ARB-issued California GHG emissions allowances or ARB offset credits. In addition to these compliance instruments, California and the Canadian province of Quebec have signed an agreement to link their cap-and-trade programmes commencing in 2014. As a result of this linkage, allowances and qualifying offsets generated in either jurisdiction may be used for compliance under both programmes effective starting 01 January 2014 (California Air Resources Board and *Gouvernement du Québec*, 2013).

1. Emissions Allowances

California emissions allowances represent one tonne of carbon dioxide-equivalent emissions, and are auctioned off by ARB once each quarter. Each quarterly auction includes a tranche of current-vintage allowances as well as an advance auction of a tranche of future-vintage allowances. Current-vintage allowances may be banked for surrender by regulated entities during future compliance periods, but future-vintage allowances from advance auctions may not be used for compliance before their vintage year (California Air Resources Board, 2012b). To bid in an auction, eligible parties must first register their intent to participate. In general, all covered entities, opt-in covered entities, and most voluntarily associated entities are eligible to register for participation in an auction. "Covered entities" are those whose emissions are covered under the cap,

while "opt-in covered entities" are those who are not obligated by law to have their emissions covered but have voluntarily elected to do so. "Voluntarily associated entities" are those parties that intend to purchase, sell, retire, hold, or clear allowances or offset credits but are not covered under the cap either by law or voluntary opt-in (California Air Resources Board, 2012b).

To reduce market volatility, the California programme includes several price containment mechanisms associated with GHG allowances. The first is an auction reserve price, which is set for each year's auctions to establish an overall allowance price floor. The auction reserve price is a predetermined price at which allowances will be sold to auction participants, and was established at $12.22 NZD in 2013. By regulation, the auction reserve price must increase by 5% annually plus the rate of inflation (California Code of Regulations §95911).

The second price-containment mechanism under California's programme is known as the Allowance Price Containment Reserve (APCR). The APCR is a set-aside allowance reserve that is used to establish a price ceiling through a fixed-price sale of allowances conducted six weeks after each quarterly allowance auction. Unlike general auctions, allowance sales from the APCR are open only to those entities whose emissions are covered under the cap, or those that have voluntarily "opted-in" to have their emissions covered (California Code of Regulations §95913). Allowances in the APCR are not auctioned, but rather are offered for sale in three fixed-price, equal-size tiers (California Air Resources Board, 2012b). If market conditions necessitate a reserve sale following a quarterly auction, allowances in the APCR are offered for sale in the lowest-price tier until the supply in that tier is exhausted, then the allowances in the next, higher-priced tier are made available (California Air Resources Board, 2012b). In 2013, the price of allowances in the three tiers was set at $45.65, $51.36, and $57.06, respectively. By regulation, the price of allowances in each tier must also increase at 5% annually plus the rate of inflation (California Code of Regulations §95913).

2. Offsets

In addition to qualifying GHG emissions allowances, offset credits issued by ARB also may be used by regulated entities for compliance under California's programme. While offsets present another form of price-containment by providing regulated entities with greater flexibility in satisfying their emissions requirements, California's programme limits the use of offsets to 8% of an entity's compliance obligation per compliance period (California Air Resources Board, 2012b). California's cap-and-trade regulations include provisions for three types of offset credits (California Air Resources Board, 2012b):

    (1) ARB offset credits;

    (2) Registry offset credits; and

    (3) Early action offset credits.

ARB offset credits are issued by the state, and are the only offsets that may be surrendered by regulated entities for compliance under California's law. Registry offset credits are issued by a recognized credit registry, and may be converted into ARB credits to be used for compliance if these credits meet all applicable regulatory criteria. Early action offset credits are offsets that have been created under qualifying early action protocols before the establishment of the ARB compliance protocols, and these offset credits may be converted to ARB credits to be used for compliance under California's trading programme (California Air Resources Board, 2012b).

In theory, offsets under California's programme should constitute a less-expensive compliance mechanism than actual GHG emissions allowances. This is due to the fact that the two are not perfectly fungible; unlike emissions allowances, offsets may only be used to satisfy up to 8% of an entity's compliance obligation in any given compliance period (California Air Resources Board, 2012b). In addition, there can be considerable heterogeneity among different types of offsets themselves, and these differences can alter the respective values of these compliance instruments. Certain offsets, such as those from forests, may carry the risk of reversal, and additional expenses might be necessary to

address this risk before an offset can be issued (California Air Resources Board, 2011b). Further, under California's program, offsets may be subject to invalidation even after issuance if they are found to be in breach of the terms governing the offset program. In such cases, invalidation generally occurs for three main reasons (California Air Resources Board, 2012c):

(1) An overstatement of greater than 5% is found with respect to the GHG emissions reductions or sequestration claimed for an offset;

(2) The emissions reductions or sequestration created has already been credited to another offset, leading to a double-counting of the credit claimed; or

(3) An offset project is found to be in regulatory non-compliance.

Under California's regulations, buyers are held liable in instances of offset invalidation (California Code of Regulations §95985). This means that regulated entities must finance additional measures to ensure that they are able to replace an offset in the event of invalidation. Despite these buyer-liability provisions however, measures can be taken by sellers to diminish invalidation risk by reducing the period in which an offset may be invalidated by ARB (California Air Resources Board, 2012c). Offsets issued by ARB are only subject to potential invalidation for an established period after their issuance (California Code of Regulations §95985). If a project undergoes verification only once, the offsets it generates will be liable to potential credit invalidation for a period of 8-years after issuance. However, if a second verification is conducted by a different ARB-approved verifier after an offset is issued, the liability period for potential invalidation is reduced from 8 years to just 3 years.

The verifier's primary role is to ensure that the quantity of offsets claimed is equivalent to the actual reductions or sequestration generated by a project. However, the double-verification requirements for the reduction of the potential invalidation risk period are currently problematic for forest projects created under early-action protocols. Credits from early-action projects are not eligible to be traded to satisfy compliance obligations

under California's programme until they are converted to ARB offsets by undergoing an additional, second verification process. As a result, early-action credits must already undergo verification twice to be eligible for compliance trading in California's market. At this point, these credits carry a potential invalidation risk period of 8 years. However, to reduce this period to 3 years, the credits would have to undergo a third, *additional* verification by a different, third verifier—but currently only two companies exist to offer verification for forest projects, meaning that it currently is not possible to convert forest offsets from early-action projects to higher-value credits with only a 3-year potential invalidation risk (California Carbon Info, 2014b).

C. Carbon Pricing in New Zealand and California

The lack of restrictions on the use of international credits and falling global prices were largely attributable to a drop in the price of carbon in New Zealand's market from about $20 NZD in May 2011 to around $7 by the end of that year (World Bank, 2012). In 2012, the price further dropped to about $2 as global carbon prices continued to plummet (World Bank, 2014). These global market conditions corresponded with amendments to the New Zealand ETS under the Climate Change Response (Emissions Trading and Other Matters) Amendment Act in 2012 that extended transitional measures designed to reduce the initial economic shock of compliance obligations for regulated entities. This included allowing certain trade-exposed sectors to continue surrendering compliance instruments at a 2:1 emissions-to-credit ratio as well as a continuation of the $25 "fixed-price" option for purchasing credits from the government, although the influx of international credits into the programme effectively mooted this price ceiling by driving the price of carbon in New Zealand's market to only a fraction of the cost of this option (World Bank, 2012).

In California, carbon prices have been closely tied to the annual auction reserve prices established for emissions allowances, which was set at $12.94 NZD for 2014 (California Air Resources Board, 2013b). On 14 May 2014, the seventh allowance auction was held by ARB, and a total of 26.2 million allowances were offered. Of these, the 16.9 million 2014 vintage allowances cleared for $13.12, while the 9.26 million 2017 vintage

allowances offered in the advance auction sold at the reserve price. The spot market for allowances of the 2014 vintage has hovered just below $13.70 a tonne throughout early 2014 (California Carbon Info, 2014a).

For offsets in California, pricing has been contingent in part on considerations surrounding invalidation, and the various levels of risk associated with these provisions have resulted in some price differentiation of offset credits in California's market. In some instances, sellers have fully assumed the risk of invalidation by including guarantees to buyers to replace any invalidated offset credits. While differences in invalidation periods and buyers' perceived risk make it difficult to determine the exact spread between prices for these guaranteed credits (which have been dubbed "golden offsets") and those subject to potential invalidation, prices for the so-called golden offsets have hovered around $11.41 NZD since the last quarter of 2013, and the spread with other offsets has been between $1.14 and $1.26 a tonne (California Carbon Info, 2014b).

VI. DISCUSSION

In general, transaction costs in environmental regulation depend on the nature of the problem in question and the policy design used to address it (McCann, 2013). With respect to environmental markets, the more difficult it is to define and quantify a given environmental good, the more transaction costs tend to increase. Standardizing quantification procedures can substantially reduce transaction costs, but may require a trade-off in terms of the accuracy with which an environmental good is quantified. Diminished accuracy may lead to an overstatement of environmental goods in some cases or an understatement in others, and can have implications for equity among participants (Fang et al., 2005; Cacho et al., 2013). If these errors are biased, reductions in accuracy may impact the overall environmental performance of a policy (Kerr, Brunton & Chapman, 2004). Risk and uncertainty also can threaten the integrity of a policy, and may increase the potential for *ex post* "hold up" problems. This can harm markets by deterring risk-averse trading partners and reducing trading opportunities (Williamson, 1985). A policy may be designed to include features that reduce risk and uncertainty, such as

monitoring, verification, and insurance requirements, but these design features add complexity to the institutional environment, and can increase transaction costs for participants.

The divergent policy designs adopted by New Zealand and California reflect contrasts between the two forest sectors as well as differences in the overall goals and objectives of the respective policies. In terms of defining and quantifying forest sequestration for its exchange in a carbon market, New Zealand's approach to forests is much less complex than that taken by California, and both the initial and on-going transaction costs incurred are considerably less. The simplicity and relatively low transaction costs of New Zealand's programme are facilitated by a forest sector that produces nearly all of its harvested timber—and carbon sequestration—on largely homogeneous, even-aged plantation forests. As a result, forests may be categorized by date of establishment with relative ease, allowing for an application of Kyoto rules and the use of a 1990 baseline for the overall forest sector. This design virtually eliminates any participant transaction costs associated with baseline establishment in New Zealand, and makes defining additional forest sequestration simply a matter of demonstrating that a forest was established after 31 December 1989. In contrast, plantation forestry in California is relatively limited, and most harvesting occurs on uneven-aged, mixed-species forests. As a consequence, using the date of forest establishment as a baseline is not practicable under California's programme, and each individual forest project must model its own business-as-usual baseline against which to quantify additional sequestration. This represents a considerable initial fixed cost for forest landowners interested in becoming participants under California's climate law.

In addition to facilitating baseline establishment, the relative homogeneity of New Zealand's plantation forests also enables a comparatively standardized approach to carbon stock quantification through the use of look-up tables. In New Zealand, the FMA look-up tables for exotic forests are expected to provide an estimate of forest carbon stocks within about 10% of the true value at a 90% confidence level (Ministry for Primary Industries, 2012). Reduced accuracy is expected on indigenous forests owing to their greater variability, but given the smaller carbon gains and higher sampling costs,

this is considered a cost-effective compromise (Ministry for Primary Industries, 2012). The generic look-up tables, which are required for projects of 100 hectares or less, may not provide as high a degree of accuracy at the individual project level, but because these tables are based on national averages, errors at the project level are expected to average out when reported for the entire sector (Kerr, Brunton & Chapman, 2004). While this might affect equity among participants by overstating carbon stocks in some cases and understating it in others, the transaction costs associated with this approach are substantially less than those incurred with the full inventory, modelling, and verification required by California.

In contrast to New Zealand's approach, forests in California are not included as a capped sector. As a result, carbon quantification methodologies under California's programme focus on the individual project level. For participating forest projects, carbon stocks must be quantified to a "high degree of accuracy" at a 95% confidence level or better to avoid confidence deductions (California Air Resources Board, 2011b). However, the smaller the area where accuracy is required, the more costly and difficult it is to meet such requirements (Kerr, Brunton, & Chapman 2004). The comparative heterogeneity of the forests under California's programme further increases these challenges by reducing the degree to which quantification methodologies can be standardized. This has a predictable effect on transaction costs, and the initial inventory/baseline determination necessary under this approach represents the largest initial outlay for forest participants in California. While some organizations currently are exploring the possibility of aggregating smaller forest parcels to help defray these initial fixed costs, California's programme still presents considerable scalar barriers to entry for smaller projects (personal communication, R. Holderman, 01 June 2014). Further, because forests are not included as a capped sector under California's programme, forest projects are susceptible to domestic emissions "leakage," or the shifting of harvesting activities from the project area to other forestlands where emissions are not regulated. To compensate for leakage, a deduction must be assessed to the total amount of sequestration generated by a project according to its leakage risk, which reduces a project's overall benefits for participants. The problem of domestic leakage is obviated in New Zealand by the inclusion of its forests under the national emissions cap.

While certain policy design decisions may be closely influenced by the characteristics of an environmental good itself, other design components may be less driven by these factors. Mechanisms to reduce uncertainty and risk are one example of this, and these decisions largely are dictated by the degree of risk acceptable to the policymakers designing a particular programme. In New Zealand and California, the different approaches taken to ensuring the "permanence" of sequestered carbon have considerable implications for transaction costs. For instance, while participants in both programmes are obligated to submit periodic monitoring reports, the frequency and complexity of these requirements are considerably greater under California's policy. Forest projects in California's programme must undergo third-party verification every 6 years while a project is actively generating sequestration credits, and participants must submit monitoring reports every year over the 100-year lifetime of a forest project. In New Zealand, forest participants also must submit monitoring reports in the form of emissions returns, but these are only required at the end of each 4-year emissions return period, or upon the reversal of credited stocks. Although an external compliance audit may be undertaken of any New Zealand emissions return to ensure that the number of units claimed is correct, California's approach provides considerably more oversight of credited sequestration and its permanence—but the on-going costs of doing so are not insubstantial.

Some differences in transaction costs may arise simply from the unique goals and objectives of a policy itself. Both New Zealand and California seek to harness the sequestration potential of forests in addressing global climate change, but the environmental goods created by the two programmes are not necessarily identical. Under California's compliance protocols, forest projects located anywhere in the contiguous United States are eligible to generate offsets (and this could potentially expand to include Quebec with the recent linkage of the two markets). In some regions, particularly in the southeastern United States, even-aged plantation forestry is the predominant means of timber production. A programme that focused solely on forests like these could lend itself to a less complex, less costly approach for generating forest sequestration. However, the requirements of California's programme extend beyond merely the creation of carbon sequestration; projects must perpetuate native forest species as well as demonstrate

sustainable "natural forest management" practices and uneven-aged stocking (California Air Resources Board, 2011b). This design promotes carbon sequestration while attempting to maintain the relatively natural state of the forests generating it, and reflects both the unique goals and objectives in California's policy as well as the different character of the forests on which it seeks to promote carbon projects. However, while this approach might produce more so-called "co-benefits" along with sequestered carbon, it also adds complexity—and transaction costs—for forest participants under California's programme.

In theory, the origin of carbon sequestration is immaterial to the environmental performance of a policy seeking to reduce atmospheric GHG concentrations. However, the apparent importance of these co-benefits arising from offsets projects, and their retention within the state, is an oft-raised concern in regard to the implementation of California's overall climate policy. Discussions regarding expanding offset eligibility to include international credits often are greeted with considerable concern over the "leakage" of environmental co-benefits (as well as revenue to offset project developers) to sources abroad (California Carbon Info, 2014b). Indeed, similar sentiments are reflected in legislation such as Senate Bill 605, which was introduced by State Senator Ricardo Lara in 2013. If passed, this bill would amend the Global Warming Solutions Act to limit the use of offsets in California to only those generated in North America by member states of the former Western Climate Initiative. Such attitudes suggest that, while California's law is aimed broadly at addressing global climate change, the full scope of its benefits are viewed (and valued) in perhaps a broader light.

Finally, while transaction cost implications undoubtedly are an important consideration for potential participants in New Zealand and California, the divergent approaches to price containment under each programme also have proved to be a factor for forest participants. For post-1989 forest landowners in New Zealand, the transaction costs involved in becoming a participant in the country's ETS are relatively insubstantial when compared to the costs that must be incurred by entities seeking to enter California's forest offset market. However, the unlimited eligibility of certain international credits under New Zealand's carbon market has lead to a considerable erosion of the price of carbon

since the establishment of the New Zealand ETS (World Bank, 2012). Although a price ceiling does exist in the form of a fixed-cost purchasing option at $25 NZD a credit, the ready availability of international credits has driven carbon prices so low as to make this of little consequence. While the lack of a price floor in New Zealand's market may be beneficial for regulated entities obligated to purchase compliance instruments to cover their GHG emissions, it has drastically altered the economics for entities generating sequestration credits under New Zealand's programme, especially the post-1989 forest landowners that initially registered to participate in New Zealand's carbon market. In contrast, despite the higher transaction costs for participants under California's forest offset programme, these greater costs have in part been compensated for by relatively stringent restrictions on credit eligibility as well as provisions for price containment. These components have reduced market volatility under California's carbon market and maintained higher prices for carbon overall.

Conclusion

The policies of New Zealand and California provide a unique opportunity to investigate the transaction cost implications of two different approaches to forests in climate policy. These programmes suggest that the nature of the forest sector itself can be important to influencing policy design decisions—and thus transaction costs—but that many of these costs may also be driven by the specific goals and objectives of a particular policy. Designing policies to increase simplicity and standardization can diminish participant transaction costs, but the nature of the forest sector itself might limit the ability of policymakers to do this. In other cases, the goals and objectives for a policy may result in a more complex design being chosen even if a simpler approach is possible. The experiences of New Zealand and California offer insight on how these factors affect transaction costs, and provide a useful starting point for future policies seeking to include forests in climate policy.

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