Policy design for forest carbon sequestration: A review of the literature

Ing-Marie Gren *, Abenezer Aklilu Zeleke

Department of Economics, Swedish University of Agricultural Sciences, Sweden

A R T I C L E   I N F O

Article history:
Received 16 June 2015
Received in revised form 7 May 2016
Accepted 13 June 2016
Available online 18 June 2016

Keywords:
Policy design
Forest carbon sequestration
Survey

A B S T R A C T

Forest carbon enhancement provides a low-cost opportunity in climate policy, but needs efficient policy design to be implemented. This paper reviews studies in economics on efficient design of policies for forest carbon sequestration and compares their findings against design systems in practice. Specific design problems are associated with the heterogeneity of landowners, uncertainty, additionality, and permanence in carbon projects. Different types of discounting of the value of the forest carbon sink compared with emissions abatement are suggested in the literature for management of most design problems, together with optimal contract design and emissions baselines for managing additionality and permanence in carbon sequestration. Design systems in practice, where forest carbon corresponds to 0.5% of all carbon volume subject to a pricing mechanism, mainly rely on additionality tests by approved standards on a project-by-project basis, and on buffer credits for management of permanence. Further development of forest carbon sinks as offsets in voluntary and compliance markets can be facilitated by applying tools for contract design and offset baseline management recommended in the literature.

© 2016 The Authors. Published by Elsevier B.V. This is an open access article under the CC BY license (http://creativecommons.org/licenses/by/4.0/).

1. Introduction

The damaging impacts of anthropogenic emissions of greenhouse gases (GHG) have been demonstrated in a number of studies (e.g., IPCC, 2014). Recognition of the need to stabilize the carbon content in the atmosphere has been manifested in a number of international and national agreements and policies, such as the Kyoto Protocol, the Paris Agreement, and the EU climate policy. The main focus of these agreements and policies is on reducing GHG emissions, but the carbon content in the atmosphere can also be offset by carbon sink enhancement. Carbon sequestration occurs in above-ground growing biomass and in below-ground soil from e.g., soil biomass and decomposition (e.g., IPPC, 2014). These two forest carbon pools are linked, as felling of trees releases carbon to the soil and this is incorporated in successional biomass production. Global carbon absorption in these two forest carbon pools in the period 2000–2007 amounted on average to 4.1 Pg C/year (Pan et al., 2011). This corresponds to approximately 30% of the emissions from fossil fuels in 2010 (IPCC, 2014). This absorption is counteracted by the release of carbon in the soil by deforestation. The global release from forest conversion amounts to approximately 2.8 Pg C/year (Pan et al., 2011).

Thus, the potential reduction in release of carbon from avoided deforestation and increased above-ground sequestration through forest plantation and improved forest management can be significant for climate policy. The Kyoto Protocol allows for carbon sequestration by afforestation and reforestation under the Clean Development Mechanism (CDM) within the LULUCF (Land Use, Land Use Change, and Forestry) activities, but this was limited to a small fraction of emissions in 1990. More recently, the need for taking action against carbon releases from forest was recognized in the Paris Agreement. In practice, carbon sequestration has been introduced under different national regulations on GHG emissions and voluntary systems (Peters-Stanley et al., 2012; Kerr, 2013; Goldstein et al., 2014). A majority of these carbon sink offset projects have been incorporated in different voluntary systems, in particular under the Reducing Emissions from Deforestation and forest Degradation (REDD) program, which was created by the United Nations in 2008 to enhance use of carbon sinks (UNFCCC, 2008). Despite these efforts, in 2013 total carbon sequestration accounted for only 0.5% of the total volume of carbon trade (Goldstein et al., 2014; Kossoy et al., 2014).

The potential of carbon sequestration to help meet climate targets depends not only on the size of carbon sink enhancement, but also on the cost compared with that of other measures, in particular fossil fuel reductions. The large body of literature calculating the cost of carbon sequestration shows that the marginal costs of carbon sink enhancement can be considerably lower than those of carbon emissions reduction (see reviews by Sedjo et al., 1995; Stavins, 1999; Richards and Stokes, 2004; van Kooten et al., 2004, 2009; Manley et al., 2005; Phan et al., 2014). Benefits in terms of cost savings from introducing carbon sinks into climate programs have been reported in studies on the cost of meeting global or EU-level climate targets, which conclude that costs can be reduced by up to 40% (e.g., Tavoni et al., 2007; Anger and...
Sathaye, 2008; Bosetti et al., 2011; Michetti and Rosa, 2012; Gren et al., 2012). However, whether these cost savings can be achieved depends on the policy design. Examples of policies are carbon sink as offset within a carbon emissions trading scheme or tax scheme, or compensation payments for afforestation to enhance carbon sink. An important economics question arises is how these policies can be designed so as to foster changes in land users’ behavior at minimum cost to society. This study investigated the answers provided in the literature and how carbon exchange systems have been designed in practice.

Any type of policy targeting carbon sequestration has to deal with specific design problems; heterogeneity, uncertainty, additionality, and permanence. Heterogeneity refers to the fact that carbon sequestration of a certain area of land differs between regions because of differences in climate and geo-hydrological conditions, which means that carbon sequestration per unit land depends on the location of the project. This is in contrast to emissions from fossil fuel products, which are quite similar per unit use of, e.g., gasoline. Uncertainty occurs because of stochastic weather conditions affecting biomass growth, and from errors in monitoring and measuring sequestration (e.g., Houghton, 2005). Although there is some uncertainty in the conversion of fossil fuel products to carbon dioxide equivalents, it is negligible compared with that in carbon sequestration (Gren and Carlsson, 2013). Additivity refers to the difficulty in assessing whether the project would be implemented without the policy in question, e.g., whether a piece of land would be converted to forest without a compensation payment for carbon sequestration. Permanence in carbon sequestration during the project period can be hampered by natural causes, such as variations in temperature and precipitation, storms and wildfires, but also by intentional violation of the project rules, e.g., harvesting before the project period expires. Another aspect of permanence is the use of harvested wood products. Carbon sequestration lasts for a longer period when forest products are used for building houses, rather than for heating as bioenergy. These policy challenges for carbon sink projects can be partly addressed by increasing the transaction costs through monitoring and verification of carbon sink enhancement projects (e.g., Cacho et al., 2013), but also through clever policy design, mitigating high transaction costs.

The economics literature on policy design for forest carbon sink enhancement is relatively limited compared with that on the cost of carbon sequestration and cost savings from introduction of carbon sink into climate policy programs, where there are a number of review studies containing over 35 studies (van Kooten et al., 2004, 2009; Manley et al., 2005; Sedjo and Sohngen, 2012; Phan et al., 2014). There are also reviews on policies for carbon sequestration, but they focus on the structuring of policy design without survey of studies (Angelsen, 2008), or on specific forest carbon projects such as REDD + (Hufty and Haakenstad, 2011), or a specific policy instrument such as contract design (Fortmann et al., 2014), or only include a few studies (less than 10) (Capon et al., 2010).

In our view, the main contribution of the present study is that it extends the existing survey literature on policy design. We focus on studies with the explicit aim of analyzing policies directed at improving forest carbon sequestration. This includes studies investigating side-effects of forestry and agricultural policies on carbon sequestration or release from soil, such as subsidies on bioenergy or taxes on emissions of carbon dioxide (e.g., van Kooten et al., 1995). It also excludes studies calculating the cost of carbon sequestration in terms of necessary compensation to forest owners (e.g., Lubowski et al., 2006; Yu et al., 2014). Studies considering uncertainty in carbon sink where policy makers allocate risk among different abatement and carbon sink options (e.g., Benitez et al., 2007; Fuss et al., 2013; Gren and Carlsson, 2013; Haim et al., 2014) are also excluded, unless they contain explicit policy design for carbon sinks.

When searching for studies with the explicit aim of analyzing policies for forest carbon sequestration, we used common search engines such as Scopus, Thomsen Reuters Web of Science, and Google, and applied key words such as ‘forest carbon sequestration’, ‘policy’, ‘policy design’, ‘economics’, and ‘incentives’. In total, we found 45 studies which address one of the key policy design challenges or describe and/or evaluate carbon policies in practice. However, we cannot exclude the possibility that we overlooked interesting and relevant studies, and the review is thus not exhaustive.

The study is organized as follows. We start by presenting the literature within economics addressing one or several of different types of the specific policy design problems with carbon sink enhancement. Next, studies on carbon sink policies in practice are presented, followed by a comparison and discussion of the main findings in the literature and main design features in practice. The study ends with a brief summary and concluding remarks.

2. Efficient policy design

It can be argued that the policy design for mitigating emissions from combustion of fossil fuel is relatively easy, since the effect on the content of carbon in the atmosphere is the same irrespective of location of the emissions sources. This is not the case with carbon sequestration. The impact of carbon sequestration is site-specific and depends on factors such as soil quality, tree species, and local climate (e.g., Houghton, 2005; Pan et al., 2011). A cost-effective policy design requires policies, such as subsidies for afforestation, to take this heterogeneity into account and adjust to the site-specific sink enhancement. In principle, this would not pose much of a challenge if the policy maker and agents had information on carbon sequestration in each plot.

The complicating factors are associated with different types of uncertainty in carbon sequestration, which can lie with the policy maker and agents, or with asymmetric information, which lies with only one party, usually the policy maker. One uncertainty common to both parties is the variability in weather conditions which affects biomass growth and thereby carbon sequestration in above-ground and below-ground living biomass. Another is the uncertainty created by errors in measuring, monitoring, and verifying carbon sequestration. A third uncertainty factor relates to permanence in a created sink, which can be turned into a source through natural events such as wildfires, storms, and insect and pathogen outbreaks. Asymmetric information on, e.g., baseline emissions and costs of carbon sequestration, is a source of uncertainty for the policy maker but not the agent, who implements a project with full information. Another type of asymmetric information is associated with intentional harvesting of planted trees before expiration of the project period and with the absence of due care to avoid or mitigate carbon reversal from natural causes, which is known by the agent but not the policy maker. These asymmetric allocations of uncertainty between buyer and seller make it difficult to ensure additionality and permanence in carbon sink projects because of the need to measure and establish a baseline and to monitor and verify the carbon sink enhancement by the project under a long period of time.

In the following, we review suggestions in the literature on policy design for managing the challenges associated with heterogeneity and uncertainty in carbon sequestration common to both parties in forest carbon exchange, and asymmetric information with respect to forest carbon cost and sequestration.

2.1. Heterogeneity and uncertainty in biomass sequestration

Instant uncertainty arises from the biological process of carbon sequestration, which depends on stochastic weather conditions, and this uncertainty is common to both the buyer and seller of carbon credits. The literature dealing with this type of uncertainty can be classified into two main categories. One category compares total abatement costs under a system making carbon payments per ton forest carbon with those of systems using other payment bases, such as unit area of land or forest practice (Parks and Hardie, 1995; Kim and Langpap, 2014). The other category regards uncertainty as costly for society and
seeks to develop methods for price discrimination between certain and uncertain carbon emissions reduction, where fossil fuel reduction is regarded as relatively certain and carbon sequestration as uncertain (Kim et al., 2008; Kim and McCarl, 2009; Gren et al., 2012; Munnich Vass et al., 2013; Gren and Carlsson, 2013).

Empirical studies on efficiency losses in a uniform payment system show differing results. Cost-effective allocation of carbon sequestration requires the cost of a marginal increase in carbon sequestration to be equal for all landowners. This condition is not fulfilled when landowners with different carbon sequestration rates per unit area of land receive the same payment. Efficiency losses are then created from the higher total cost of achieving a certain carbon sink enhancement compared with a cost-effective solution. These losses can be relatively low, as shown by Parks and Hardie (1995) for forests planted on marginal agricultural land in the US. Using simulations, they calculate and compare total sequestration obtained under a per ha and per ton carbon-based policy for a given budget and demonstrate relatively small difference in total carbon sequestration between the policies; 10% higher carbon sequestration in the cost-effective policy with differentiated payments. On the other hand, relatively large efficiency losses are reported by Kim and Langpap (2014), who compare the costs of different carbon sink policies for improved forest management practices (thinning, fertilization, and fire hazard reduction) in forests held by non-industrial private forest owners in the US. They develop an econometric model to estimate the probability of adopting a certain forest practice, which is used to simulate effects on carbon sequestration from carbon-based and practice-based incentive schemes. The results show that, for a given total cost, annual carbon sequestration can increase between approximately 40% and 300% on moving from a practice-based to a carbon-based payment system. The variation is caused by the practice chosen as a basis for payments.

With respect to uncertainty in carbon sink, the value of a unit uncertain carbon sink is lower than that of a unit certain emissions reduction when the policy maker is risk-averse. This risk discount is measured by risk aversion and risk in carbon sink. Kim and McCarl (2009) measure the risk discount by mean and standard deviation in carbon sink and the buyers' risk attitude, expressed as acceptable confidence interval in sequestration. They point out that data on mean and variance can be obtained from field measurements, biophysical data, or by proxies such as crop yield. In an application to the East Texas region in the US, they show that the discount is about 20% of the carbon price at the 90% confidence interval.

A similar approach is applied by Gren et al. (2012); Munnich Vass et al. (2013), and Gren and Carlsson (2013), who calculate the optimal discounting of stochastic forest carbon sink in the EU 2020 climate policy, which includes both a trading market for emissions reductions (ETS) and national allocation plans. Gren et al. (2012) show that the discounting of forest carbon sink on the ETS market is relatively low, approximately 5%. This is due to low supply of forest carbon credits on the market because of the main use of the credits for reducing the relatively high cost of fulfilling the national allocation plans. Munnich Vass et al. (2013) examine the impact on equity from including uncertain carbon sequestration in the EU climate policy and show that equity increases when carbon sink is treated as uncertain instead of certain. The reason is that the costs increase for the rich countries, which gain the most from introduction of a low-cost carbon sink option under deterministic conditions. Gren and Carlsson (2013) consider abatement of carbon emissions from fossil fuel and forest carbon sinks as uncertain. Since the uncertainty in fossil fuel abatement is lower than that of carbon sinks, there is still a risk discount of carbon sinks, but it is lower than when fossil fuel reduction is regarded as certain.

Kim et al. (2008) develop uncertainty discounting by adding a time perspective. They investigate the impact of uncertainty in carbon sequestration on carbon prices by comparing prices under risk-free conditions with those under risky conditions. They calculate the discount by equalizing the value of carbon offsets from a perfect offset, without any uncertainty, with that of forest carbon sequestration under risk. The cost of a perfect offset is calculated as the discounted purchasing costs of current and future offsets, where the time period is determined by the contract length. The effective price is then defined as the discounted outlay divided by total amount of offsets. The effective price for the impermanent project is calculated in a similar way, but adds future costs in terms of buyback of offsets in the event of impermanence before the contract expires, maintenance costs of carbon sequestration, and variation in carbon sequestration over time. The constant permanence discount then increases in buyback, maintenance costs, and growth rate in carbon prices.

2.2. Additionality

As mentioned in the introduction, problems with additionality arise when projects funded with carbon credits would have been undertaken without the credits. If carbon sink projects can be used as offsets by a firm included in a compliance market, the firm can buy a non-additional carbon sink and reduce its own abatement. The overall abatement is then decreased, but without a compensating increase in the carbon sink. Thus, inability to separate additional from non-additional carbon sinks may lead to higher costs and lower emissions reduction. Two main approaches are suggested to manage this problem. One is to make the individual carbon seller reveal their real cost of the carbon project (MacKenzie et al., 2012; Mason and Plantinga, 2013; Cordero-Salas et al., 2013; Delacote et al., 2014; Mason, 2015). The other is to accept the asymmetric information and instead mitigate the problems created by its existence (Murray et al., 2013; van Benthem and Kerr, 2013; Bento et al., 2015). Common to both approaches is the assumption that the principal, i.e., the buyer of credits, knows the distribution of business—as-usual (BAU) deforestation and opportunity costs, but cannot identify those of an individual agent.

Under the first approach, contracts are developed which give landowners with relatively low opportunity costs some extra compensation, information rents, in order to accept a contract designed for them. Mason and Plantinga (2013) compare the costs of such a contract design for private forest landowners in the US with the costs under a unit subsidy system. The results indicate that the total cost of the optimal contract system would be half the cost of a unit subsidy for a given increase in forest area. Instead of differences in opportunity costs between agents, Mason (2015) considers differences in agents' tolerance of risk, where risk occurs in the opportunity cost of land from e.g., stochastic output prices on forest products. Agents with relatively high risk aversion and, hence, low risk tolerance, are likely to accept a lower certain payment from a principal than less risk-averse agents. Mason (2015) derives the optimal allocation of a fixed transfer, which can be regarded as an insurance, and a unit payment per area of forest land set aside in order to make the agents reveal their risk preferences. However, as considered by Cordero-Salas et al. (2013), a particular landowner can sometimes face relatively high and other times relatively low opportunity costs, depending on changes in product prices. They show in a theoretical setting that a two-part tariff contract with a base and performance payment is needed to make the agents reveal their true opportunity costs in each period. The base payment corresponds to the information rent needed for ensuring first-best forest conservation of the low-cost type.

While most studies maximize net social welfare when identifying optimal contracts, Delacote et al. (2014) investigate the consequences of contract design and improved information on the agent's opportunity cost when the principal has different decision rules. Given the multiple aims of REDD to ensure carbon sequestration and alleviate poverty, the principal can be guided by these concerns, or simply maximize their own net income. Improved information for the profit-maximizing principal allows for less rent payments to the agent, but has no effect on avoided deforestation. In contrast, a principal with environmental objectives uses the gains made from less payment to low-cost agents to
pay agents with higher opportunity costs that can opt in, and the deforestation area decreases. Effects on deforestation for a principal guided by poverty alleviation are ambiguous, since they depend on the weight assigned to different poor agents, but the income transfer to the poor increases.

The second approach to non-additionality considers two sectors, capped and uncapped, where the capped sector, which can be within a carbon trading sector, can buy offsets from the uncapped sector as compensation for its own emissions. In principle, upscaling of a capped sector will result in equal or lower costs, since more low-cost options become available. The risk of introducing non-additional projects that would have been undertaken without the offset credit hampers the cost-effectiveness of an enlarged capped sector. The literature suggests and discusses seven main approaches to mitigate this threat; additionality tests of offset credits, increased monitoring and verification, increasing the baseline scale of offset projects, lower emissions baselines for the offset sector, more stringent emissions targets for the capped sector, trading ratios between the capped and non-capped emissions reductions, and a limit on the offset sector.

No study to date has analyzed and compared all these options, but all studies include trading ratio between emissions reductions and carbon sink as an option, i.e., that trading between the two options is made at a certain trading ratio. A trading ratio higher than unity (i.e., more than one unit of carbon offset for one unit emissions in the capped sector) reduces demand for offsets, which in turn decreases welfare for the uncapped sector (Murray et al., 2013). This can alleviate non-additionality, since ‘real’ projects will opt out because of the price decrease (van Bentham and Kerr, 2013). On the other hand, a trading ratio below unity mitigates the problem of non-additionality, since more ‘real’ projects will be supplied because of the price increase (Bento et al., 2015). Increasing the abatement requirement for the capped sector will raise its costs, but the net welfare of the uncapped sector will increase because of the higher demand for offsets (Murray et al., 2013; Bento et al., 2015). Increasing baseline emissions on a project-by-project basis increases efficiency and transfers to sellers of offsets, since more projects opt in (van Bentham and Kerr, 2013). However, this also implies that more non-additional offsets opt in (van Bentham and Kerr, 2013; Bento et al., 2015). A limit on the amount of offset credits has no effect on non-additional carbon sink, but raises the cost of abatement and reduces the transfers to the non-capped sector (Bento et al., 2015).

Additionality tests can be made by scrutinizing each project or by relying on secondary data and common practices in the region and sector. Murray et al. (2013) show that total abatement costs are lowest under the project-based additionality test, but the transaction costs are high, because of the many small and spatially dispersed offset projects and the existence of a fixed cost component for trade with each project.

A key finding by van Bentham and Kerr (2013) is that it is almost always efficient to scale up offset programs as an entity to a political jurisdiction at the regional or national scale with one emissions baseline. Sellers then have to participate with all their forested land, instead of only some with plot-specific baselines. This makes it more difficult for them to exercise their informational advantage. However, if a spatial correlation exists between plots which jointly determine baseline emissions among plots, these efficiency gains are reduced.

2.3. Permanence

Non-permanence arises from intentional felling of trees, e.g., to make a profit from selling timber, and from natural disturbances. In addition to market risks associated with carbon prices and opportunity costs of land, compliance failures of partners, and political risks, there are specific natural risks associated with carbon sequestration (e.g., Dutschke and Angelsen, 2008; Cooley et al., 2012). Cooley et al. (2012) make a profound investigation of the measurement of these carbon reversal risks and point out the importance of spatial correlation in risks among projects. Such correlation may exist among projects in a similar ecological zone, where a fire event can spread to neighboring parcels and affect a large part of the forest area and number of nearby projects. Correlation can also exist between events, such as insect outbreaks and wildfire, depending on temperature. When these risks are negatively or positively correlated, the total risk of the program decreases or increases, respectively. Fire risks can be reduced by forest management practices such as mechanical thinning and controlled burns, which can be promoted by carbon credits (e.g., Daigleault et al., 2010). However, the specific policy design problem arises from the buyers’ difficulty in observing the agents’ management and protective measures against risk of carbon reversals and the associated risk of moral hazard.

Dutschke and Angelsen (2008) and Palmer (2011) provide a number of suggestions on how to deal with carbon reversal within the REDD framework. Currently, Annex I countries purchasing REDD credits are liable for the release of carbon in violation of the contract (UNFCCC, 2005). Liability in cases of carbon release can be shared between host and buyer governments, as suggested by Eliasch (2008), or in a bilateral project-based setting. Palmer (2011) discusses pros and cons of liability at the government or individual level in the host country, and emphasizes the need for defining carbon property rights. In addition to liability management, Dutschke and Angelsen (2008) present and discuss advantages and disadvantages of several options which include temporary credits, credit buffers, pooling of reversal risks, and commercial insurances. However, neither Dutschke and Angelsen (2008) nor Palmer (2011) provide in-depth analysis of efficient design of suggested policies.

Economics studies analyzing the properties of policies mitigating non-permanence can be classified into two categories; contract design under moral hazard for promoting due care of the forest (Gulati and Vercammen, 2005; Cordero-Salas and Roe, 2012; MacKenzie et al., 2012; Cordero-Salas et al., 2013; Pana and Gheyssens, 2015; Veronesi et al., 2015; Engel et al., 2015) and duration of the credits (Feng et al., 2002; Olschewski et al., 2005; Maréchal and Hecq, 2006; van Kooten, 2008). The studies on contract design investigate how to allocate a fixed initial payment and a performance payment, which depends on discounted current and future net benefits for entering the contract and alternatives for both parties. This is made under different assumptions on principal behavior (Cordero-Salas and Roe, 2012), liability allocation between buyer and seller (MacKenzie et al., 2012), choice of emissions baseline (Pana and Gheyssens, 2015), and existence of uncertainty in future net benefits (Engel et al., 2015).

Cordero-Salas and Roe (2012) show that the optimal contract design, a close to zero fixed payment and almost all upon delivery, is the same irrespective of whether the principal is altruistic or not. In both cases, relatively high opportunity cost for the agent requires relatively high net future value for sustainable cooperation. The necessary discounted net benefits for sustaining cooperation are reduced when the principal is altruistic, since he/she puts a weight on the agent’s wellbeing. This can be of specific relevance for REDD projects, which usually aim at achieving poverty reduction in addition to carbon sink enhancement.

In principle, there are three types of liability regimes; the principal, the agent, or nobody is responsible for the carbon sink reversal. MacKenzie et al. (2012) show that a switch from a practice of buyer to seller liability would improve enforcement of the contract and hence increase investment in carbon sink. Investment may also be higher under no liability compared with buyer liability, since the buyer does not have to pay any penalty for carbon releases. On the other hand, the carbon sink may be lower, since nobody is held responsible for its realization and associated creation of incentives.

With respect to the role of baseline, Pana and Gheyssens (2015) note that the practice so far for REDD projects has been to use historical information, such as average deforestation, which has the advantage of being transparent. They suggest and analyze this system with three alternatives; model-based projection of BAU in the absence of REDD, a fixed
or variable system, and a ‘corridor’ system with upper and lower bounds of historical averages. The ‘corridor’ sets payments that depend on distance from the lower bound of deforestation; the closer to the upper bound (above which there is no payment) the lower the payment, and the payment is zero when the upper bound is reached. The system has the advantage of accounting for fluctuations in the opportunity cost of keeping forest caused by e.g., changes in market prices and weather conditions. They also show that a modest upward-biased increase in the corridor band width increases efficiency, but not a symmetrical or downward-biased increase. The reason is that the upward-biased increase gives higher incentives to stay in the program, thereby avoiding deforestation.

However, permanence is not only uncertain because of climate events, but also because of the opportunity cost of land. Prices of agricultural commodity crops are volatile, which makes the net benefits from conversion of forest uncertain. Engel et al. (2015) investigate the payment needed for ensuring permanence in forest cover with a minimum probability when the variable part of payment is indexed on either carbon prices or agricultural commodity prices. The former means that payments vary with carbon prices and the latter that payments vary with the opportunity cost of land. They show that payments are low for a high correlation between carbon prices and opportunity costs, since the price of carbon and the opportunity cost are then high or low at the same time. On the other hand, the payments for ensuring permanence are high when there is large relative volatility in the indexed component. Similarly to Mason (2015), they also show that increases in the fixed payment reduce total payment cost for ensuring permanence, since this reduces total risk for the landowner.

The studies on temporary credits approach the permanence problem by investigating how to value and compare temporary and permanent offsets (McCarl and Schneider, 2000; Marland et al., 2001; Olschewski et al., 2005; Maréchal and Hecq, 2006; van Kooten, 2008) or how to optimally design policies over time (Feng et al., 2002; Gulati and Vercammen, 2005). As suggested by McCarl and Schneider (2000); Marland et al. (2001), and van Kooten (2008), it would be most straightforward to treat the project as a carbon sink when there is sequestration and as a source when there is carbon reversal. If this is not possible, one way of comparing temporary and permanent credits is to convert the associated duration of carbon sequestration into permanent abatement (McCarl and Schneider, 2000; Marland et al., 2001; Dutschke, 2002; van Kooten, 2008). This is similar to the discounting with respect to risk described in Section 2.1 of this paper, but now the value is determined by the duration of the temporary credit and, hence, carbon sequestration. The shorter the duration of the temporary credit, the lower the value compared with a permanent credit. The conversion rate is also related to forest growth and varies between 42 and 150 ton-years of temporary sequestration to cover 1 ton of permanent carbon. They show that carbon sequestration should be used as early as possible in the planning period, since this reduces emissions abatement costs and allows carbon emissions to flow into a carbon sink. In a world of no uncertainty, they show how different incentive systems can achieve first-best outcome; sales of contracts with different contract periods to landowners, and sales of carbon offsets for sequestration but purchases of credits for carbon releases.

3. Design of forest carbon exchange systems in practice

Forestry in carbon markets is a relatively new phenomenon, the trading volume of which increased from 2.1 MtCO₂e in 2005 to 32.7 MtCO₂e in 2013 (Goldstein et al., 2014). This covers approximately 0.5% of the carbon emissions subject to pricing instruments, including both tax and cap-and-trade systems, in the world (Kossoy et al., 2014). A vast majority of the forest carbon was exchanged under voluntary systems in both 2012 and 2013, see Table 1.

There was a slight increase in total trade in forest carbon between 2012 and 2013, which was due to the increase in voluntary forest carbon offsets. Exchange in two compliance markets, CDM and New Zealand ETS, almost ceased in 2013. The CDM decline was a result of the ending of the first Kyoto commitment period. The reason for the decline in New Zealand ETS was a government decision allowing

| Compliance and voluntary markets | Volume, MtCO₂e | Average price, USD/ton CO₂e |
|----------------------------------|----------------|--------------------------|
|                                  | 2012           | 2012                     |
| CDM/JI                           | 0.5            | 1.1                      |
| California cap-and-trade         | 1.5            | 8.2                      |
| Australia CAPM CIF               | 2.9            | 13.3                     |
| New Zealand ETS                  | 0.2            | 7.9                      |
| Others (REGG, J-VER, T-VER)      | 0.6            | 25.3                     |
| Voluntary OTC (over the counter) | 22.3           | 7.6                      |
| **Total**                        | **28**         | **7.8**                  |

Source: Goldstein et al. (2014).
international offsets in the emissions trading market, which resulted in a large supply of cheap offsets from China and Russia (Goldstein et al., 2014).

In 2013, forest carbon exchanges accounted for almost 50% of all voluntary carbon offsets, which also included investment in renewable energy (Goldstein et al., 2014). Most (85%) of the voluntary forest carbon exchanges invested in REDD projects. The allocation of carbon exchange on the compliance markets was almost equally divided between the Australia Carbon Pricing Mechanism and the California cap-and-trade systems. In both systems, national forest carbon can be used as offset by compliance firms. Another compliance system which allows for forest carbon offsets is the Regional Greenhouse Gas Initiative (RGGI) in the US, which involves cooperation among 10 federal states for the purpose of running a cap-and-trade system, the first system of its kind in the US (Streck et al., 2009).

The average prices on the compliance and voluntary markets differ, USD 9.7/ton CO₂e and USD 4.8/ton CO₂e, respectively (2013 prices). However, the prices also differ within the respective markets, between USD 6.0 and 20.8/ton CO₂e for compliance markets and between USD 1 and 100/ton CO₂e for voluntary projects (Goldstein et al., 2014). The range in prices on compliance markets may reflect the marginal cost of carbon emissions reductions of other measures, which shows buyers’ willingness to pay for carbon offset. Buyers of offset on voluntary markets are mainly driven by corporate social responsibility, and the variation in prices can be large. Approximately two-thirds of the total trade volume is purchased by buyers in Europe and a majority of the sellers are found in South America (55% of the total trade volume), followed by Africa (17%) (Goldstein et al., 2014).

3.1. Heterogeneity, uncertainty and additionality

With respect to the analyses in the literature of uniform versus differentiated carbon payments, the practice has been to use differentiated payments with negotiations on carbon delivery and payments for each project. Independent third-party standards have been applied on a project-by-project basis on the forest carbon markets to ensure certain and additional carbon sinks. The treatment of uncertainty with risk discounting of forest carbon offsets has been applied in a compliance market, the New Zealand ETS. It was established in 2008 and originally only comprised forest, where all forest owners had to participate (Jiang et al., 2009). Two years later, all GHG emitting sectors were included. The New Zealand Unit is comparable to the Kyoto Unit and can thus be bought and sold not only on the New Zealand market, but also on the international market. Risk discounting of forest carbon offsets corresponding to 2:1 applies, i.e., 2 ton forest carbon are required to offset 1 ton CO₂e emissions reduction (Kossyev et al., 2014).

Non-additionality is managed by means of independent standards, which carry out additionality tests. The most common standard for voluntary projects is the Verified Carbon Standard (VCS), which accounted for 46% of the transaction volume in 2013 (Goldstein et al., 2014). VCS was set up in 2005 by non-government organizations including the Climate Group, the International Emissions Trading Association, The World Economic Forum, and the Business Council for Sustainable Development (Asciu and Neef, 2013). The land management segment includes almost all types of projects; afforestation/reforestation (A/R), REDD+, forest management, avoided conversion of grassland and scrubland, and wetland restoration and conservation. Other independent standards are the Gold Standard (GS), Plan Vivo, Climate Action Reserve (CAR), and American Carbon Registry (ARC), which together accounted for 3% of total forest carbon trade in 2013 (Goldstein et al., 2014).

Australia’s Carbon Pricing Mechanism uses Carbon Farming Initiative (CFI) and California’s cap-and-trade system applies the Compliance Offset Protocol (COP), which account for 5% and 3%, respectively, of the total forest carbon transactions (Goldstein et al., 2014). The Australian CFI was established in 2011 and allows for forest sequestration as offset for compliance firms having to pay carbon taxes under Australia’s Carbon Pricing Mechanism. From 2015, there is no limit on the maximum amount of offsets, which include A/R, improved forest management (IFM), and deforestation. The California cap-and-trade system started in 2013 and forest carbon is one of four allowable offset types. The maximum forest carbon offset, which includes A/R and IFM, is set at 8% of the capped emissions, and it must follow California’s COP.

Non-additionality can be mitigated not only by tests, but also by upsaling baseline forest carbon, which reduces the ability of each project owner to exercise information advantages on their true BAU level (van Benthem and Kerr, 2013). The guidelines released in 2012 for developing VCS from project-based standard to jurisdictional and regional levels, VCS JNR (jurisdictional and nested REDD), can be seen as a step towards upsaling (Goldstein et al., 2014). In addition to upsaling of baseline, the extended standard rests on payments by performance. The state of Acre in Brazil, where deforestation has been extensive, hosts a piloting project in this respect where the government aims at reducing 12.5 million ton CO₂e by 2020.

3.2. Permanence

In practice, two main approaches have been used for managing carbon reversal risks; buffer credits and distinction of temporary and permanent credits. The establishment of a buffer, for one or several projects, means that part of the credits issued for a project are set aside for future use in an eventual reversal. The project can claim back the offsets in the pool if there is no carbon reversal. This system is used by VCS, RGGI, California COP, and Australia CIF. The VCS standard deduction is project-specific and ranges between 5% and 80%. In RGGI, the deducted offsets, corresponding to 10% of total offsets, can be waived if the landowner insures against natural carbon reversal. The Australian CIF has a deduction of 5%, and the Californian COP assigns project-based risk to assess the buffer requirement (Global Canopy Programme, 2015a,b).

The system of separate temporary and permanent credits has been applied within the CDM framework and in the New Zealand ETS. During the 2008–2012 commitment period, the CDM allowed for carbon sequestration as offset for temporary and long-term certified emissions reduction (UNFCCC, 2003). New Zealand ETS initiated the Permanent Forest Sink Initiative (PFSI) for ensuring long-term carbon conservation (Belton, 2012). The PFSI was designed to meet the permanence problem by setting minimum sustainable timber harvest during 99 years, liability in case of carbon release, and permission to withdraw after 50 years subject to replacement of carbon stock. The landowner is responsible for carbon releases during the contract period, and insurance systems have been developed to cover unintentional releases.

As discussed in the literature, the allocation of fixed and performance payments can be designed to give incentives to take due care of the forest. This allocation of payments in practice is very much related to type of forest activity. The most common contract types for REDD and IFM projects are performance payments either instantaneously when offsets are issued, or with payment on delivery for projects under development (Goldstein et al., 2014). In contrast, contracts with fixed payments account for the largest share (almost 75%) of A/R projects because of the need to fund expenses for tree plantation.

4. Discussion

Comparing the main findings in the literature as reported in Section 2 and the common design principles of actual forest carbon sink instruments presented in Section 3 reveals several similarities, but also some differences, Table 2.

With respect to uncertainty and the cost of uniform compared with differentiated carbon payments, the practice has been to issue offsets on a project-by-project basis and apply approved standards for monitoring and verification. Payments and carbon delivery are then differentiated
among projects and there are no costs in terms of efficiency losses from a uniform system. On the other hand, the differentiation is associated with transaction costs. The cost of monitoring and verification can vary between 3% and 10% of the carbon prices for forest projects in the US (Mooney et al., 2004; Kile, 2009). Extended scope of transaction costs (including cost of project development and management) are presented by Milner (1999), who shows that the transaction costs of forest carbon projects located in Latin America, Asia, and the Russian Federation range between 6% and 45% of total cost. According to Galik et al. (2012), similar types of transaction costs for improved carbon forest management in the US amount on average to 25% of the total cost.

Risk discounting as a means of reducing the value of a forest carbon offset because of uncertainty has been applied under the New Zealand ETS. Concerning additionality, tests by approved standards have been applied in practice. Contract design for information asymmetry on carbon sequestration costs, as suggested in the literature, seems to be applied in practice. Contract design for information asymmetry on carbon sequestration costs, as suggested in the literature, seems to be applied in practice. Contract design for information asymmetry on carbon sequestration costs, as suggested in the literature, seems to be applied in practice. Contract design for information asymmetry on carbon sequestration costs, as suggested in the literature, seems to be applied in practice. Contract design for information asymmetry on carbon sequestration costs, as suggested in the literature, seems to be applied in practice.

Table 2

| Main findings in the literature | Main design features in practice |
|---------------------------------|---------------------------------|
| Heterogeneity and uncertainty  | Project-based offset and payment with standards for monitoring and verification |
| Treatment of uncertainty as a risk discount increases the carbon sink enhancement cost. | Risk discounting (New Zealand ETS) |
| Addirionality | Additionality tests on a project basis (VCS, California's Compliance Protocol, Australia's Carbon Farmer Initiative) |
| In a two-sector system, with firms in compliance and carbon sinks as offsets, higher stringency in the carbon project’s baseline emissions can foster additionality. Risk discount of offset reduces efficiency. | Upscaling of BAU baseline from individual projects to jurisdictions (VCS JNR) |
| Permanence | Temporary and permanent credits (CDM, New Zealand) |
| Liability on sellers | Buffer credits (VCS, California’s Compliance Offset Protocol, Australia’s Carbon Farmer Initiative) |
| Optimal contract design with low base payment and high performance payment, or the opposite if uncertainty in landowner income is included | Mainly performance payments for REDD+ and IFM, but fixed payments for A/R |
| Temporary credits | Discounting |
| Buffer credits (VCS, California’s Compliance Offset Protocol, Australia’s Carbon Farmer Initiative) | }

Jindal et al. (2008) survey 23 REDD projects in Tanzania, Kenya, and Uganda, which differ with respect to project size, benefit sharing with the community, potential carbon sequestration, and forest activity (forest conservation, IFM, A/R). They show that seemingly promising projects with respect to carbon sequestration and income provision to the local communities are constrained by insecure land tenure. Similar results are reported by Gong et al. (2010), who investigated why so few areas are reforested under a CDM project in China, the first of its kind. They found that, in addition to the uncertainty associated with carbon sequestration as such, farmers face risks with vaguely defined property rights, uncertain government policies, and carbon market prices. Mbatu (2015) shows that the government signing other agreements, such as the Convention on Biological Diversity and Forest Principles, may have helped implementation of REDD in Cameroon.

The problem with insecure property rights can also be an impediment to promising carbon projects in other countries, since it has been found that relatively much of the REDD funding is distributed to countries with weak enforcement institutions (Ebeling and Yasué, 2008; Kronenberg et al., 2015). This finding supports the hypotheses of ‘ecosystem service curse’ and ‘REDD+ paradox’, where the recipient countries face more problems as a result of the payments. The reason is that, even if deforestation decreases and forest management improves, payments may foster corrupt government agencies who may receive most or all benefits that are not shared with the rural populations, and thereby give no or little incentive for conservation initiatives.

The role of price risks in compliance markets as an impediment to landowners entering a contract is highlighted by Manley and Maclaren (2012), who evaluated the effect on forest management of the New Zealand emissions trading system, according to which forest owners receive units for increases in carbon stocks of their plantations. They point out the need for risk reduction measures, such as a hedging policy mitigating carbon price risks. Kerchner and Keeton (2015) investigated the financial viability of family landowner projects in California’s Air Resource Board. They found that carbon stocking and size of the property were the main determinants of return for the forest owners, but also that uncertainty in carbon prices reduces the expected profits.

Whether or not uncertainties in a buyer’s perspective affect actual prices on carbon offsets is analyzed by Conte and Kotchen (2010) in an econometric analysis of a hedonic model of carbon offset prices on the voluntary markets. Based on project data from the Carbon Catalog (CC), they were able to regress prices of offsets on different attributes; type of offset (wind, solar, forestry carbon, etc.), developed, developing, and least developed country, CDM, Gold Standard (GS), or VCS verification. They found that a forestry carbon project reduces the carbon price, that VCS verification also reduces it, and that CDM and GS increases it. This can be explained by the fact that certification of the two latter qualifies for emissions reductions under the Kyoto Protocol. Further analysis by Conte and Kotchen (2010) of the determination of only forestry carbon prices, which amount to 40% of all projects included, showed that forestry projects in developing and least developed countries are subject to substantial price reductions, up to 70%, which may reflect the particular uncertainty associated with weak enforcement institutions in these countries.

5. Conclusions

This study reviewed the economics literature on efficient policy design for promoting forest carbon sequestration and compared the recommendations with policy design in practice. The specific difficulties associated with policy design for carbon sequestration are associated with the site-specific sequestration conditions, uncertainty in sequestration, additionality, and permanence. The literature calculates and compares the costs of uniform and differentiated carbon schemes, but the practice to date has been to use a differentiated system where...
approved standards are used to verify carbon sequestration and additionality for each project. Most studies show relatively small differences in costs, i.e., efficiency losses, of a second-best policy, e.g., a system with uniform payments per area of land or management practice. However, transaction costs, for REDD projects in particular, are considerable and can amount to 25% of the total cost.

In principle, the literature describes two approaches for management of additionality and permanence. One is to accept the magnitude of non-additionality and non-permanence and design policy instruments accounting for the deficiencies, an approach applied in particular for design of forest carbon as offset on compliance markets, such as cap-and-trade systems. One ton of carbon sequestration is then compared with a certain permanent emissions reduction and a discount or price ratio that depends on the magnitude of uncertainty, non-additionality, or non-permanence in carbon sequestration is established. This approach is used in the New Zealand ETS. Another instrument employed within this approach in practice by several standards and carbon pricing systems is the use of buffer credits as insurance against carbon reversals during the contract period. The other approach is to find policy instruments for mitigating the magnitude of non-additionality and non-permanence, on a project-by-project basis, by creating incentive-compatible contracts. Non-additionality for carbon offsets on compliance markets can also be mitigated by setting baselines for projects to a larger scale. Neither suggestion has been implemented in practice, but the recent extension of VCS to jurisdictional level, which includes several projects, represents a move in this direction. The role of improved systems for monitoring and verification for reducing uncertainty and non-additionality in carbon sequestration is not analyzed in the literature on policy design for forest carbon sequestration, but has proven efficient for carbon offsets elsewhere. Interdependence of above-ground and below-ground carbon pools complicates measurement and monitoring of carbon sequestration, but no study addresses policy design for this interdependence.

One impediment to introduction of forest carbon offset projects, in particular REDD + projects, is weak institutions and insecure property rights. Design of contracts creating incentives for self-enforcement can be of more practical relevance in the future, in particular since many of the low-cost options for carbon offsets exist in the developing countries. Voluntary funding for REDD + projects is already large, 88% of total forest carbon trade volume in 2013, and the market for these projects may be saturated. (Goldstein et al., 2014). However, the demand for carbon offsets on compliance markets is expected to increase when formally introduced compliance markets, such as the California carbon offset-projects ab 32 (November 10, 2015, date of access).)

Belton, M., 2012. New Zealand’s permanent forest sink initiative: experience from a functioning carbon forestry mechanism. Silviculture Magazine (at http://www.silviculturemagazine.com/sites/silviculturemagazine.com/files/issues/2012104216/SilviM201222final_med/pdf/silviM201222final_med2ores.pdf (September 29, 2014, date of access)).

Benitez, P., McCallum, I., Obersteiner, M., Yamagata, Y., 2007. Global potential for carbon sequestration: geographical distribution, country risk and policy implications. Ecol. Econ. 60, 572–583.

Bento, A.M., Kanbur, R., Leard, B., 2015. Designing efficient carbon markets for offset with distributional constraints. J. Environ. Econ. Manag. 70, 51–71.

Bosetti, V., Lubowski, R., Golub, A., Markandya, A., 2011. Linking reduced deforestation and a global carbon market: implications for clean energy technology and policy flexibility. Environ. Dev. Econ. 16, 479–505.

Cacho, O.J., Lipper, L., Moss, J., 2013. Transaction costs of carbon offset projects: a comparative study. Ecol. Econ. 88, 232–243.

Capon, T., Harris, M., Reeson, A., 2010. Soil carbon sequestration market-based instruments: a literature review. BEEM, Faculty of Agriculture, Food and Natural Resources. The University of Sidney (At http://www.dpi.nsw.gov.au/__data/assets/pdf_file/0004/350275/literature-review-soil-carbon-sequestration-MBs.pdf (November 12, 2015, date of access)).

Conte, M., Köch, M., 2012. Explaining the price of voluntary carbon offsets. Clim. Chang. Econ. 1, 93–111.

Cooley, D.M., Galik, C.S., Holmes, T.P., Kousky, C., Cooke, R.M., 2012. Managing dependencies in forest offset projects: toward a more complete evaluation of reversal risk. Mitig. Adapt. Strateg. Glob. Chang. 17, 17–24.

Cordero-Salas, P., Roe, B., 2012. The role of cooperation and reciprocity in structuring carbon sequestration contracts in developing countries. Am. J. Agric. Econ. 94, 411–418.

Cordero-Salas, P., Rod, B., Sobek, J., Roe, B., 2013. Addressing additionality in REDD contracts when formal enforcement is absent. Policy Research Working Paper 6502. The World Bank, Washington, DC (At http://elibrary.worldbank.org/pdf>Please do not use 

Daigouaud, A., Miranda, M., Sohngen, B., 2010. Optimal forest management with carbon sequestration credits and exogenous fire risk. Land Econ. 86, 155–172.

Delacote, P., Palmer, Balkegga, R.K., BJ, T., 2014. Unveiling information on the opportunity costs in REDD: who gains the surplus when policy objectives differ? Resour. Energy Econ. 36, 508–527.

Dutschke, M., 2002. Fraction of permanence - squaring the cycle of sink carbon accounting. Mitig. Adapt. Strateg. Glob. Chang. 7, 381–402.

Dutschke, M., Angelsen, A., 2008. How do we ensure permanence and assign liability? In: Angelsen, A. (Ed.), Moving Ahead With REDD: Issues, Options and Implications. Center for International Forestry Research (CIFOR), Bogor, Indonesia (At http://www.cifor.org/publications/pdf_files/Books/BAngelsen0801.pdf (November 16, 2015, date of access)).

Ebeling, J., Yaaf, M., 2008. Generating carbon finance through avoided deforestation and its potential to create climatic, conservation and human development benefits. Philos. Trans. R. Soc. B 363, 1917–1924.

Elsach, J., 2008. Elsach Review: Climate Change: Financing Global Forest. (At https://www.gov.uk/government/system/uploads/attachment_data/file/228831/9780108507632.pdf (November 12, 2015, date of access)).

Engel, S., Palmer, C., Taschini, L., Urech, S., 2013. Conservation payments under uncertainty - a case study. Land Econ. 91, 36–50.

Feng, H., Zhao, J., Kling, C., 2002. The time path and implementation of carbon sequestration. Am. J. Agric. Econ. 84, 134–149.

Fortmann, L., Cordero, P., Sohngen, B., Roe, B., 2014. Incentive contracts for environmental services and their potential in REDD. Policy Research Working Paper 6825. Washington: The World Bank.

Fuss, S., Reuter, W.H., Szolgayová, J., Obersteiner, M., 2013. Optimal mitigation strategies with negative emission technologies and carbon sinks under uncertainty. Clim. Chang. 118, 73–87.

Galik, C., Cooley, D., Baker, J., 2012. Analysis of the production and transaction costs of forest offsets in the USA. Environ. Manag. 112, 128–138.

Global Canopy Programme, 2015a. California U.S. forestry offset projects (AB 32). The REDD Desk (At http://theredddesk.org/markets-standards/california-us-forestry-offset-projects-ab-32 (November 10, 2015, date of access)).

Global Canopy Programme, 2015b. Australian carbon farming initiative. The REDD Desk (At http://theredddesk.org/markets-standards/australian-carbon-farming-initiative-redhdesign-feature-additionality (November 10, 2015, date of access)).

Goldstein, A., Gonzales, G., Peters-Stanley, M., 2014. Turning Over a New Leaf. State of the Forest Carbon Markets in 2014. A Report by Forest Trends’ Ecosystem Marketplace, Washington D.C. (At http://www.forest-trends.org/documents/files/doc_4770.pdf (November 16, 2015, date of access)).

Gong, Y., Bull, G., Baylis, K., 2010. Participation in the world’s first clean development mechanism forest project: the role of property rights, social capital and contractual rules. Clim. Econ. 69, 1292–1302.

Gren, L.-M., Carlsson, M., 2011. Economic value of carbon sequestration in forests under multiple sources of uncertainty. J. For. Econ. 19, 174–189.

Gren, L.-M., Carlsson, M., Munnich, M., Elfsönn, K., 2012. The role of stochastic carbon sink for the EU emission trading system. Energy Econ. 34, 1523–1531.

Gulati, S., Vercaemmen, J., 2005. The optimal length of an agricultural carbon contract. Can. J. Agric. Econ. 53, 359–373.

Anger, N., Satyade, J., 2008. Reducing deforestation and trading emissions: economic implications for the post-Kyoto carbon market. ZEW Centre for European Economic Research Discussion Paper, pp. 08–016.

Asoci, F., Neef, T., 2013. Future options for forest carbon markets in Scotland and the UK. Forestry Commission, UK (At http://forestry.gov.uk/pdf/Forest_Carbon_Markets_in_Scotland_and_UK.pdf?SL=FILE/Forest_Carbon_Markets_in_Scotland_and_UK.pdf (October 2, 2014, date of access)).
Milner, M., Plantinga, A. T., Thomann, E., 2014. The optimal time path for carbon abatement and carbon sequestration under uncertainty: the case of stochastic targeted stock. Resour. Energy Econ. 36, 151–165.

Houghton, R.A., 2005. Aboveground forest biomass and the global carbon balance. Glob. Chang. Biol. 11, 945–958.

Hutyra, M., Halaasenstad, A., 2011. Reduced emissions for deforestation and degradation: a critical review. Consilience: J. Sustain. Dev. 5, 1–24.

IPCC (Intergovernmental Panel on Climate Change), 2014. Climate Change 2014: A Synthesis Report. (At http://www.ipcc.ch/pdf/assessment-report/ar5/syr/AR5_SYR_FINAL_All_Topics.pdf (May 29, 2015, date of access)).

Jiang, N., Sharp, B., Sheng, M., 2009. New Zealand’s emissions trading scheme. N. Z. Econ. Pap. 43, 69–79.

Jindal, R., Swallow, B., Kerr, J., 2008. Forestry-based carbon sequestration projects in Africa: potential benefits and challenges. Nat. Res. Forum 32, 116–130.

Kerr, S., 2013. The economics of international policy agreements to reduce emissions from deforestation and degradation. Rev. Environ. Econ. Policy 7, 47–66.

Kim, I., Langpap, C., 2014. Incentives for carbon sequestration using forest management. Ecol. Econ. 94, 147–171.

Kerr, S., 2013. The economics of international policy agreements to reduce emissions from deforestation and degradation. Rev. Environ. Econ. Policy 7, 47–66.

Kile, J., 2009. Use of Agricultural Offsets to Reduce Green House Gases: Congressional Testimony. Congressional Budget Office, Washington DC, p. 10.

Kim, I., Langpap, C., 2014. Incentives for carbon sequestration using forest management. Ecol. Econ. 94, 147–171.

Kim, M., McCarl, B., 2009. Uncertainty discounting for land based carbon sequestration. J. Agric. Appl. Econ. 41, 1–11.

Kim, M., McCarl, B., Murray, B.C., 2008. Permanent discounting for land-based carbon sequestration. Ecol. Econ. 64, 763–780.

Kosoy, A., Oppenmann, K., Plattanin-Oquah, S., Suphacalasai, S., 2014. State and Trends of Carbon Pricing 2014. World Bank Group, Climate Change, Washington DC. (at http://www-wds.worldbank.org/external/default/WDSContentServer/WDSP/IB/2015/06/01/090224b082b8b0cde/1_0/Rendered/PDF/StateAndTrendsofCarbonPricing2014.pdf (November 16, 2015, date of access)).

Kronenberg, J., Örflögur-Samkowska, E., Cembrowski, P., 2015. REDD + and institutions. Sustainability 7, 10250–10263.

Lubowski, R.N., Plantinga, A.J., Stavins, R.N., 2006. Land-use change and carbon sinks: econometric estimation of the carbon sequestration supply function. J. Environ. Manag. 51, 135–152.

MacKenzie, I.A., Ohndorf, M., Palmer, C., 2012. Enforcement-proof contracts with moral hazard in precaution: ensuring ‘permanence’ in carbon sequestration. Oxf. Econ. Pap. 64, 350–374.

Manley, B., Maclaren, P., 2012. Potential impact of carbon trading on forest management in New Zealand. Forest Policy Econ. 24, 35–40.

Manley, J., van Kooten, G., Moeltner, K., Johnson, D., 2005. Creating carbon offsets in agriculture. Can. J. Agric. Econ. 52, 257–287.