Analysis

Trade-offs between ecosystem services: Water and carbon in a biodiversity hotspot

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A R T I C L E   I N F O

Article history:
Received 8 October 2009
Received in revised form 4 May 2010
Accepted 5 May 2010
Available online 16 June 2010

Keywords:
Trade-offs between ecosystem services
Afforestation
Carbon sequestration
Water resources
Fynbos
Pinus radiata

A B S T R A C T

Carbon sequestration by afforestation can help mitigate global climate change but may have adverse environmental and economic impacts in some regions. For example, economic incentives for carbon sequestration may encourage the expansion of Pinus radiata timber plantations in the Fynbos biome of South Africa, with negative consequences for water supply and biodiversity. I built a dynamic ecological-economic model to investigate whether afforestation of a Fynbos catchment with Pinus radiata is economically viable when the potential benefits of carbon sequestration and timber production are balanced against the losses to water supply. I found that afforestation appears viable to the forestry industry under current water tariffs and current carbon accounting legislation, but would appear unviable if the forestry industry were to pay the true cost of water used by the plantations. I also found that under various plausible future economic scenarios, afforestation can be associated with either large future economic gains or losses, suggesting a need for future analyses based on branches of decision theory that deal with severe uncertainty. I conclude with a general recommendation that climate legislation should be explicit about the conditions under which afforestation for carbon sequestration of native vegetation is a legitimate climate mitigation strategy.

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1. Introduction

Afforestation for carbon sequestration is a potential strategy for the mitigation of global climate change in regions of the world where the net carbon storage of a tree plantation would be greater than that of the ecosystem it replaces. The Kyoto Protocol’s Clean Development Mechanism (CDM), for example, allows countries to offset a portion of their greenhouse gas emissions through carbon sequestration projects such as afforestation. Although afforestation can be used to ease the eventual transition to a low-carbon economy, it is not a long-term solution to the climate change problem and it may have negative impacts on other ecosystem services including water supply, soil fertility (Jackson et al., 2005) and biodiversity values (Caparrós and Jacquemont, 2003; Caparrós et al., 2009). Understanding these trade-offs between ecosystem services is a necessary precondition to sensible decision-making about when and where afforestation for carbon sequestration is an economically and ecologically justified land-use option.

Potential trade-offs between carbon sequestration and other ecosystem services are particularly stark in regions where water is scarce, biodiversity values are high, and/or the biomass of native vegetation is lower than that which the landscape can sustain. One region that meets all of these criteria is the Fynbos biome in the Cape Floristic region of South Africa. The Cape Floristic Region is one of the world’s 25 biodiversity hotspots (Myers et al., 2000; Forest et al., 2007): the Fynbos biome alone covers 46,000 km² and contains about 7000 plant species, 80% of which are endemic. Fynbos plant forms are generally short in stature: the largest species (mostly of the family Proteaceae) grow to less than 5 m (Midgley and Rebelo, 2008). Nevertheless, several alien tree species (mostly of the genera Acacia, Hakea, and Pinus), the largest of which grow to over 30 m, have successfully become established in the Fynbos as both plantation trees and invaders.

There is a strong tradition of ecosystem-service research and practice in South Africa, largely because widespread poverty means that government expenditure on environmental programmes must be justified in economic and social terms. Many previous ecological-economic analyses of alien trees in the Fynbos have focussed on the costs and benefits of clearing invaded water catchments (e.g., van Wilgen et al., 1996; Higgins et al., 1997; Currie et al., 2009). These studies have found that the benefits of clearing (in terms of greater water yields, tourism revenue, wildflower harvests and biodiversity value) generally outweigh the costs (in terms of labour and machinery) and have served to justify major government initiatives aimed at clearing alien invasive trees (Turpie et al., 2008; Koenig, 2009). These government-sponsored environmental restoration programmes also have significant social benefits and have served as...
models for payments-for-ecosystem-services systems worldwide (Turpie et al., 2008).

Whereas many studies, such as those described above, have focussed on the costs and benefits of clearing invasions of alien trees in the Fynbos biome, fewer studies have focussed on the costs and benefits of plantations of alien trees in this region, and fewer still have considered the potential benefits of carbon sequestration in these plantations. An assessment of *Acacia mearnsii* plantations in South Africa concluded that the benefits exceeded the costs providing that appropriate control strategies were implemented to prevent the alien trees from invading surrounding vegetation, but carbon sequestration accounted for only a few percent of the total value of ecosystem services in this case (de Wit et al., 2001). *Pinus* plantations have a much greater potential for carbon sequestration than *Acacia* plantations because of the higher biomass of *Pinus* plantation species. Although the potential carbon sequestration benefits of *Pinus* plantations in the Fynbos have been recognised for some time (Lavery and Mead, 1998), there has, to my knowledge, been no economic analysis of this problem. One ecological study found that these plantations are associated with large negative environmental impacts but did not attempt quantify these in economic terms (García-Quijano et al., 2007).

The aim of this study is to assess the economic viability of *P. radiata* afforestation in the Swartboskloof catchment in the Jonkershoek Valley near the town of Stellenbosch, South Africa with a dynamic ecological-economic model that incorporates timber, carbon and water values. Most of the Jonkershoek valley, including Swartboskloof, is dominated by native mountain Fynbos vegetation, but several catchments have been afforested with *P. radiata* (D. Don) since the 1930s. The Jonkershoek valley and the Swartboskloof catchment in particular have been the subject of dozens of ecological, hydrological, geological and forestry studies since the establishment of the plantation (van Wilgen et al., 1992b). The ecological model in this study consists of a six-pool carbon-flux model (with two biomass pools, two litter pools and two soil pools). The carbon pools are periodically adjusted according to harvesting and fire regimes. The economic model consists of timber, carbon and water valuation sub-models: it estimates the value of these three ecosystem services from the outputs of the ecological model. Initially, I conduct a cost–benefit analysis based on current estimates of the values of timber, carbon and water and current CDM accounting rules. I then explore the sensitivity of the results to future changes in these economic parameters, to changes in the fire regime, to different carbon accounting schemes and to different social discount rates. I discuss the implications of the results for land-use patterns in the Fynbos biome and for climate change policy as it relates to carbon afforestation.

2. Methods

2.1. Study Site

The Jonkershoek Valley lies approximately 50 km due east of Cape Town, South Africa. The Jonkershoek Forestry Research Centre on the valley floor near the entrance to the forest plantation is located at 33°58′05″S, 18°56′05″S at an elevation of 244 m. The Dwarsberg plateau at the head of the valley is at 1220 m. An area of 1700 ha of the valley is considered the potential benefits of carbon sequestration in these plantations (Turpie et al., 2008).

The climate of the valley is Mediterranean, with hot, dry summers and cool, wet winters. Rainfall varies across the catchment from about 1110 mm yr⁻¹ at the Forestry Research Centre on the valley floor to 3625 mm yr⁻¹ on the plateau. Most rainfall (75%) occurs from April to September. Near the valley floor, the mean annual temperature is 16.2 °C. Mean monthly maximum temperature is 27.1 °C in January. Mean monthly minimum temperature is 6.1 °C in July. Temperatures decrease by about 0.5 °C with each 100 m of elevation (van Wilgen and McDonald, 1992).

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Most of the native vegetation of the valley is mountain Fynbos, with smaller areas of native moist forest in riparian corridors and sites protected from fire. The two main types of Fynbos in the valley are ericoid–restioid shrublands and proteoid shrublands. Ericoid–restioid shrublands are dominated by restioids (reed-like plants of the family Restionaceae), sedge, and shrubs (up to 2 m in height) in the families Ericaceae, Bruniaceae and others. Proteoid shrublands are associated with areas of lower rainfall (Cowling and Holmes, 1992) and lower elevation (van Wilgen and McDonald, 1992) than ericoid–restioid shrublands. Proteoid shrublands are characterised by an upper stratum of large shrubs (up to 4–5 m) of the Proteaceae family, which have a canopy cover of 25–70% (van Wilgen and McDonald, 1992).

Fire in Fynbos communities can occur at intervals from 4 to 45 years, but intervals of 10 to 25 years are recommended to maintain species richness (van Wilgen et al., 1992a). The Jonkershoek Nature Reserve is actively managed on a 12–15-year fire cycle. A large unplanned fire in March 2009 burnt most of the nature reserve.

The *P. radiata* plantation at Jonkershoek is managed on a 25–35 year burning cycle. The trees grow more slowly on the warmer, drier south side of the valley (where the Swartboskloof catchment lies) than on the north side. To encourage growth, trees are pruned at 6, 8 and 12 years of age and stands are thinned by about 40% at 8 and 13 years of age (Cocks, 2002). Pruned material is not removed from the site. Thinned material is sold to recover some of the operational costs of thinning. Roughly 90% of harvested timber is sold (mostly as construction timber with a mean product lifetime of 20 years) and the remaining 10% is burnt as boiler fuel (Aubrey Thompson, pers. comm.). Native Fynbos species can coexist with young *P. radiata* stands, but few plants of any kind are observed under the canopies of older stands (>15 years).

Fires generally kill the *P. radiata* trees in the plantation and burn the entire litter layer. MTO Forestry aims to protect the plantation from fire, and fire breaks are maintained around it. Nevertheless, a 1996 fire destroyed the entire south side of the plantation, and the 2009 fire destroyed about half of the south side and about half of the north side. Following fire, trees are salvage harvested and seedlings are replanted.

2.2. Study Species

*P. radiata* is native to a small area of coastal California in the United States (Lavery and Mead, 1998), where the climate is Mediterranean with a dry summer and cool, wet winter. *P. radiata* regenerates profusely from wind-dispersed seeds after fire, but individual trees do not resprout (Richardson and Brown, 1986).

Despite its narrow endemic range, *P. radiata* shows broad climate tolerance and has become the most widely planted exotic pine species and one of the most invasive woody plant species in the world (Richardson and Brown, 1986; Lavery and Mead, 1998; Yan et al., 2006). Extensive *P. radiata* plantations now exist in many parts of the world, including Australia, New Zealand, Chile and South Africa. In South Africa, *P. radiata* was introduced in 1865 and began to be used
widely for forestry in 1910 (Richardson and Brown, 1986). *P. radiata* plantations now cover about 550 km² of South Africa most of which is in the Fynbos biome (Lavery and Mead, 1998). *P. radiata* is a fast-growing tree; at Jonkershoek, trees grow roughly one metre per year until harvesting age.

2.3. Ecological Model

The ecological model in this study is based on a standard carbon-flux sub-model parameterised for Fynbos and *P. radiata* plantations at Jonkershoek. The carbon-flux sub-model is linked to fire and harvesting sub-models, according to which the carbon stocks are periodically adjusted.

2.3.1. Carbon-flux Sub-model

To describe the dynamics of ecosystem carbon in vegetation, litter and soils at Jonkershoek I used a six-pool carbon-flux model based on the Century model (Parton et al., 1987, 1988; Bolker et al., 1998) and other carbon-flux models (Jenkinson, 1990; Roughgarden, 1998). The six carbon (C) pools are metabolic biomass, structural biomass, metabolic litter, structural litter, active soil organic matter and slow soil organic matter. Net primary productivity (NPP) is partitioned between the active and slow soil C pools, with a greater allocation to the slow soil pool for litter with high lignin contents (the slow turnover time makes it unimportant for estimating changes in C stocks over a timescale of centuries (Parton et al., 1987)). I excluded the passive pool from the models of soil organic C decomposition include a third soil organic matter pool. I then adjusted the parameters within these ranges to match empirically observed biomass (Kruger, 1977; van Wilgen, 1981; van Wilgen and le Maitre, 1981; van Wilgen, 1982; van Wilgen and van Hensbergen, 1992; le Maitre et al., 1996), litter (Kruger, 1977; van Wilgen, 1982; van Wilgen and van Hensbergen, 1992) and soil C stocks (Muys et al., 2002; Garcia-Quijano et al., 2007). Empirical observations indicate that aboveground biomass in Fynbos reaches approximately 35 t/ha at 20–25 years post fire (van Wilgen and le Maitre, 1981; van Wilgen, 1982) and then gradually decreases with the senescence of the dominant Protea shrubs after 25 years of fire exclusion (van Wilgen and van Hensbergen, 1992). The model ignores the long-term decrease with the justification that this is a rare event (the probability of a fire interval being longer than 25 years is less than 1% according to the fire sub-model; see next subsection) and that the decline is gradual.

To parameterise the model for *P. radiata* plantations, I first established plausible ranges for the parameters based on studies from Swartboskloof, other catchments at Jonkershoek or, in some cases, from other Fynbos sites (see Table 1 for details of parameterisation and sources). I then adjusted the parameters within these ranges to match empirically observed biomass (van Laar and van Lill, 1978; van Laar, 1982; Snowdon et al., 2000; Garcia-Quijano et al., 2007), litter (Versfeld, 1981) and soil C stocks (Cocks, 2002; Muys et al., 2002; Garcia-Quijano et al., 2007), I used the same values of soil parameters for both the Fynbos and *P. radiata* models.

To convert published estimates of biomass and litter stocks to C stocks, I assumed that C content of all vegetative matter was 50% by mass (Stock and Allsopp, 1992).

2.3.2. Fire Sub-model

I modelled fire occurrence as a stochastic process by drawing fire intervals from a Weibull distribution, which has been used previously in fire models of the Fynbos (Polakow and Dunne, 1999) and other ecosystems (Gill and McCarthy, 1998). This is a phenomenological rather than a mechanistic model of fire. A fire causes the aboveground portions of the two litter C pools and the two biomass C pools to burn and be lost from the system. The aboveground portions are calculated as proportions 1/(1 + R), where R is the root:shoot ratio (see Tables 1 and 2 for parameter values). The belowground portions of the biomass C pools enter the litter pools; a proportion F_0(R)/(1 + R) of the metabolic biomass C pool enters the metabolic litter C pool; a proportion (1 − F_0)R/(1 + R) of the metabolic biomass C pool and a proportion R/(1 + R) of the structural biomass C pool enters the structural litter C pool. Fire does not affect the soil organic matter pools in the model. Soil CO₂ evolution at Jonkershoek has been observed to recover within a few months of fire (van Reenen et al., 1992), suggesting that at least the fast soil organic matter pools are robust to perturbation by fire.

I parameterised the fire model in the Fynbos with fire occurrence data from the Swartboskloof catchment at Jonkershoek (van Wilgen and McDonald, 1992) (Table 1; corresponding mean fire interval is 16 years with standard deviation 4 years).

In view of the uncertainty associated with fire intervals in the *P. radiata* plantation, I repeated the analysis with three different mean fire intervals: 16 years (corresponding to a scenario with a complete lack of fire management relative to the Fynbos), 32 years
(corresponding approximately to recent fire history) and 100 years (corresponding to a largely successful fire exclusion policy). I set the shape parameter of the Weibull distribution for the fi re sub-model in the *P. radiata* plantation to the same value as for the Fynbos.

### 2.3.3. Harvesting Sub-model

MTO Forestry harvests trees from *P. radiata* plantations at Jonkershoek at intervals of roughly 30 years or when trees are killed by fire. After harvesting, *P. radiata* seedlings are replanted. For the *P. radiata* parameterisation of the model, I simulated harvesting events by removing the aboveground portion of the structural biomass C pool from the system and transferring the metabolic biomass C pool and the belowground portion of the structural biomass C pools to the litter C pools.

Although there is potentially a large social and economic advantage associated with the carbon in long-lived wood products (see the Discussion), I did not explicitly model these in my study of the Swartboskloof catchment because when the timber is sold from the plantation the rights to capturing any carbon value in wood products would presumably be sold also. From the perspective of the land manager, the economic value of carbon in wood products would be reflected by rising timber prices, which I explore in the sensitivity analysis (see below).

### 2.3.4. Simulations

I ran the model 2000 times for 100 years for each of the two parameterisations of the model (i.e., the Fynbos parameterisation and the *P. radiata* parameterisation) to produce two sets of 2000 time series of C stocks (the output of each run of the model varies because of stochasticity in the fire sub-model).

#### 2.4. Economic Valuation Model

In the economic valuation model I considered three ecosystem services: carbon sequestration, water supply, and timber production. I did not include tourism because current estimates of the value of tourism at Jonkershoek are negligible compared to other ecosystem services (see the Discussion). I did not consider wildflower harvesting because Cape Nature has a no-take policy in the Jonkershoek Nature Reserve. I did not attempt to quantify existence value (see the Discussion). All currency values are reported in United States dollars ($), using a conversion rate of 8 South African Rand to the dollar where appropriate.

I discounted future costs and benefits by applying exponential discounting with a discount rate of $\delta = 0.04$. I then tested the robustness of the results to different discounting schemes by repeating the analysis with two different exponential discount rates ($\delta = 0.01$ and $\delta = 0.10$) and with gamma discounting (with $\mu = 4$ and $\alpha = 3$), according to which the effective discount rate declines into the future (Weitzman, 1998, 2001). Gamma discounting has been proposed, based on empirical and theoretical grounds, as a superior alternative to tradition exponential discounting, especially for long-term environmental projects (Weitzman, 1998, 2001).

### Table 1

Values used to parameterise the carbon-flux model for the Fynbos.

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Value</th>
<th>Sources</th>
</tr>
</thead>
<tbody>
<tr>
<td>Net primary productivity (NPP) [tC ha$^{-1}$ yr$^{-1}$]</td>
<td>$p = 4$</td>
<td>(Specht, 1969; Kruger, 1977; Stock and Allsopp, 1992)</td>
</tr>
<tr>
<td>Proportion of NPP allocated to metabolic biomass</td>
<td>$q_M = 1/3$</td>
<td>(Roughgarden, 1998)</td>
</tr>
<tr>
<td>Proportion of NPP allocated to structural biomass</td>
<td>$r_M = 2/3$</td>
<td>(Kruger, 1977; van Wilgen and le Maitre, 1981; van Wilgen, 1982; Richardson and van Wilgen, 1986; Versfeld and van Wilgen, 1986; van Wilgen and van Hensbergen, 1992)</td>
</tr>
<tr>
<td>Metabolic litter production rate [yr$^{-1}$]</td>
<td>$\alpha_M = 0.5$</td>
<td>(model results insensitive to this parameter; set to same value as for <em>P. radiata</em>)</td>
</tr>
<tr>
<td>Structural litter production rate [yr$^{-1}$]</td>
<td>$\alpha_S = 0.1$</td>
<td>(van Wilgen and van Hensbergen, 1992)</td>
</tr>
<tr>
<td>Decay rate of metabolic litter [yr$^{-1}$]</td>
<td>$\beta_M = 0.07$</td>
<td>(van Wilgen and van Hensbergen, 1992)</td>
</tr>
<tr>
<td>Decay rate of structural litter [yr$^{-1}$]</td>
<td>$\beta_S = 0.4$</td>
<td>(van Wilgen and van Hensbergen, 1992)</td>
</tr>
<tr>
<td>Decay rate of active soil C pool [yr$^{-1}$]</td>
<td>$\epsilon_M = 1/1.5$</td>
<td>(Parton et al., 1987, 1988)</td>
</tr>
<tr>
<td>Decay rate of slow soil C pool [yr$^{-1}$]</td>
<td>$\epsilon_S = 1/100$</td>
<td>(Parton et al., 1987)</td>
</tr>
<tr>
<td>Fraction of plant residue that is metabolic</td>
<td>$\beta_M = 0.05$</td>
<td>(Parton et al., 1987)</td>
</tr>
<tr>
<td>Lignin-to-nitrogen ratio of metabolic biomass</td>
<td>$L/N = 35$</td>
<td>(Mitchell and Coley, 1987)</td>
</tr>
<tr>
<td>Lignin content of structural biomass</td>
<td>$A = 0.35$</td>
<td>(unknown; set to same value as for <em>P. radiata</em>)</td>
</tr>
<tr>
<td>Fraction of C lost as CO2 when <em>S</em>$_5$ decomposes to <em>S</em>_5</td>
<td>$E_M = 0.85-0.68$ T</td>
<td>(Parton et al., 1987)</td>
</tr>
<tr>
<td>Silt + clay content of soil</td>
<td>T = 0.15</td>
<td>(Mitchell and Coley, 1987; Witkowski and Mitchell, 1987)</td>
</tr>
<tr>
<td>Mean root:shoot ratio</td>
<td>R = 0.33</td>
<td>(Higgins et al., 1987; Smith and Higgins, 1990)</td>
</tr>
</tbody>
</table>

### Table 2

Values used to parameterise the carbon-flux model for the *P. radiata* plantation.

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Value</th>
<th>Sources</th>
</tr>
</thead>
<tbody>
<tr>
<td>Net primary productivity (NPP) [tC ha$^{-1}$ yr$^{-1}$]</td>
<td>$p = 4$</td>
<td>(Arneth et al., 1998; Coops et al., 1998; Waring et al., 1998)</td>
</tr>
<tr>
<td>Proportion of NPP allocated to metabolic biomass</td>
<td>$q_M = 1/3$</td>
<td>(Roughgarden, 1998)</td>
</tr>
<tr>
<td>Proportion of NPP allocated to structural biomass</td>
<td>$r_M = 2/3$</td>
<td>(Versfeld, 1981; van Laar, 1982; Versfeld and Donald, 1991; Garcia-Quijano et al., 2007)</td>
</tr>
<tr>
<td>Metabolic litter production rate [yr$^{-1}$]</td>
<td>$\alpha_M = 0.0042$</td>
<td>(Versfeld, 1981; Versfeld and Donald, 1991; Ganjegunte et al., 2004)</td>
</tr>
<tr>
<td>Structural litter production rate [yr$^{-1}$]</td>
<td>$\alpha_S = 0.0042$</td>
<td>(Versfeld, 1981; Versfeld and Donald, 1991; Ganjegunte et al., 2004)</td>
</tr>
<tr>
<td>Decay rate of metabolic litter [yr$^{-1}$]</td>
<td>$\beta_M = 0.4$</td>
<td>(Parton et al., 1987, 1988)</td>
</tr>
<tr>
<td>Decay rate of structural litter [yr$^{-1}$]</td>
<td>$\beta_S = 0.1$</td>
<td>(Parton et al., 1987, 1988)</td>
</tr>
<tr>
<td>Decay rate of active soil C pool [yr$^{-1}$]</td>
<td>$\epsilon_M = 1/1.5$</td>
<td>(Parton et al., 1987)</td>
</tr>
<tr>
<td>Decay rate of slow soil C pool [yr$^{-1}$]</td>
<td>$\epsilon_S = 1/100$</td>
<td>(Parton et al., 1987)</td>
</tr>
<tr>
<td>Fraction of plant residue that is metabolic</td>
<td>$\beta_M = 0.05$</td>
<td>(Parton et al., 1987)</td>
</tr>
<tr>
<td>Lignin-to-nitrogen ratio of metabolic biomass</td>
<td>$L/N = 14$</td>
<td>(Girlisha et al., 2003)</td>
</tr>
<tr>
<td>Lignin content of structural biomass</td>
<td>$A = 0.35$</td>
<td>(weighted average of values for stem wood, bark and branches; Ganjegunte et al., 2004)</td>
</tr>
<tr>
<td>Fraction of C lost as CO2 when <em>S</em>$_5$ decomposes to <em>S</em>_5</td>
<td>$E_M = 0.85-0.68$ T</td>
<td>(Parton et al., 1987)</td>
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<tr>
<td>Silt + clay content of soil</td>
<td>T = 0.15</td>
<td>(Mitchell and Coley, 1987; Witkowski and Mitchell, 1987)</td>
</tr>
<tr>
<td>Mean root:shoot ratio</td>
<td>R = 0.33</td>
<td>(Heth and Donald, 1978; van Laar and van Lill, 1978; Watson and O’Loughlin, 1990; Snowdon et al., 2000)</td>
</tr>
</tbody>
</table>
2.4.1. Carbon Sequestration

I estimated the value of carbon sequestration in the *P. radiata* plantation from the time series of C stocks generated by the ecological model (baseline C stocks were estimated from the Fynbos parameterisation of the model). I applied accounting rules for temporary certified emissions reductions (CERs) (Dutschke et al., 2004; Olschewski and Benitez, 2005). Temporary CERs and long-term CERs are both types of expiring CERs, which apply to projects in which the carbon sequestered is not permanent. Both lead to the same discounted payment flow, but the long-term approach differs in that it sacrifices flexibility for higher payments earlier in the project (Olschewski and Benitez, 2005; see Appendix A). According to the CDM accounting rules, project developers can choose between either a single crediting period of 30 years, or up to three consecutive crediting periods of 20 years with a renewable baseline at the end of each period. For this study, I found that the former option produced a higher discounted revenue stream than the latter (Appendix B, Fig. B1) and I therefore used a single crediting period of 30 years in the main simulations.

In August 2009 the price of carbon on international compliance markets was roughly US$20/tCO2 (www.pointcarbon.com). Because the price of carbon is extremely volatile and difficult to predict into the future, I tested the sensitivity of the results to a range of plausible carbon prices up to US$200/tCO2 (based on an estimate of the carbon price in 2050 under a scenario in which emissions allowances are cut to 80% of 1990 levels by 2050; Paltsvev et al., 2007).

A limitation to the accounting system described above is that it ignores the value of carbon in the second and subsequent rotations (because it is based on a single crediting period of 30 years), I explored the sensitivity of the results to an alternative accounting scheme, in which the single crediting period is extended to the lifetime of the simulation (100 years). This corresponds to a hypothetical scenario under which carbon credits from CDM projects could be renewed indefinitely (see Discussion).

2.4.2. Water Supply

To calculate the economic value of water from the Swartboskloof catchment, I first estimated streamflow from the Swartboskloof catchment from the following equations:

\[ Q = Q_0 - \Delta Q \]

\[ \Delta Q = 0.02 \times 100 \times \left( \frac{1}{1+R} \right) \times \left( \frac{B_M + B_S}{0.5} \right) + 37 \]

where \( Q = 1140 \text{ mm/yr} \) is the baseline streamflow for Swartboskloof and \( \Delta Q \) is the streamflow reduction associated with aboveground biomass (le Maitre et al., 1996) (the factor of 100 is for unit conversion; the factor \( 1/(1+R) \) converts biomass to aboveground biomass; and the factor 1/0.5 converts mass of C to total biomass).

I then multiplied the streamflow estimates (converted to m\(^3\) ha\(^{-1}\) yr\(^{-1}\)) by the unit value of water, which I estimated to be $0.10 based on water prices paid by the town of Stellenbosch (see Table 3 for details). No explicit projections for future water prices in South Africa were available, but it seems plausible that present water scarcities (Turpie et al., 2008) will be exacerbated by rising demand for water associated with population growth and increasing per-capita consumption and by diminishing supply associated with changes in precipitation patterns (global climate models consistently predict lower rainfall in the Fynbos biome by the end of the 21st century; IPCC, 2007), leading to a higher unit value of water. In view of the uncertainty associated with this parameter, I tested the sensitivity of the results to values of water between $0.04 \text{ m}^3 \text{ m}^{-3}\) and $0.50 \text{ m}^3 \text{ m}^{-3}\). The lower bound corresponds to a different estimate of the current value of water, which I calculated based on the cost and expected water supply rate of the region’s new Berg River Dam (see Table 3 for details). The upper bound corresponds to the alternative cost of desalination (Withtholz et al., 2008).

2.4.3. Timber Production

To calculate the value of timber, \( x \) ($ha\(^{-1}\)), harvested from a stand of age \( t \) (yr), I used the following regression derived from data on timber yields from 11 *P. radiata* compartments at Jonkershoek harvested at ages 10 to 32 years between 2006 and 2009, and on sawlog prices from Jonkershoek timber in 2009 ($R^2 = 0.81$; raw data from Aubrey Thompson, MTO Forestry, pers. comm.; 88% by volume of the timber sold from these stands was sold as sawlogs, 6% as pulp and 6% as veneer):

\[
x = \begin{cases} 
0 & \text{if } t < 10 \\
883.4(t - 9.7961) & \text{if } t \geq 10 
\end{cases}
\]

![Fig. 1. Carbon stocks in biomass, litter and soil for a single realisation of the stochastic carbon-flux model parameterised for the Fynbos (mean fire interval = 16 years).](image)

### Table 3

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Value</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>Cost of Berg River Dam [$]</td>
<td>187,500,000.00</td>
<td>(Bathebule, 2008)</td>
</tr>
<tr>
<td>Water supply from Berg Dam [m(^3)/yr]</td>
<td>81,000,000.00</td>
<td>As above</td>
</tr>
<tr>
<td>NPV of water from dam [$/m(^3)] (exponential discounting with ( \mu = 0.04 ))</td>
<td>0.09</td>
<td>Calculated from above values</td>
</tr>
<tr>
<td>NPV of water from dam [$/m(^3)] (gamma discounting with ( \mu = 0.04, \sigma = 0.03 ))</td>
<td>0.04</td>
<td>Calculated from above values</td>
</tr>
<tr>
<td>Steenbosch raw water supply (internal) [m(^3)/yr]</td>
<td>6,080,157</td>
<td>(Anic Smit, Stellenbosch City Council, pers. comm.; draft of council report, Chapter 5, Functional Service Delivery Reporting)</td>
</tr>
<tr>
<td>Cost of raw water supply (internal) [$/yr]</td>
<td>0.00</td>
<td>As above</td>
</tr>
<tr>
<td>Steenbosch raw water supply (external) [m(^3)/yr]</td>
<td>2,293,841</td>
<td>As above</td>
</tr>
<tr>
<td>Cost of raw water supply (external) [$/yr]</td>
<td>89,246.00</td>
<td>As above</td>
</tr>
<tr>
<td>Steenbosch treated water supply [m(^3)/yr]</td>
<td>3,476,976</td>
<td>As above</td>
</tr>
<tr>
<td>Cost of treated water supply [$/yr]</td>
<td>1,093,079</td>
<td>As above</td>
</tr>
<tr>
<td>Estimated value of water in Stellenbosch [$/m(^3)]</td>
<td>0.10</td>
<td>Calculated from above values</td>
</tr>
</tbody>
</table>
Between 1979 and 2007 the average price of sawlogs in South Africa (in real terms) fluctuated between a minimum of about 50% of the 2007 value in the late 1990s and a maximum just above the 2007 value in the early 1980s (Brian van Wilgen, CSIR, pers. comm.). In view of the long-term volatility of timber prices, I tested the sensitivity of the model's results to timber prices that are 50% higher and 50% lower than those obtained in 2009 at Jonkershoek. I also estimated the increase in timber price that could occur if future carbon accounting schemes included carbon in wood products and if 17% (Garcia-Quijano et al., 2007), 54% or 90% (Aubrey Thompson, pers. comm.) of the harvested timber from Jonkserhoek was eligible for permanent credits under these schemes (see the Discussion), and I tested the sensitivity of the model's results to these timber prices.

I accounted for the management costs of a plantation based on data from Jonksherhoek (Aubrey Thompson, pers. comm.). The cost of establishing a plantation is $500 ha$ \(^{-1}\) (for pitting, planting and fertilising). The cost of tending is $69 ha$ \(^{-1}\) for the first three years after planting. The cost of pruning is $81 ha$ \(^{-1}\) in years 6, 8 and 12. Other maintenance and administration costs total $31 ha$ \(^{-1}\). To calculate the costs associated with harvesting, I multiplied the unit cost of harvesting ($31 m^3$; Aubrey Thompson, pers. comm.) by the harvested volume $V$ (m$^3$ ha$^{-1}$) (this regression is derived from data from the same 11 plots as (1); $R^2 = 0.70$):

$$V = \begin{cases} 0 & \text{if } t < 10 \\ 13.581(t-7.231) & \text{if } t \geq 10 \end{cases}$$

I did not account for thinning of *P. radiata* stands. The inclusion of thinning treatments would have complicated the ecological model but would not have greatly affected the final cost–benefit analysis because revenue from thinned timber largely balances the operational costs of thinning (Aubrey Thompson, pers. comm.).

3. Results

According to the ecological model, the *P. radiata* plantation stores roughly 90 tC/ha more than the Fynbos averaged over the 100-year simulation (Figs. 1 and 2). Most of the carbon accumulation in the plantation is attributable to carbon in biomass, although carbon stocks in litter are important for several years after harvesting and carbon in soil organic matter becomes important towards the end of the simulation period (Fig. 2).

Under the current CDM accounting scheme, for current economic values of water, carbon and timber, and for a mean fire interval of 32 years, afforestation is unviable (expected change in NPV after afforestation = $-1454 ha$ \(^{-1}\); Fig. 3A). Under a high-water-value scenario afforestation is associated with large losses ($-25,676 ha$ \(^{-1}\); Fig. 3B). Under a high-carbon-value scenario afforestation is associated with large losses ($-25,676 ha$ \(^{-1}\); Fig. 3B). Under a high-carbon-value scenario afforestation is associated
with large gains ($18,766 ha$^{-1}$; Fig. 3C). Under a high-water-value high-carbon-value scenario afforestation is again associated with large losses ($-$5711 ha$^-$1; Fig. 3D). The soil C accumulation has little impact on the economic model because most of it occurs after the 30-year CDM project lifetime.

Considering a broader range of economic scenarios shows that the null cline corresponding to marginal viability of afforestation coincides with a roughly linear trade-off between the values of carbon and water (Fig. 4). For instance, for a fire interval of 32 years and the current value of timber, afforestation is viable when the value of a tonne of CO$_2$ is roughly 400 times the value of a cubic metre of water. Lower timber values shift the null cline upwards along the carbon axis; higher timber values shift the null cline downwards along the carbon axis (Fig. 4; Appendix B, Fig. B4). Allowing the timber price to reflect the potential value of carbon in wood products increased it by 11%, 39% or 67% (assuming 17%, 54% or 90% of the harvested timber is used for wood products that are eligible for carbon offsets) with attendant changes in the null cline of marginal viability (Appendix B, Fig. B4). If the mean fire frequency in the plantation is increased to 100 years, most of the marginal economic scenarios become viable, but the unviable scenarios mostly remain unviable. If the mean fire frequency in the plantation is decreased to 16 years (the same as in the Fynbos), many of the previously viable economic scenarios become unviable. A lower discount rate makes afforestation appear more attractive relative to Fynbos conservation at low timber prices and high-carbon prices, but less attractive at high timber prices and low-carbon prices (Appendix C, Fig. C1). A higher discount rate has the opposite effect (Appendix C, Fig. C2). The use of gamma discounting has little effect on the results (Appendix C, Fig. C3).

Under the alternative carbon accounting scheme with a 100-year crediting period, afforestation is viable for current values of water and carbon, but not for the high-water-value scenario (Appendix B, Fig. B2). Exploring a broader range of scenarios under the alternative accounting scheme shows that the slope of the trade-off between carbon and water is skewed such that the threshold carbon price for afforestation to be viable is lower than under the current accounting scheme (especially at high values of water; Appendix B, Fig. B3).

4. Discussion

I have investigated the economic viability of afforestation of a mountain Fynbos catchment with alien $P$. radiata trees. The case study is an example of the trade-offs between water supply, carbon sequestration and other ecosystem services that will emerge as global economic incentives for climate mitigation grow. The main finding was that afforestation of Swartboskloof is currently unviable according to a standard cost–benefit analysis using current CDM accounting rules and current estimates of the economic values of carbon and water. However, there are several caveats and limitations to this analysis. In particular, the future viability of afforestation is uncertain, because of severe uncertainty associated with future carbon accounting rules and the future economic values of carbon and water. I now discuss these issues, and then turn to some recommendations that follow from this analysis.

4.1. Caveats and Limitations

The internalisation of environmental values into economic markets is an essential step towards the sustainable use of environmental resources. However, the internalisation of some environmental values (e.g., carbon) but not others (e.g., biodiversity and scenic values) can lead to suboptimal outcomes (Caparrós et al., 2009). In the present study, I did not internalise some costs and benefits because of a lack of adequate data. Most of these external costs and benefits would probably make the afforestation option less attractive if internalised: they include tourism value, biodiversity value and the potential impacts of tree invasions originating from the plantation.

![Fig. 4. Economic viability of afforestation of the Swartboskloof catchment under current CDM accounting rules for different values of water (horizontal axes), carbon (vertical axes), and timber (panels), and for different fire frequencies (panels) using exponential discounting with discount rate $\delta = 0.04$ (white regions indicate combinations of parameters for which afforestation is viable with greater than 80% probability; black regions indicate combinations of parameters for which afforestation is not viable with greater than 80% probability).](image-url)
Annual income from tourist fees in the Jonkershoek Nature Reserve in 2008 was $10,661, and estimated annual income for 2009 based on data up to and including July was $11,154 (Donna Laloux, Cape Nature, pers. comm.). Based on these values, the estimated tourism value of the nature reserve is $1.11 ha$^{-1}$ yr$^{-1}$ (based on a total area of 9,800 ha), under the assumption that tourism value scales linearly with area. This figure may under- or overestimate the marginal value of tourism at Jonkershoek (the linear scaling assumption may lead to overestimates because the conversion of small areas of Fynbos to pines is unlikely to deter most tourists, whereas the exclusion of revenue from year-round inter-park “Wild Card” passes may lead to underestimates). In any case, the direct revenue from tourism at Jonkershoek is small relative to the value of other ecosystem services included in the model.

Existence value is conceptually related to tourism value, but is more difficult to measure. The existence value of biodiversity in the entire Fynbos biome was estimated in a contingent valuation study of households in the Western Cape and found to be between $3.3 million and $15 million per year (depending on whether respondents were faced with absolute or partial biodiversity losses; Turpie, 2003). Estimating marginal per-hectare existence values from these figures is problematic. Scaling existence value linearly with area gives marginal values of $0.72–3.26$ ha$^{-1}$ yr$^{-1}$, but this is difficult to justify because of embedding biases: it is likely that a contingent valuation study focussed solely on Jonkershoek would lead to much higher existence values (see below). Scaling existence value linearly with species richness (and applying published species–area relationships for the Fynbos; Cowling et al., 1992; Keeley and Fotheringham, 2003) gives lower marginal values of $0.18–0.83$ ha$^{-1}$ yr$^{-1}$ (Appendix D), because the per-hectare rate of species loss is initially small. However, scaling by species richness is also difficult to justify because the public has poor quantitative knowledge of Fynbos biodiversity (fewer than 15% of respondents to a contingent valuation study guessed the number of Fynbos plant species to within an order of magnitude; Turpie et al., 2003).

A better estimate of the existence value of Jonkershoek Nature Reserve could in principle be attained from a site-specific contingent valuation study. In a study of the trade-offs between carbon sequestration and biodiversity values in a forest ecosystem in Spain, for instance, Caparrós et al. (2009) used stated preference methods to elicit park visitors’ valuations of reforestation by native versus alien tree species. However, even if an independent contingent valuation study of Swartboskloof were conducted to estimate existence value explicitly at this scale, there is a further issue of malleability: a proposed afforestation programme would likely trigger widespread media coverage and marketing campaigns, which would in turn raise public awareness and willingness to pay for conservation of Swartboskloof, although the media coverage and willingness to pay might be diluted if other afforestation programmes were proposed contemporaneously in multiple locations across the Fynbos. For now, I note that the only estimates of per-hectare existence value of Fynbos are substantially lower than those of other ecosystem services at Swartboskloof: the inclusion of current estimates of existence value in the model would not affect the conclusions. If arguments about existence value are to convince policy-makers of the benefits of biodiversity conservation in the face of other pressing social and economic priorities, then more research on this topic is required.

Another externality associated with the plantations that I did not internalise because of a lack of data was the cost associated with plantation trees invading adjacent native Fynbos vegetation (Richardson, 1998). It seems likely that, since most P. radiata germination occurs after fire, most of the control costs will be incurred in the years following a fire. The March 2009 fire at Jonkershoek provides an opportunity for collecting data on these costs.

An essential aspect of the complete forest carbon cycle that I did not model explicitly is carbon in wood products. Wood products can reduce atmospheric carbon dioxide via three mechanisms: by physically storing carbon, by substituting for more carbon-intensive products such as steel and concrete, and by substituting directly for fossil fuels (Schlamadinger and Marland, 1996; Sathe and O’Connor, 2008). Carbon offsets associated with wood products can either be temporary (if wood products decay at the end of their lifetime) or permanent (if wood products are burnt to substitute for fossil fuel or are stored in sealed landfills at the end of their lifetime). The physical storage of carbon alone can be substantial: in the United States, for example, it was estimated that in 1990 the national wood-product carbon pool was $7\%$ of the size of the total forest carbon pool and rising (Skog and Nicholson, 1998). My justification for not modelling carbon in wood products explicitly, despite its scientific and economic importance, was that any workable accounting scheme would presumably transfer the rights to carbon credits along with the wood products themselves, and that the owner of a timber plantation would perceive the value of carbon in timber products only through a rise in the timber price. Therefore, I incorporated carbon in wood products implicitly in the sensitivity analysis of the timber price. Under the most realistic scenario in which only $1\%$ of the harvested timber is turned into long-term wood products (García-Quijano et al., 2007), I found that the sale value of timber would increase by about $1\%$, corresponding roughly to a doubling of the net value of timber, in line with observations from Norway and Netherlands, where the introduction of a carbon tax led to a $2–3$ fold increase in the net value of timber (Nabuurs and Sikkema, 2001).

The uncertainties discussed above are mostly associated with economic parameters, but there is also uncertainty in the model’s ecological parameters, particularly the fire interval and soil organic matter input and decay rates. According to the results of the model, shorter fire intervals make forestry less viable (Fig. 4), which makes sense because shorter fire intervals allow less time for carbon and timber stocks to accumulate. The uncertainty in the soil parameters (results not shown) is important from a scientific perspective but not from a current economic perspective, because the crediting period of a CDM project within the temporary CER framework (30 years; Dutschke et al., 2004) is less than the time it takes for substantial soil carbon accumulation to occur (Fig. 2).

Even under the alternative accounting scheme based on a 100-year crediting period (Appendix B), the long-term soil carbon accumulation is essentially discounted into irrelevance. From a scientific perspective, it is interesting that modelled soil carbon stocks continue to accumulate up to 100 years after the plantation is established, but this result should be taken tentatively because the model is largely phenomenological rather than mechanistic: the relative sizes of the carbon pools in the Fynbos and the plantation 70 years after afforestation are consistent with field data (Chisholm unpublished results; Cockx, 2002), but the absolute sizes of the pools are somewhat arbitrary because I excluded the passive organic soil organic matter pool (which may constitute up to 50% of total soil organic matter; Trumbore, 1997).

### 4.2. Streamflow Reduction Fees

The finding that afforestation of the Swartboskloof catchment in Jonkershoek is not currently economically viable raises the immediate question of why 760 ha of neighbouring catchments are nevertheless maintained as P. radiata plantations. The answer is that the forestry company does not pay the full cost of water used by their plantation. Therefore, the plantation imposes a negative externality on other water users, and the interests of the forestry company are not aligned with those of society. Such water-use conflicts are by no means unique to South Africa (Jackson et al., 2005). In Australia, market-oriented water-trading schemes aim to resolve such conflicts, but in South Africa, where the government is trustee and custodian of the nation’s water resources (DWAF, 1998; Blignaut et al., 2007) such solutions may not be appropriate. Current South African water legislation assigns water rights to two sectors: basic human needs...
and aquatic ecosystems (King and Brown, 2006). Other sectors, including the forestry industry, must pay tariffs based on estimates of streamflow reductions to Department of Water Affairs and Forestry (DWAF) (Turpie et al., 2008). These tariffs include charges for alien plant clearing, pollution control, water allocation and other management activities (Turpie et al., 2008).

Although South Africa’s water tariffs constitute a progressive payment-for-ecosystem-services scheme by world standards, they nevertheless understate the true value of water. The DWAF calculates the fee for a plantation by multiplying the estimated streamflow reduction volume (based on tables in Gush et al., 2002) by a nominal water value (which varies nationally across 19 Water Management Areas). The DWAF’s streamflow reduction estimate for Jonkershoek is roughly 90 mm (Rester Lyons, DWAF, pers. comm.), which is about one third of the true mean streamflow reduction (Le Maitre et al., 1996). The water value used to calculate the streamflow reduction fee at Jonkershoek (which is part of the Berg River Water Management Area) in 2009 was $0.00075 m$^{-3}$ (rates subject to annual review; Bertrand van Zyl, DWAF, pers. comm.), which is two orders of magnitude smaller than the value of water estimated in this and previous studies (van Wilgen et al., 1996; Higgins et al., 1997). Multiplying these two values shows that the streamflow reduction fees paid by the forestry company at Jonkershoek represent only a fraction of one percent of the true value of water used.

Conceptually, a straightforward approach to correcting the water externality at Jonkershoek (and elsewhere) would be to base calculations of streamflow reduction fees on something closer to the true value of water. However, given that the forestry industry resisted the introduction of the current modest water-use fees (Jewitt, 2002; Turpie et al., 2008), such a large fee increase is likely to be politically problematic.

4.3. Future Scenarios and Decision-Making under Severe Uncertainty

As mentioned above, the net present value of Fynbos appears to be higher than that of a new P. radiata plantation. However, as one projects further into the future the comparison becomes highly uncertain, mostly because of uncertainty in the future values of carbon and water. One widely anticipated future scenario involves a substantial rise in the carbon price. Carbon prices on the EU compliance market were roughly $20/tCO$_2$ in August 2009, but these are expected to rise to $40–110/tCO$_2$ by 2030 and $70–200/tCO$_2$ by 2050 (Paltsev et al., 2007). An extreme scenario puts the price at $257/tCO$_2$ per 2025 (NWPC, 2009). The model predicts that even a modest increase in the carbon price to $30/tCO$_2$ makes afforestation of Swartboskloof viable, assuming that the other parameters in the model remain constant. Thus, the viability of afforestation is highly sensitive to the value of carbon and it is realistic to expect that in the near future afforestation might appear economically viable based on standard cost–benefit analysis.

Another realistic future scenario involves an increase in the value of water. Water is already a scarce resource across South Africa (Turpie et al., 2008). In the future, demand for water is likely to rise with increasing population size and increasing per-capita consumption, and supply is likely to fall (global climate models consistently predict lower rainfall in the Fynbos biome by the end of the 21st century; IPCC, 2007). The model predicts that, if the price of water were to reach an upper limit corresponding to the alternative costs of desalination, then afforestation would be economically unviable for most plausible combinations of the model’s parameters under the current CDM accounting rules (Fig. 4), and for most combinations of the model’s parameters with carbon prices below $100/tCO$_2$ under alternative carbon accounting scheme based on a 100-year crediting period (Appendix B). Thus, the viability of afforestation is highly sensitive to the value of water. At scales much larger than the single catchment considered here, there is also the possibility for negative feedbacks: large-scale afforestation would deplete water resources, increase water scarcity, raise the value of water and ultimately lower the viability of afforestation itself.

In addition, there are other scenarios not included in the model, under which the potential gains from afforestation carbon credits could be lost. For instance, the fungal pathogen Sphaeropsis sapinea has devastated P. radiata plantations in many areas of South Africa and is apparently excluded from the Western Cape only by currently prevailing climate conditions (van Staden et al., 2004). It is even possible that afforestation will prove ineffective for climate change mitigation in this region because of the lower albedo (and hence higher positive radiative climate forcing) generally associated with plantations relative to treeless landscapes in the temperate zone (Betts, 2000).

Given that afforestation could be associated with either large economic gains or large economic losses under alternative realistic future scenarios, and that the probabilities associated with the various scenarios are very difficult to estimate, how should the decision-making process of whether or not to afforest proceed? One possible approach would be the application of tools from non-probabilistic decision theories, such as info-gap decision theory (Ben-Haim, 2006). Whereas classical decision theory approaches generally optimise the expected value of the decision variable, the info-gap approach instead minimises the probability of failing below a certain threshold (i.e., it maximises robustness to failure). McCarthy and Lindenmayer (2007) applied info-gap decision theory to a water/timber trade-off problem in Australia and found that, after a recent fire in a P. radiata-afforested catchment, restoring a catchment’s native vegetation was more robust to uncertainty than re-establishing a plantation, provided that the costs of revegetation were sufficiently small. A similar application of decision theory to the Fynbos/P. radiata problem is beyond the scope of this paper but is a priority for future research. In the meantime, it may be prudent to rely on the precautionary principle, according to which a project (such as afforestation of native vegetation) should not proceed if there is the threat of serious or irreversible environmental damage and a severe lack of scientific certainty regarding future costs and benefits.

4.4. General Recommendations

Afforestation and native Fynbos conservation are competing land uses not just at Swartboskloof but in mountain catchments across much of the Fynbos biome. It seems probable that the per-hectare values of carbon, water and timber will be positively correlated across different catchments, because the per-hectare volumes of all three are positively correlated with rainfall (most catchments have lower rainfall than Swartboskloof, and most plantations have smaller mean annual increments than Jonkershoek; Aubrey Thompson, pers. comm.) and because the unit values of all three are fairly constant across the Fynbos biome (water is largely fungible across catchments because most are connected to the same water supply infrastructure (David le Maitre, pers. comm.;) and timber and carbon are fungible because they are traded on national and international markets). These observations suggest that, although the model would need to be parameterised separately for different catchments, the qualitative conclusions are fairly general.

At a global scale, there are many other shrubland, grassland and even forest (e.g., Caparrós et al., 2009) ecosystems in which carbon sequestration may be disadvantageous with biodiversity conservation, water supply and other ecosystem services (Jackson et al., 2005). This is in stark contrast to the situation in many tropical (Venter et al., 2009) and temperate (e.g., Creedy and Wurzbacher, 2001) forests, where carbon sequestration is synergistic with other ecosystem services and is even viewed as a means for achieving biodiversity conservation goals. The difficulties of making land-use...
decisions when different ecosystem services are antagonistic and when severe uncertainty in future economic conditions prevails points to a need for the application of more sophisticated techniques in decision theory, as discussed earlier, but more immediately to the need for general guidelines as to the conditions under which afforestation of native vegetation for carbon sequestration is a viable climate mitigation strategy. Ideally, such guidelines would be incorporated into future climate change legislation. Although the accreditation procedures for CDM projects are meticulous and complex (Basu, 2009), they focus almost entirely on carbon assessment, and make scant reference to the potential impacts of CDM projects on other ecosystem services or to biodiversity conservation (Caparrós and Jacquemont, 2003). The situation for accreditation procedures for carbon sequestration projects in smaller voluntary offset markets is similar. The provision of clear guidelines and regulations on these matters would facilitate effective land-use decision-making across the globe in regions that face ecosystem service trade-offs similar to those in the Fynbos.

Acknowledgements

I thank Simon Levin, Geoff Heal and Jim Adelman for reading the manuscript and providing helpful comments. I thank Stephen Pacala, David Wilcove, Lars Hedin, Henry Horn, David le Maitre, Brian van Wilgen, Mark Wilmot, William Bond, Jane Turpie, Robert Jackson, Will Harvey and Emily Sung for helpful discussions. I thank Aubrey Thompson for information about timber harvesting. I thank Donna Laloux for information on tourism revenue. I thank Bertrand van Zyli and Rester Lyons for information about streamflow reduction fees. I thank Timm Hoffman and the members of the Plant Conservation Unit at the University of Cape Town for their support. I thank two anonymous reviewers for comments that greatly improved the manuscript. This project was funded by the David A. Gardner ’69 Magic Grant, the Andrew W. Mellon Foundation through the Emergence and Evolution of Ecosystem Functioning Grant, and the High Meadows Foundation through the Africa Grand Challenge program.

Appendix A. Equivalence of Temporary and Long-term CER Accounting Schemes

Here I prove that Eqs. (5) and (8) in (Olschewski and Benitez, 2005) are equal (using their Eq. (10)), thereby demonstrating that accounting using temporary CERs and long-term CERs gives the same net present value assuming that the crediting period is the same. In the following, \( B_t \) is the net present value of long-term CERs, \( B_t \) is the net present value of short-term CERs, \( C_t \) is the net cumulative CO2 emissions up until year \( k \), \( p_0 \) is the price of a long-term CER unit in year \( k \), \( d \) is the discount rate and \( n \) is the number of accounting periods.

\[
\begin{align*}
\beta^f &= \sum_{i=1}^{n} \frac{C_t - C_{i-1}}{(1 + d)^{i-1}} \cdot p_0 + 1 - p_0 \\
&= \sum_{i=1}^{n} \left( C_t - C_{i-1} \right) \frac{p_0}{(1 + d)^{i-1}} + \frac{1}{(1 + d)^{n-1}} \\
&= \frac{p_0}{1} \sum_{i=1}^{n} \left( C_t - C_{i-1} \right) \frac{1}{(1 + d)^{i-1}} + \frac{1}{1 + d^{n-1}} \\
&= \frac{p_0}{1} \sum_{i=1}^{n} \left( C_t - C_{i-1} \right) \frac{1}{(1 + d)^{i-1}} + \frac{1}{1 + d^{n-1}} \\
&= \frac{p_0}{1} \left( C_t - C_{i-1} \right) \frac{1}{(1 + d)^{i-1}} + \frac{1}{1 + d^{n-1}} \\
&= p_0 \left( \frac{C_t}{1 + d^{i-1}} + \frac{\sum_{i=1}^{n} \left( C_t - C_{i-1} \right)}{1 + d^{n-1}} \right) - C_t \frac{1}{1 + d^{n-1}} \\
&= p_0 \left( \frac{C_t}{1 + d^{i-1}} + \frac{\sum_{i=1}^{n} \left( C_t - C_{i-1} \right)}{1 + d^{n-1}} \right) - C_t \frac{1}{1 + d^{n-1}} \\
&= p_0 \left( \frac{C_t}{1 + d^{i-1}} + \frac{\sum_{i=1}^{n} \left( C_t - C_{i-1} \right)}{1 + d^{n-1}} \right) - C_t \frac{1}{1 + d^{n-1}} \\
&= p_0 \left( \frac{C_t}{1 + d^{i-1}} + \frac{\sum_{i=1}^{n} \left( C_t - C_{i-1} \right)}{1 + d^{n-1}} \right) = \beta^f
\end{align*}
\]

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**Fig. B1.** As for Fig. 4 in the main text but with a carbon accounting scheme that is based on three consecutive crediting periods of 20 years each.
Appendix B. Sensitivity Analysis: Carbon Accounting Schemes and Timber Prices

Here I present the results of sensitivity analyses that explore the impact of different carbon accounting schemes (Fig. B1–B3) and different timber prices (Fig. B4) on the outputs of the ecological-economic model described in the main text.

Appendix C. Sensitivity Analysis: Discount Rates

Here I present the results of sensitivity analyses that explore the impact of different exponential discount rates (Figs. C1 and C2) and the use of gamma discounting (Fig. C3) on the outputs of the ecological-economic model described in the main text.

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Fig. B2. As for Fig. 3 in the main text but with a carbon accounting scheme based on a single crediting period of 100 years.

Fig. B3. As for Fig. 4 in the main text but with a carbon accounting scheme based on a single crediting period of 100 years.
Fig. B4. As for Fig. 4 in the main text but with different timber values (current value + 17%, +54% and +90%).

Fig. C1. As for Fig. 4 in the main text but with discount rate $\delta = 0.01$. 
Fig. C2. As for Fig. 4 in the main text but with discount rate $\delta = 0.10$.

Fig. C3. As for Fig. 4 in the main text but with gamma discounting (with $\mu = 4$ and $\sigma = 3$).
Appendix D. Species–Area Relationships and Existence Value

Here I estimate the marginal per-hectare existence value of Fynbos assuming that existence value scales linearly with species richness. The first step is to parameterise the relationship $S = A^\alpha$, where $A$ is area km$^2$, $S$ is the number of species, and $c$ and $z$ are constants (Preston, 1962).

At small scales (1–1000 m$^2$), a proposed relationship is $S = 287.24^{0.215}$ (Keeley and Fotheringham, 2003). At large scales, proposed relationships are $S = 384.4^{0.253}$ for the south-west Cape Floristic Region and $S = 163.3^{0.274}$ in the south-east (Table 4.4 and Fig. 4.9 in Cowling et al., 1992). The small scale relationship closely matches the south-west relationship at small scales and closely matches the south-east relationship at large scales. The south-west relationship also leads to reasonable estimates for the entire biome (5814 spp. in 46,000 km$^2$). The large-scale species–area relationship is more relevant to the problem at hand, because I am interested in marginal losses to species diversity. So, take:

$$S = 384.4^{0.253}$$

where $c$ is the number of species remaining based on $A_0 = 46,000$ km$^2$. Annual existence value of Fynbos biodiversity is $3.3$ million to $15$ million (Turpie et al., 2003) = $V_0$. The value of remaining biodiversity can be calculated as:

$$V = cV_0$$

$$\frac{dV}{dA} = cV_0 \times 0.0661A^{0.253}$$

For $A = A_0$, this gives:

$$\frac{dV}{dA} = 0.016723V_0A^{0.253} = 5.5 \times 10^{-6}V_0\text{km}^{-2}$$

So the marginal existence value of Fynbos biodiversity, assuming that existence value scales linearly with species richness, is $0.18$ to $0.83$ ha$^{-1}$.

References


