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A sustainable agricultural landscape for Australia: A review of interlacing carbon sequestration, biodiversity and salinity management in agroforestry systems

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Abstract

Transformation of the south-western Australian landscape from deep-rooted woody vegetation systems to shallow-rooted annual cropping systems has resulted in the severe loss of biodiversity and this loss has been exacerbated by rising ground waters that have mobilised stored salts causing extensive dry land salinity. Since the original plant communities were mostly perennial and deep rooted, the model for sustainable agriculture and landscape water management invariably includes deep rooted trees. Commercial forestry is however only economical in higher rainfall (>700 mm yr^{-1}) areas whereas much of the area where biodiversity is threatened has lower rainfall (300–700 mm yr^{-1}). Agroforestry may provide the opportunity to develop new agricultural landscapes that interlace ecosystem services such as carbon mitigation via carbon sequestration and biofuels, biodiversity restoration, watershed management while maintaining food production. Active markets are developing for some of these ecosystem services, however a lack of predictive metrics and the regulatory environment are impeding the adoption of several ecosystem services. Nonetheless, a clear opportunity exists for four major issues – the maintenance of food and fibre production, salinisation, biodiversity decline and climate change mitigation – to be managed at a meaningful scale and a new, sustainable agricultural landscape to be developed.

Keywords: Ecosystem services; Salinity; Soil carbon; REDD; Reforestation; Watershed management

Introduction

The pre-European Australian landscape had a complex mosaic of vegetation that supported a rich biodiversity.[1] Following European settlement in 1788, there was initially an expansion of pastoral based activity followed by extensive clearing of land for cropping and pasture and further intensification of agriculture and urbanisation (Burvill, 1979 and Australian Greenhouse Office, 2001).

This transformation of the Australian landscape from deep-rooted woody vegetation to shallow-rooted annual crops and pastures has come at a huge environmental cost. Two major consequences of this land-cover change are dry land salinity (Peck and Hatton, 2003) and loss of biodiversity (Myers et al., 2000), both directly attributed to land clearing and the subsequent hydrological changes. Such changes have had a particularly profound impact in areas such as the south-west of Western Australia where approximately 20 million hectares have been cleared for farming; a region which is classed as a global biodiversity hotspot (Myers et al., 2000), with up to 500 species at threat of extinction (Keighery et al., 2004).

Since the original plant communities that preceded most of Western Australian farmlands were mostly woodlands and other deep rooted perennials, the model for sustainable agriculture in this region often includes some form of forestry practices (George et al., 1999 and Smettem and Harper, 2009) which are more profitable in higher rainfall (>700 mm yr^{-1} annual rainfall) areas where they take the form of plantations of pulp-wood and timber producing species (Harper et al., 2009a and Harper et al., 2009b). Such forestry is less profitable in areas with rainfall from 300 to 700 mm yr^{-1} and a range of new forestry options are being examined including agroforestry (systems that integrate forestry into existing agricultural systems; Zorzetto and Chudleigh, 1999, Stirzaker et al., 2002, Bartle and Abadi, 2010, Harper et al., 2010a and Harper et al., 2010b). Agroforestry initiatives are often aimed at meeting multiple objectives, such as limiting salinity related land degradation (Harper et al., 2005) and water quality decline (Townsend et al., 2012), enhancing agricultural sustainability and reducing pressure on public native forests (Powell, 2009). An emerging focus is to incorporate trees in farm land for climate change mitigation through carbon sequestration (Harper et al., 2007 and Powell, 2009) or bioenergy production (Bartle and Abadi, 2010, Harper et al., 2010a and Harper et al., 2010b), with this recognised in international treaties such as the Kyoto Protocol (Schlamadinger and Karjalainen, 2000 and Canadell and Raupach, 2008) and national carbon management legislation such as recently enacted in Australia (Mitchell et al., 2012).

Payment for carbon, either as part of a formal emissions trading scheme (a market-based scheme allowing parties to trade permits for emissions or credits for reductions in emissions of certain pollutants) or through voluntary arrangements, will provide landowners contemplating tree planting with a financial incentive to reforest. The Australian Government has for example passed the Carbon Credits (Carbon Farming Initiative) Act 2011 which will establish a trade in carbon credits from reforestation (Mitchell et al., 2012). In addition, such plantings could be made more attractive by introducing payments for other environmental services provided such as biodiversity enhancement or water quality improvement (Townsend et al., 2012) since currently the economic rationale for biodiverse reforestation or maintaining remnant forest patches is not strong. Although reforestation can provide a range of environmental services these are often not brought to account and land-holders are not rewarded for their decisions to change land-use. This limitation becomes particularly important in the lower rainfall areas where commercial returns from timber or wood products are limited and reforestation represents a non-income producing land-use.

A functional relationship between biodiversity and carbon sequestration could have important implications for the management of carbon-sink (a natural or artificial reservoir that accumulates and stores carbon-containing compound for a long period) projects, not only for reforestation and afforestation projects, that are currently supported under international agreements such as the Kyoto Protocol's Clean Development Mechanism (CDM) and Joint Implementation (JI) (Schlamadinger and Karjalainen, 2000), but also for emissions reductions projects that focus on forest management (UNFCCC, 1997 and UNFCCC, 2005). In the former case, the relationship of tree species diversity to carbon sequestration is likely to be of greatest concern for managers interested in optimising the amount of carbon sequestration and maintaining this carbon on a site for periods of up to 100 years. The balance of interest will depend on the payments for the different components (i.e. carbon, water quality improvement, biodiversity) that accrue from the project. In the latter case, understanding the relationship of tree-species diversity to rates of carbon sequestration and carbon storage will be critical to maintaining carbon stocks of protected forests over the long term.

Together with planned offsets (a reduction in emissions of carbon dioxide or greenhouse gases made in order to compensate for or to offset an emission made elsewhere) funded by governments and individuals, the carbon market could potentially develop into a large business (Capoor and Ambrosi, 2007 and Galatowitsch, 2009), with the depth of this depending on both international and national carbon policies. In the past, carbon off setting schemes have largely relied on investment in monoculture plantations (Lamb et al., 2005 and Glenday, 2006) with relatively low conservation value by providing cash-flow from timber and other products. Integrating biodiversity and carbon sequestration initiatives will provide carbon offsetters the opportunity to accrue biodiversity credits and reverse biodiversity decline (Western, 1992, Diaz et al., 2009, Koziell and Swingland, 2002 and Swingland et al., 2002). Currently, the scale of uptake has been modest; Mitchell et al. (2012), for example, identified 65,000 ha of carbon reforestation projects in Australia, with 22% or 14,000 ha having a biodiversity focus, contrasted with almost one million hectares (Parsons and Gavran, 2010) of commercial reforestation established for wood production over the same period.

Although there are presently three compelling reasons to incorporate trees in the Australian landscape (climate change amelioration, secondary salinisation abatement, and biodiversity conservation), currently these issues are all viewed separately. A unified approach is needed to attain sustainable agriculture and biodiversity conservation by retaining and replanting native vegetation. Moreover, recently formulated policy may give rise to financial incentive sin making such an integrated approach financially viable and therefore at an environmentally meaningful scale. This review thus aims to assess agro forestry land use systems that mimic natural systems and in particular explore how markets for carbon, biodiversity and salinity improvement can be used to fund transformational, landscape-scale changes. It draws on a range of key recent scientific reports (Lawson et al., 2008, Polglase et al., 2008 and Eady et al., 2009) that have influenced some of the recently legislated climate change policy in Australia (Garnaut, 2008, Australian Government, 2010b and Garnaut, 2011).

Aligning carbon sequestration, biodiversity and salinity alleviation ecosystem services in Australian agroforestry practice

Agroforestry, which combines forestry with agricultural production (Nair, 1993) is also termed “farm forestry” in Australia (Powell, 2009). Agroforestry encompasses plantations on farms, woodlots, timber belts, alleys, wide-spaced tree plantations and sustainably managed private native forests and is of smaller extent than commercial plantations (DAFF, 2005). Farm forestry sits in the middle of the plantation continuum (Fig. 1), and this review is confined to agroforestry systems where reforestation activities are integrated with farm systems.

Industrial plantation forestry is larger in scale than agroforestry with the majority of hardwood plantations consisting of *Eucalyptus globulus* Labill., *E. grandis* W. Hill ex Maiden, and *E. nitens* H. Deane & Maiden grown for short-rotation pulpwood, while softwood plantations (*Pinus radiata* D. Don, *P. patula* Schiede ex Schltdl. & Cham.) are grown for sawn-wood as well as pulp-wood products (Harper et al., 2009a, Harper et al., 2009b and Parsons and Gavran, 2010). The Australian National Plantation Inventory estimated the area of industrial plantations as 1.97 million ha in 2008 (Parsons and Gavran, 2010). Farm forestry occupied an area of 155,000 ha of which 67,000 ha was under plantations under various managed investment schemes (URS, 2008). However, these plantations may provide minimal direct benefits in terms of biodiversity. A low vertebrate habitat biodiversity benefit of only 15–25% over agricultural land was reported by Hobbs et al., 2003a and Hobbs et al., 2003b and Lindenmayer and Hobbs (2004) based on a study from *E. globulus* plantation in south-west Western Australia which was attributed to structural simplicity of such monoculture plantations. Private native forests in farmlands, were estimated at around 38 million ha (URS, 2008), and are also important contributors to biodiversity conservation.

Several styles of agroforestry are common in Australia. Alley farming to address dry land salinity commonly consists of belts of mallees eucalypts (*Eucalyptus* spp.) in the wheat belt of Western Australia (Wildy et al., 2004 and Robinson et al., 2006), whereas mixed species environmental plantings are common in eastern Australia (Powell, 2009), with these often being established as a consequence of Government land conservation grant schemes. Shelterbelts are planted usually in a linear configuration of perennial woody vegetation and are maintained primarily to conserve soil moisture, reduce wind erosion (Sudmeyer and Flugge, 2005), and provide protection for crops, pastures and livestock (Carberry, 1997 and Brouwer, 1998). Economic analyses for the National Windbreaks Program across southern Australian agricultural regions found that windbreaks either lead to only a small financial gain or were cost neutral (Cleugh, 2003); carbon-offsetting benefits were not considered in this analysis. Alley farming systems are being used in early examples of carbon-offsetting schemes in the northern wheat growing area of Western Australia (Ogawa et al., 2006 and Harper et al., 2011) and various salt-tolerant species used for carbon mitigation of salt affected land (Archibald et al., 2006 and Sochacki et al., 2012).

Currently, Australian agroforestry systems are more developed in higher productivity, high rainfall regions that are close to timber markets or wood processing facilities and less advanced in lower rainfall areas where generally markets and processing facilities are less developed (Powell, 2009). Except for short-rotation pulpwood plantations, the profitability of ‘traditional’ farm forestry is generally marginal in higher rainfall areas and mostly unprofitable in medium to low rainfall areas (Flugge and Abadi, 2006). A carbon-credit market will provide an additional source of income and may thus allow the expansion of agroforestry particularly in medium to low rainfall areas (Harper et al., 2007 and Polglase et al., 2008). Indeed, this bio-region has seen the emergence of a range of early carbon forestry projects (Mitchell et al., 2012). Such projects have occurred in an essentially voluntary basis, although reforestation was a component of the New South Wales Greenhouse Gas Abatement Scheme (MacGill et al., 2006), a state based emissions trading programme. Several authors suggest that carbon sequestration will transform the economic prospects of large parts of remote rural Australia (Garnaut, 2008 and Polglase et al., 2008) a conclusion which concurs with global analyses (Nabuurs et al., 2007). A regional analysis in south-western Australia, examined the economic return from reforestation established for carbon sequestration for a range of price scenarios (Harper et al., 2007).

A synergistic approach to carbon sequestration and biodiversity

The functional significance of biodiversity has been attributed to Elton's (1958) hypothesis linking its role in ecosystem stability and to the less studied role of diversity and ecosystem productivity as put forward by Charles Darwin (Hector and Hooper, 2002). Biodiversity has strong effects on ecosystem properties and the goods and services obtained from them (Hooper et al., 2005 and Naeem et al., 2009). Based on a series of reviews and meta-analyses highlighting the prevalence and importance of biodiversity effects on ecosystem function, it was concluded that a reduction in biodiversity can lead to a reduction in ecosystem process rates and these effects can get magnified over time (Hooper et al., 2005, Balvanera et al., 2005, Stachowicz et al., 2007, Cardinale et al., 2007 and Hillebrand and Matthiessen, 2009). Opportunities for more efficient resource use are possible with the presence of higher biodiversity – a measure of the probability of the presence of species with wider array of traits (Chapin et al., 1997) (Fig. 2).

There is an increasing interest among ecologists in the potential functional relationship between biodiversity and carbon sequestration (Chapin et al., 2000, Tilman et al., 2001 and Srivastava and Vellend, 2005). Strassburg et al. (2010) found a high congruence between species richness (biodiversity indices mapped on a two-dimensional scale; Williams and Gaston, 1998) and biomass carbon (mapped using global biomass carbon map; Ruesch and Gibbs, 2008) on a global scale. Experiments in grassland and aquatic systems have found monotonical increases in productivity with increasing species diversity (Tilman et al., 1996, Hector et al., 1999 and Chase and Leibold, 2002). This relationship is primarily explained by the 'niche-complementarity hypothesis,' wherein a larger array of species in a system utilises a broader spectrum of resources resulting in the system becoming more productive (Lehman and Tilman, 2000). Such functional complementarity in trees results in an uneven contribution of species to carbon sequestration with higher rates of carbon sequestration by fast growing species, greater storage of carbon in large, long-lived species and in species with dense wood (Pinard and Cropper, 2000, Caspersen and Pacala, 2001 and Balvanera et al., 2005).

Floristic assemblage studies point to a declining relationship between long-term sequestered carbon with a reduction in plant diversity pools (Fang et al., 1998 and Pacala et al., 2001) interpreted as the ability of highly diverse species mixtures with differing functional traits allowing species to efficiently exploit resources (Hector et al., 1999 and Spehn et al., 2005). Recent plant productivity studies in natural ecosystems are revealing that the diversity of functional groups/traits related to the acquisition, processing and use of key resources had more pronounced effects than species numbers (Diaz and Cabido, 2001, Lavorel and Garnier, 2002 and Hooper et al., 2005). While experiments have generally proved that niche differentiation leads to complementary resource use leading to higher resource consumption with increasing diversity (Scherer-Lorenzen, 2005), a caveat needs to be added that the shape of the productivity–diversity relationship is scale dependent on: data viewed at a more homogenous local scale is ‘hump-shaped’, where diversity peaks at intermediate productivity; when the same data are viewed at a more heterogeneous regional scale, diversity increases linearly with productivity (Chase and Leibold, 2002). Care must be also taken in quantifying biological diversity since there is a growing realisation that biodiversity cannot be expressed along a unifying dimension unless its facets can be quantified (Purvis and Hector, 2000).

Does the carryover effect of increased productivity (carbon biosequestration) associated with higher diversity translate to increased long-term soil carbon sequestration? Currently only anecdotal evidence exists to support this proposition (Tibbett, 2008, Tibbett, 2010 and George et al., 2010). George et al. (2010) studying the variation in the stable isotopic soil carbon (δ^{13}C) signature following de novo establishment of diverse vegetation in post-mined restored landscapes of South-Western Australia revealed that with increasing forest age (productivity) there was a re-establishment of the labile to refractory carbon continuum with soil depth which was comparable to undisturbed native forest (Fig. 3). For reforestation with monocultures, evidence suggests that soil carbon can either increase or decrease depending on the species selected (Polglase et al., 2000 and Guo and Gifford, 2002), whereas in the lower rainfall zone of Western Australia, there were no significant changes in soil carbon following reforestation with several *Eucalypt* monocultures after

26 years (Harper et al., in press). These studies provide an interesting contrast in soil carbon between floristically diverse and monoculture forests, but a clear relationship is yet to be established.

Most of Australia is projected to have low biosequestration rates (Roxburgh et al., 2004) with biosequestration generally related to site water balance (Harper et al., 2007). Many areas with potential for higher rates of sequestration are already under agriculture or urbanised (Polglase et al., 2008). The profitability of a carbon project, however, will depend on more than growth rates and in many cases lower growth rates in drier environments will be counterbalanced by lower land costs (Harper et al., 2007).

Currently, much of the carbon sequestration that appears in Australia's national accounts is from commercial monoculture plantations, that have been established for timber or pulp production with relatively little contribution from biodiversity based schemes (Mitchell et al., 2012). This bias towards promoting establishment of monoculture plantations has been inherited from concerns about ensuring wood supply on a sustainable basis (Plantation 2020 Vision Implementation Committee, 1997) and the relative profitability of wood producing systems compared to biodiversity plantings. Mitigating climate change through reforestation opens the possibility of a new form of forestry – where the main product is carbon and issues with timber markets and transport do not inhibit the rates of establishment (Harper et al., 2007 and Bottcher and Lindner, 2010).

From a long-term sustainable management point of view, future reforestation schemes establishment with diverse vegetation would benefit from carbon accrual at an increased magnitude, turnover and long-term carbon stock in vegetation and soil. This biodiversity could potentially be brought to account and sold as an environmental service (Swingland et al., 2002), thus providing an additional source of income for the reforestation activity. Future reforestation based CO2 mitigation initiatives will thus greatly benefit from incorporation of a biodiversity component into the design, implementation, and regulatory framework. This will also bring the two quite different entities of biodiversity conservation and carbon sequestration together. Whereas a price for carbon is becoming evident through both formal (see for example Australian Government, 2010b) and voluntary schemes, a similar construct is needed for biodiversity. Such an additional mechanism will provide supplementary financial returns for biodiversity planting thus providing additional competitive returns for reforestation programmes.

The development of a carbon market using reforestation has required the development of metrics that allow the prediction of likely rates of sequestration, methods for measuring and reporting carbon increments (Srivastava, 2010) and also underpinning legislation that provides a legal title to the carbon that represents a real financial asset that can be readily traded (Galatowitsch, 2009 and Mitchell et al., 2012). A major difference between carbon and biodiversity will be the depth of the market; carbon markets already exist with carbon considered economically feasible across a range of sources in the economy, and indeed worldwide. Thus carbon emitted from a power station, or from transport, can be mitigated via reforestation. No such market exists for biodiversity, and whereas carbon is a clearly defined unit that can be measured and traded, no similar entity exists for biodiversity.

Congruence of carbon sequestration, biodiversity and salinity mitigation in Australian agroforestry practice: opportunities and case studies

In the Australian agroforestry context, land protection plantings (Fig. 1) have a greater scope of integrating biodiverse planting to accrue higher carbon sequestration benefits. Spatial-scale predictions of above and below-ground carbon sequestration using the 3-PG growth model (Landsberg and Waring, 1997) for a suite of planting options (Fig. 2) have highlighted the opportunities for large-scale biodiverse agroforestry plantings for carbon sequestration across a wide area of Australia (Polglase et al., 2008). For various modelled scenarios, Polglase et al. (2008) concluded that biodiverse carbon plantings could be profitable across 9 million ha in south-eastern Australia, southern and south-eastern South Australia and parts of Tasmania and south-west Western Australia. These plantings were estimated to achieve an annual increment of 143 Tg CO_2-e yr^{-1}at an average rate of 16 Mg CO_2-e ha^{-1} yr^{-1}, although this estimate is considerably more than the actual rates of sequestration measured for two sites in the 400 mm rainfall zone of Western Australia (Archibald et al., 2006 and Harper et al., in press) of 1–2 Mg CO_2-e ha^{-1} yr^{-1}. Nonetheless, the total annual increment would be equivalent to about a quarter of Australia's overall 2005 greenhouse gas emissions.

Modelled outcomes for equilibrium carbon stock in the 600 mm mean annual precipitation region of Western Australia were estimated to be 183, 259 and 457 Mg CO_2-e ha^{-1} for biodiverse forest, *E. globulus* and *P. pinaster* respectively (Harper et al., 2007). The Polglase et al. (2008) study concluded that biodiverse carbon farming is promising due to the relatively low cost of production (no harvesting or transport costs) compared with a possibly high product price, and annual payment of carbon sequestered and may have more long-term sustainability.

Townsend et al. (2012) describes the bundling of multiple products from reforestation in a ‘payments for environmental services’ (PES) framework, with the sum of returns from this being more profitable than the existing land-use (agriculture). Although Townsend et al. (2012) describe payments for timber, carbon mitigation and water quality improvement the same principle of bundling applies to biodiversity, if suitable metrics for this can be developed.

There may be additional scope for accruing higher conjoint benefits by establishing biodiverse planting that re-establish connectivity between existing biodiverse remnant forests or are established in proximity of biodiverse forests. Hence, biodiverse planting in south-west Western Australia, south-east South Australia and south-east Queensland are of higher combined biodiversity and carbon sequestration value. The present condition of this native vegetation and the area in proximity will be a key determinant in the type of biodiverse community that can be established as detailed by Hobbs et al. (2009). The extent of biotic and environmental site modification will determine if communities resembling ‘historical’ native forest can be re-established. Excessive physico-chemical alteration due to secondary salinisation can render the soil and the local hydrology fundamentally altered thereby establishing an alternative ecosystem may be the only possible alternative (Hobbs et al., 2009).

Along with the current wide range of market opportunities for farm forestry there is a potential for some emerging markets to provide very large-scale opportunities in the future including carbon sequestration (Fig. 4; URS, 2008). We propose that the potential scale of opportunity (relative size of the circle) estimated for carbon sequestration will shift when biodiversity considerations are included in carbon sequestration schemes (Fig. 4).

Based on the potential for incorporating sustainable biodiverse planting for increased carbon sequestration outcomes, three options in the plantation continuum (Fig. 1) integrated into farm forestry practice are explored in further detail.

Integrating biodiverse perennials to alleviate salinity affected farming systems

Landscape scale changes in water balance occur as a result of replacing predominantly perennial, deep-rooted native vegetation with shallow-rooted annual crops and pastures (Peck and Hatton, 2003 and Bari and Smettem, 2006). This results in of the development of ground water systems within the regolith which in turn leads to the mobilisation of regolith salt stores, and discharge of these salty waters on the land surface. This is the secondary dry land salinity which has mostly occurred across southern Australia (National Land and Water Resources Audit, 2001 and Peck and Hatton, 2003). To offset this changed hydrological regime (Hatton and Nulsen, 1999), an introduction of deep-rooted perennial species, and/or a significant enhancement of groundwater discharge and pumping has been advocated (Schofield, 1990 and Smettem and Harper, 2009) although George et al. (1999) suggest that the amount of reforestation needed to control water tables and markedly impact salinity may be as high as 80% in many catchments. Nonetheless, the introduction of deep-rooted native biodiverse perennial species into catchments dominated by annual crops and pastures can be part of a more sustainable solution to rectify this hydrological imbalance and provides an opportunity to remediate other environmental problems such as erosion and the loss of biodiversity and also provides carbon offsetting opportunities. One example is the establishment of vegetation on areas that are already salinised (Marcar et al., 1995) and effectively abandoned to productive agriculture.

Block planting, hills lope belts, woodlot rotation/phase farming, alley planting, and high water table planting are a set of agroforestry-based options suggested to tackle salinity issues (Stirzaker et al., 2002). Except for woodlot rotation/phase planting, whereby perennials are used to deplete soil moisture stores following which the trees are harvested and a new location can be replanted (Harper et al., 2010a and Harper et al., 2010b) the rest of the tree-based remedial measures have the scope for direct biodiverse planting for carbon sequestration. All techniques, however, could indirectly contribute to the protection of biodiversity in existing remnants in lower landscape positions, by controlling hydrology at a landscape scale.

The Goulburn Broken Catchment (GBC; 2.3 million hectares, covering 17% of Australia's state of Victoria), provides a typical case study for the need to alleviate dryland salinity issue similar to that faced by many Australian catchments (Loockwood et al., 2002). Extensive research has highlighted the opportunity for addressing concomitant issues of salinity mitigation and biodiversity conservation through conserving and re-planting of key native vegetation (Curtis et al., 2001). Wide adoption of plant-based salinity management systems is currently hampered due to the un profitability of currently available financial instruments (DNRE, 2000). With a limited opportunity for commercial monoculture plantations in most of the high saline dry land areas of the GBC catchment, the designated zones can be brought under native forest conservation/planting with at least 30% of the pre-European cover in order to achieve a sustainable biodiversity goal (Loockwood et al., 2002). Although most of the landholders intended to undertake some revegetation, the total revegetation would not be sufficient to meet biodiversity conservation targets for several high priority vegetation types and likely to achieve only a marginal improvement to the amount of tree cover in areas of high priority for salinity mitigation (Loockwood et al., 2002). Additional economic incentives that assign higher value to ecosystem services such as biodiversity and sequestered carbon would be essential to bridge this gap in planting.

Reforestation of catchments to achieve carbon sequestration and improve water quality

The impact of reforestation on water supplies is often considered in terms of impacts on water yield. In specific circumstances, such as the restoration of salinity, reforestation will improve water quality to the extent that previously unusable water can be utilised. Townsend et al. (2012) examined the multiple values from reforestation in the Warren-Tone, a large (408,000 ha) agricultural catchment with between 500 and 700 mm yr^{-1} annual rainfall in southern Western Australia. Around a quarter of this catchment (105,000 ha) had been previously cleared, with a resultant deterioration in water quality (Smith et al., 2006); 25,000 ha subsequently reforested with commercial pulpwood (*E. globulus*) plantations. Water yield and quality outcomes of various reforestation scenarios were estimated using LUCICAT, a calibrated hydrological model (Bari and Smettem, 2006), and the costs and benefits of different land-uses examined at a whole catchment level and returns from water, wood and carbon estimated.

Various land-use change scenarios were examined, with these suggesting that 70% reforestation was required to restore stream salinity to a potable standard (500 mg L^{-1} total dissolved salts). Although it was estimated that this would reduce annual water yields from 260 GL yr^{-1} to 237 GL yr^{-1} water would be restored to a potable condition and thus have value. The sale of potable water following reforestation could thus provide a new source of income for landholders. The economic modelling suggested that the sale of 100 GL yr^{-1} of water at AUD$150,000 GL^{-1} would result in a net water value of $285 ha^{-1} yr^{-1} when applied across the areas reforested (Table 1). Reforestation was unprofitable when only wood revenues from reforestation were considered, with a discount rate of 9.5% but was profitable at lower discount rates and with carbon prices of at least $26 Mg CO_2-e. Additional income would come from the sale of timber and carbon, and the bundled return from timber, carbon and water was more profitable than the existing agricultural system (Table 1).

Enhancing carbon and biodiversity value of on-farm remnant native forests

Conserving the carbon stocks within natural forests should be one of the foremost priorities from a climate change mitigation point of view (Canadell and Raupach, 2008) and as seen earlier there are around 38 million hectares of remnant forests in Australian farmlands (URS, 2008). Depending on the biome and forest condition, the total biocarbon in native forest can be substantial (Keith et al., 2009); recouping this would take decades to achieve. Indeed, a substantial reduction in the Australia's rate of land clearing, from rates of around 1 million ha per year in the 1980s, has been counted against Australia's Kyoto Protocol carbon account under Article 3.7 of that Treaty (Hamilton and Vellen, 1999). This has allowed Australia's 2008–2012 emission target to be met by reducing annual emissions from deforestation from 132 to 50 Tg CO_2-e yr^{-1}(Australian Government, 2010a). Other options for enhancing carbon management from natural forests include fertilisation and changing fire and harvesting regimes (Kirschbaum, 2000).

Over much of Australian agricultural lands, as is the case in most cropped areas the world over, the initial condition to which the native plant communities were so well adapted no longer prevail hence an alternative community structure which suits the altered water and nutrient regimes needs to used (Pate and Bell, 1999). In the south-western Australian context, Pate and Bell (1999) applied a four-stage species selection process based on growth phenology root morphology life form and fire response resulting in a minimum set of eight plant-types required for a functional mimic. Wherever possible, at a landscape scale, a patch dynamics model designed as a structural and functional mimic of natural ecosystems can be used to improve connectivity of these remnant native forest (Lefroy, 2009) thereby increasing ecosystem functionality. Such mimics of natural ecosystems may better utilise resources resulting in higher sustainable carbon sequestration on the long-term (especially when considering stable soil carbon also).

Conservation of such on-farm remnant native forests patches and increased functionality through connectivity of these remnant patches could be supported by a scheme similar to Reducing Emissions from Deforestation and Degradation of tropical rainforests in developing countries (REDD; Gullison et al., 2007). Even though the REDD scheme is devised for tropical rainforest conservation, a local version of REDD-styled framework could be developed in the Australian context aimed to reduce and eliminate emissions from deforestation and degradation of remnant native forests and securing their permanent protection, legally and ecologically. In addition, studies have shown that benefits of REDD based schemes will sharply increase by explicitly incorporating biodiversity values into carbon payments (Venter et al., 2009) which can be a justification for attracting higher incentives for conserving more diverse remnant vegetation with greater conservation value.

Conclusions

Australia faces mounting pressures, like other countries, for structural changes to deal with both climate change and its mitigation. Reforestation is a relatively cheap C-abatement strategy when compared to many of the available and conceived future options and can be implemented with immediate effect, while other mitigation technologies are being developed. There is theoretical and experimental evidence linking aboveground ecosystem productivity and biodiversity but only a few observational studies illustrating this trend (even though only anecdotal evidence linking soil carbon sequestration and aboveground productivity). The convergence of carbon sequestration with biodiversity conservation and salinity management presents a unique opportunity to tackle three environmental problems together. Moreover, the potential scale of carbon investment provides the opportunity to produce landscape scale changes that would not otherwise be achievable due to lack of capital. Similar approaches could be developed around this concept in other regions. Using an agroforestry approach will allow the integration of trees into farmland and not displace food production, but rather stabilise agricultural systems and alleviate dryland salinity.

Government policies that support biodiverse reforestation as part of carbon reforestation programmes are essential and these could include the development of predicting biodiversity benefit, overcoming technical difficulties for establishing trees on farmland, systems of payment and providing a title to the biodiversity credits in a manner similar to land, water and carbon titles. A local version of REDD framework focusing primarily on existing natural ecosystems and well-established mimics could be developed in the Australian context aiming to reduce and eliminate emissions from deforestation and degradation of remnant native forests.

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Fig. 1. Agroforestry/farm forestry, land protection planting and industrial forestry based on the emphasis on timber production. Shaded portion encompasses agroforestry practices with potential for conjoint carbon sequestration and biodiversity benefits and salinity amelioration outcomes.

Modified from Prosser (1995) and Mitchell et al. (2012).

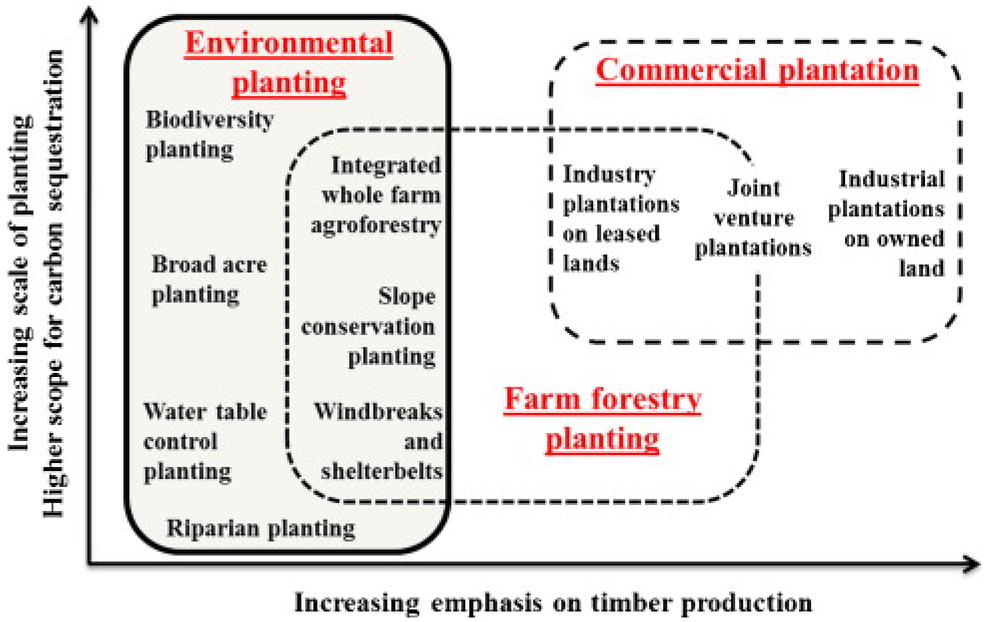

Fig. 2. Annual carbon sequestration (Tg CO_2-e) over a 40 year time period (2010–2050) based on estimates from Garnaut (2008) and Eady et al. (2009). *Definitional and/or estimation of carbon sequestration rates are reasons for significant difference between Garnaut (2008) and Eady et al. (2009) estimates.

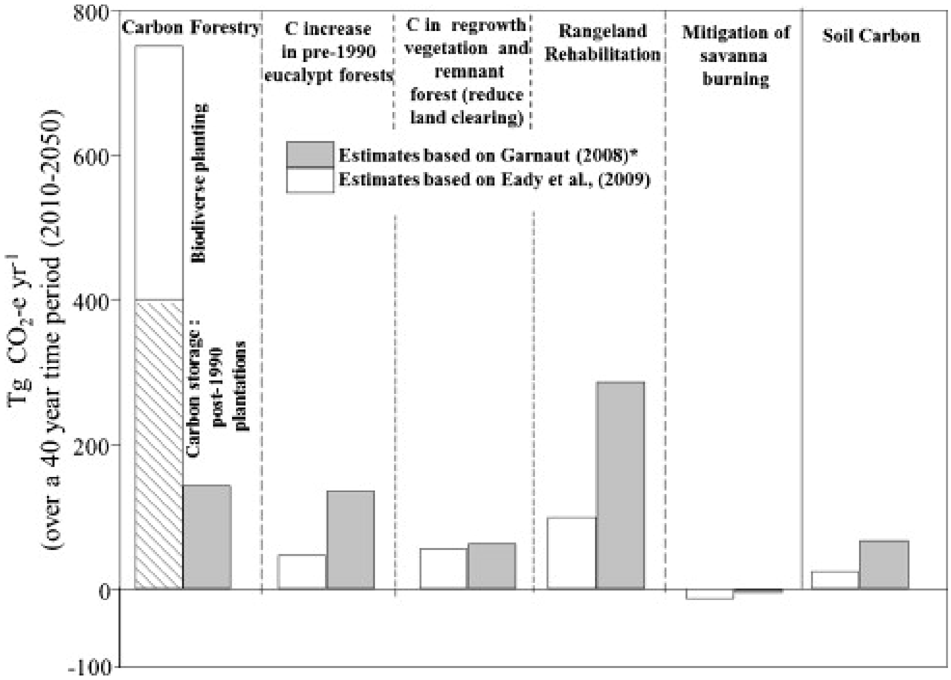

Fig. 3. Re-establishment of complex soil organic matter relationship with increasing restoration age following disturbance in an establishing biodiverse forest. Linear correlations of the δ^{13}carbon and log-transformed carbon concentrations in the whole soil for increasing age since restoration (2, 5, 8, 11, and 18 years) compared to benchmarked native forest site (grey panel) at the four sampling depths.

Modified from George et al. (2010).

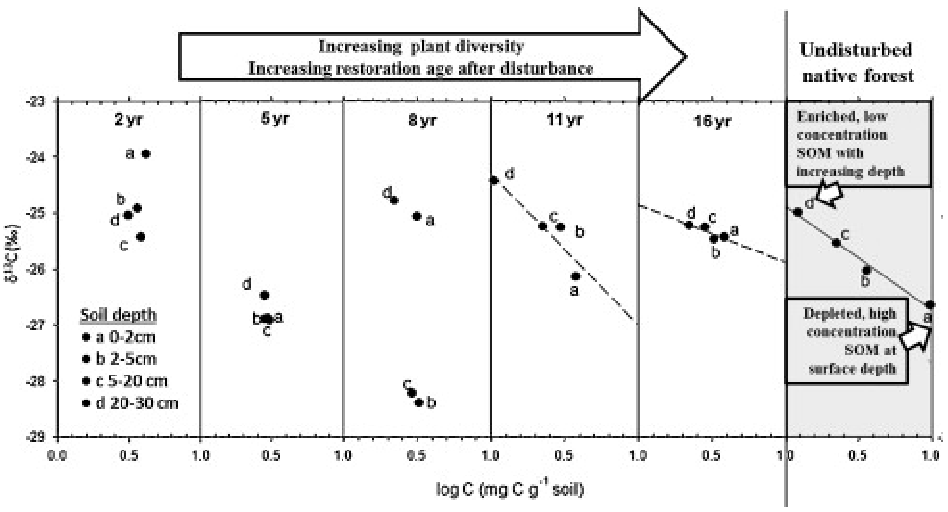

Fig. 4. Summary of farm forestry market opportunities by major wood-based forest product.

Modified from URS (2008).

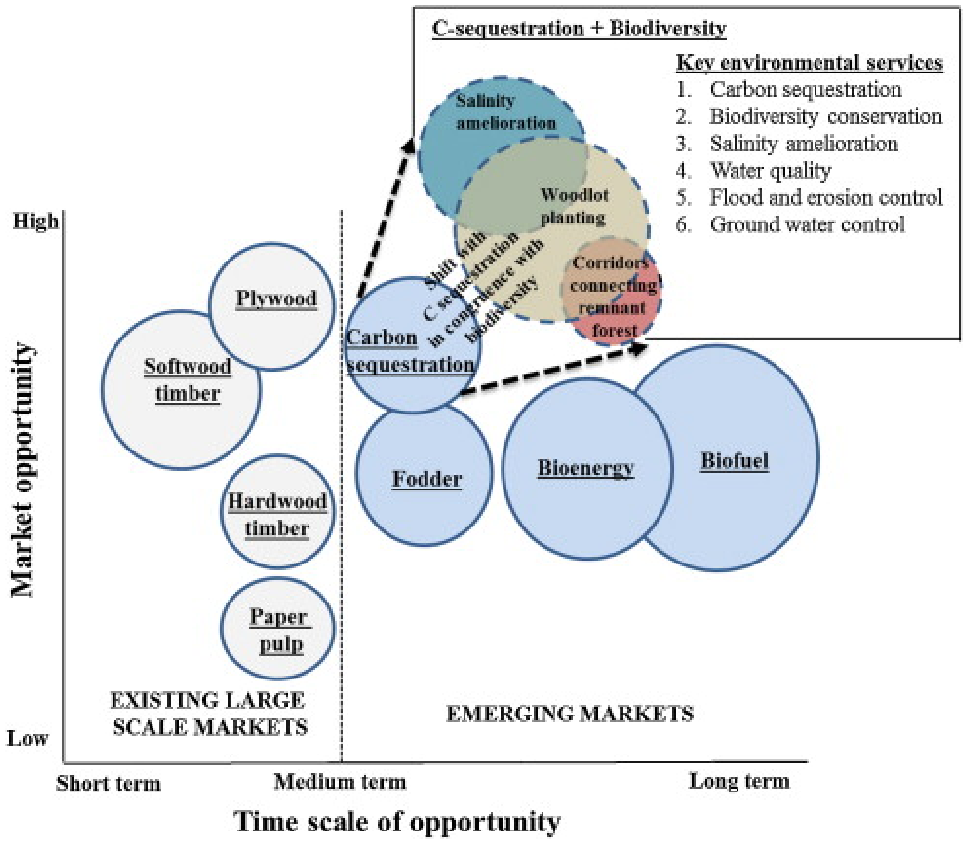

Table 1. Agricultural returns and externalities and forestry returns from both timber andcarbon (AUD\$ $ha^{-1}yr^{-1}$) in the Warren Tone catchment. Net Present Values were calculated with a discount rate of 7%.

	Annual rainfall (mm yr^{-1})	
	500	**700**
	Returns (\$ ha yr^{-1})	
Agricultural returns	150	190
Externality (salinity) costs of agriculture	−50	−30
Net value of agriculture	100	160
Timber return	−200	−113
Carbon return	354	357
Water return	285	285
Timber + carbon + water	439	529
Net benefit of forestry over agriculture	339	369