Chapter 19

Carbon and Water Tradeoffs in Conversions to Forests and Shrublands

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19.1 Introduction

Carbon sequestration programs on land and in the oceans are gaining attention globally as a means to offset increasing fossil fuel emissions and atmospheric carbon dioxide concentrations (e.g., DeLucia et al. 1999; Caldeira and Duffy 2000; Schimel et al. 2000; Pacala et al. 2001; Hoffert et al. 2002; Jackson et al. 2002; Hungate et al. 2003; McNeil et al. 2003). Many industrialized nations now have national plans to foster land-based sequestration; Australia's Plantations 2020 program is one example of such a national effort (Polglase et al. 2000). Ocean based sequestration, particularly deep ocean pumping of CO_2 and iron fertilization, is also receiving considerable attention, although it remains even more controversial than land-based programs (e.g., Chisholm et al. 2001; Lawrence 2002; Buesseler and Boyd 2003; Tsuda et al. 2003). Despite uncertainties about the size and sustainability of sinks and markets, programs for emissions trading and carbon credits are underway, including the Chicago Climate Exchange and the European Union Greenhouse Gas Emission Trading Scheme.

On land, many biological sequestration programs emphasize storing carbon in soil organic matter in agricultural fields, in woody encroached sites, and in the soils and wood of plantations. Land-based sequestration in agricultural soils restores all or part of the soil organic carbon (SOC) lost with plowing and intensive agriculture (Gebhart et al. 1994; Lal et al. 1999). No-till and low-till management are additional approaches proposed for increasing soil organic carbon in croplands.

For plantations, the most controversial factors for carbon sequestration and management include the feasibility and permanence of the carbon sequestered, the scale of management needed to offset anthropogenic emissions, and the accompanying biogeochemical changes that would occur. As an example, a carbon sequestration rate of 3 000 kg C ha^{-1} annually in plantations would require an area the size of Texas or Pakistan to offset 0.2 Pg C yr^{-1} of emissions; such a rough calculation ignores economic and biophysical limitations to storage and downstream losses of carbon as the wood is processed. A more complete evaluation of the feasibility of carbon storage by vegetation management, both scientifically and economically, is needed, including a more complete biogeochemical accounting of the consequences. The biogeochemical interaction that we examine in this chapter is water availability; other key interactions, such as with nitrogen, are beyond the scope of this chapter (e.g., Vejre et al. 2001; Dalal et al. 2003).

Here we will examine some of the potential benefits of biological sequestration programs on land, some of the uncertainties surrounding them, and some unintentional consequences if they are initiated broadly. We will also address a related land-cover change, woody plant encroachment, which has important consequences for carbon and water cycling. Woody plant encroachment differs from afforestation and abrupt land-cover changes because it occurs over many decades. However, its global extent, potential for carbon sequestration, and similarities to afforestation make it important to address. For these land cover and land-use changes we will estimate potential carbon sequestration rates, explore key biophysical interactions, and discuss examples of other biogeochemical and hydrological changes that may occur. For example, plantations may be the most beneficial environmentally when they are used to ameliorate groundwater upwelling, but they may also decrease water yield (defined as the amount of water from a unit area of watershed) (Herron et al. 2002; Farley et al. 2005). Our long-term goal is to identify these biogeochemical and hydrological costs and benefits that accompany sequestration scenarios.

19.2 Afforestation

19.2.1 Afforestation: Carbon Storage Potential

For purposes of this chapter, afforestation is defined as the conversion of rangelands and agricultural lands to tree plantations. Forest plantations grown on rangelands or agricultural lands have large sequestration potentials because carbon can be stored rapidly in biomass pools that are large, have relatively slow turnover times compared to the previous vegetation, and have wider carbon to nutrient ratios (e.g., Paul et al. 2002; Halliday et al.

2003). Depending on land-use history, plantations can also increase or decrease carbon storage in the soil (e.g., Post and Kwon 2000; Guo and Gifford 2002; Farley et al. 2004). Summary data across plantations for total carbon storage in soil and wood reveal an average rate of C storage of ~3 600 kg C ha^{-1} yr^{-1} (e.g., Hamilton et al. 2002), an order of magnitude larger than that in agricultural soils (Post and Kwon 2000). Based on this coarse estimated rate of carbon storage, ~280 million ha of plantations would be needed globally to reach a target of 1 Pg C yr^{-1}, approximately one-sixth of global carbon emissions in fossil fuels.

Sequestration rates a little higher than 3 600 kg C ha^{-1} yr^{-1} are possible in some locations and in the short term, but significantly higher rates are unlikely over large areas (Jackson and Schlesinger 2004). Moreover, back-of-the-envelope estimates such as the one described above do not address issues such as permanence of carbon storage and leakage, defined as activities shifted to locations outside of a sequestration program that counteract some of its carbon benefits (e.g., Murray et al. 2004). One alternative that acknowledges these uncertainties is carbon "rental" payments, where landowners contract to store carbon for specific periods of time (Lewandrowski et al. 2004).

Additional costs also need to be included to estimate the technical and economic potentials of carbon storage, including site preparation and planting, potential carbon losses from disturbance (e.g., storms, pests, and fires), and post-harvest losses in timber use and processing (Jackson and Schlesinger 2004; Murray et al. 2004). Carbon stored in the soil or wood must be protected from plowing, fire, storm damage, and decomposition to keep the carbon from returning to the atmosphere. If long-term uses for the wood are not found, some of the plantation carbon will almost certainly return to the atmosphere after harvesting through uses such as burning for fuel or pulp and paper supply. Economic competition for land use must also be considered. Recent economic models for U.S. agriculture and forestry suggest that C prices would need to be ~U.S.$125 to $400 per metric ton C equivalents for potential sequestration in plantations to approach 0.2 Pg C yr^{-1} (McCarl and Schneider 2001; Lewandrowski et al. 2004).

19.2.2 Afforestation: Evapotranspiration and Water Yield

National policies promoting afforestation can lead to considerable carbon storage for decades, but the amount stored, the economic subsidies needed, and the environmental changes that would result require careful evaluation (e.g., McCarl and Schneider 2001). If economic incentives are provided to convert agricultural or range lands to plantations, what other social, economic, or biogeochemical changes might occur?

To provide a concrete example of such "ancillary" changes with afforestation, we examine changes in water yield. We explore the evidence for this interaction at a range of scales, including long-term catchment measurements and potential atmospheric feedbacks operating over large areas. The outcomes of those feedbacks depend on the initial land use, climate, and location of afforestation in complex but predictable ways.

Large changes in evapotranspiration (ET) and water yield are common with shifts between herbaceous and woody vegetation (Bosch and Hewlett 1982; Zhang et al. 2001; Brown et al. 2005; Farley et al. 2005). The effects of deforestation on water balance have been examined extensively at a range of scales that include eddy covariance measurements of canopy fluxes, multi-year catchment manipulations, and general circulation models examining regional mass and energy fluxes (e.g., Bosch and Hewlett 1982; Rannik et al. 2002; Werth and Avissar 2002). As just one example, Grace et al. (1998) found that daily evapotranspiration from a Brachiaria pasture in Brazil was 39% lower than in a nearby undisturbed rain forest, 2.74 mm d^{-1} compared with 4.48 mm d^{-1}, respectively. However, research has clearly shown that deforestation and afforestation are not opposite and reversible processes in terms of water yield (Robinson et al. 1991; Scott and Lesch 1997; Vertessy 1999).

In grasslands and agricultural lands, afforestation can reduce water yield from rivers and streams because canopy interception and evapotranspiration generally both increase (e.g., Herron et al. 2002; Farley et al. 2005). Interception tends to be relatively small in grasslands but can account for 10–20% of rainfall in hardwood systems and 20–40% in conifer plantations (e.g., Rao et al. 1979; Le Maitre et al. 1999; Levia and Frost 2003). For evapotranspiration, afforestation can alter leaf area, stomatal characteristics, surface roughness (a measure of the aerodynamic properties of the land surface), albedo, and the depth and density of root systems (Vertessy 2001; Brooks et al. 1997; Jackson et al. 2000). Holmes and Sinclair (1986) and Zhang et al. (1999) suggested that annual evapotranspiration from catchments planted to eucalypts or other plantation species could increase anywhere from ~50–250 mm compared to a grassland catchment. This change in ET could then affect other terms in the water budget, potentially decreasing annual stream flow, deep drainage, and rates of baseflow, as well as altering salinity.

In humid grasslands like those in the Pampas of Argentina, evapotranspiration is less than precipitation (P), and hydrological recharge takes place through deep drainage. Where groundwater is available, trees reverse the relationship between ET and P, initiating a net discharge regime (ET > P) in which groundwater use exceeds deep drainage (Fig. 19.1). The discharge regime is sustained by groundwater recharge and lateral transport from the surrounding grassland. Ground-
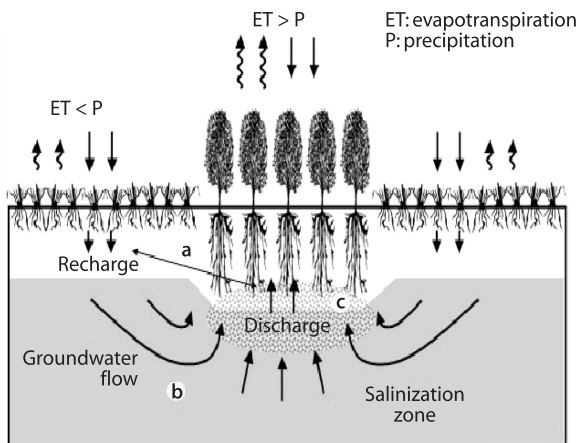
Fig. 19.1. Mechanism of groundwater use and salinization after afforestation of grasslands. Tree establishment reverses the net vertical flux of groundwater when absorption exceeds recharge. Groundwater exits the system through transpiration, leaving salts in the afforested stand (from Jobbagy and Jackson 2004)

water use through transpiration leaves dissolved salts behind that accumulate in the system if this net discharge regime is sustained. In the framework in Fig. 19.1, salinization is predicted to occur when three ecohydrological conditions are met: *(a)* grasses allow a moderate net hydrological recharge that is interrupted by trees, *(b)* the terrain and soil conductivity allow a sustained horizontal flow of groundwater and dissolved salts towards the tree stands, and *(c)* trees access groundwater. In the Pampas, afforested plots (10–100 ha in size) showed 2–19-fold increases in groundwater salinity (Jobbágy and Jackson 2004) compared to adjacent grasslands; in addition, surface soil pH typically decreased by a full pH unit, a change comparable in magnitude to the effects of acid rain (Jobbágy and Jackson 2003).

Globally, one of the best tools for examining the effects of afforestation on streamflow is long-term catchment datasets (e.g., Holmes and Sinclair 1986; Zhang et al. 2001). More recently, Farley et al. (2005) compiled catchment data in afforested grasslands and shrublands from 26 catchment studies with 504 annual observations, in part to stratify changes by original vegetation type and climate, and to quantify the effects of plantation age. Across the dataset, annual runoff decreased consistently in afforested catchments (Fig. 19.2, grassland data only; $P < 0.001$ for both grasslands and shrublands). Reductions in annual runoff in afforested grasslands and shrublands were similar in the first 5 years after tree establishment (–16% and –15%, respectively) but diverged as the plantations aged. Afforested grasslands reached a 50% reduction in runoff within a decade compared with a one-third decrease in afforested shrublands. Moreover, the data from grasslands show faster losses of runoff with eucalyptus afforestation, sometimes within a decade (Fig. 19.2 and Farley et al. 2005).
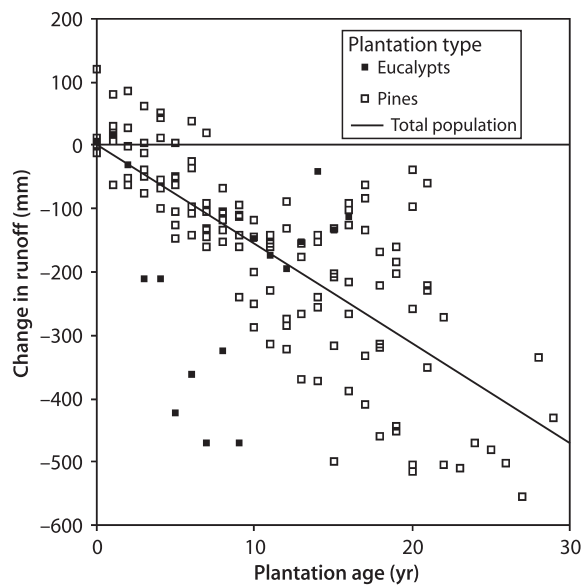

Fig. 19.2. Change in runoff with plantation age ($r^2 = 0.77$; $P < 0.001$). Data come from 26 catchment studies of afforestation globally and a database of 504 yearly observations (from Farley et al. 2005)

This synthesis clearly demonstrates that a loss of 100 to 500 mm (or one-third to three-quarters) of annual streamflow can be expected with afforestation at sites with mean annual precipitation >750 mm. Eucalypts tend to reduce streamflow more than pines do (Fig. 19.2 and Farley et al. 2005). Dye (1996) compared eucalypts and pines in South Africa and found that the rate of increase in evapotranspiration following afforestation was more rapid under eucalypts because of their faster growth and canopy closure; better data for poplar plantations would be helpful to compare with the more extensive pine and eucalypt datasets. While there is likely to be some variation in species effects by region, generalizations regarding the effects of different plantation species on runoff should be useful for planning afforestation projects and the tree species that will be used in them.

19.2.3 Afforestation: Potential Atmospheric Feedbacks

While afforestation typically increases evaporative water losses and reduces runoff locally, climate feedbacks at regional scales could also affect precipitation and water use. These regional effects on water availability depend on the location, extent, and patchiness of afforestation and operate through changes in albedo, roughness length, and water transport properties from soil to the atmosphere, including leaf area index, stomatal conductance, and rooting depth. These variables influence climate because they help determine the total amount of energy transferred between the vegetation and the atmosphere as well as the fraction of that energy driving evapotranspiration (latent heat) and warming of the

air (sensible heat). Since little attention has been given to the direct climatic effects of afforestation, our understanding here must derive primarily from studies of deforestation.

Most simulations of tropical deforestation have concluded that the effect of the forest, relative to pasture, on precipitation is greater than its effect on evapotranspiration (Hahmann and Dickinson 1997; Hoffmann et al. 2003), such that runoff should be higher under forest than pasture. Similar effects of tree cover were observed in simulations of tropical savanna regions (Hoffmann and Jackson 2000, 2002; Clark et al. 2001). Analogously, at coarse scales the conversion of agricultural lands and grasslands to homogeneous forest cover might be predicted to increase precipitation slightly, in part because of increased water recycling through convective storms. Whether this increase is enough to offset completely the increased evapotranspiration of the forests is unclear.

In temperate regions, however, published simulations suggest otherwise. Although simulation studies have sometimes found that tree cover increases precipitation in temperate regions, this effect has been consistently smaller than the effect on ET, indicating that complete afforestation should result in reduced runoff (e.g., Xue et al. 1996; Bounoua et al. 2000). Furthermore, under some conditions, afforestation is likely to *reduce* precipitation, particularly near bodies of water, where actively transpiring vegetation might reduce the strength of the sea breeze and associated convection and cloud formation (Pielke and Avissar 1990). Simulations of land-use change in the US indicate that summer precipitation should decline with afforestation of the southern coastal plain, Florida, and the Great Lakes region (Copeland et al. 1996; but see Pielke et al. 1999). In such cases, the climate feedbacks resulting from afforestation should magnify the local effects of plantations on water yield, resulting in even lower water yield than predicted by catchment studies alone.

The pattern and scale of land-cover change in some cases may be as important as the switch between trees and herbaceous vegetation. Circulation patterns similar to a sea breeze can arise at the interface between forest and non-forest vegetation (Pielke and Avissar 1990; Pielke et al. 1997), which is particularly relevant over heterogeneous landscapes typical of afforestation. Where heterogeneity promotes these mesoscale circulations, convection cells develop where ET is lowest (Pielke and Avissar 1990; Pielke et al. 1997), leading to the rather counterintuitive observations that cloud formation can be greater over nonforest patches (Roy and Avissar 2002; Weaver et al. 2002). A patchwork of plantation and non-forest vegetation may therefore generate more precipitation than homogeneous forest vegetation, particularly under conditions of weak synoptic forcing, such as when prevailing winds are weak (Pielke et al. 1997).

19.3 Woody Encroachment and Agriculture

19.3.1 Grassland Conversions with Woody Plant Encroachment and Agriculture

Shifts among grasslands, woody communities, and croplands represent another form of vegetation change important for carbon cycling today. The conversion of croplands to perennial vegetation consistently leads to carbon stored in soil organic matter (Post and Kwon 2002; Guo and Gifford 2002). This carbon storage benefit is recognized in the U.S. Conservation Reserve Program (CRP), which has provided economic incentives for farmers to convert crops to perennials in ~15 million ha to date (Skold 1989; Jackson and Schlesinger 2004).

At the regional scale, potential carbon storage in croplands can be estimated by combining observed sequestration rates through programs like the CRP or no-till agriculture with the extent of agricultural lands in a region. Recent reviews of more than 100 observations show that SOC increased ~450 kg C ha^{-1} yr^{-1} after croplands converted to pastures or no-till management (Post and Kwon 2000; West and Post 2002), peaking five to ten years after conversion and slowing considerably within two decades (West and Post 2002). Taking the U.S. as an example, if all ~132 000 000 ha of croplands in production in 2001 were converted to no-till agriculture or, less likely, were retired through the CRP, potential sequestration rates of 0.059 Pg C yr^{-1} might be possible for several decades. This amount would be ~4% of total U.S. fossil fuel emissions of ~1.6 Pg yr^{-1}. Currently, the ~15 million ha enrolled in the CRP store an order of magnitude less than this technical potential, ~0.005 Pg C yr^{-1} (Jackson and Schlesinger 2004).

Conversion of croplands back to perennial grassland is not the only change in land use likely to affect C pools in grasslands. Many regions of the world are currently undergoing woody plant encroachment, a process enhanced by management practices such as intensive grazing and fire suppression and possibly by interactions with climate change (Kucera 1960; Bragg and Hulbert 1976; Briggs et al. 2002). The soil C consequences of woody encroachment into native grasslands vary considerably, ranging from gains of soil C in some ecosystems (Archer et al. 2001) to no net change or losses in others (Kieft et al. 1998; Schlesinger and Pilmanis 1998; Gill and Burke 1999; Tilman et al. 2000; Smith and Johnson 2003). Recent work suggests that the changes in soil carbon storage are related to climate, with drier grasslands more likely to gain soil C and productive sub-humid/humid grasslands to lose it (Jackson et al. 2002).

From a soil C perspective, the primary difference between woody plant encroachment into native grasslands and into old agricultural fields is the quantity, quality, and distribution of soil C at the onset of woody plant
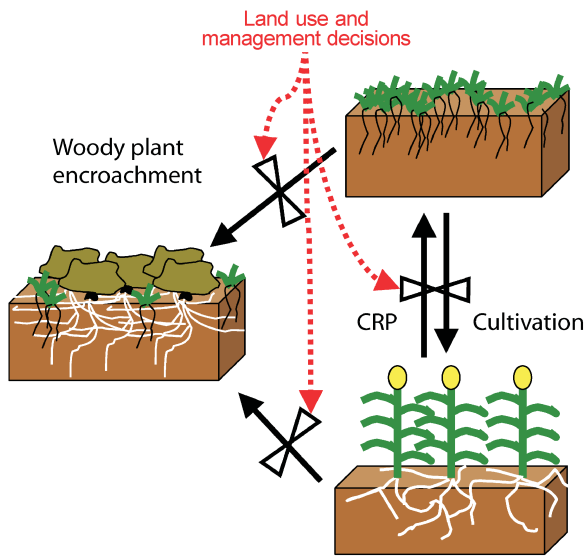
Fig. 19.3. Conceptual diagram of how land-use and management decisions alter the structure and function of grassland ecosystems (*CRP* = Conservation Reserve Program)

establishment (McCulley et al. 2005). In tallgrass prairie, for example, agricultural cultivation significantly reduces soil C pools (Burke et al. 1989; Huggins et al. 1998; Knops and Tilman 2000). Thus, unlike woody plant encroachment into native grasslands where the soil C response varies depending on the climate, land management, and species involved, woody plant establishment and dominance in many old agricultural fields should replenish soil C pools. Indeed, this hypothesis is supported by Guo and Gifford (2002) who found that converting cropland to secondary forest or plantations increased soil C pools by 53% and 18%, respectively.

19.3.2 Processes Controlling Soil C Storage: Grassland Vs. Woodland

The amount of C stored within the soil and the rate at which C accumulates and turns over is determined largely by the quantity and quality of C inputs, soil properties such as clay or sand content, and possibly the microbial community present. Edaphic properties such as texture are unlikely to change dramatically within the time-scale of plant establishment on abandoned cropland or conversion of grassland to woodland. However, major changes in other controlling variables, such as the quantity and quality of C inputs through net primary productivity as well as microbial community composition are likely.

In terms of carbon inputs to the soil, many differences between herbaceous and woody species exist. For example, grassland species tend to allocate a higher proportion of fixed C to belowground root systems than woody species do (Jackson et al. 1997). In addition, the

two vegetation types differ in root morphologies and distributions, with grasses having a shallower, more densely fibrous root system and woody plants typically having deeper, coarser roots (Canadell et al. 1996; Schenk and Jackson 2002a,b). Thus, woody dominated systems may deliver more C than grasslands to the soil surface via litter while increasing C inputs deep within the soil profile. The quality of litter inputs, both above- and belowground, often differs significantly between the two types as well, with woody material containing more lignin and secondary compounds. All of these factors interact to alter the rate of decomposition, the soil microbial community, and ultimately the quantity, quality, and depth distribution of soil C storage.

Previous studies showing that differences in carbon inputs between grasslands and woodlands alter soil C storage come primarily from global analyses and woody encroachment studies in rangelands. Global analyses of root distributions and soil organic C contents suggest that converting croplands to grasslands or woodlands alters soil C profiles through changes in the amount and depth of root C inputs (Jackson et al. 1996; Jobbágy and Jackson 2000). More specifically, a comparison of grasslands and forests in sub-humid climates, such as tallgrass prairies experience, found that the proportion of total soil organic C stored in the top 20 cm of the soil profile was higher in forests; in contrast grasslands stored proportionally more soil C deeper within the profile (Jackson et al. 2000). Using the fact that woody and herbaceous plants often differ in photosynthetic pathways and produce organic matter with different ^{13}C signatures (Smith and Epstein 1971), woody plant encroachment studies have shown that C contributions to the soil are highest at the surface soil layers (Gill and Burke 1999; Smith and Johnson 2003), perhaps resulting from greater inputs of litter at the soil surface (Connin et al. 1997; Boutton et al. 1999; Hibbard et al. 2001). Moreover, in woodlands or brush, the presence of a thicker litter layer may result in a relatively high proportion of aboveground litter decomposing before entering the mineral soil. Alternatively, if such material had high C/N ratios, it might be slow to decompose and enhance longer-term soil C sequestration.

Besides altering the depth distribution of soil C storage, grasslands and woodlands differ in soil C quality, as measured by soil organic matter C/N ratios, organic C chemical constituents, and the relative percentage residing in active, slow, and passive soil C pools. In general, woody dominated areas have higher total soil C/N ratios, greater quantities of complex organic molecules within soil organic matter, higher percentages of slowly turning over soil C pools, and, consequently, longer mean residence times for C within the soil than grasslands have (Connin et al. 1997; Boutton et al. 1998; Gill and Burke 1999; Martens et al. 2003; Smith and Johnson 2003; McCulley et al. 2005).

19.3.3   Uncertainties in Water and Carbon Balances with Woody Plant Encroachment

The ecohydrological consequences of woody plant encroachment are much less clear than for afforestation (Wilcox 2002; Petheram et al. 2002; Huxman et al. 2005). Because the ratio of evaporation to transpiration increases with aridity, plants have a proportionally smaller role in regulating total water fluxes in arid systems. In addition, streamflow and evapotranspiration are differentially affected by woody encroachment, depending on the degree and seasonality of aridity and the availability of subsurface water (Huxman et al. 2005).

Despite uncertainties, a few generalizations are possible. Streamflow may decline with woody encroachment in semi-arid landscapes dominated by subsurface flow, including systems with available groundwater or karst systems where water flows quickly into the subsoil (Jackson et al. 1999; Seyfried et al. 2005). There, the deeper root systems of woody plants can capture water that is otherwise unavailable to herbs. Similarly, woody encroachment may increase bare soil evaporation in semiarid ecosystems, because of the increasingly patchy vegetation cover (Huxman et al. 2005). Explicitly considering the ecohydrology associated with vegetation change provides important information on the consequences of woody plant encroachment for carbon storage.

In addition to woody plant encroachment, woody plant or brush "thickening" is likely to decrease stream flow. Brush thickening is defined as an increase in woody plant density at sites where woody species were already present. Like woody encroachment into grasslands, thickening has occurred in many regions of the world, including the western U.S. and Australia (Van Auken 2000; Fensham and Fairfax 2003; Hicke et al. 2004). In practice, brush thickening will affect stream flow through similar mechanisms to those occurring with afforestation and woody encroachment. All three land-cover changes can increase leaf area, evapotranspiration, and rainfall interception, potentially decreasing stream flow. However, the more arid a system is, the more uncertain the outcome for water yield will be, as evapotranspiration from the soil plays an increasingly important role relative to plant transpiration at drier sites.

How much more carbon is stored in shrublands and woodlands compared to grasslands remains uncertain. Decades of research have described the formation of "islands of fertility" under single plants and plant clusters as woody species invade grasslands. This process increases heterogeneity in cover and net primary productivity, concentrating soil nutrients and carbon under the woody plants and depleting them by erosion and other processes in the interspaces. Using such shrub islands and herbaceous interspaces as endpoints to compare encroached and intact grassland systems may underestimate the carbon stored in the original grasslands and overestimate it in the expanding woodlands. This is especially true when the microsites first invaded by the woody plants are the deepest, most productive soils. When examined for entire ecosystems, data from 242 sites show increasingly more SOC for grasslands than for shrublands/woodlands as precipitation increases to 1000 mm (Fig. 19.4).

Land-use history is a crucial lens for viewing the consequences of vegetation and land-use change. Degraded rangelands can be actively restored, and store carbon in turn, using any type of vegetation. Transformations in the opposite direction, however, from shrubland or woodlands to grasslands, are also enlightening. Recent meta-analysis of >100 studies emphasizes the potential for SOC gains at sites originally dominated by woodlands when converted to grasslands (Conant et al. 2001) and soil carbon losses when pastures are converted to plantations (Guo and Gifford 2002). Grasslands are deceptively productive ecosystems at both short and long time scales (e.g., Retallack 2001).

U.S. carbon budgets estimate the sequestration due to woody plant encroachment to be $\sim$0.12 to 0.13 Pg C yr^{-1} but acknowledge large uncertainties in this estimate (e.g., Houghton et al. 1999). We believe the correct estimate will eventually be positive but smaller, both because of the relationship between soil carbon stor-
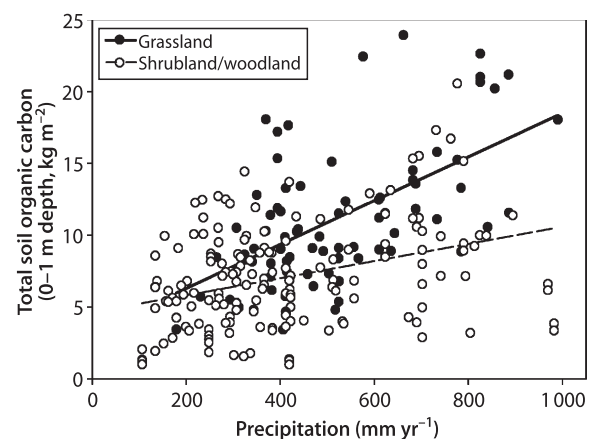

Fig. 19.4. An analysis of soil organic carbon (kg m^{-2}) vs. mean annual precipitation (mm) for the top meter of soil. The data are derived from the National Soil Characterization Database of the U.S. Department of Agriculture and the World Inventory of Soil Emission Potential Database of the International Soil Reference and Information Centre. Across the global dataset, the slope of the relationship of SOC vs. precipitation was 2.6 times higher for grassland vegetation than for shrublands/woodlands ($p = 0.001$). Whereas grassland SOC was statistically indistinguishable from values for woody plants at 200 mm MAP (Mean Annual Precipitation), woodlands had 43% less total SOC than grasslands at 1000 mm ($p < 0.01$) (from Jackson et al. 2002)

age and precipitation described earlier (Jackson et al. 2002) and because the land area attributed to woody encroachment is over-estimated – 220 million ha, or two-thirds of the continental U.S. excluding forests and croplands (Houghton et al. 1999; Pacala et al. 2001). Better constraining the carbon sink for woody encroachment will entail a combination of field research, remote sensing, and spatially explicit modeling that takes into account precipitation (and climate in general), land-use history, erosion, and other disturbances. Patch-based comparisons of vegetation yield important ecological insight but must be scaled up appropriately for conclusions at ecosystem, landscape, and regional scales.

19.4 Summary and Conclusions

The environmental consequences of afforestation and other biologically based sequestration programs have received much less attention than their carbon sequestration potential (e.g., Herron et al. 2002; Jackson and Schlesinger 2004; Farley et al. 2005). Potential sequestration rates are high but will require managing enormous land areas. One of the most important issues in how to implement land-use change projects within the Clean Development Mechanism of the Kyoto Protocol is to understand their total effects on local livelihoods and environments (Pedroni 2003). For plantations, the synthesis described above clearly shows that a reduction in runoff can be expected after afforestation of grasslands and shrublands. In a few locations such as part of Australia where lower runoff can ameliorate salinity and groundwater upwelling, this may be a positive change. In other regions, the reduction in runoff could cause water shortages, a tradeoff that should be acknowledged before policies promoting afforestation are implemented. One challenge and opportunity to the research community is to help policy makers choose locations for afforestation that are as economical and environmentally beneficial as possible (Pimentel et al. 2004).

The ability to predict the likely effects of afforestation in specific places will be the biggest challenge to zoning and planning for these projects (Farley et al. 2005). Catchment data are collected over decades and are unavailable in most places. However, we can use indicators, such as the change in runoff as a percent of mean annual precipitation at a site, to gain insight into the likely severity of the loss of runoff. Synthesis data suggest that trees are able to use 15 to 20% more precipitation than grasses, so that in a region where natural runoff is less than 10% of mean annual precipitation, afforestation may result in a complete loss of runoff, whereas in places where natural runoff is 30% of precipitation, it may be cut in half when trees are planted (Farley et al. 2005). This information can be useful to land managers and policy makers in guiding the location of plantations with respect to other water demands. There are also many other changes that need to be examined through sequestration scenarios. For example, the potential effects on biodiversity would depend in part on the previous land use. If landowners choose to afforest agricultural lands, the effects on biodiversity would probably be minimal; if instead they choose to afforest native grasslands and rangelands, the consequences would be larger.

For woody plant encroachment in arid and semi-arid systems, the biggest uncertainties are the consequences for the water balance and groundwater recharge and its regional and global role in the carbon balance. Understanding the consequences for water yield is especially important because essentially all of the places where woody encroachment occurs are water limited for both plants and people.

Finally, this book summarizes some of the scientific progress in more than a decade of GCTE's existence (GCTE-Global Change and Terrestrial Ecosystems). As GCTE ends and the scientific community looks forward, the biggest need for global change research is to do a better job of combining research in the social and natural sciences. The natural science community cannot hope to predict the consequences of global change without input from economic and policy models. The plantation scenarios discussed above are a good example. In turn, economists and other social scientists need input on the environmental changes that are likely to affect people and economies in the future. This need is reflected in the next generation of GCTE, the Global Land Project (see Ojima et al. 2007, Chap. 25 of this volume). The Global Land Project is a joint effort of the International Geosphere-Biosphere Programme (IGBP) and the International Human Dimensions Programme (IHDP).

Fortunately, many good examples of such coordination are underway. One of these is the global desertification network, as presented in such projects as ARIDnet, a research network for studies of global desertification (Reynolds and Smith 2003; Reynolds et al. 2007, Chap. 20 of this volume; http://www.biology.duke.edu/aridnet/). ARIDnet and other related projects recognize the important dual effects of humans and the environment in understanding global change. Another example is SAHRA (Sustainability of semi-Arid Hydrology and Riparian Areas) based in Arizona (http://www.sahra.arizona.edu/). This project seeks to develop an integrated, multidisciplinary understanding of the hydrology of semi-arid regions, and to build partnerships with a broad spectrum of stakeholders (both public agencies and private organizations). With the multi-disciplinary philosophy that these and other projects display, we are well on our way to making our research stronger and more relevant to society.

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