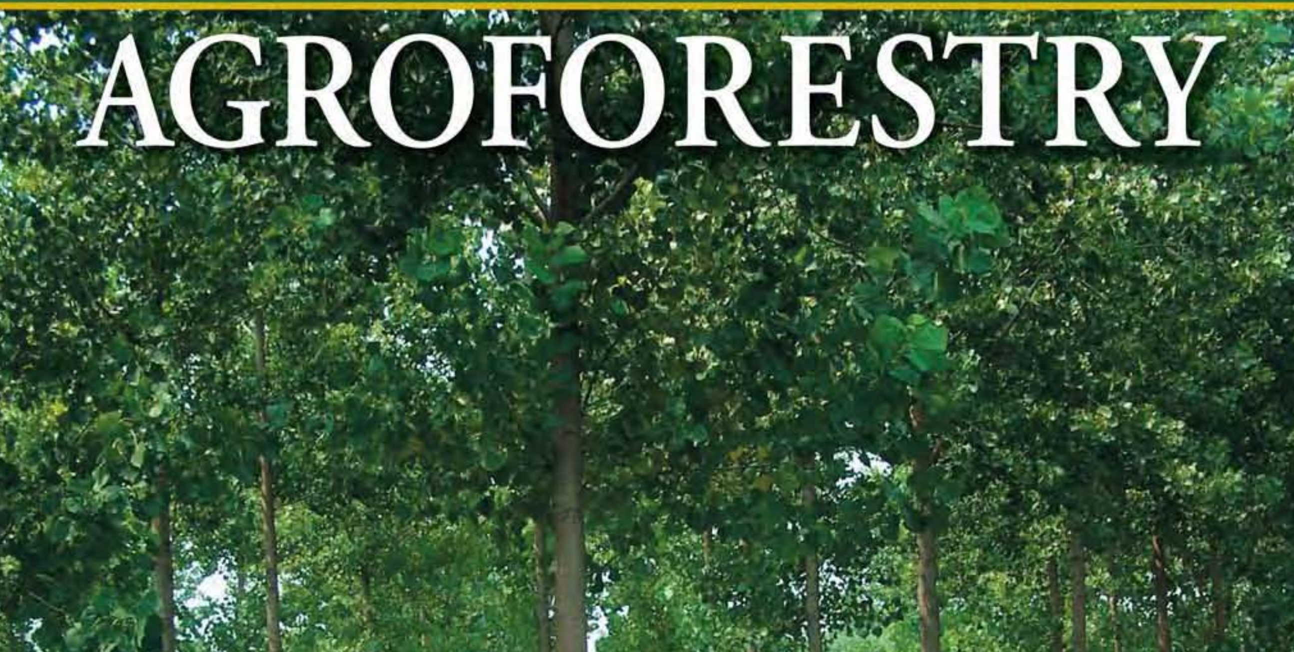
ECOLOGICAL BASIS OF AGROFORESTRY



Edited by

Daizy Rani Batish
Ravinder Kumar Kohli
Shibu Jose
Harminder Pal Singh

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Preface

The world at present is facing innumerable problems such as burgeoning population, ecosystem degradation, particularly in the tropics, declining agricultural productivity, and changing environment. In order to sustain in the future, it is essential to find solutions to these problems, particularly with regard to ensuring food security and coping with the changing environment. Existing approaches to enhance productivity and mitigate environmental degradation are inadequate. Proper land-use patterns, sustainable agroecosystems, and resource management are possible alternatives to these problems. Agroforestry—a traditional practice of combining trees with agricultural crops or pasture—can contribute substantially in this direction through its multiple benefits and ecosystem services. If properly designed, agroforestry may help in alleviating poverty, providing food security and livelihood, maintaining ecosystem health, managing pest and weeds, conserving biodiversity, and mitigating greenhouse effects by carbon sequestration. Conversely, a poorly designed agroforestry system may lead to problems such as loss of productivity due to resource competition and allelopathy or negative effects of shading, aggravated problems of pest and weed infestation, loss of diversity, and ecosystem degradation due to the introduction of invasive species.

For an agroforestry system to be profitable, better understanding of various ecological processes that govern these complex systems is required. This volume aims at providing knowledge as to how ecologically sustainable agroecosystems can meet the challenges of enhancing crop productivity, soil fertility, and environment sustainability. The topics of the 19 chapters were carefully selected to accomplish the above objectives. These are divided into four sections—Ecological Interactions: An Overview (seven chapters), Belowground Ecology (six chapters), Models in Agroforestry (two chapters), and Ecological Economics (four chapters).

Part I focuses on various tree–crop interactions in different ecoregions of the world. Various above- and belowground interactions, especially in alley-cropping systems in temperate zones, have been critically analyzed and will be of immense help to readers. Among various interactions that affect crop productivity, allelopathy—a chemical-mediated interplant interaction—has often been rejected because of lack of sufficient field demonstration. A chapter is devoted to this important aspect of chemical ecology, which also highlights how allelopathy and the chemicals involved therein can be put to some practical use. The proof of attempt has also been made to include other important issues such as tri-trophic interactions and ecologically based pest management in agroforestry and how crop production can be enhanced. Part II is devoted to root-mediated belowground interactions in agroforestry systems and their role in enhancing crop productivity, soil fertility, and sustainability. An exhaustive study on litter dynamics in plantation and agroforestry systems and various factors affecting nutrient release may be beneficial to readers. Part III provides insight into the role of ecological modeling of complex agroforestry systems such as shelterbelts and how they help in choosing suitable computer-based designs to gain profitability. Part IV deals with various socioeconomic aspects of agroforestry and technological tools that benefit society in different eco-regions of the world. It also intends to supply in-depth knowledge on various farming systems and technologies that help enhance the socioeconomic status of farmers and provide environmental benefits to land users.

In sum, efforts have been made to integrate the relevant information on various ecological processes in the agroforestry system into a single comprehensive volume that will be useful to university teachers, students, researchers, agroforestry specialists, landscapists, agriculture and forestry extension workers, scientists, and farmers.

We offer our sincere thanks to all the authors and reviewers for their commendable contributions and cooperation.

Daizy Rani Batish
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Part I

Ecological Interactions: An Overview

1 Ecological Interactions in Agroforestry: An Overview

Ravinder Kumar Kohli, Harminder Pal Singh,
Daizy Rani Batish, and Shibu Jose

CONTENTS

1.1 INTRODUCTION

Agroforestry is one of the sustainable approaches to land-use management where both agriculture and forestry combine into an integrated production system to get maximum benefits (Kidd and Pimentel, 1992; Nair, 1998). As per ICRAF (International Centre for Research in Agroforestry, now World Agroforestry Centre), "agroforestry is a deliberate integration of woody components with agricultural and pastoral operations on the same piece of land either in a spatial or temporal sequence in such a way that both ecological and economic interactions occur between them." Incorporation of the trees under agroforestry systems (AFS) to harvest potential benefits of trees offers a good option under Low Input Sustainable Agriculture (LISA). In fact, it is an age-old practice revived in the recent past with a renewed scientific interest to maintain the sustainability of agroecosystems (Noble and Dirzo, 1997). The revival of agroforestry became inevitable to meet growing demands of increasing population, to compensate forests in the wake of fast increasing rate of deforestation and soil degradation, both in the tropics and temperate regions of the world, and to conserve biodiversity. Agroforestry provides one of the best alternatives for planting trees outside forests. In other words, it is a collective name for sustainable land-use system to get social, economical, and environmental benefits (Sanchez, 1995). It leads to a more diversified and sustainable system than other croplands without trees. Griffith (2000) considers agroforestry as an ecologically sustainable land-use option alternative to the prevalent subsistence farming patterns for conservation and development, particularly in the tropics. Though practiced in the majority of ecoregions, agroforestry is more common in the tropics. According to a report of the World Bank, around 1.2 billion rural people currently practice agroforestry the world over (World Bank, 2004). There are more than 2000 tree species used in agroforestry (Rao et al., 2000). AFS have been classified based on structural, functional, physiognomy, floristics, socioeconomic, and ecological aspects (Nair, 1993; Ffolliott, 2003). However, classification based on structural components is very common.

Nair (1998) pointed that the concept of agroforestry, which popularized during the 1990s, has passed through stages of hypothesis making and experimentation and now focuses on science and technology to get a better and wider applicability. Sanchez (1995) opined that the science of agroforestry centers around four factors—competition, complexity, sustainability, and profitability and there should be a balance among all these factors to get fruitful results. In fact, agroforestry is substantially assisting in meeting the United Nations Millennium Development Goals (MDG) such as eradication of poverty and hunger, better health, nutrition and education to people, gender equality, and environmental sustainability, particularly in developing countries (Garrity, 2004, 2006). In other words, agroforestry is an integrated science that helps in bridging the gap between the need for conservation and meeting people's demand at the same time.

However, there are several limitations linked with agroforestry. These include competition of trees with crops for resources, allelopathic effects of trees on crops, rapid growth of some tree species within agricultural fields occupying the space of crops, entry of invasive species in the agricultural land, and trees serving as habitat for harmful pests and diseases. To gain maximum benefits from AFS, it is essential to minimize the negative concerns linked to it. In fact, the ecological sustainability and success of any AFS depend on the interplay and complementarity between positive and negative interactions. It can yield positive results only if positive interactions outweigh the negative interactions.

1.2 ECOLOGICAL INTERACTIONS UNDER AFS

An ecological interaction refers to the major impact of one species on the other or on the same type of species. In general, there are three types of interactions: neutral, positive, and negative. Among these, neutral interactions are very rare and happen only when the niches are wide apart. Specifically, the interactions in AFS can be complementary (positive), supplementary (neutral), or competitive (negative) (van Noordwijk and Hairiah, 2000). Further, these can be belowground or aboveground. In agroforestry, particularly simultaneous systems, trees (being perennial, large-sized, and dominant) have a major and continuous influence on the crops and determine the extent of interactions (Ong, 1995). Further, due to well-developed root systems and better adaptability toward environmental stresses, trees are able to modify AFS for their own benefit. Additionally, trees generally have their roots well below the crop zone, use water from the lower soil layers, and thus do not affect crop. Rather, tree roots act as safety nets and capture the nutrients that are lost because of leachation (van Noordwijk and Hairiah, 2000). Swift et al. (2006) pointed that incorporation of trees within any land-use system results in a large number of secondary interactions. AFS are much more complex than the sole cropping system because of the nature and arrangement of the components and their unequal size. Initially, the research on tree–crop interactions in AFS received little attention of scientists and researchers; however, it has recently gained the momentum world over (Rao et al., 1998). Various positive and negative interactions of trees with crops, particularly under simultaneous agroforestry system (SAFS), are given in Table 1.1. Competition and allelopathy, in addition to the shading, harboring of the enemies of the crops, and invasive potential of some of the introduced tree species, are the predominant negative interactions. On the other hand, positive interactions include improvement of soil fertility through addition of tree litter, natural weed and pest management through allelochemicals of trees or through chemical signaling,

TABLE 1.1
Various Types of Positive and Negative Impacts of Trees on Crops under AFS

Positive Effects	Negative Effects
Soil fertility enrichment	Shading
Improvement of microclimate	Resource competition
Maintenance of water quality	Allelopathy (chemical interference)
Weed and pest management	Invasive behavior of some of the introduced species
Biodiversity conservation	Harboring of harmful pathogens and pests
Enhancing food security	
Alleviating poverty	
Carbon sequestration and greenhouse gas mitigation	
Habitat for wildlife	
Phytoremediation	

modification of microclimate, environmental mitigation and phytoremediation, habitat for wildlife, and conservation of soil, moisture, and biodiversity through the protective roles of trees. The direction and magnitude of these interactions, however, may depend on patterns of resource sharing and the time at which these patterns are determined (Rao et al., 1998).

1.2.1 POSITIVE EFFECTS (COMPLEMENTARITY)

1.2.1.1 Improvement of Soil Fertility and Microclimate

Land degradation and declining soil fertility pose a major threat to agricultural productivity. Use of synthetic fertilizers to replenish soil nutrients fails to provide adequate solution. Incorporation of trees in the croplands can help in maintaining the nutrient pool and enhance soil fertility both under sequential and simultaneous agroforestry (Young, 1997; Rao et al., 1998; Giller, 2001; Thevathasan and Gordon, 2004; Jama et al., 2006b). Tejwani (1994) reported that AFS are an excellent strategy for reclamation of salt-affected soils. Tree litter and prunings improve soil fertility not only through the release of nutrients in the soil by mineralization but by also adding soil organic matter. However, it depends on the quality and quantity of tree litter or prunings, soil type, and climatic conditions of the area. Hulugalle and Ndi (1994) demonstrated that hedgerows of Senna (*Senna spectabilis* [DC] Irwin & Barneby) and Flemingia (*Flemingia congesta* [Willd.] Merrill) significantly improved soil properties in a newly cleared Ultisol (Typic Kandiudult) in southern Cameroon. A significant increase was observed in exchangeable Ca, CEC, and water infiltration in the alleys of both the species. Chander et al. (1998) demonstrated that adoption of *Dalbergia sissoo* Roxb. ex DC., a N-fixing tree, under agroforestry significantly increased nutrient pool, organic biomass, and activities of enzymes—hydrogenases and alkaline phosphatases—in the soil. Further, agroforestry trees also help in improving soil physical and biological properties (Rao et al., 1998). Thevathasan and Gordon (2004) reported that tree intercropping under temperate AFS significantly enhanced the diversity of birds, insects, and earthworms; increased soil organic carbon content and N cycling; and improved soil health. In general, the mechanisms by which trees improve soil physicochemical and biological properties are as follows:

1. Release of nutrients from tree litter and prunings
2. Nitrogen input through biological nitrogen fixation (through N-fixing trees)
3. Phosphorus input through mycorrhizal associations

4. Reduced soil erosion and nutrient leaching
5. Nutrient capture from the subsoil through deep-rooted trees
6. Redistribution of nutrients through lateral roots of some trees

Another positive interaction between trees and crops is the improvement of microclimate through modification of temperature to reduce heat stress and evapotranspiration, improvement of crop–water efficiency and energy balance (Brenner, 1996; Jose et al., 2004).

1.2.1.2 Maintaining Water Quality

Agroforestry can also help in improving water quality by reducing levels of pollution and soil erosion and thus landscape amelioration (Nair and Graetz, 2004; Schultz et al., 2004). For example, riparian buffer zones, if well designed and properly located, can be very helpful in this direction (Dosskey, 2002). These buffers help in reducing the transport of polluted runoffs to the rivers and streams. Agroforestry also improves water-use efficiency and increases environmental sustainability. In addition, trees increase the water-holding capacity of the soil, reduce soil evaporation, increase water infiltration into the soil (Nair, 1993), and efficiently capture rainwater compared with traditional agricultural practices (Lott et al., 2002). Of late, it has been proposed that trees can efficiently increase water productivity, particularly under semiarid regions (Ong and Swallow, 2003; Ong et al., 2007).

1.2.1.3 Weed and Pest Management

In tropical and temperate agroecosystems, weeds and pests interact and interfere with crop plants and cause enormous harm to crop productivity. Their management is a big challenge and the indiscriminate use of synthetic herbicides and pesticides for controlling them has led to a number of problems like toxicological effects on the nontarget species, environmental degradation, and loss of sustainability of croplands. Presence of trees in agricultural lands may reduce weed populations because of the shading effect of trees, availability of less space for their growth, shifts in species composition, and altered environmental conditions (Liebman and Staver, 2001; Sileshi et al., 2006). Jama et al. (1991) demonstrated that alley cropping with *Leucaena leucocephala* (Lam.) de Wit reduced weed density by 90% and increased maize yield by 24%–76%. Incorporation of trees into the cropping system, particularly in the east and west Africa, holds a good potential for the control of parasitic weeds. For example, Gworgwor (2007) observed that *Faidherbia albida* (Del.) A. Chev. trees can fully eliminate *Striga hermonthica* (Del.) Benth. from pearl millet fields.

AFS create a landscape that is important for biological pest control (Pandey, 2007). However, there are conflicting reports regarding the potential beneficial effects of trees in agroforestry for disease and pest management. Studies have indicated that due to modification of microclimate, water regime, moisture, air humidity, and surface temperature, the number of insects, pests, and pathogens increases, particularly near the tree line (Schroth et al., 2000). In contrast, other studies have indicated that trees, particularly as windbreak or hedgerow or shelterbelt, act as barrier to airborne pests and pathogens, repel them, and thus have a protective action (Rao et al., 2000; Sileshi et al., this volume, Chapter 5). In addition, trees may provide more habitats for enemies of insect pests and thus more options for pest management (Middleton, 2001).

Further, allelopathic effects of tree mulch, prunings, and residues can also be useful in weed suppression (Singh et al., 2003). Allelochemicals from trees can be used for sustainably managing the weeds on the pattern of herbicides and pesticides. For example, ailanthone from tree of heaven (*Ailanthus altissima* [Mill.] Swingle), volatile monoterpenes as well as crude oil from *Eucalyptus* species, mimosine from *L. leucocephala*, and caffeine from *Coffea arabica* L. (Rizvi et al., 1999; Singh et al., 2003). Even plant–plant signals through allelochemicals within the soil can be exploited for weed management in a practical way rather than studying their direct physiological effects on the other plants (Birkett et al., 2001). For this, desirable allelopathic trees could be intercropped with crops to achieve weed management through rhizospheric allelochemicals-based signals.

1.2.1.4 Conserving Biodiversity

Biodiversity loss, particularly due to deforestation, is one of the major causes of worry to scientists. Agroforestry helps in reducing biodiversity loss by providing a protective tree cover along agricultural fields. The presence of trees further enhances diversity by providing shelter and habitat to a diversity of other flora and fauna. It also helps in conserving genetic diversity of ethnocultivars or landraces and trees that are in danger of loss and require priority conservation (Noble and Dirzo, 1997; Pandey, 2007). Further, it also helps in conserving traditional knowledge about the conservation of wild varieties of trees and other plants. Studies have shown higher biodiversity levels and species richness in AFS than in sole cropping systems (Estrada et al., 1993; Perfecto et al., 1996; Thevathasan and Gordon, 2004). Agroforestry helps in biodiversity conservation through (1) provision of secondary habitats for species, (2) reduction in the rate of conversion of natural habitats, and (3) creation of a benign and permeable matrix between habitat remnants (Schroth et al., 2004; McNeely and Schroth, 2006). AFS enhance diversity both at the site level as well as at the landscape level. At a given site, AFS have more diversity both at above- and belowground levels than the sole cropping system (Vandermeer, 2002; Ruark et al., 2003). AFS also provide refuge to species in the event of some catastrophic fire (Griffith, 2000). Gillison et al. (2004) reported that complex AFS and shade-grown coffee had higher biodiversity levels than simple sun-grown coffee; however, it was lesser than in the primary forests.

Although AFS have less species diversity than the tropical forest, they have a variety of species diversity compared with traditional agricultural systems. Their rich diversity makes them ecologically resilient and thus gives them the ability to provide more and better ecological functions (Olson et al., 2000; Vandermeer, 2002). Altieri (1995) opined that since AFS are more diverse and have low-input strategies, these have greater biological interactions and thus are richer in biodiversity. Increased biodiversity further enhances chances of bioprospecting, that is, searching for new chemicals and plant-based products for the welfare of humanity. Guo (2000) viewed AFS as an excellent land-use practice for biodiversity conservation and sustainable development in the tropics. AFS also helps in reducing the dependence of local peasants or farmers on the natural resources of the protected areas—national parks and sanctuaries (Murniati et al., 2001).

1.2.1.5 Enhancing Food Security and Alleviating Poverty

Trees are the sources of a number of valuable and marketable products. Agroforestry helps in providing an opportunity to marginal and low-income farmers to improve their livelihood by marketing these products as household food, medicine, small timber, domestic wood supply, fiber, or fuel. It thus provides both food and economic security to farmers, particularly in the tropics (Garrity, 2004). Recently, agroforestry has been suggested to play a central role in improving food security, alleviating poverty, and natural resource management, particularly in east and central African regions (Ashley et al., 2006; Jama et al., 2006a; Leakey et al., 2006). Agroforestry adoption has also been viewed as a viable option to provide support in the form of value-added products (i.e., food, medicine, timber), livelihood, and income to HIV- or AIDS-affected communities, particularly in very poor regions of the world like sub-Saharan Africa (Garrity, 2004, 2006; Leakey et al., 2006). Leakey et al. (2006) advocated agroforestry as a new approach for sustainable rural development. However, much needs to be done in this direction to include underutilized and medicinal tree species, which can offer good economic returns to the farmers in addition to providing other benefits of AFS.

1.2.1.6 Carbon Sequestration and Greenhouse Gas Mitigation

World over, scientists are facing the challenging problem of loss of carbon (C) stocks in the terrestrial ecosystems and increase in the levels of green house gases in the atmosphere. AFS have a great scope in sequestering aboveground and belowground (soil) C and help in mitigating the greenhouse effect by reducing C emissions (Dixon et al., 1994; Wang and Feng, 1995; Batjes and Sombroek, 1997; Pandey, 2002; Albrecht and Kandji, 2003; Montagnini and Nair, 2004; Lal, 2005). Trees can store C both *ex situ* (products) as well as *in situ* (biomass and soil) and are considered as effective C sinks (Montagnini and Nair, 2004). Though the exact potential of agroforestry trees for this purpose is largely unknown, yet some preliminary reports are available. AFS, particularly in the tropics, can even ease the environmental degradation caused by deforestation and reduce the pressure on natural forests (Dixon, 1995). He estimated that AFS on 1 ha of land could compensate the loss caused by 5–20 ha of deforestation. Recently, agroforestry practices in humid tropics have been reported to reduce soil emission of N_2O and CO_2 and increase the CH_4 sink strength when compared to agricultural systems (Mutuo et al., 2005). However, extensive research is required to quantify exactly this underexploited C sequestration potential of AFS, in general, and under specific management patterns.

Similar to the impact on global C balance, AFS can also ameliorate the greenhouse gas, particularly nitrous oxide (N_2O), emission. Liang and Thevathasan (2003) demonstrated that intercropping of *Populus* into AFS reduced N_2O emissions by 0.69 kg hm^{-2} a^{-1}. Thevathasan and Gordon (2004) reported that trees intercropped in AFS reduce the N_2O emissions due to reduced fertilizer use and efficient N cycling. However, the mitigation of greenhouse gas emission under AFS varies greatly with the tree species used and depends on the C:N ratio, polyphenol content, and protein-binding capacity (Millar and Baggs, 2004).

1.2.1.7 Phytoremediation and Environmental Clean-Up

Garrett and Buck (1997) suggested that AFS including trees as intercrops, riparian plantations, shelterbelts, and windbreaks have a good potential for cleaning up the contaminated soils. Schultz et al. (1995) reported that multispecies riparian buffer strips are very effective in stopping sediments and flow of runoff nutrients, pesticides, and fertilizers. In this direction, short-rotation woody trees like *Populus*, *Salix*, *Eucalyptus*, *Pinus*, and *Acacia* spp. incorporated under AFS hold a great potential for remediation of soil contaminated with heavy metals, pesticides, herbicides, and organic compounds (Rockwood et al., 2004).

1.2.2 NEGATIVE EFFECTS

A number of negative interactions such as shade, competition, allelopathy, harboring of harmful pests, and threat from invasive potential of trees prevail under AFS (Table 1.1).

1.2.2.1 Shading Effect

Although the reports available in the literature concerning the effects of shade or competition for light vary greatly, shading by agroforestry trees generally has negative effects on crop productivity. However, it depends on soil type, climate, crop or tree species, and the management practices (Ong and Huxley, 1996; Huxley, 1999). On the other hand, shading may have either no (Gillespie et al., 2000) or even positive effect on associated crops under a given set of environmental conditions. For example, shading by trees increased forage yield (Lin et al., 1999), reduced pest density in intercrops (Stamps and Linit, 1998), and decreased weed density and increased maize yield (Jama et al., 1991). However, it depends on the soil fertility status, especially the N content.

The physiological mechanism by which shading affects crop productivity could be the interception of photosynthetically active radiations (PAR) and thus the quantity and quality of light reaching crops (Chirko et al., 1996), and differences in carbon fixation pathways, that is, C_3 or C_4

plants (Jose et al., 2004). Pillar et al. (2002) demonstrated the shading effect of *Eucalyptus* spp. on grass communities and indicated that differences occurred in cover abundance of C_3 and C_4 species. Increased shading by tree canopy reduced the cover abundance of C_4 species and increased the number of C_3 species (Pillar et al., 2002).

1.2.2.2 Resource Competition

Competition for essential growth substances including water and nutrients is one of the most severe negative effects that trees can have on crops (Nair, 1993). If improperly selected and managed, trees in AFS strongly compete with crops for light, resources, shade, and water and thus can have a devastating effect on crop yields (García-Barrios, 2003). However, it largely depends on the climate, soil type, management practices, tree–crop combination, and fertility patterns. Its intensity and type varies with the geographical region, that is, tropical or temperate area, or arid, semiarid, or wet type (Nair, 1993; Huxley, 1999). For example, in a humid region where there is enough moisture, competition normally exists for light or nutrients, whereas in a semiarid or arid zone, the trees and crops compete for moisture and nutrients, though there is adequate light (Nair, 1993; Huxley, 1999). The choice of tree component is very important since studies have shown that fast-growing tree species are not good for hedgerow species (Broadhead et al., 2003; De Costa and Surenthran, 2005). However, trees in agroforestry, particularly in dry and semiarid regions, can be managed to optimize their water use and productivity by root and shoot pruning to decrease underground competition, avoiding fast growing evergreen species like *Eucalyptus*, and opting for deciduous tree species that use little water during dry seasons (Ong et al., 2007). Further, selection of tree species should be done keeping in mind the phenology so that there is no extra burden on the water regime, particularly during the dry seasons. For example, trees like *F. albida* should be avoided in dry areas as they produce leaves and branches during dry season and demand more water (Ong et al., 2007). The severity of the competition further depends on the architecture of the tree and crop root systems. A complementarity between tree and crop roots is essential to minimize resource competition and maximize resource use (Huxley, 1999). Cannell et al. (1996) opined that tree incorporation in crops is beneficial only if the trees can capture resources not used by crops. A number of earlier studies have reported that removal of root competition significantly increases yield (Corlett et al., 1992; De Costa and Surenthran, 2005). Management of competition between tree and crops is very important, especially under SAFS; and if properly managed, it can lead to a successful system.

1.2.2.3 Allelopathy

Allelopathy is another negative interaction between trees and crops that operates under SAFS. It mediates through the release of chemicals by one plant into the surrounding environment and retards or suppresses the growth of other plants. Allelopathy causes crop losses under conditions of unsuitable tree–crop combination, for example, eucalypts (*Eucalyptus* sp.), poplar (*Populus deltoides* Bartr. ex Marsh), and black walnut (*Juglans nigra* L.) planted under SAFS. Allelochemicals—the chemicals responsible for allelopathic effects—may be present in any part of the tree (Rice, 1984). However, their effects under field conditions are a function of their bioactive concentrations in the soil, and depend upon prevailing environmental conditions (Rice, 1984). Studies on allelopathy are available from both under temperate as well as tropical AFS (Rao et al., 1998; Rizvi et al., 1999; Jose et al., 2004).

Allelopathic implications of trees in AFS have been discussed in detail in Chapter 3 of this book.

1.2.2.4 Exotic Invasive Species

One of the major problems linked with agroforestry trees is that some of them, particularly exotics, have a tendency to become weedy and invade other ecosystems. Such trees when incorporated in AFS can negate the perceived economic returns (Richardson et al., 2004). Further, they escape into the nearby ecosystems, outcompete the native vegetation, and threaten native plant communities. It has been estimated that of the 2000 trees frequently used under agroforestry programs, at least 135 acquired weedy character under some situations, whereas 25 were frequently weedy, which included *L. leucocephala* and *Prosopis* sp. (Richardson, 1998). Recently, in a review of invasive trees by CAB International, 194 species used in agroforestry have been classified as invasive. Prominent agroforestry tree species such as *Pinus*, *Eucalyptus*, *Acacia, Sesbania, Crotalaria,* and *Senna* also possess weedy character outside their natural range (Richardson et al., 2004). A number of *Pinus* species are serious invaders and colonizers in the southern hemisphere. Several species of *Acacia* introduced from Australia for agroforestry purposes have become invasive (Richardson et al., 2004). The reasons for their acquiring weedy habits include fast growth rate, remarkable adaptability in the alien environment, rapid ability to colonize, high reproductive rate, and ability to outcompete or suppress other plants. *L. leucocephala*—one of the most important agroforestry tree species—is also a serious invader and a noxious weed in 20 countries (Hughes, 2006). It is a prolific seed producer and forms its own monospecific thickets that are difficult to eradicate. It has also been included in the list of 100 worst invaders of the world (Hughes, 2006). Thus, there is an urgent need to predict and assess the risks of agroforestry tree species becoming weedy before their introduction and widespread promotion into new environment; however, it is very challenging.

1.3 CONCLUSIONS AND WAY FORWARD

From the above discussion, it is pertinent that agroforestry has a great scope and potential in terms of social, economic, and environmental services. Bene et al. (1977) rightly pointed out that agroforestry has a great potential to improve the life of people within a reasonably short time, particularly in the developing countries. McNeely (2004) advocated AFS as a unique ecological system that favours both crop productivity and biodiversity conservation, and thus is a best example of ecoagriculture. Garrity (2006) viewed agroforestry as a science and practice in achieving the United Nations MDG eradicating hunger and poverty, thus improving the livelihood of farmers and advancing health and nutrition. However, it depends on the complementarity between negative and positive interactions (effects) of AFS and minimization of negative concerns. However, the problem is where and how to integrate these strategies to achieve a balance between potential conservation benefits, on the one hand, and the sustainable rural development, on the other (van Noordwijk et al., 1997). In other words, there is a need to develop agroforestry as an ecologically sustainable land-use system that involves interplay between various positive and negative interactions leading to human development, conservation, management, and development of natural resources in an efficient manner. However, to achieve these goals, further research is required on the following lines:

1. Careful evaluation of various social, economic, environmental, biophysical, and developmental concerns linked with incorporation of trees into AFS and the diversification of existing AFS into new agroecological regions, particularly degraded lands.
2. Integration of environmental services and concerns linked with tree crops with the research and development initiatives to have an ecologically sustainable AFS.
3. Identification, formulation, development, and adoption of new technologies involving native multipurpose tree species keeping in mind the perception and needs of local stakeholders.
4. Developing, evaluating, and promoting innovative synergistic agroforestry technologies that provide multiple environmental benefits in synergism with economic returns.
5. Innovative AFS designs for large-scale biodiversity conservation including birds, animals, and wildlife.
6. Incorporating indigenous knowledge into the existing and future AFS to enhance overall sustainability.

7. Need of rigorous testing for the invasive and weedy nature of a tree species before incorporation into AFS.
8. Development of efficient management plans for potential invasive agroforestry tree species.
9. Selection and promotion of native tree crop species with multipurpose roles to prevent the introduction and spread of potential invasive tree species.
10. Developing strategies and programs to foster a more efficient relationship between researchers, entrepreneurs, and local stakeholders and providing access to agroforestry technology and benefits to all stakeholders.

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2 Tree–Crop Interactions: Lessons from Temperate Alley-Cropping Systems

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CONTENTS

2.1 INTRODUCTION

Individuals and institutions in the world's temperate regions are increasingly taking notice of the science and art of alley cropping. This is due in part to growing concerns over the long-term sustainability of intensive monocultural systems. In the temperate context, alley cropping involves the planting of timber, fruit, or nut trees in single or multiple rows on agricultural lands, with crops or forages cultivated in the alleyways (Garrett and McGraw, 2000). Major purposes of this type of agroforestry system include production of tree or wood products along with crops or forage; improvement of crop or forage quality and quantity by enhancement of microclimatic conditions; improved utilization and recycling of soil nutrients for crop or forage use; control of subsurface water levels; and provision of favorable habitats for plant, insect, or animal species beneficial to crops or forage (USDA, 1996; Garrett and McGraw, 2000).

As an association of plant communities, alley cropping is deliberately designed to optimize the use of spatial, temporal, and physical resources by maximizing positive interactions (facilitation) and minimizing negative ones (competition) between trees and crops (Jose et al., 2000a). For example, trees in these systems are capable of improving site-growing conditions for crops in terms of soil and microclimate modification, thus improving productivity (Wei, 1986; Wang and Shogren, 1992). Trees are also capable of capturing and recycling lost soil nutrients (Nair, 1993; Palm, 1995; Rowe et al., 1999), and are thus a potential moderating factor in groundwater pollution caused by leaching of nitrates and phosphates (Williams et al., 1997; Garrett and McGraw, 2000). Trees also provide producers an opportunity to utilize idle growing area during the early stages of tree stand establishment, thus providing a more immediate return on land investment (Williams et al., 1997). Likewise, government incentive programs promote tree planting on private lands (Zinkhan and Mercer, 1997; Garrett and McGraw, 2000). In addition, trees on agricultural lands offer landowners the possibility of accruing carbon credits via the sequestration of stable carbon stock, an added incentive for adopting alley cropping (Dixon, 1995; Williams et al., 1997; Sampson, 2001). Moreover, new technologies for agroforestry modeling, such as the WaNuLCAS (Water, Nutrients, Light Capture in Agroforestry Systems) model (van Noordwijk and Lusiana, 1999, 2000) and the SBELTS (ShelterBELT and Soybeans) model (Qi et al., 2001), are shedding light on the potential for applying agroforestry techniques in new locales. However, trees also compete with plants for available light, water, nutrients, and other resources, which can negatively impact productivity. Thus, more understanding is needed of tree–crop interactions in temperate settings to design agroforestry systems that make best use of the various resources at hand to increase both productivity and sustainability. This is the subject of this chapter.

2.2 ALLEY CROPPING IN THE TEMPERATE REGIONS

Alley cropping, like any other agricultural practice, has been shaped by the environmental and sociocultural contexts in which it has been applied. In the temperate zones, where agriculture has generally been driven by high-input, large-scale production and, more recently, on management for environmental sustainability, alley cropping has naturally tended to mirror these practices. Although much of its foundation has been derived from tropical zone applications, temperate zone alley cropping nevertheless remains a distinct practice. Generally, trees in temperate systems are planted at comparatively wider spacings than those in the tropics, to allow for mechanical cultivation of crops in the strips or alleys (Williams et al., 1997; Gillespie et al., 2000). In addition, temperate systems do not typically rely on the direct reintroduction of prunings from trees or shrubs to maintain soil fertility and productivity (Garrett and McGraw, 2000). To provide a better understanding of temperate alley cropping, we first examine how it is practiced in various regions of the world.

In the mid-western United States and parts of Canada (e.g., Ontario), many of the alley-cropping systems in use are based on the production of high-value hardwoods (Garrett and McGraw, 2000). Perhaps the most widely planted species in such systems is black walnut (*Juglans nigra* L.) (Williams et al., 1997; Garrett and McGraw, 2000; Jose et al., 2000a). Companion crops that are typically grown with black walnut include winter wheat (*Triticum aestivum* L.), barley (*Hordeum vulgare* L.), corn (*Zea mays* L.), sorghum (*Sorghum bicolor* L.), and forage grasses. Black walnut systems have been useful in shedding light on various biophysical parameters, including water and nutrient competition, crop productivity, and crop response to juglone, an allelopathic compound (Williams et al., 1997; Jose and Gillespie, 1998; Garrett and McGraw, 2000; Jose et al., 2000a).

Fruit and nut production are also important components of alley cropping in various parts of North America. For example, in southern Canada, producers are growing vegetables and other crops among their fruit and nut trees during orchard establishment (Williams and Gordon, 1992). For example, peach (*Prunus persica* L.) trees have been intercropped with tomatoes (*Lycopersicon* spp.), pumpkins (*Cucurbitaceae* spp.), strawberries (*Fragaria* spp.), sweet corn (*Z. mays* L. var. *rugosa* Bonaf.), and other vegetables. Similarly, chestnut (*Castanea* spp.) trees have been intercropped with soybeans, squash (*Cucurbitaceae* spp.), and rye (*Secale cereale* L. subsp. *cereale*) (Williams and Gordon, 1992). Other species such as red oak (*Quercus rubra* L.), Norway spruce

(*Picea abies* L. Karrst.), White ash (*Fraxinus americana* L.), White cedar (*Chamaecyparis thyoides* L.), Red maple (*Acer rubrum* L.), and Carolina poplar (*Populus canadensis* Moench.) have been intercropped with soybeans, corn, and barley (Williams and Gordon, 1992).

Systems involving softwood production are more important in the southern United States and have involved silvopastoral systems for cattle grazing, and alley-cropping systems for forage production (Mosher, 1984; Zinkhan and Mercer, 1997). Pine species such as loblolly pine (*Pinus taeda* L.), longleaf pine (*P. palustris* Mill.), and slash pine (*P. elliottii* Engl.) have been intercropped with forage crops such as crimson clover (*Trifolium incarnatum* L.), subterranean clover (*T. subterraneum* L.), ryegrass (*Lolium perenne* L.), bahiagrass (*Paspalum notatum* Flugge.), coastal Bermuda grass (*Cynodon dactylon* L. Pers.), tall fescue (*Festuca arundinacea* Schreb.), and other species (Davis and Johnson, 1984; Clason, 1995; Morris and Clason, 1997; Zinkhan and Mercer, 1997). Pines have also been intercropped with row crops such as cotton (*Gossypium* spp.), peanuts (*Arachis hypogaea* L.), soybean, corn, wheat, and watermelon (*Citrullus lanatus* Thumb. Monsaf.) (Zinkhan and Mercer, 1997; Allen et al., 2001; Ramsey and Jose, 2001). Pecan (*Carya illinoensis* L.), an important nut-bearing species, has been intercropped with soybeans, grains, squash, potatoes (*Solanum tuberosum* L.), peaches, raspberries (*Rubus* spp.), and other crops (Nair, 1993; Williams et al., 1997; Zinkhan and Mercer, 1997; Cannon, 1999; Long and Nair, 1999; Reid, 1999; Ramsey and Jose, 2001).

Other species of current or potential application to North American alley cropping include trees such as honeylocust (*Gleditsia triacanthos* L.), basswood (*Tilia* sp.), silver maple (*Acer saccharinum* L.), oak (*Quercus* spp.), ash (*Fraxinus* spp.), poplar (*Populus* spp.), birch (*Betula* spp.), alder (*Alnus* spp.), and black locust (*Robinia pseudoacacia* L.), as well as speciality crops such as ginseng (*Panax quinquefolium* L.) and goldenseal (*Hydrastis canadensis* L.) (Garrett and McGraw, 2000; Miller and Pallardy, 2001).

In temperate regions of South America (e.g., southern Chile and Argentina), silvopastoral systems are a prevalent form of agroforestry. These may involve tree species such as Radiata pine (*Pinus radiata* D. Don.), nire (*Nothofagus antarctica* G. Foster Oerst.), and lenga (*N. pumilio* Poepp. & Endl. Krasser) (Somlo et al., 1997; Amiotti et al., 2000). Such species may be intercropped with forage grasses or legumes such as subclover (Balocchi and Phillips, 1997).

Alley cropping in the Australian or New Zealand sector has tended to focus on large-scale timber production with forage production and grazing of sheep or cattle underneath (Mosher, 1984; Hawke and Knowles, 1997; Moore and Bird, 1997). Common tree species in these systems include Radiata pine and various eucalypts (e.g., *Eucalyptus accedens* W. Fitzg., *E. globulus* Labill., *E. maculata* Hook, *E. saligna* Sm.), and forage grasses include ryegrass, white clover (*Trifolium* spp.), and other species (Hawke and Knowles, 1997; Moore and Bird, 1997). Planting of poplar with row and vegetable crops has also been reported in Australia (Garrett and McGraw, 2000).

Various systems have also been developed in Europe over the years. English walnut (*Juglans regia* L.), for example, is a common species for intercropping systems, which might include alfalfa or forage grasses (Dupraz et al., 1998; Mary et al., 1998; Paris et al., 1998; Pini et al., 1999). In addition, poplar has been grown with vegetable and row crops, as reported for the former Yugoslavia area (FAO, 1980; Garrett and McGraw, 2000). Another tree–crop combination of scientific interest is hazel (*Corylus avellana* L.), interplanted with cocksfoot (*Dactylis glomerata* L.) (de Montard et al., 1999). Lastly, forest grazing, an ancient silvopastoral system in which thinned stands of species such as Scots pine (*P. sylvestris* L.) and European larch (*Larix decidua* Mill.) are oversown with grasses and grazed by sheep and cattle, is also reported to be in use in various parts of Europe (Dupraz and Newman, 1997).

Agroforestry is also popular in China, and its practice dates back many centuries (Wu and Zhu, 1997). Various types of intercropping systems are in use today, with biomass and nut–tree intercropping systems being common. Intercropping systems based on paulownia (*Paulownia* spp.), a fast-growing species, are popular (Wu and Zhu, 1997). Scientific study of this species has focused on paulownia–winter wheat intercrops in north central China (Chirko et al., 1996). Planting of poplar with vegetable and row crops has also been reported in China (Kai-fu et al., 1990; Garrett and McGraw, 2000).

Alley cropping is also practiced in the mid-elevation regions of the Himalaya mountains of India, with fruit trees and other species (Nair, 1993). For example, citrus is grown with gram (*Cicer arietinum*) and winter vegetables, and beans and peas are grown under dwarf-apple (*Pyrus* sp.), peach, plum (*Prunus domestica* L.), apricot (*P. armeniaca* L.), and nectarine (*P. persica* L.) (Tejwani, 1987; Nair, 1993). These and other systems point to the uniqueness and complexity of tree–crop interactions in each geographic location.

2.3 INTERACTIONS BETWEEN TREES AND CROPS

A guiding principle of agroforestry is that productivity can increase if trees capture resources that are underutilized by crops (Cannell et al., 1996). Thus, alley cropping may be viewed as a complex series of tree–crop interactions guided by utilization of light, water, soil, and nutrients. An understanding of the biophysical processes and mechanisms involved in the mutual utilization of these resources is essential for the development of ecologically sound agroforestry systems (Ong et al., 1996). The following section discusses important above- and belowground interactions occurring between trees and crops in temperate alley-cropping systems.

2.3.1 Aboveground Interactions

2.3.1.1 Light Availability, Competition, and Facilitation

Light is the major aboveground factor affecting photosynthesis and biological yields within agroforestry systems. Trees and crops capture light in the form of photosynthetically active radiation, or PAR (400–700 nm wavelength). The degree of light capture is dependent on the fraction of incident PAR that each species intercepts and the efficiency with which the intercepted radiation is converted by photosynthesis (Ong et al., 1996). These factors, in turn, are influenced by time of day, temperature, CO_2 level, species combination, canopy structure, plant age and height, leaf area and angle, and transmission and reflectance traits of the canopy (Brenner, 1996; Garrett and McGraw, 2000).

The effect of light interception on biological productivity has been widely studied (e.g., Monteith et al., 1991; Monteith, 1994; Chirko et al., 1996; de Montard et al., 1999; Gillespie et al., 2000). When water or nutrients are not limiting factors, biomass production may be limited by the amount of PAR that tree and crop foliage can intercept (Monteith et al., 1991; Monteith, 1994). Chirko et al. (1996), for example, in their study of a *Paulownia*–winter wheat intercropping system in northern China found that low PAR levels resulting from overhead shading significantly reduced yield of winter wheat near tree rows (Figure 2.1). However, they also found that, with a wide interrow spacing, late leaf flush, north–south tree arrangement, and long clear boles, wheat was able to receive higher levels of PAR in the morning and afternoon. Lin et al. (1999), in a greenhouse experiment on the effects of shade on forage crop production, found that shading significantly reduced the mean dry weights (MDW) of various warm-season grasses and legumes (Table 2.1).

On the other hand, studies have pointed to minimally negative or even positive effects (facilitation) of moderate shading on crop growth in some cases. In theory, crop photosynthesis levels may remain unchanged under shade, provided that the understory species becomes "light saturated" at relatively low levels of radiation (Wallace, 1996). Lin et al. (1999), in the same greenhouse study cited earlier, found that 50% shading did not significantly reduce MDW of cool-season grasses. Interestingly, two native warm-seasons legumes, Hoary Tick-clover and Panicled Tick-clover, exhibited shade tolerance and had significantly higher MDW at 50% and 80% shade than in full sunlight (Lin et al., 1999; Garrett and McGraw, 2000). These authors also reported that total crude protein content of some of the forage species was greater under 50% and 80% shade than in full sun (Table 2.2). It is likely that shading has caused a reduction in cell size, thereby concentrating nitrogen content per cell as speculated by Kephart and Buxton (1993).

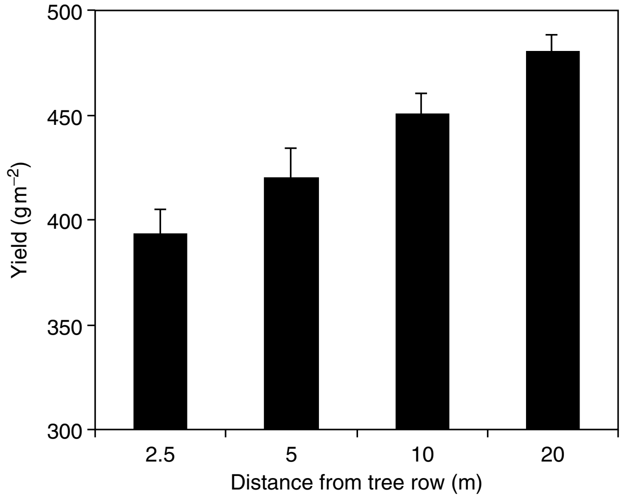

FIGURE 2.1 Winter wheat grain yield as influenced by distance from the tree row in a *Paulownia*–winter wheat alley-cropping system in northern China. (Adapted from Chirko, C.P., M.A. Gold, P.V. Nguyen and J.P. Jiang, *For. Ecol. Manage.*, 83, 171, 1996.)

Research by Jose (1997) and Gillespie et al. (2000) indicated that shading did not have a major influence on the yield of maize in two mid-western United States alley-cropping systems with black walnut and red oak. These researchers found that, in general, the eastern-most row of maize in the black walnut alley cropping received 11% lower PAR than the middle row (Figure 2.2). Shading was greater in the red oak alley cropping because of higher canopy leaf area, where a 41% reduction was observed for the eastern row. Similarly, western rows were receiving 17% and 41% lower PAR than the middle rows in the black walnut and red oak systems, respectively. Irrespective of the shading, no apparent yield reduction was observed when belowground competition for nutrients and water was eliminated through trenching and polyethylene barriers.

2.3.1.2 Microclimate Modification

The presence of trees in an alley-cropping system modifies site microclimate in terms of temperature, relative humidity, and wind speed, among other factors. Figure 2.3 summarizes the microclimatic modifications that occur when trees are introduced into an agricultural field. Serving as windbreaks, trees slow the movement of air and thus in general promote cooler, moister site conditions. Temperature reductions in the alleys can help to reduce heat stress of crops by lowering rates of foliar evapotranspiration and soil evaporation. Together, these factors have a moderating effect on site microclimate.

Crops such as cotton and soybean have higher rates of field emergence when grown at moderate outdoor temperatures. For example, Ramsey and Jose (2001), in their study of a pecan–cotton alley-cropping system in northwest Florida, observed earlier germination and higher survival rate of cotton under pecan canopy cover, due to cooler and moister soil conditions. Similarly, a study in Nebraska showed earlier germination, accelerated growth, and increased yields of tomato (*Lycopersicon esculentum* L.) and snap bean (*Phaseolus vulgaris* L.) under simulated narrow alleys compared with wider alleys (Bagley, 1964; Garrett and McGraw, 2000). In addition, studies on *Paulownia*–wheat intercropping in temperate China showed increased wheat quality due to enhanced microclimatic conditions (Wang and Shogren, 1992). Wind speed was also substantially reduced under a Radiata pine silvopastoral system in New Zealand due to increased tree stocking (Hawke and Wedderburn, 1994).

TABLE 2.1
Total Aboveground Dry Weight of 30 Forages under Three Levels of Shade during 1994 and 1995 at New Franklin, Missouri, U.S.A.

Species	Scientific Name	Full Sun (g)	50% Shade (g)	80% Shade (g)
Introduced cool-season grasses				
Kentucky bluegrass	*Poa pratensis* L.	12.5 a	12.3 a	8.0 b
Orchardgrass "Benchmark"	*Dactylis glomerata* L.	13.8 a	11.7 a	6.4 b
Orchardgrass "Justus"	*Dactylis glomerata* L.	11.7 a	11.2 a	9.5 a
Ryegrass "Manhattan II"	*Lolium perenne* L.	12.7 a	11.1 ab	8.6 b
Smooth bromegrass	*Bromus inermis* Leyss.	9.6 a	12.0 a	9.5 b
Tall Fescue "KY31"	*Festuca arundinacea* Schreb.	13.3 a	16.2 a	8.0 b
Tall Fescue "Martin"	*Festuca arundinacea* Schreb.	12.4 a	11.8 a	6.0 b
Timothy	*Phleum pratense* L.	10.2 a	9.0 a	5.5 b
Introduced warm-season grasses				
Bermuda grass	*Cynodon dactylon* (L.) Pers.	56.1 a	37.0 b	8.6 c
Native warm-season grasses				
Big Bluestem	*Andropogon gerardii* Vitman	45.3 a	33.4 b	17.8 c
Buffalograss	*Buchloe dactyloides* (Nutt.) Engelm.	29.9 a	13.7 b	6.1 b
Indiangrass	*Sorghastrum nutans* (L.) Nash	42.3 a	30.2 b	16.9 c
Switchgrass	*Panicum virgatum* L.	79.5 a	57.6 b	26.5 c
Introduced cool-season legumes				
Alfalfa "Cody"	*Medicago sativa* L.	6.2 a	5.3 ab	3.8 b
Alfalfa "Vernal"	*Medicago sativa* L.	9.4 a	7.1 b	4.2 c
Alsike clover	*Trifolium hybridum* L.	17.0 a	9.8 b	5.4 c
Berseem clover	*Trifolium alexandrinum* L.	16.0 a	7.0 b	2.9 c
Birdsfoot trefoil hybrid "Rhizomatous"	*Lotus corniculatus* L.	15.0 a	9.8 b	5.3 c
Birdsfoot trefoil "Nocern"	*Lotus corniculatus* L.	19.6 a	12.6 b	6.0 c
White clover	*Trifolium repens*	16.0 a	13.0 a	9.5 b
Red clover	*Trifolium pratense* L.	19.9 a	12.1 b	5.9 c
Introduced warm-season legumes				
Korean lespedeza	*Kummerowia stipulacea* (Maxim.) Mankino	42.7 a	29.7 b	13.5 c
Korean lespedeza "Summit"	*Kummerowia stipulacea* (Maxim.) Mankino	34.1 a	12.7 b	7.3 c
Striate lespedeza "Kobe"	*Kummerowia striata* (Thumb.) Schindler	28.5 a	23.6 a	14.7 b
Serecia lespedeza	*Lespedeza virginica* L.	55.9 a	37.9 b	24.6 c
Native warm-season legumes				
Hoary Tick-clover	*Desmodium canescens* L.	16.8 b	22.2 a	21.9 a
Panicled Tick-clover	*Desmodium paniculatum* L.	21.0 b	26.2 a	23.0 ab
Hog peanut (overwintered)	*Amphicarpaea bracteata* L.	8.8 b	28.9 a	31.0 a
Slender lespedeza (overwintered)	*Lespedeza virginica* L.	18.7 a	19.4 a	9.6 a

Source: Adapted from Lin, C.H., R.L. McGraw, M.F. George, and H.E. Garrett, *Agroforestry Syst.*, 44, 109, 1999.

Note: Means followed by the same letter within a row do not differ significantly from each other (Tukey's studentized range test, $\alpha = 0.05$).

TABLE 2.2

Percent Crude Protein (CP%) and Total Crude Protein/Pot (TCP) of Selected Grasses and Legumes When Grown under Three Levels of Shade during 1994 and 1995 at New Franklin, Missouri, U.S.A.

	CP%			TCP (g)		
Species	**Full Sun**	**50% Shade**	**80% Shade**	**Full Sun**	**50% Shade**	**80% Shade**
Introduced cool-season grasses						
Kentucky bluegrass	20.3 b	20.7 b	22.7 a	2.45 A	2.58 A	1.57 B
Orchardgrass "Benchmark"	12.6 c	15.7 b	19.6 a	1.80 A	1.84 A	1.19 B
Orchardgrass "Justus"	19.8 a	16.7 a	18.5 a	1.60 A	1.92 A	1.79 A
Ryegrass "Manhattan II"	15.3 b	16.0 b	18.5 a	1.74 A	2.06 A	1.62 A
Smooth bromegrass	16.7 c	18.1 b	20.2 a	1.64 A	2.25 A	1.94 AB
Tall Fescue "KY31"	14.0 b	15.0 b	18.1 a	1.83 B	2.43 A	1.43 C
Tall Fescue "Martin"	14.3 b	15.5 b	18.5 a	1.75 A	1.84 A	1.12 B
Timothy	15.4 c	17.6 b	20.4 a	1.60 A	1.59 A	1.12 A
Introduced cool-season legumes						
Alfalfa "Cody"	19.4 a	19.9 a	19.4 a	1.49 A	1.48 A	1.00 A
White clover	20.1 a	20.6 a	19.9 a	2.49 A	2.03 A	1.23 B
Introduced warm-season legumes						
Striate lespedeza "Kobe"	13.2 a	13.0 a	12.5 a	3.34 A	2.65 B	1.56 C
Native warm-season legumes						
Slender lespedeza	11.0 a	10.5 a	10.8 a	2.04 A	2.04 A	1.04 A
Panicled Tick-clover	11.6 b	11.7 b	12.9 a	2.57 B	3.53 A	3.38 A
Hoary Tick-clover	13.0 a	13.2 a	12.8 a	2.19 B	2.98 A	2.88 A
Hog peanut	9.1 ab	8.7 b	9.7 a	0.80 B	2.51 A	2.97 A

Source: Adapted from Lin, C.H., R.L. McGraw, M.F. George, and H.E. Garrett, *Agroforestry Syst.*, 53, 269, 2001.

Note: Means followed by the same letter within a row do not differ significantly from each other (Tukey's studentized range test, $\alpha = 0.05$).

2.3.1.3 Weed Density

The presence of a tree canopy alters the growing environment for any species that may find its way into the understory, including weeds. The abundance of weed species in the environment ensures that some species will likely invade an intercropped area, and, through natural selection, adapt to the spectrum of existing growing conditions present. Generally, this condition results in a change in weed density or weed species composition, depending on distance from tree component. Ramsey and Jose (2001), in their study of a mature pecan–cotton intercrop in Florida, observed that, unlike monocrop plots, plots under pecan trees were heavily infested with Asiatic dayflower (*Commelina communis* L.), an exotic, summer annual that appeared to be shade loving. The presence of this weed was attributed to the nutrient-rich soil of the understory, as well as the moist conditions of the soil due to shading. In this case, weeds (e.g., Bermuda grass) that were prevalent in the cotton monoculture were less prevalent within the alleys of the intercrop due to niche specificity.

2.3.1.4 Insect Density

Plant–insect interactions are another important factor in the design of agroforestry systems, as variations in tree–crop combinations and spatial arrangements have been shown to have an effect on insect population density (Vandermeer, 1989; Altieri, 1991; Nair, 1993). According to Stamps and Linit (1997), agroforestry is a potentially useful technology for reducing pest problems because tree–crop combinations provide greater niche diversity and complexity than polycultural systems of

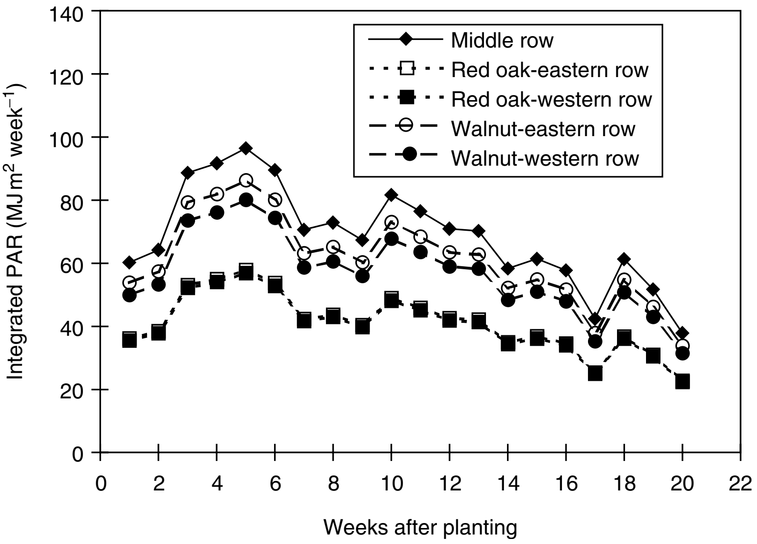

FIGURE 2.2 Seasonal variation in weekly incident PAR (June 1 through October 15, 1996) at three different locations (eastern row, middle row, and western row) in black walnut and red oak alley-cropping systems in mid-western United States. (Adapted from Jose, S., *Interspecific Interactions in Alley Cropping: The Physiology and Biogeochemistry*, Ph.D. Dissertation, Purdue University, West Lafayette, IN, 1997.)

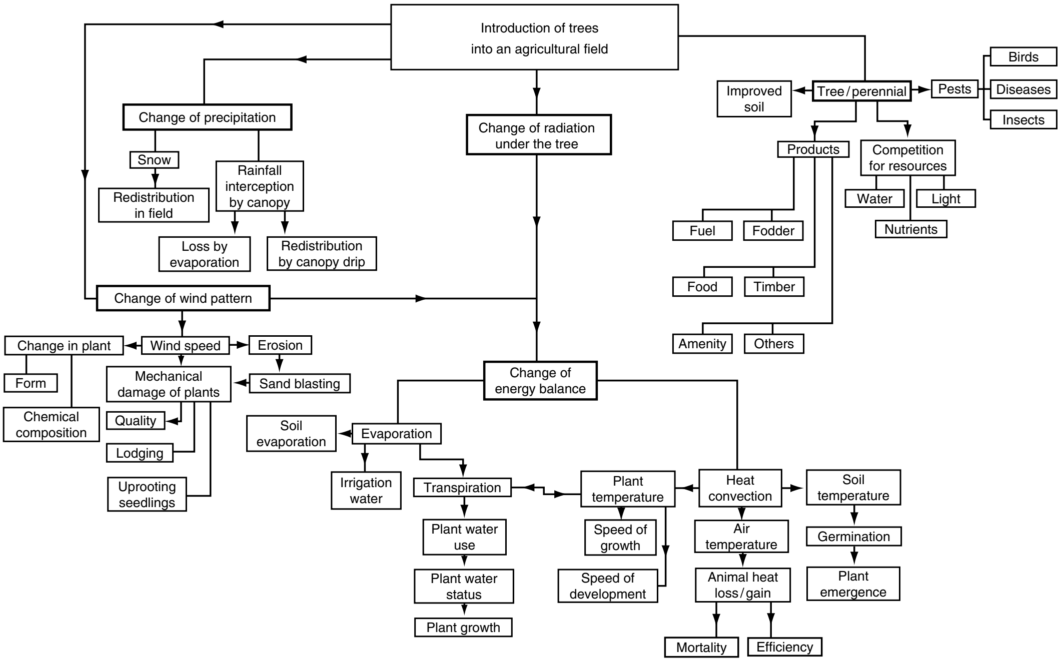

FIGURE 2.3 The changes in a predominantly agricultural-based landscape following introduction of trees. The flow diagram shows causal relationships by lines with arrows and subdivisions by lines without arrows. (From Brenner, A.J., *Tree-Crop Interactions: A Physiological Approach*, C.K. Ong and P. Huxley, eds., CAB International, Wallingford, UK, 1996. With permission.)

annual crops. This effect may be explained in one or more of the following ways: (1) wide spacing of host plants in the intercropping scheme may make the plants more difficult to find by herbivores; (2) one plant species may serve as a trap-crop to detour herbivores from finding the other crop; (3) one plant species may serve as a repellent to the pest; (4) one plant species may serve to disrupt the ability of the pest to efficiently attack its intended host; and (5) the intercropping situation may attract more predators and parasites than monocultures, thus reducing pest density through predation and parasitism (Root, 1973; Vandermeer, 1989).

Various studies have shed light on plant–insect interactions. Studies with pecan, for example, have looked at the influence of ground covers on arthropod densities in tree–crop systems (Bugg et al., 1991; Smith et al., 1996). Bugg et al. (1991) observed that cover crops (e.g., annual legumes and grasses) sustained lady beetles (*Coleoptera: Coccinellidae*) and other arthropods that may be useful in the biological control of pests in pecan (Bugg et al., 1991; Garrett and McGraw, 2000). However, Smith et al. (1996) found that ground cover had little influence on the type or density of arthropods present in pecan. Although beyond the scope of this discussion, the competitive activity of belowground pests is another important consideration (Ong et al., 1991).

2.3.2 BELOWGROUND INTERACTIONS

2.3.2.1 Soil Structure Modification

Trees play an important role in soil structure and subsequent soil-holding capacity. The presence of trees on farmlands can improve the physical conditions of the soil—permeability, aggregate stability, water-holding capacity, and soil temperature regimes—the net effect of which is a better medium for plant growth (Figure 2.3; Nair, 1987). In addition, various factors work to protect soil from the damaging effects of rain and wind erosion. Tree canopies, for example, intercept and rechannel rainfall and wind in patterns that tend to be less damaging to soil (del Castillo et al., 1994). Ground-level physical barriers in the form of stems, roots, and litterfall also help to protect the soil from surface runoff (Kang, 1993; del Castillo et al., 1994; Sanchez, 1995; Garrett and McGraw, 2000). Further, agroforestry systems can add significant amounts of organic matter to the soil, which can aid in providing cover as well as improving soil physical and chemical properties. In a recent study, Seiter et al. (1999) demonstrated that soil organic matter could increase by 4%–7% in alley-cropping systems with red alder (*Alnus rubra* Bong.) and maize in comparison with maize monoculture following 4 years of cropping (Figure 2.4). The presence of abundant organic matter serves to reduce soil compaction and increase infiltration and porosity (del Castillo et al., 1994). The net effect of soil structure modification is reflected in the degree to which roots are able to permeate the soil and exploit water and nutrient resource pools.

2.3.2.2 Water Availability, Competition, and Facilitation

Water is a major limiting factor in plant growth and productivity. The presence of trees in an agricultural system alters the soil water availability of the system, with repercussions for all associated plants. Trees generally have deeper roots and a higher fine root biomass than crop plants, and thus are in a more favorable position for water uptake than neighboring crops (Jose et al., 2000a). Fine roots are generally concentrated in the top 30 cm of the soil, where water fluctuation is greatest (Nissen et al., 1999; Gillespie et al., 2000; Jose et al., 2000a, 2000b) and severe water and nutrient competition takes place (Rao et al., 1993; Lehmann et al., 1998). In some cases, trees and crops may utilize separate soil water resource pools due to differences in rooting depth and intensity (Wanvestraut et al., 2004). However, in many cases, trees and crops compete directly for water. When this happens, soil water availability tends to be lower for the associated agronomic or forage crop due to competitive disadvantages in water acquisition (Rao et al., 1998; Jose et al., 2000a). Ultimately, the impact of soil moisture depletion on crops is expressed in terms of lower emergence rate, diminished plant size, and decreased yield (Jose et al., 2000a).

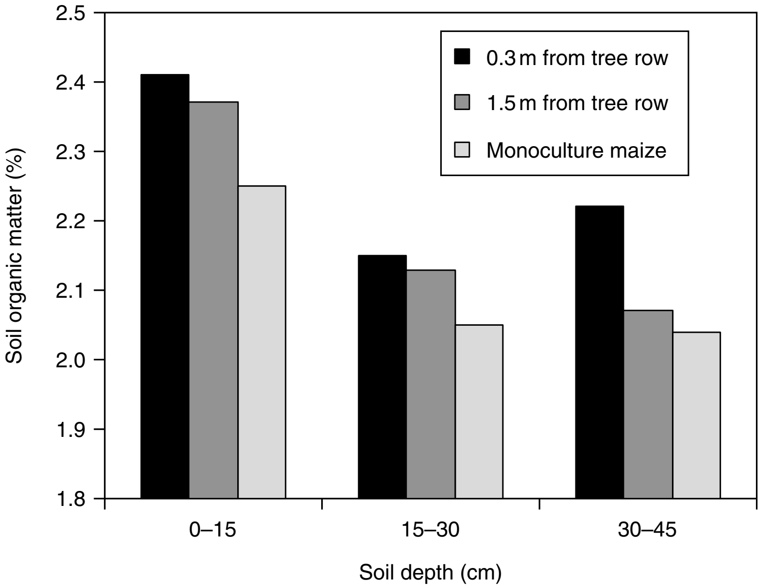

FIGURE 2.4 Soil organic matter as influenced by depth, distance, and cropping practice in western Oregon, United States. Soil organic matter in red alder–maize alley-cropping system (0.3 and 1.5 m from tree row) was significantly ($\alpha = 0.05$) higher than the soil organic matter in monoculture maize (specifically in the 0–15 cm soil layer) following 4 years of cropping. (Adapted from Seiter, S., R.D. William and D.E. Hibbs, *Agroforestry Syst.*, 46, 273, 1999.)

Competition for water is a major limiting factor in temperate alley-cropping systems (Garrett and McGraw, 2000; Jose et al., 2000a; Miller and Pallardy, 2001). In a silver maple–maize alley cropping in Missouri, United States, Miller and Pallardy (2001) observed greater soil water content in the alleys when tree–crop interaction was excluded via a root barrier treatment (Figure 2.5). The barrier treatment also had a higher maize yield than the nonbarrier treatment. Jose et al. (2000a) reported similar findings and attributed the lower soil water content and maize yield in nonbarrier treatments to greater rooting intensity of component tree species. In another study, water competition in a hazel–cocksfoot system in central France, for example, began after 4 years of intercrop establishment when roots of both species started to expand and concentrate at the 0–50 cm soil depth (de Montard et al., 1999). Competition for soil moisture was also a major constraint in a black locust and barley intercropping system (Ntayombya and Gordon, 1995). The effects of water competition were also observed in a recent study of a pecan–cotton alley cropping in northwest Florida by Wanvestraut et al. (2004), in which cotton lint yield was reduced by 21% because of belowground competition for water.

The facilitative role of trees in soil–water relations is also important. For example, trees can benefit nearby understory plants through the mechanism of hydraulic lift, wherein water from deep moist soils is transported to drier surface soils through the root system of trees, thus providing more moisture for surrounding vegetation during dry periods (Dawson, 1993; Chirwa et al., 1994b; van Noordwijk et al., 1996; Burgess et al., 1998; Lambers et al., 1998; Ong et al., 1999). For example, in an Orange wattle (*Acacia saligna* Labill. H. Wendl.) and sorghum intercrop, Orange wattle penetrated deeper soil strata to avoid competition in soil zones of high root density (Lehmann et al., 1998). High nitrogen levels along with moisture brought by hydraulic lift of the tree roots stimulated growth of the intercropped sorghum (Lehmann et al., 1998). Facilitation has also been shown in favorable stand establishment of conifers (*Austrocedrus chilensis*) grown under nurse shrubs during dry periods in Patagonia, Argentina (Kitzberger et al., 2000). Trees can also improve

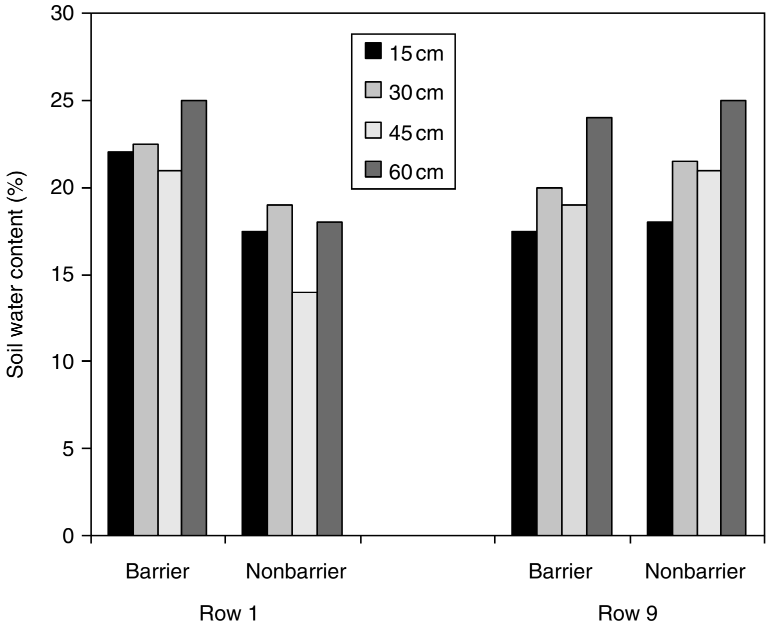

FIGURE 2.5 Mean seasonal volumetric soil water content at four depths for rows 1 and 9 within a silver maple–maize alley-cropping system in north-central Missouri, U.S.A. The differences between barrier and nonbarrier treatments were significant at 45 and 60 cm depths for row 1 ($\alpha = 0.05$). Barrier treatment was subjected to belowground trenching and a polyethylene barrier to separate tree and crop root interaction, whereas nonbarrier represents control treatment with no belowground trenching or root barriers. (Adapted from Miller, A.W. and S.G. Pallardy, *Agroforestry Syst.*, 53, 247, 2001.)

net productivity by providing for more effective use of rainfall in sequential systems. For example, Ong et al. (2002) postulated that agroforestry systems could be used to harness residual water remaining in the soil after harvest of crops and during off-season. Trees show facilitation in other ways as well. As mentioned earlier, a tree canopy, for example, acts to reduce soil and air temperature, wind speed, and irradiance, which influence soil water evaporation and humidity within the system (Rao et al., 1998).

2.3.2.3 Nutrient Availability, Competition, and Cycling

Alley-cropping systems modify the availability of soil nutrients in various ways. Generally, the inclusion of woody species on farmlands improves soil fertility. For example, trees help to increase the organic matter content of soil through the addition of leaf litter and other parts from trees (Table 2.3; Figure 2.4). In addition, they generally provide for more efficient cycling of nutrients within the system (Nair, 1987; Palm, 1995). The system can also moderate extreme soil reactions via the increased soil organic matter (Nair, 1987), improve nutrient release and availability patterns (Nair, 1987), and provide a more suitable environment for increased activity of beneficial microorganisms in the rooting zone (Lee and Jose, 2003).

Nitrogen is usually the most limiting soil nutrient in alley-cropping systems. Because N is lost via harvests of crop biomass and removal of limb prunings, N supplements are needed in alley-cropping systems to maintain favorable growth of trees (Garrett and McGraw, 2000). In temperate agricultural settings, nitrate is primarily introduced into the environment in the form of solid fertilizer compounds such as ammonium nitrate, calcium nitrate, and potassium nitrate, or as a solution of ammonium nitrate. N may also be introduced as chicken litter or some form of organic

TABLE 2.3
Nitrogen from Leaf Litter of 12 Temperate Tree Species with Potential for Alley-Cropping Systems

Tree Species	N Addition (kg ha^{-1} yr^{-1})	Source
Alder (*Alnus* sp.)	48–185	Tarrant et al. (1969), Daniere et al. (1986)
Autumn olive (*Elaeagnus umbellata* Thunb.)	236	Paschke et al. (1989)
Black locust (*Robinia pseudoacacia* L.)	30–35	Boring and Swank (1984)
Black walnut (*Juglans nigra* L.)	75–100	Jose (1997)
European beech (*Fagus sylvatica* L.)	40	Duvigneaud and Denaeyer-DeSmet (1970)
Loblolly pine (*Pinus taeda* L.)	58	Wells and Jorgensen (1975)
Mesquite (*Prosopis glandulosa* Torrey)	45	Rundel et al. (1982)
Paulownia (*Paulownia* sp.)	68–90	Yin and He (1997)
Pecan (*Carya illinoensis* L.)	70	Allen and Jose (unpublished data)
Poplar (*Populus* spp.)	68	Thevathasan and Gordon (1997)
Red oak (*Quercus rubra* L.)	50	Jose (1997)

mulch. Litterfall from trees and crops also supplements the N supply in the soil as explained earlier. Plants of the same species and growth stage compete heavily for N because of the high mobility of water and nitrate ions along the root surface, resulting in zones of depletion in the soil that overlap with neighboring plants. Similarly, roots of trees and crops possess a high potential for interspecific competition for nitrate in the topsoil, depending on rooting depth, water availability, and tree species phenology (Jose et al., 2000b).

Available literature indicates that interspecific competition for nutrients is generally of minor importance to system productivity, although certain factors can work to increase nutrient competition. For example, Jose et al. (2000b) observed that competition for fertilizer nitrogen was minimal in a black walnut–maize alley-cropping system, since nutrient acquisition was not simultaneous among the system's components. However, water availability was a factor in nutrient competition, as competition for water by tree roots was responsible for reduction in biomass in intercropped maize, resulting in decreased efficiency of fertilizer use (Figure 2.6; Jose et al., 2000b). Similarly, in a poplar–barley system in southern Ontario, associated trees and crops utilized different sets of soil nutrient resource horizons (Williams et al., 1997). However, competition for available nutrients cannot be avoided when fertilizer is not supplied. This is shown in the fact that the addition of nutrients to an alley-cropping system can increase yields compared with an alley-cropping system that utilizes only the available nutrients in the soil (Chirwa et al., 1994a; de Montard et al., 1999; Immo and Timmer, 2000).

The effect of trees in alley-cropping systems is of interest, in part, due to the mechanism of nutrient capture, in which deep roots of trees serve as a "safety net" for capturing nutrients that leach below the root zone of crops (van Noordwijk et al., 1996; Rowe et al., 1999). At lower depths, tree roots can exploit subsoil nitrate and other nutrients beyond the rooting depths of crops. A portion of these nutrients that are absorbed by the trees are later returned to the soil surface through decomposition of fine roots and litterfall, representing a gain to the soil nutrient pool (Nair, 1993; Jose et al., 2000b). This phenomenon is of importance because it serves as a possible mechanism for groundwater cleanup. Because nitrates are highly soluble, they are easily transported through the soil matrix (Aelion et al., 1997), where they may be carried away by runoff, or leached through the soil profile into the water table (USDA, 1998a; Nair et al., 1999). Such contamination can lead to pollution of drinking water wells, as well as create conditions for eutrophication and related ecological disruptions of rivers, lakes, estuaries, and aquifers (Johnson and Raun, 1995; USDA, 1998a, 1998b; Bonilla et al., 1999; Ng et al., 2000). From a human health standpoint, nitrate

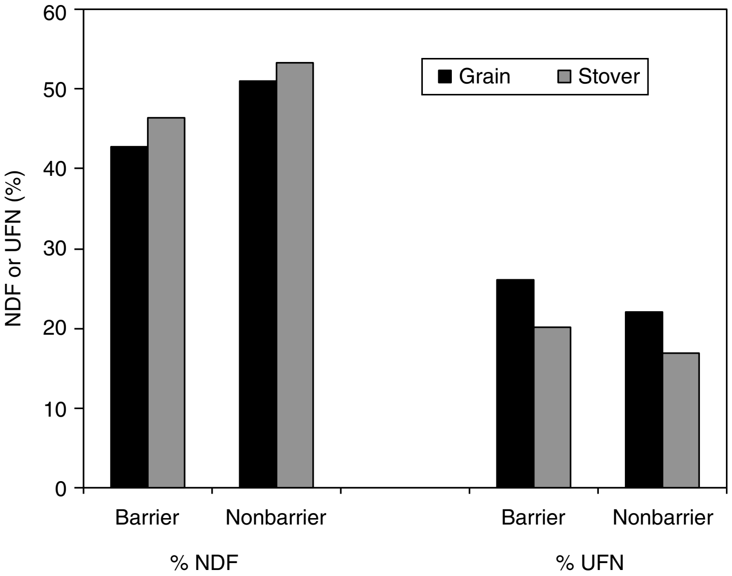

FIGURE 2.6 Percent nitrogen derived from fertilizer (%NDF) and percent utilization of fertilizer nitrogen (%UFN) for maize grain and stover in a black walnut–maize alley-cropping system in the mid-western United States. The differences between barrier and nonbarrier treatments were significant at $\alpha = 0.05$. Barrier treatment was subjected to belowground trenching and a polyethylene barrier to separate tree and crop root interaction, whereas nonbarrier represents control treatment with no belowground trenching or root barriers. (Adapted from Jose, S., A.R. Gillespie, J.R. Seifert, D.B. Mengel and P.E. Pope, *Agroforestry Syst.*, 48, 61, 2000b.)

is of concern in drinking water because it can cause a respiratory deficiency known as methemoglobinemia (blue baby syndrome) in infants under 6 months of age, and similar problems in adults of frail constitution (Sawyer et al., 1994; Baker, 1998; Ng et al., 2000; Reddy and Lin, 2000).

Preliminary data from a pecan–cotton alley-cropping project at the University of Florida suggest that pecan is able to extract nitrate from the alleys that cotton was not able to absorb, as seen from lower groundwater nitrate levels in the nonbarrier treatment than in the barrier treatment (Figure 2.7; Allen, 2003).

2.3.2.4 Allelopathy

Another important consideration is the allelopathic relationships that may exist in a tree–crop combination. Allelopathy refers to the process by which one plant inhibits the growth of another plant through the release of chemical compounds in the soil (Rice, 1984). Allelopathy has been studied by various researchers in both tropical and temperate agroforestry systems (e.g., Garrett and Kurtz, 1983; Gordon and Williams, 1991; Williams and Gordon, 1992; Jose and Gillespie, 1998; Thevathasan et al., 1998; Rizvi et al., 1999; Sasikumar et al., 2002). The degree to which these allelopathic chemicals inhibit growth depends on their concentrations as well as the combinations in which they are released into the ecosystem (Nair, 1993). One example of allelopathy can be seen in alley-cropping systems involving black walnut, a valuable timber- and nut-producing species recognized for its allelopathic traits (Jose and Gillespie, 1998). Juglone, a phenolic compound (5-hydroxy-1,4-napthoquinone) exuded mainly by black walnut roots, bark, and leaves was found to accumulate in high concentrations in soil, with inhibiting effects on juglone-sensitive species nearby. Entering the soil through root exudation, precipitation throughfall, and litter decay, juglone acts to suppress root growth of neighboring plant species. However, Thevathasan et al. (1998) found

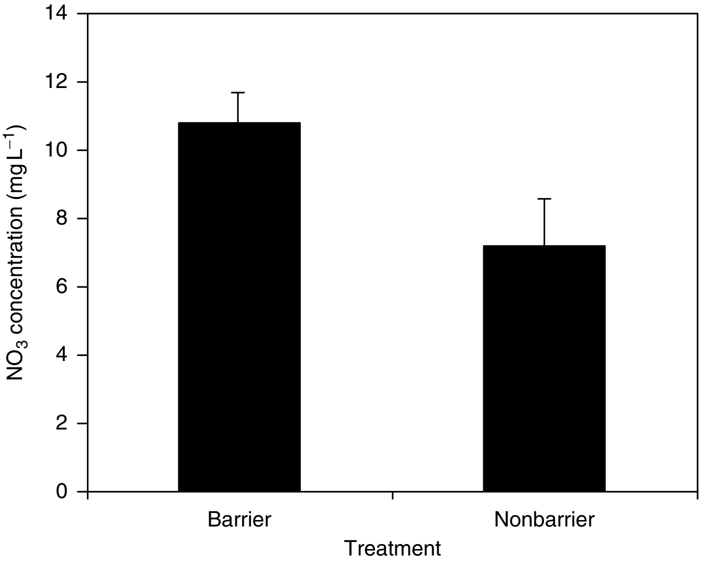

FIGURE 2.7 Concentration of NO_3 ions in soil water at a depth of 0.9 m with and without tree–crop root interaction in a pecan–cotton alley-cropping system in the southern United States. Barrier treatment was subjected to belowground trenching and a polyethylene barrier to separate tree and crop root interaction, whereas non-barrier represents control treatment with no belowground trenching or root barriers. (Adapted from Allen, S.C., *Nitrogen Dynamics in a Pecan (Carya illinoensis K. Koch)-Cotton (Gossypium hirsuitum L.) Alley Cropping System in the Southern United States*, Ph.D. Dissertation, University of Florida, Gainesville, FL, 2003.)

that juglone did not inhibit nitrification in their stands of black walnut and poplar. Jose and Gillespie (1998) suggested measures for limiting the effects of soil allelochemicals, such as root pruning, root disking, fertilizer injection, and installation of man-made root barriers.

Interestingly, certain annual species may exhibit allelopathic effects on trees as well. For example, Smith et al. (2001) administered allelochemical-containing leachates to container-grown pecan trees, and found that tall fescue, Bermuda grass, and cutleaf evening primrose (*Oenothera*: *Laciniata* Hill) leachate decreased pecan trunk weight by 22%, root weight by 17%, and total tree dry weight by 19%, respectively, compared with the control.

2.4 TREE–CROP INTERACTIONS: A MODELING APPROACH

Modeling of agroforestry systems and tree–crop interactions provides a means for predicting the degree of resource competition, productive yield, and sustainability in different growth environments. Recent years have seen the development of a variety of computer-based models with application to agroforestry and alley cropping (Table 2.4). Although space does not permit a discussion of all current models, two models of note are the WaNuLCAS model (van Noordwijk and Lusiana, 1999, 2000) and the SBELTS model (Qi et al., 2001). WaNuLCAS was developed to deal with various agroforestry systems, including hedgerow intercropping, fallow-crop mosaics, and other systems. Simulation runs have shown its ability to integrate spatial and management regimes into the model, as well as test for nutrient capture of deep tree roots (van Noordwijk and Lusiana, 1999, 2000). SBELTS has been used to test for microclimatic effects of agroforestry shelterbelts and resulting effects on crop yield. Through simulation, soybean production was determined to be greater in areas where trees are taller and provide a greater degree of leeward sheltering (Qi et al., 2001).

TABLE 2.4
Some Recent Models Relevant to Agroforestry and Alley-Cropping Systems

Model Name	Source
WaNuLCAS (Water, Nutrients, Light Capture in Agroforestry Systems)	van Noordwijk and Lusiana (1999, 2000)
SBELTS (ShelterBELT and Soybeans)	Qi et al. (2001)
SCUAF (Soil Changes Under Agroforestry)	Young et al. (1998)
APSIM (Agricultural Production Systems Simulator)	McCown et al. (1996)
Vector competition analysis model	Immo and Timmer (2000)
Soil–root water transport model	Sillon et al. (2000)

2.5 SYSTEM MANAGEMENT: OPPORTUNITIES AND CONSTRAINTS

As an agroforestry system, alley cropping offers potential for alleviating numerous environmental problems, but, as with other systems, its success or failure depends on how it is designed and managed. Some beneficial management practices for alley cropping are discussed in the following sections, followed by certain constraints to its success.

2.5.1 Spatial Factors

An alley-cropping system must reconcile the need for arable growing space for both crop and tree components. In the case of a pecan–cotton intercrop, for example, various biophysical factors must be considered, such as row spacing, planting density, and optimal arrangement of trees and crops. Proper measurements of cultivator and seeder clearances need to be made in relation to tree distances, to ensure smooth establishment and maintenance of the cotton, as well as minimal disturbance to tree roots. Trees should be planted with an eye toward long-term pecan production at optimal tree distances, so the cotton component would thus need to be viewed as a shorter-term component, say, for 15 years, or until the tree canopy prevents acceptable cotton production. Likewise, crop rotation with other crops such as maize or soybean would likely be recommended for the cotton area. One appeal of this spatial arrangement (cotton and pecan on the same land) lies in the fact that a landowner can theoretically have a higher economic return for a given land area while the pecan trees are maturing. Thus, side-by-side growth can be facilitative, assuming that there is ample growing area, and that neither crop is experiencing a significant loss in production. Improper spatial arrangement, however, could result in lost productivity, inefficient cotton maintenance, or even damage or destruction of valuable pecan trees and farm equipment (Zinkhan and Mercer, 1997). Management options are also complicated with regard to chemical and pesticide interactions. Since pesticides have specific labeling restrictions, their use near crops for which they are not labeled may create risk for violating label restrictions (Ramsey and Jose, 2001).

One method used to control root density and distribution (as well as competition for light) is aboveground pruning of trees. Chirwa et al. (1994b) observed no apparent competition for water in a semi-arid system where the tree species was heavily pruned, which reduced plant surface area evapotranspiration. Another mechanism possibly responsible for the decrease in competition could be root shedding following tree pruning. Lehman et al. (1998) cited a root-shedding response to aboveground pruning of the hedgerow species, which effectively reduced belowground competition. Nonetheless, this reaction to pruning can possibly reduce subsoil root length density and increase the danger of nutrient losses by leaching (Peter and Lehmann, 2000). A second potential drawback to tree pruning is the reduction of aboveground biomass production, which can make moving organic carbon through an agroforestry system somewhat difficult. However, the manipulation of alley density and width can reduce the pressure to prune (Schroth and Zech, 1995).

Root pruning, usually by way of trenching, has been used as a means to separate root systems of trees and crops, thereby reducing belowground competition significantly in alley-cropping systems in both the tropics (McCune, 1986; Ong et al., 1991; Kang, 1993) and temperate regions (Gillespie et al., 2000; Miller and Pallardy, 2001). However, this practice can also cause increases in root length density at the pruning site as the root system regenerates, resulting in increased soil moisture depletion in that area (Jose et al., 2000a). Although periodic root pruning has been used in the management of fruit production (Miller and Pallardy, 2001), the long-term effects of repeated root prunings are not known. Other options include deep disking to a depth of 0.6–0.9 m to sever roots (Garrett and McGraw, 2000).

2.5.2 Temporal Factors

Alley-cropping design must also reconcile temporal (timing) factors related to each species, such as dates for stand establishment, fertilization, pesticide application, irrigation, and harvest of trees and crops. Improper timing can create a range of problems. In our pecan–cotton system, for example, late season cotton is not well suited for growing with mature pecan trees, since both species are ready to be harvested around the same time—October and November of each year. It is difficult to harvest pecans when they are in a stand of 1.5 m high cotton. Likewise, it is difficult to harvest cotton without destroying much of the pecan harvest (through defoliation or mechanical injury or burial by crop residue). These practices would appear to be mutually exclusive, and thus are in conflict due to timing factors. Such a situation is difficult to avoid, however, if weather conditions preclude establishment of cotton earlier in the season. Similarly, wrong timing and spacing can create situations in which, though pesticides are applied appropriately to one species, the other species is subject to exposure to the same chemical. Not only this is illegal, but it can also damage the untargeted plant. On the other hand, it is possible to have complementary temporal interactions. An irrigation or fertigation system serving pine trees, for example, could be adapted to service nearby crops during irrigation times. In addition, peanuts, for example, would seem to pose less risk of creating a timing-related problem, since they have a shorter growing season and take up less area than crops such as cotton. Spatial and temporal interactions are thus important factors in the design and management of agroforestry systems.

A number of constraints to alley cropping exist in the temperate regions of the world. For example, alley cropping has the potential to diminish production and yield of any plant component in the system. Additionally, some agroforestry species can become invasive and ultimately disrupt the natural ecosystem of the area. Finally, most alley-cropping systems in the United States and elsewhere in the temperate region have not been sufficiently researched at this time, and are not well known by the general public, and thus pose a potential for being misapplied. To overcome these handicaps is no easy task.

A key strategy for preventing unanticipated negative outcomes is judicious selection of tree and crop species and varieties. At a minimum, tree and shrub species should possess the following traits: adaptation to the soil and climate of the planting site; production of wood, fruit, or fodder suited to the purpose of planting; resistance to pests and herbicides; tolerance of sediment deposition and pollutant-laden runoff; and resistance to stem and branch breakage from high winds, ice, and snow (USDA, 1996). Once species are selected, due consideration should be given to timing of operations, spatial arrangement, potential problem areas, and overall goals of the system. Tree or crop species that possess the potential for becoming "weed" species should either not be introduced, or carefully monitored to ensure containment. Additionally, if there seems to be an inherent problem with timing of two species, then perhaps one of the species should be replaced with another crop. Unintended contamination during pesticide application is also not acceptable, which again is ground for replacing one of the crops. In spite of these constraints, the benefits of alley cropping are worth a calculated risk, which will vary depending on the degree of risk aversion held by the landowner.

2.6 RESEARCH NEEDS

The preceding discussion illustrated the need for further research into temperate alley-cropping systems, particularly with regard to their biogeochemical interactions and resulting effects on system sustainability and profitability. Since many such systems are in the formative stages of adoption and research, much baseline information is needed to establish guidelines for tree–crop interactions of specific species. To optimize production and sustainability of these systems, various major research imperatives are needed. These include: examination of alley-cropping system components in terms of tree spacing, tree–crop configurations, and their role in system productivity; determination of biophysical interactions between system components and their impact on the system; evaluation of alley-cropping systems for their environmental benefits; and determination of the economic benefits of alley cropping. Results from such studies can be used to educate farmers and landowners about the potential for alley cropping as an alternative land-use practice and aid researchers and extension personnel in the design of more sustainable farming systems for the world's temperate regions.

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3 Allelopathic Tree–Crop Interactions under Agroforestry Systems

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CONTENTS

3.1 INTRODUCTION

In agroforestry system (AFS), where both woody and nonwoody components occur together, a variety of positive and negative interactions occur (García-Barrios and Ong, 2004). Trees, owing to their perennial habit, larger size, and better adaptability, have a major influence on crops and also modify the biophysical environment to favor their own growth (Ong et al., 1996). Trees improve soil quality by providing organic matter, reduce soil erosion, conserve soil moisture, provide protection against wind, reduce weed population and composition, help in nutrient recovery from the deeper soil layers, and add more amounts of phosphorus and nitrogen (Young, 1997; Rao et al., 1998; Giller, 2001). In addition, a number of negative or antagonistic interactions, competitive and allelopathic, may also prevail in the AFS (Rao et al., 1998). The key to success in agroforestry is the minimization of negative interactions and maximization of positive interactions to get the best results (Thevathasan and Gordon, 2004). Among various negative interactions that occur under agroforestry practices, least attention has been paid to allelopathy, though some workers have highlighted its role in AFS (Rizvi et al., 1999; Singh et al., 2001a; Kohli et al., 2006). This chapter focuses on and summarizes various aspects about the potential and role of allelopathy in agroforestry systems.

The term *allelopathy* refers to a type of chemical-mediated interaction in which one plant releases chemicals into environment that are detrimental to the growth of other plants growing in its vicinity. Rarely, allelopathy can be beneficial; however, it depends on the concentration of chemicals involved and the response of recipient species (Rice, 1984). Though the term was coined by

Hans Molisch in 1937, the concept of allelopathy is very old and dates back to the writings of Theophrastus (ca. 300 BC)—a Greek philosopher. The chemicals responsible for allelopathic activity are called as allelochemicals and belong to a diverse array of chemical nature and structure. These are synthesized within plants as secondary metabolites and released through leachation from the fresh and decaying plant parts or on microbial decomposition of the fallen plant parts or litter, or as root exudates and volatilization in case of aromatic plants (Rice, 1984). On release, these may either accumulate in the soil or may undergo transformation or detoxification by the soil microbes (Blum et al., 1999). Their toxic effects thus depend on several biotic and abiotic factors and are a function of their bioactive concentrations in the soil.

3.2 TREE ALLELOPATHY AND ITS IMPLICATIONS UNDER AFS

Allelopathy occurs in diverse taxonomic groups including microbes, algae, fungi, ferns, gymnosperms, and angiosperms (Rice, 1984, 1995). It has, however, been largely reported in angiosperms. Different life-forms like herbs, shrubs, and trees exhibit allelopathy. Among these, the allelopathy of trees assumes greater significance since trees being larger in size and perennial in habit may serve as a major and continuous source of allelochemicals in soil. Several reports indicate that tree allelopathy is responsible for many visible effects such as bare forest floor or poor vegetation under canopy of some trees, regeneration problem of propagules of some of the forest tree species, and gradual loss of species diversity (Rice, 1984). Allelopathy is operative under both plantations and natural forests. Trees incorporated in agroecosystems in various ways may also bring about significant effects on the associated crops and result in reduction in crop productivity (Kohli et al., 2006). In the tropics, particularly under simultaneous agroforestry systems, allelopathy plays an important role in influencing both the negative effects and the positive benefits and is a major factor in determining tree–crop–soil interactions (Rao et al., 1998). A number of tree species are allelopathic and exert their effect through various parts like leaves, litter, stem, roots, and even fruits (Rice, 1984, 1995; Lisanework and Michelsen, 1993; Rizvi et al., 1999). However, the contribution of leaves and litter is more as reported by several workers under laboratory, greenhouse, and field conditions (Rice, 1984, 1995; Rizvi et al., 1999). Rizvi et al. (1999) extensively reviewed the role of allelopathy under AFS and listed over 80 trees exhibiting allelopathic influence on the crops. However, most of these reports are based on laboratory experiments using ecologically unrealistic concentrations, and their demonstration and practical utility under field conditions is largely lacking. Nevertheless, the allelopathy of certain tree species like *Eucalyptus*, *Populus*, *Leucaena*, and *Juglans* is well documented under laboratory and field conditions and provides a strong basis for conducting allelopathic research under AFS. The allelopathic studies on these trees have been discussed separately in the following sections.

A number of studies have shown that mulches and prunings of trees may release allelochemicals and thus suppress crop growth. Kamara et al. (1999) studied the effect of leaf extracts and mulch from 5 year old multipurpose trees (MPTs) viz. *Gliricidia sepium* (Jacq.) Walp., *Leucaena leucocephala* (Lam.) de Wit., *Enterolobium cyclocarpum* (Jacq.) Griseb., *Senna siamea* (Lam.) H.S. Irwin & Barneby, *Tetrapleura tetraptera* (Schum & Thonn.) Taub., *Milletia thoningii* (Schum & Thonn.) Bak., *Lonchocarpus sireceus* (Poir.) H.B. & K., *Pithecelobium dulce* (Roxb.) Benth., *Terminalia superba* Engl. & Diels, *Gmelina arborea* Roxb., *Grewia pubescens* P. Beauv., *Pterocarpus santalinoides* Pherex, *Nuclea latifolia* Sm., and *Alchornia cordifolia* (Schum & Thonn.) Mull. Arg. on cowpea (*Vigna unguiculata* L.). Leaf extracts and mulch from *Gliricidia sepium*, *S. siamea*, *M. thoningii*, *Grewia pubescens*, and *Tertapleura tetraptera* significantly reduced germination and early growth of cowpea. Further, mulch from the trees with fast decomposing leaves viz. *Leucaena leucocephala*, *Gliricidia sepium*, and *Grewia pubescens* was more phytotoxic. These workers postulated allelopathy, in addition to nitrogen immobilization, as the possible reason for the observed effect. Later, Kamara et al. (2000b) tested these 14 MPTs for their phytotoxic effects against *Zea mays* L. (maize). Maize germination and initial growth was severely affected by leaf extracts; however, under field and pot conditions the growth of maize was dependent on nitrogen supply rather than phytotoxic effects. These workers reported that trees like *Gliricidia sepium* and *L. leucocephala* with fast decomposing foliage supplied more nitrogen to the soil and thus growth of maize was better in their mulches. However, the growth of maize was stunted when mulch decomposed slowly and released lesser nitrogen. The study concluded that allelopathic effects of leaf extracts of MPTs diminished under field conditions and are thus ecologically irrelevant. Anthofer et al. (1998) tested the effect of leaf prunings of nine agroforestry trees viz. *Grevillea robusta* A. Cunn. ex R. Br., *Erythrina abyssinica* Lam. ex DC., *Gliricidia sepium*, *Albizia schimperiana* Oliv., *Acacia nilotica* L., *A. polyacantha* Willd., *L. leucocephala*, *L. pallida* Britton and Rose, and *Entada abyssinica* A. Rich. on the growth of wheat (*Triticum aestivum* L.) under a pot trial experiment. Leaf prunings of *L. leucocephala*, *L. pallida*, *G. sepium*, and *E. abyssinica* proved good for the growth of wheat, whereas those of *Grevillea robusta*, *A. polyacantha*, *A. nilotica*, and *Erythrina abyssinica* adversely affected the growth of wheat. The adverse effect of the leaf prunings of these agroforestry trees was attributed to allelopathy in addition to N immobilization.

3.3 ALLELOPATHIC AGROFORESTRY TREES: SOME EXAMPLES

3.3.1 *EUCALYPTUS* SPECIES

Eucalyptus is one of the most important agroforestry trees that was promoted largely because of its fast growth rate and commercial value. The trees are widely planted along field boundaries as shelterbelts and windbreaks, as alleys, or as field bund plantations under the simultaneous agroforestry. Both laboratory and field experiments support that many *Eucalyptus* species are allelopathic (Rizvi et al., 1999, and references cited therein). Igboanugo (1988a) reported that growth and yield of *Capsicum annuum* L. was reduced near the tree line of *Eucalyptus citriodora* Hook., and the effect diminished with increasing distance from tree line (Igboanugo, 1988b). Under field conditions, Kohli and his associates (1990) observed that density, root and shoot length, biomass, and economic yield of crops viz. *Cicer arietinum* L. (chickpea), *Lens esculentum* Medik.(lentil), *Triticum aestivum* L. (wheat), *Brassica oleracea* L. var. botrytis (cauliflower), *B. campestris* L. (toria), *Trifolium alexandrinum* L. (berseem), *Cajanus cajan* (L.) Huth. (pigeon pea), *Oryza sativa* L. (rice), *Sorghum vulgare* Pers. (sorghum), and *Zea mays* L. (maize) were reduced up to a stretch of 12 m to the south of *Eucalyptus* (*Eucalyptus tereticornis* Sm.) shelterbelts compared to control. Singh and Kohli (1992) reported that economic yield of lentil, chickpea, wheat, cauliflower, berseem, and toria was significantly reduced in the fields sheltered by *Eucalyptus* compared to unsheltered control fields (Figure 3.1). The reduction in yield was severe in crops growing closer to the tree line. However, with the increase in distance from the tree line, the yield reduction was lesser and at distance 11 m or beyond there was not much difference in yield compared to unsheltered control (Figure 3.1). The poor crop performance was attributed to the presence of allelochemicals in the soil near the tree line (Singh and Kohli, 1992). The amount of allelochemicals varied with distance from the tree line as well as depth from the surface (Figure 3.2). At 1 m distance from tree line, where yield was the minimum, the amount of phytotoxins was the maximum. However, their amount declined with increasing distance from tree line and was almost negligible at 11 m from tree line (Figure 3.2). Thus, there was a strong reciprocal correlation between yield and amount of phytotoxins (Singh and Kohli, 1992). These workers concluded that under natural conditions, the allelochemicals of *Eucalyptus* continuously enter the soil environment and thus affect the other plants including crops. This study serves as one of the best examples in which the role of allelopathy, including allelochemicals, has been demonstrated under field conditions in a simultaneous agroforestry system.

Kohli (1990) identified a number of volatile and nonvolatile allelochemicals from the leaves, bark, litter, and understory soil of the *Eucalyptus*. The volatile monoterpenes of *Eucalyptus* like

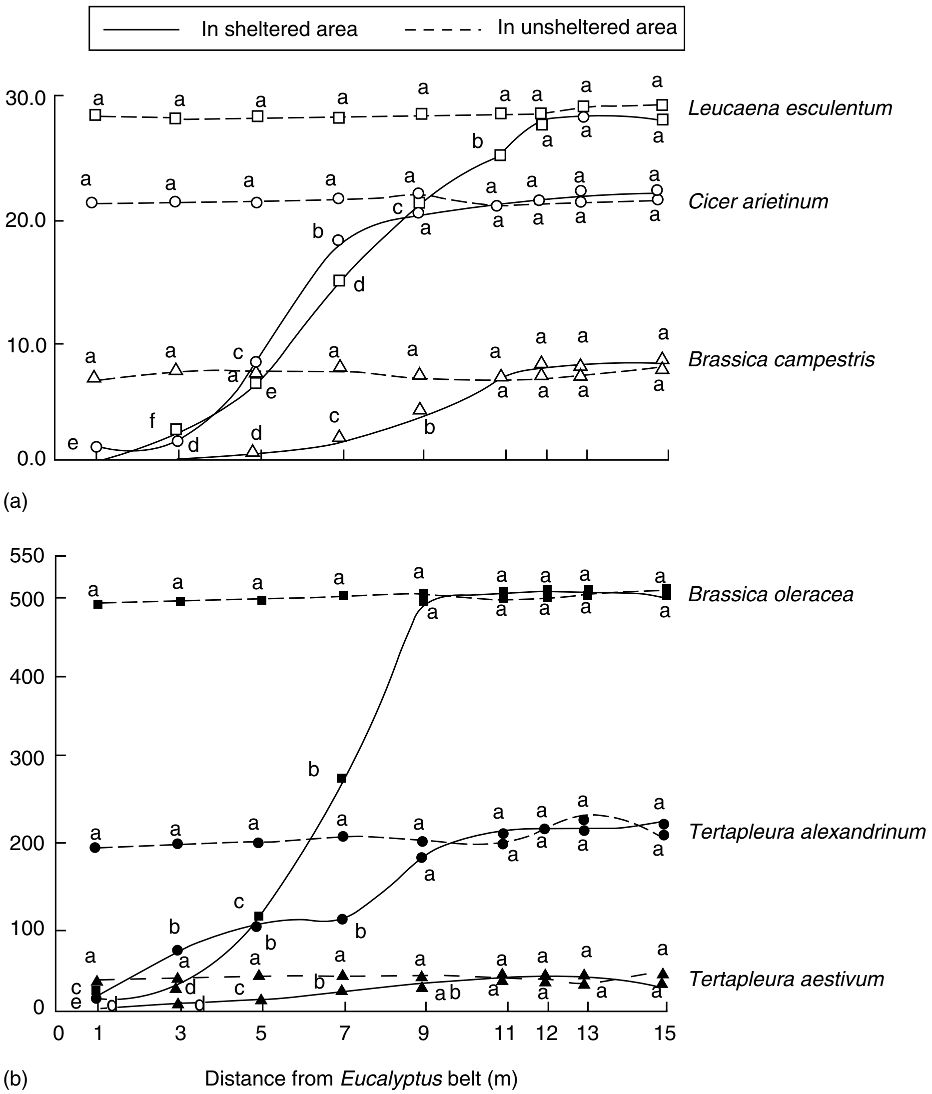

FIGURE 3.1 Mean economic yields in 100 kg ha^{-1} of (a) *Cicer arietinum*, *Lens esculentum*, and *Brassica campestris* and (b) *Triticum aestivum*, *Trifolium alexandrinum*, and *Brassica oleracea* grown in areas sheltered by *Eucalyptus* (solid line) or unsheltered control areas (dotted line). Similar superscript symbols along each curve represent insignificant difference at $p < 0.05$ applying Duncan's multiple range test. (From Singh, D. and R.K. Kohli, *Agroforest. Syst.*, 20, 253, 1992. With permission from Springer.)

limonene, 1,8-cineole, citronellal, citronellol, α-pinene are highly toxic and are potential allelochemicals (Singh et al., 2002, 2006a, 2006b). These are heavier than air, move downward, adsorb to soil particles, and affect the associated vegetation (Kohli, 1990). The nonvolatile allelochemicals include various phenolic acids viz. *p*-coumaric acid, ferulic acid, *p*-hydroxybenzoic acid, caffeic acid, gallic acid, and vanillic acid that have been identified from the leaves and bark of the tree and are highly phytotoxic (Kohli, 1990).

3.3.2 POPULUS DELTOIDES

Populus deltoides Bartr. ex Marsh. is another fast-growing tree extensively promoted under agroforestry programs as shelterbelts or windbreaks or field bund plantations. Its significance as a

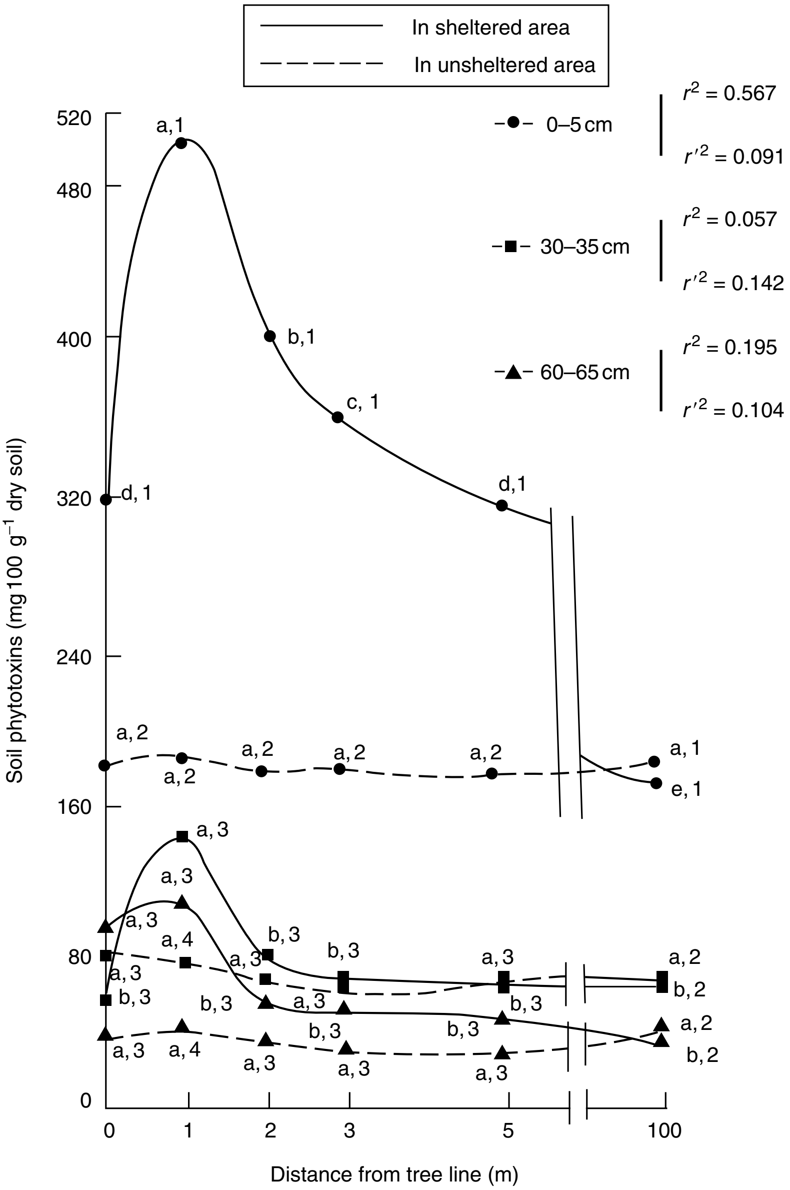

FIGURE 3.2 Amount of soil phytotoxins (mg 100 g^{-1} dry soil) at varying depths and distances from tree line of the *Eucalyptus* sheltered and unsheltered areas. Different alphabets or numerals along a curve represent significance at different distances or depths, respectively, at $p < 0.05$ applying Duncan's multiple range test. r^2 represents correlation coefficient between amount of phytotoxin and distance from tree line at respective soil depths. (From Singh, D. and R.K. Kohli, *Agroforest. Syst.*, 20, 253, 1992. With permission from Springer.)

fast-growing tree has increased tremendously owing to the negative campaigning of *Eucalyptus* plantations in India. However, it also exerts a negative influence on crops and native vegetation through its leaf litter. Singh et al. (1998) assessed the effect of *P. deltoides* shelterbelts on wheat. These workers reported that density, growth, biomass, and yield of wheat were significantly reduced up to 12 ± 1 m in the sheltered fields compared with unsheltered control fields and comparatively the effect was more on biomass and yield of wheat (Figure 3.3). However, there was a considerable

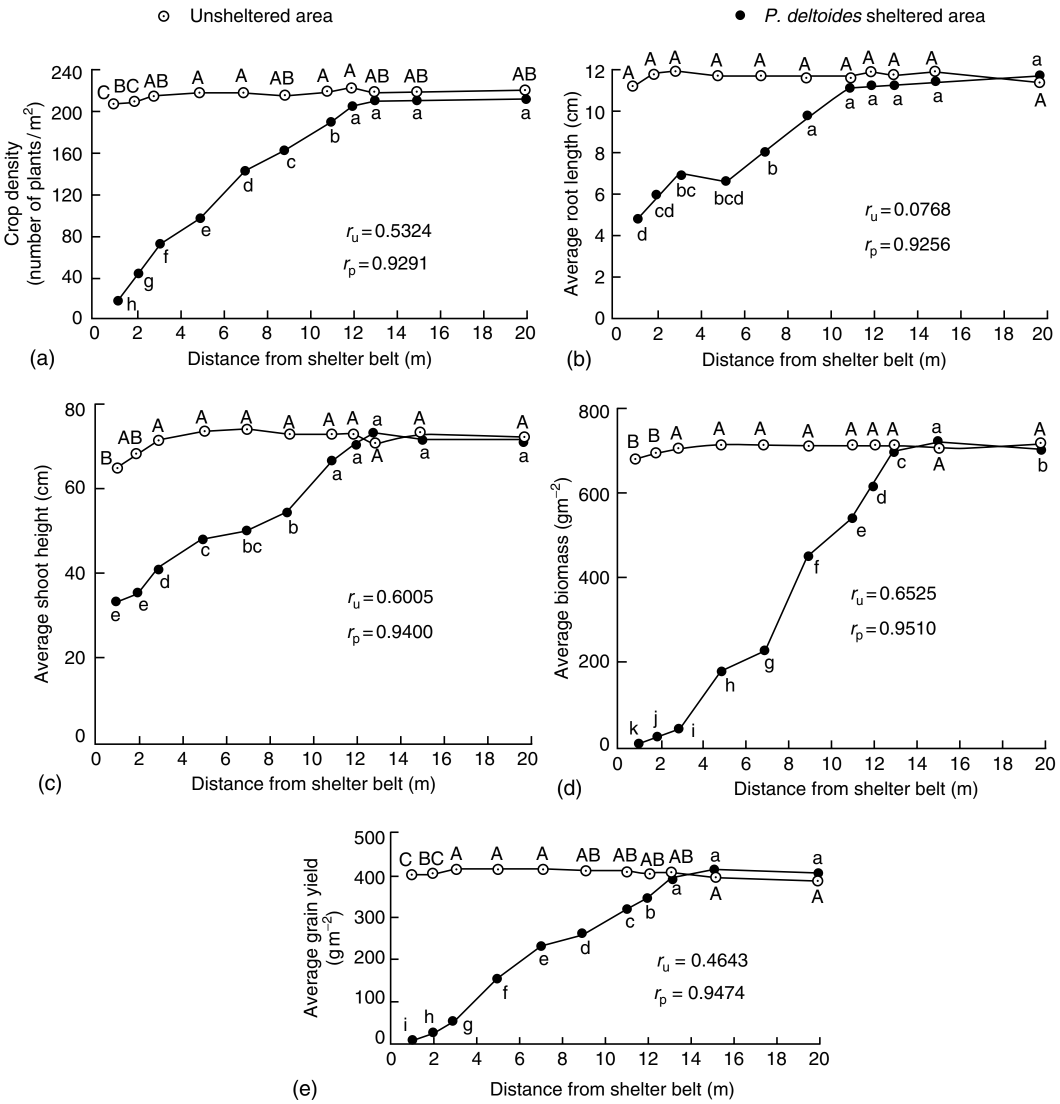

FIGURE 3.3 Mean values of (a) crop density, (b) root length, (c) shoot height, (d) biomass, and (e) grain yield of wheat at different distances from the shelterbelt of *Populus deltoides* or in the unsheltered fields in Punjab, India. Similar letters along respective curves represent insignificant difference at 5% level applying DMRT. "*r*" represents the value of correlation coefficient between respective parameters and different distances in unsheltered area (r_u) or in *Populus deltoides* sheltered area (r_p). (From Singh, H.P., R.K. Kohli and D.R. Batish, *Agroforest. Syst.*, 40, 208, 1998. With permission from Springer.)

improvement in growth and yield of wheat beyond 12 m. Further, in the 20 m belt from the tree line, there occurred a significant reduction (~41%) in the wheat yield in *P. deltoides* sheltered fields compared with unsheltered control fields (Table 3.1). However, beyond 20 m there was not much change in the wheat yield between sheltered and unsheltered fields. Overall, there was a reduction of nearly 8% in the *P. deltoides* sheltered fields compared with unsheltered control fields (Table 3.1). The growth reduction in wheat in the sheltered fields was attributed to the phenolic allelochemicals released by the tree. The amount of phenolics estimated at different distances from the *P. deltoides* shelterbelts showed a negative correlation with wheat performance, that is, amount of phenolics

TABLE 3.1
Grain Yield of Wheat in Fields Sheltered by *Populus deltoides* Trees in Punjab, India

Fields	In the 1 × 20 m Block (from 1 m to 20 m)	In the 1 × 40 m Block (from 20 m to 60 m)	Total Yield (kg ha^{-1})	Reduction in Yield over Control	
				Total (kg ha^{-1})	Percent
Unsheltered (control)	8.27 ± 0.84 (0.4136)	16.69 ± 0.24 (0.4172)	4165 ± 131.20	—	—
Sheltered by *Populus deltoides*	4.86 ± 0.55 (0.2432)	16.66 ± 0.79 (0.4165)	3819 ± 133.01	346.0	8.31

Source: From Singh, H.P., R.K. Kohli and D.R. Batish, *Agroforestry Syst*., 40, 208, 1998. With permission from Springer.

Note: Values in parentheses represent grain yield in kg m^{-2} and ± represents standard error of the mean.

decreased and the wheat yield increased with increasing distance from the tree belt (Singh et al., 1998). Singh et al. (1999a) made a comparative analysis of shelterbelts of *P. deltoides* and *Dalbergia sissoo* Roxb. ex DC.—a native AF species of India. These workers reported that wheat growth and yield was very poor in the fields sheltered by *P. deltoides* compared with those sheltered by *D. sissoo*. In the former, the growth reduction in the wheat continued up to 12 m, whereas in the latter there was some growth retardatory effect up to a distance of 6 m from the tree line. Based on their observations, these workers concluded that though shelterbelts, in general, retard growth of crops especially at a distance nearer to the tree line, allelopathic trees had a greater detrimental effect than the nonallelopathic trees. Selection of suitable agroforestry tree is thus very important. Singh et al. (2001b) studied the performance of some winter season crops viz. wheat, lentil, mung bean, oat, clover, rapeseed, and sunflower in association with alley-cropped *P. deltoides* under two sets of conditions—when parent soil of *P. deltoides* was retained and when it was replaced with soil collected from an area devoid of *P. deltoides*. The germination and growth of crops measured 30 and 60 DAS (days after sowing) was very poor in alley-cropped *P. deltoides* where parent soil was retained compared with the other situation in which the parent soil was replaced with control soil as well as compared with the control (without alley-cropped *P. deltoides*). The reasons for such a poor growth in the first situation were attributed to allelopathic interference of tree through its litter that releases phytotoxic phenolics in the soil. Allelopathy of *P. deltoides* has also been demonstrated under laboratory conditions. Different parts of the tree exhibited a significant phytotoxic effect on mung bean (*Phaseolus aureus* Roxb.), but the leaves were found to be more phytotoxic compared with other parts (Kohli et al., 1997). A number of allelochemicals including phenolic acids and salicin (a phenolic glucoside) were detected in the aqueous leaf extracts and soil collected from under the canopy of *Populus deltoides* (Singh, 1996).

3.3.3 *LEUCAENA LEUCOCEPHALA*

L. leucocephala is one of the most important multipurpose tree (MPT) that has been widely promoted under various forestry programs including agroforestry, especially as hedgerows and under alley-cropping systems (Brewbaker, 1987). A native of Central America and Mexico, it has been introduced in India and other tropical and subtropical countries under various afforestation programs because of its characteristics like fast growth rate, fodder and fuel value, and nitrogen-fixing ability. However, the plantations of *L. leucocephala* have little understory vegetation and it was attributed to allelopathy (Chou and Kuo, 1986). It has been introduced into the agricultural fields under various agroforestry programs because of its nitrogen-fixing property and serves as a good source of green manure. Singh et al. (1999b) reported that leaves and litter of *L. leucocephala* collected from top soil (A_0 and A_{00} horizon) severely retarded the growth of maize under laboratory as well as greenhouse conditions. Both leaf and litter mulched (at 2.5–10 ton ha^{-1}) on the surface and amended in soil reduced the growth (root and aboveground) and biomass of maize. The inhibitory effect was more when these were amended in soil as a mixture than as mulch, and the litter was more phytotoxic than fresh leaves. The inhibitory effects were attributed to the presence of phenolics, which were also present in A_0 and A_{00} soil horizons (Singh et al., 1999b). Mimosine (a nonprotein amino acid) found in leaves, litter, and seeds of the tree and phenolic acids have been held responsible for the its allelopathic effect (Kuo et al., 1983; Suresh and Rai, 1987; Tawata and Hongo, 1987; Rizvi et al., 1990; Chaturvedi and Jha, 1992). Mimosine, however, does not seem to cause any damage to *L. leucocephala* seedlings as these abundantly grow on its floor. Kamara et al. (1999) reported that leaf extracts of *L. leucocephala* reduced germination, initial growth, and nodulation in cowpea. They attributed growth retardatory effects of *L. leucocephala* to allelopathy and nitrogen immobilization. In a later study, these workers observed that though extracts of *L. leucocephala* negatively affect the growth of maize under laboratory or greenhouse conditions, yet under field conditions the growth of maize was better in the mulched fields than in unmulched fields. These workers attributed better growth of maize to the better nitrogen supply from *L. leucocephala* and ruled out any allelopathic effects (Kamara et al., 2000b). These conflicting reports thus indicate that allelopathic effects of this tree may be site specific or may depend on the response of target crops.

3.3.4 *JUGLANS NIGRA*

J. nigra L., commonly known as black walnut, is extensively cultivated in agroecosystems of temperate North America under alley-cropping systems (Jose and Gillespie, 1996). It is often alley cropped with maize, soybean, and wheat because of commercial value. However, the tree is strongly allelopathic. In fact, it is one of the classical examples where allelopathy has been demonstrated. It was reported to interfere with the growth and establishment of neighboring plants since first century AD (Rice, 1984). Its proximity to other plants like vegetables, ornamentals, or legumes leads to several visible effects like wilting, necrosis, browning of tissue, and even death of plants (Weston and Duke, 2003). The phytotoxicity of the tree is attributed to the presence of a potent allelochemical juglone—a colored phenolic compound identified as 5-hydroxy-1,4-naphthoquinone (Davis, 1928). Within the plant, juglone occurs in an inactive and reduced state, whereas on exposure to air, it activates and becomes toxic (Lee and Campbell, 1969; Rietveld, 1983). Crist and Sherf (1973) reported that several plants like tomato, potato, pea, pear, apple, and many ericaceous plants are sensitive to walnut. Though juglone is found in various parts of the tree, yet, the amount is more in the buds, nut hulls, and roots. Rietveld (1983) tested sensitivity of many plant species to juglone at concentrations ranging from 10^{-6} to 10^{-3} M under hydroponic system and soil culture. He found that shoot elongation and dry weight of many plants were seriously impaired. Jose and Gillespie (1998a) observed that in a walnut–maize alley-cropping system the concentration of juglone in soil decreases as the proximity from the tree line increases, though there is no change in its concentration seasonally. At a distance of about 4.25 m from the tree line, nearly 80% decrease in the juglone content was noticed, thus showing a clear spatial pattern of the allelochemical. Installation of polythene root barriers resulted in a decline in juglone concentration in the alleys where the crops are planted, and this may protect sensitive crops from the toxicity of juglone. Jose and Gillespie (1998b) further established that the observed growth reduction of maize and soybean alley cropped with walnut is due to the presence of juglone in the soil. A significant reduction in growth and physiology of hydroponically grown test crops was observed when treated with 10^{-6}, 10^{-5}, and 10^{-4} M juglone. Soybean was found to be more sensitive than maize.

3.3.5 *GLIRICIDIA SEPIUM*

It is another important MPT species planted under agroforestry programs, particularly in the humid tropics. In fact, it is the second most important multipurpose legume tree after *L. leucocephala*.

Tian and Kang (1994) evaluated the effect of leaf prunings of *Gliricidia sepium* on maize and cowpea under laboratory and field conditions. The growth of maize reduced significantly in response to the aqueous leachates of leaf prunings of the tree. Under field conditions, leaf chlorosis in both the target species occurred when sown in soil mulched with leaf prunings of *G. sepium*. Maize was more susceptible than cowpea. However, when the prunings are mulched a week before sowing of crops, all phytotoxic effects were eliminated. Ramamoorthy and Paliwal (1993) identified various phenolic allelochemicals viz. gallic acid, protocatechuic acid, *p*-hydroxybenzoic acid, gentisic acid, β-resorcyclic acid, vanillic acid, syringic acid, *p*-coumaric acid, *m*-coumaric acid, *o*-coumaric acid, ferulic acid, sinapic acid (both trans- and cis-forms), coumarin, and myricetin from the extracts of the tree. These adversely affected the growth of *Sorghum vulgare* under laboratory conditions. In the *Sorghum* fields, application of different quantities of *G. sepium* leaf mulch (viz. 400, 800, and 1200 g m^{-2}) effectively controlled weeds and improved the total yield of *Sorghum*.

From the above discussion pertaining to allelopathic interactions of trees with crops under agroforestry systems, the following important conclusions can be drawn:

- Allelopathy could be an important mechanism of growth retardatory effects of some trees on the associated crops under simultaneous agroforestry. However, it should be separated from the competition.
- While assessing allelopathic effects of the trees, suitable field experiments should be designed to demonstrate this under realistic conditions. It is important because in many cases a species showing phytotoxicity under laboratory conditions fails to do so under field conditions.
- Efforts should be made to manage and reduce the amount of phytotoxic litter or plant residues that fall on the soil surface and bring about undesirable effect on the crops due to released allelochemicals.
- Mulches and tree prunings placed on the agricultural soil with a view to get positive effects on the crops should be carefully monitored as these might release large quantities of allelochemicals through leachation or decomposition.

3.4 PRACTICAL UTILITY OF ALLELOPATHY FOR WEED AND PEST MANAGEMENT UNDER AFS

Although allelopathic interactions of some agroforestry trees may be harmful to crops, it could be suitably manipulated to reduce pest and weed populations and help in enhancing crop productivity. Trees could serve as storehouses of useful chemicals and provide a continuous bulk resource because of their perennial nature (Birkett et al., 2001). There are a number of reports highlighting that the phenomenon of allelopathy or the allelochemicals could be exploited for the management of weeds and pests (Duke et al., 2002; Singh et al., 2003). This could be attributed to various reasons:

1. Allelochemicals involved in the phenomenon of allelopathy are natural plant products and are thus biodegradable. Their use for the weed and pest management would thus be environmentally safe unlike synthetic herbicides that are fraught with various toxicological implications on human health and environment as a whole.
2. Allelochemicals found in some tree species may serve as invaluable sources of lead compounds for the synthesis of new herbicides. Availability of the trees in the farmland thus provides a basis for bioprospecting. Further, the large size of trees and their perennial nature may provide greater quantity of these chemicals for a longer time.
3. Allelochemicals possess novel target sites of action different from synthetic herbicides and can be tapped for synthesis of bioherbicides under biorational approach.

Rizvi et al. (1999) reviewed the prospects of allelopathy of agroforestry tree species for the management of weed and pests like pathogens, insects, and nematodes. Several allelochemicals or phytotoxic principles have been identified from trees that possess pesticidal potential and examples include ailanthone, a quassinoid from *Ailanthus altissima* (Mill.) Swingle, caffeine from *Coffea arabica* L., mimosine from *L. leucocephala*, azadirachtin from *Azadirachta indica* A. Juss., and volatile monoterpenes like cineoles, citronellal from *Eucalyptus* spp. (Rizvi et al., 1999; Singh et al., 2003).

Use of tree prunings as mulch on the companion crops under agroforestry system is an important area of research in the tropics. Kamara et al. (2000a) studied the effect of mulches from the woody fallow species like *L. leucocephala*, *Gliricidia sepium*, and *Senna siamea* on the weed composition, biomass, and grain yield of maize. Both *G. sepium* and *S. siamea* significantly reduced weed density and biomass compared with the unmulched control plots and plots mulched with *L. leucocephala*. Less weed smothering effect of *L. leucocephala* was attributed to the fast decomposition of its litter compared with the other trees. However, the study did not provide any evidence of allelopathy.

Hong et al. (2003) screened a number of weeds, shrubs, and trees for their allelopathic effects to use them as mulch in the agricultural land. They found that trees like *Morus alba* L., *Melia azedarach* L., and *Leucaena glauca* Benth. were very promising. Later, Hong et al. (2004) reported that *Morus alba* and *L. glauca* applied in the paddy fields at a rate of 2 ton ha^{-1} significantly reduced weed density and enhanced rice productivity owing to their allelopathic effects and can thus serve as an alternate source of weed management in the paddy fields. Likewise, mulching of *Melia azedarach* applied in the paddy fields at the rate of 1 ton ha^{-1} reduced weeds by 90% and can thus be used for weed management (Hong et al., 2004).

One of the best utilities of allelopathic agroforestry trees could be to exploit this property for the control of parasitic weeds like *Striga*, *Orobanche,* and *Cuscuta*, which are very difficult to control. Allelopathy could be included as one of the approaches under integrated weed management strategies to manage these parasitic weeds (Singh et al., 2003). Marley et al. (2004) used seed and leaf powders of neem (*A. indica*) and fruit and fruit peel of *Parkia biglobosa* (Jacq.) R. Br. ex G. Don. under screen house and field conditions to manage purple witchweed (*Striga hermonthica* [Del.] Benth.) in the fields of *Sorghum bicolor*. Neem powder was more effective in reducing the emergence of *Striga* than the products of *Parkia*. It resulted in a significant increase in the grain yield of *Sorghum*. Neem alone can be used for weed control, especially the parasitic weeds like *Cuscuta, Orobanche*, and *Striga* (Malkomes, 2006). However, its effect on the nontargets should be thoroughly investigated before use.

The essential oils from the lemon-scented eucalypt (*E. citriodora*) possess species selectivity and could serve as an excellent source of bioherbicides (Batish et al., 2004). These were found to suppress the germination of ragweed parthenium (*Parthenium hysterophorus* L.)—a noxious weed of waste and arable lands in India and several other countries (Kohli et al., 1998). Batish et al. (2004) reported that oils from lemon-scented eucalypt not only suppressed germination of weeds but also adversely affected 4 week old plants of *Cassia occidentalis* L. and *Echinochloa crus-galli* (L.) P. Beauv. In both these plants, the spray treatment of eucalypt oil caused significant visible injury and adversely affected photosynthetic and respiratory machinery. At low doses of the eucalypt oil, though enormous visible injury coupled with physiological changes were evident, the plants recovered 3 weeks after the treatment. Higher concentrations were, however, more injurious and caused complete mortality even a day after the treatment and showed no signs of recovery (Batish et al., 2004). Later, these essential oils were tested against *P. hysterophorus* and a similar effect was observed on the 4 week old plants spray treated with the aqueous solution of eucalypt oil (Singh et al., 2005). The plants exhibited significant injury in the form of wilting and necrosis followed by rapid leakage of ions showing thereby that oil affects membrane integrity. The essential oils of lemon-scented eucalypt not only possess herbicidal potential but these were also found to suppress the growth of phytopathogenic fungi harmful to important agricultural crops like wheat and rice (Ramezani et al., 2002).

Xuan et al. (2006) reported that mimosine—a nonprotein amino acid and a major constituent of *L. leucocephala*—suppresses germination and growth of weedy species like *Mimosa pudica* L. and *Bidens pilosa* L. and thus could be used as a bioherbicide. Gworgwor (2007) reported that trees could serve as a sustainable and efficient alternative to control weeds in millet crops. *Faidherbia albida* (Del.) A. Chev. was found to inhibit the growth of *Striga hermonthica*—a serious parasitic weed in millet fields. Not even a single emergence of the weed was seen under the tree (Gworgwor, 2007).

3.5 SOME RECOMMENDATIONS AND FUTURE DIRECTIONS

Keeping in mind the allelopathic interactions of tree species with crops under agroforestry systems, efforts should be made to minimize their negative effects on the associated crops and to manipulate them for practical use. This could be achieved by suitable combinations of allelopathic trees and crops to get the desirable effects. Even the mulch and residues of allelopathic tree species can be managed to reduce weed populations. One of the recent approaches of using allelopathy as a tool for weed management is by exploiting signaling effects of potential allelochemicals in the rhizosphere (Birkett et al., 2001). *Desmodium* species (*Desmodium unicinatum* DC. and *Desmodium intorum* [Mill.] Urb.) intercropped in maize crop suppressed parasitic weed *Striga hermonthica* and repelled insect pests, particularly stem borers, and it was attributed to the allelopathy of *Desmodium* spp. (Khan et al., 2002). However, further research in this direction is required. Efforts should also be made to understand the role of roots in various underground interactions. Roots could also release allelochemicals through exudates, leachation, or decomposition on death and decay. This aspect is highly neglected in the allelopathic studies, and further efforts are required to understand belowground interactions involving roots.

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4 Approaches to Tree–Environment–Crop Interactions

Ramun M. Kho

CONTENTS

4.1 INTRODUCTION

By integration of trees in farmland and rangeland, agroforestry (AF) has in general three objectives (Anderson and Sinclair, 1993). First, increasing total productivity, that is, increasing output of valuable products per unit of land and labor. Second, increasing stability, that is, reduced sensitivity to short-term fluctuations by spreading of risk through species (microsite and product) diversity. Third, increasing sustainability, that is, maintaining long-term productivity by protection of the resource base. In the 1980s, when AF came to the attention of researchers, policy makers, and extensionists, these benefits were often taken for granted based on a superficial comparison of monocultures to natural ecosystems. However, frequently, yields in AF systems reduced instead of increased (Sanchez, 1995). From the early 1990s, this stimulated research on the biophysical interactions in AF systems. The main question to be answered was: given a specific situation (soil, climate, and topography), which particular AF technology (tree and crop species, tree–crop arrangements in time and space and management) likely realizes certain benefits? This chapter focuses on the biophysical aspects related to especially the first objective of AF (Section 4.2).

Even if increased productivity is not the main objective, interaction research on (crop) productivity remains necessary, because trees influence crop production anyway and a reduced production is in general not acceptable to farmers.

Interaction refers to the influence of one or more components of a system on the performance of another component and of the overall system. A two-way interaction involves two components (e.g., the effect of the tree component on production of the crop). A three-way interaction involves three components (e.g., the effect of the tree on crop production depends on the environment). Table 4.1 aims to give a general overview of the approaches to interaction research.

The early approaches to AF interaction research (Section 4.3) were strongly empirically oriented. Ideas and concepts from related disciplines such as community ecology and intercropping were used. Several types of interaction were distinguished based on the net result of the component species (Anderson and Sinclair, 1993). Effects on other components were separated into positive (production enhancing or "fertility") effects versus negative (production decreasing or "competi-

TABLE 4.1
Overview of Approaches to Tree–Environment–Crop Interactions

Approach	Separating Simple Tree Effects	Resource Capture (Modeling)	Resource Balance
Rationale	Trees influence the crop through positive and negative effects The overall tree effect on crop production must be the difference between the positive (i.e., "fertility") and negative (i.e., "competition") tree effects	Trees influence the crop through altering the crop's capture of the limiting resource Biomass production must be the product of the capture of the limiting resource and the conversion efficiency (ε) of captured resource into biomass	Trees influence the crop through altering the balance of available resources The relative contribution of a net tree effect on availability of a resource (T_i) must increase with the degree of limitation of that resource (L_i)
Key equation	$I = F - C$	$W = \varepsilon_{\text{conversion}} \cdot \text{Capture}$	$I = \sum_{i=1}^{n} L_i \cdot T_i$
Research methods	Quantification of effects by field experiments with: • Mulch transfer • Soil transfer • Pruning of branches • Root barriers	Detailed process measurements and modeling of (tree effects on) resource captures and conversion efficiencies	Evaluation of the equation by field experiments with changed resource limitations (e.g., by fertilization)
Main problems	Positive and negative tree effects are highly site specific and change with the environment; therefore the predictive power is limited	Integration of (sub-models of) different resources is problematic because many key component processes are not yet properly understood; heavy parameterization requirements and lack of detailed data to test models	Of most environments the degree of limitation of the resources is not yet accurately known
References	Ong (1995) Sanchez (1995) Corlett et al. (1992)	Ong and Huxley (1996) Cannell et al. (1996) Muetzelfeldt and Sinclair (1993) Lawson et al. (1995)	Kho (2000a, 2000b) Kho et al. (2001)

tion") effects (e.g., Nair, 1993). The negative effects were often separated into "aboveground" competition effects (for light) and "belowground" competition effects (for water and nutrients). Quantification of these different effects was believed to give insight. However, the sizes of these simple tree effects are highly site specific and change with the environment. This limits the predictive power of this approach.

Recognizing the need for a more mechanistic understanding, AF and intercropping researchers (Ong and Black, 1994; Ong and Huxley, 1996) moved to a more theoretical (modeling) approach. They studied the resource capture and the conversion (also called "utilization") efficiency of captured resource into biomass in mixtures compared with sole crop systems (Section 4.4). This has given a wealth of theoretical insights, but a drawback of this approach is that it requires expensive measurements (mainly because of costly equipment) and that it is focused on capture and utilization of only one specific resource that is postulated to be "major limiting."

AF as a science had difficulty with the translation of theoretical insights into practical management options (Rao et al., 1998). Especially tools to extend AF technologies to other circumstances and environments were missing. Recognizing that the balance of available resources is a major factor determining crop production (Kho, 2000a) and that trees influence crop production through altering this balance (Kho, 2000b), Kho et al. (2001) proposed an approach quantifying (tree effects on) this balance (Section 4.5).

4.2 MEASURING THE YIELD ADVANTAGE OF AF SYSTEMS

A yield advantage in AF and intercropping systems occurs if the mixture produces more yield from an area of land than can be obtained by dividing that area into pure stands. It is most frequently (Vandermeer, 1989; Ong, 1996) quantified by the land equivalent ratio (LER), which is defined as the relative land area in pure stands that is required to produce the yields in mixture. If LER > 1, then the mixture is more advantageous than separate monocultures. The LER is estimated as the sum of the relative yields, the yields in mixture expressed relative to the yields in monoculture:

$$\mathrm{LER} = \frac{\mathrm{Yield}_{\mathrm{species\ 1\ in\ mixture}}}{\mathrm{Yield}_{\mathrm{species\ 1\ in\ monoculture}}} + \frac{\mathrm{Yield}_{\mathrm{species\ 2\ in\ mixture}}}{\mathrm{Yield}_{\mathrm{species\ 2\ in\ monoculture}}}, \tag{4.1}$$

where Yield is in units product per unit area. For example, 1 ha of an AF system yields 40 units of product from the annual crop and 10 units of tree products. If 1 ha of the annual crop in monoculture yields 60 units and 1 ha of a pure tree stand yields 20 units of tree products, the LER equals: $\mathrm{LER} = 40/60 + 10/20 = 7/6$.

There are several pitfalls when using Equation 4.1. Yield strongly depends on the stand density. Consequently, the outcome of Equation 4.1 depends on the chosen stand density of the respective monocultures. Two extremes can be distinguished. In replacement series (or substitutive) designs total stand density is kept constant. By replacing each individual of one of the species in the mixture by the other, the monoculture is constructed. This leads to the maximum stand density of the monoculture that is reasonable to choose and to a lower estimate of the LER. The LER calculated this way is also called the relative yield total (RYT), which is only valid for the particular total stand density. In additive designs stand density for each species is kept constant. By adding all individuals of the monocultures together on the same piece of land, the mixture is constructed. This leads to the minimum stand density of the monoculture that is reasonable to choose and to an upper estimate of the LER. In a replacement series design, probably more plants would be grown in monoculture than is optimal. In an additive design, probably fewer plants are grown in monoculture than is optimal. If the design is not explicitly stated, it is usually assumed that stand densities of the monocultures were at their optimum (Vandermeer, 1989; Ong, 1996). Vandermeer (1989) illustrates a statistical pitfall using LER because of the fact that the LER is a sum of ratios of yields that are often lognormal distributed. Ong and Black (1994) mention a pitfall when using LER for the analysis of resource capture.

In spite of its pitfalls, the LER is a useful concept for intercropping. For AF, however, there are several reasons why LER is not used. First, farmers are often not concerned about maximizing both tree and crop components, but about maximizing the annual crop production at an acceptable growth of the tree component. Second, a sole tree comparison is often not available. Investing time and costs to determine the optimal tree density in monoculture only for the sake of having a good estimate of the LER is difficult to justify. Third, in AF systems, one component is dominant and perennial. The trees may have a big impact on crop performance, whereas the crop has little effect on tree performance. These reasons suggest evaluating the yield advantage with the yield of the crop component separated from that of the tree component. A measure frequently used is the difference in crop yields relative to the sole crop yield (Sanchez, 1995; Ong, 1996; Rao et al., 1998; Kho, 2000b):

$$I = \frac{\text{Yield}_{\text{crop in AF system}} - \text{Yield}_{\text{crop in monoculture}}}{\text{Yield}_{\text{crop in monoculture}}}, \qquad (4.2)$$

where Yield is again in units product per unit area. If $I > 0$, then the AF system has a yield advantage regarding crop production. Equation 4.2 defines I as a fraction of sole crop yield. If I is used as a percentage, Equation 4.2 must be multiplied by 100%. If the AF system yields valuable tree products as poles, fruits, fuelwood or fodder, these are simply reported in addition to I.

4.3 SEPARATING SIMPLE TREE EFFECTS

For a long time, ecologists have studied plant interactions in natural ecosystems. It has led to various classifications of interactions (see e.g., Anderson and Sinclair, 1993). The classifications are phenomenological (descriptive) in nature, that is, the net result of the interaction is leading. The net result of each species can be positive, nil, or negative. Following this phenomenological thought, the next step is to separate the overall tree effect into a positive (also called "fertility") effect and a negative ("competition") effect. This is the approach at the start of tree–environment–crop interaction research. It is formalized by the equation (Ong, 1995; see also Sanchez, 1995):

$$I = F - C, \qquad (4.3)$$

where

I is the overall interaction, that is, the percentage net increase in crop yield attributable to the presence of trees, compared with sole crop yield (cf. Equation 4.2)

F is the fertility effect, that is, the percentage of crop yield increase attributable to favorable effects of the trees on soil fertility and microclimate

C is the competition effect, that is, the percentage crop yield decrease attributable to competition with the trees for light, water, and nutrients

In an alley-cropping experiment, the equation is quantified by four treatments (Sanchez, 1995):

Co = sole crop
Cm = sole crop + mulch from pruned trees
Ho = crop + tree with mulch removed
Hm = crop + tree with its mulch (normal AF technology)

Sanchez (1995) concluded that long-term trials with these "four key treatments" need to be conducted in AF systems. He proposed to estimate F by Cm − Co and C by either Hm − Cm or

TABLE 4.2
A 2^2 Factorial Experiment to Separate Fertility and Competition Effects

	No Mulch (o)	With Mulch (m)	Difference
Absence of other tree effects (C)	Co = sole crop	Cm = sole crop + mulch	“Fertility” (i.e., mulch) effect
Presence of other tree effects (H)	Ho = AF with mulch removed	Hm = AF + its mulch	
Difference	“Competition” (i.e., other tree) effect		

$Ho - Co$. However, careful examination of the four treatments shows that they are nothing else than the 4 treatments of a 2^2 factorial design (Table 4.2), with the factor mulch (at levels absence and presence) and the factor “other tree effects” (also at levels absence and presence). The experiment can thus be analyzed by standard statistical methods (i.e., two-way analysis of variance), and it only separates the mulch effect (F) from other tree effects (C).

In addition, the separation in “aboveground” and “belowground” interactions was believed “to give a sound basis for studying the processes involved” (Nair, 1993).

Field experiments to separate the light competition effect include a comparison of sole crop treatments with and without artificial shade (using shade cloths) and AF system treatments with and without pruning of branches. These 4 treatments can again be structured according to a 2^2 factorial design (Table 4.3) and accordingly analyzed.

Root competition was experimentally separated by vertical polythene barriers or trenches (Corlett et al., 1992). Because a sole crop treatment with invaded tree roots (Table 4.4) is normally not done, these experiments miss the power of a 2^2 factorial design.

The results of these experiments are all highly site specific. Sanchez (1995, Table 4.1) has shown a dataset of long-term alley-cropping experiments with the four mulch transfer treatments of Table 4.2 in different climates and at different soils. The competition and fertility effects vary heavily (from -77% to 0% and from 0% to $+58\%$ of sole crop yield) without any correlation with soil or climate. The predictive power of this approach (when extrapolating the AF technology to other types of environment) is therefore limited, and the practical value of the gained insights is not clear.

Recognizing the shortcomings of Equation 4.3, Ong (1996) has extended it to the tree–crop interaction equation:

$$I = F - C \pm P + L, \tag{4.4}$$

where

F is the effect on soil fertility and microclimate of the soil surface
C is competition as defined earlier
M is the effect on aboveground microclimate

TABLE 4.3
A 2^2 Factorial Experiment to Separate Light Competition Effects

	No Light Comp.	With Light Comp.	Difference
Absence of other tree effects	Co = sole crop	Cm = sole crop + art. shade	Light competition effect
Presence of other tree effects	Ho = AF with pruning of branches	Hm = AF + its shade	
Difference	Other tree effects		

TABLE 4.4
A 2^2 Factorial Experiment to Separate Root Competition Effects

	No Root Competition	**With Root Competition**	**Difference**
Absence of other tree effects	Sole crop	Sole crop with invaded tree roots	Root competition effect
Presence of other tree effects	AF with root trenches or barriers	AF + its root competition	
Difference	Other tree effects		

P is the effect of changes in soil properties
L is the reduction of losses of nutrients and water

All effects are expressed as percentage of the sole crop. As noted by Ong (1996), estimates of M, P, and L are not easily separated from F and C and are difficult to obtain. He suggests estimating these effects indirectly by a combination of process measurements and interpretation of the processes involved. In other words, Ong (1996) suggests to move from a pure empirically approach to a more theoretical (modeling) approach (see Section 4.4).

Rao et al. (1998) adapted Equation 4.4 to

$$I = F + C + M + P + L + A, \tag{4.5}$$

where

F refers to effects on chemical, physical, and biological soil fertility (without the microclimate component)
C to competition for water, light, and nutrients
M to effects on microclimate
P to effects on pests, diseases, and weeds
L to soil conservation
A to allelopathy effects

As noted by the authors, many of these effects are interdependent and cannot be experimentally estimated independent of one another.

Equation 4.5 gives a categorization of, and is a framework for, the various “simple” tree effects (see Rao et al., 1998; Table 4.1). However, the interdependence, although recognized, is not specified in the equation. It is thus not clear how to quantify each category in order that their sum equals the overall effect I. This makes the equation of doubtful value for determining the relative importance of each factor for a given system. Does Equation 4.5 give the right categories that should be aimed at to quantify? Another drawback is that the dependence with the environment (soil and climate) is implicitly contained in each category, but not explicitly specified in the equation. As a consequence, it is questionable whether quantification of Equation 4.5 helps to determine the yield advantage of a particular AF technology in another environment. Quantification of Equation 4.5 for one particular technology and one specific environment is a major research effort; so more cost-effective approaches to get predictive understanding would be appropriate.

4.4 RESOURCE CAPTURE APPROACH

Cannell et al. (1996) reinterpreted Equation 4.3 in terms of resource capture. They noticed that the fertility (F) and competition (C) effects may be interdependent over time, because part of the C effect in a season may contribute to the F effect in the following seasons. To clean up the overlap, the authors introduced the concepts F_{noncomp} (resources acquired by the trees that the crop would not otherwise acquire) and $C_{\text{comp, nonrecycled}}$ (resources that the crop is deprived of which are used in tree growth and are not recycled). The overall interaction I would be the difference between these two. Under the premise of a strict law of the minimum, they could reduce the majority of the biophysical AF hypotheses given by Sanchez (1995) to this central tenet. The original hypotheses as well as the application of this central tenet postulate one resource (light, water, or nutrients) to be the "major limiting" resource. The introduced concepts are difficult to quantify and it is not clear how to translate the theoretical insight into practical management options.

Biomass production (W) can be described by (Monteith et al., 1994; Ong et al., 1996):

$$W = \varepsilon_{\text{conversion}} \cdot \text{Capture}, \tag{4.6}$$

where

Capture is the capture of a specific resource in the course of the growing season
$\varepsilon_{\text{conversion}}$ is the efficiency to converse the captured resource into biomass

The conversion efficiencies are mostly considered species specific and conservative, but Kho (2000a) has shown that this is most likely only the case within the set of environments with the same balance of available resources.

In this section, the main processes of the capture of the resources light, water, and nutrients are described. For each resource, this is first done in very general terms for whole vegetation, after which adjustments are made for a crop in an AF system. Tree effects on the most important parameters are indicated. The aim of this section is to give a picture of this approach, not to give an in-depth, well-balanced, comprehensive, overall tree–environment–crop interaction model. The models for each resource are not all on the same level of detail, reflecting the different states of knowledge of the processes involved. Methods, techniques, and equipment for the measurement of process parameters are not discussed. The section ends with some implications for species choice.

4.4.1 LIGHT

Dry matter production (W) can be described by (Azam-Ali et al., 1994):

$$W = \varepsilon_{\text{s}} \int f\, S_0\, \mathrm{d}t, \tag{4.7}$$

where

ε_{s} is the conversion efficiency of captured photosynthetic active radiation (PAR) into biomass (g dry matter MJ^{-1} of captured PAR)
S_0 is the daily incident PAR (MJ m^{-2})
f is the fraction of incident radiation captured by the canopy

According to Beer's law, f is equal to

$$f = 1 - \mathrm{e}^{(-k \cdot \text{LAI})}, \tag{4.8}$$

where

k is the light extinction coefficient
LAI is the leaf area index (m^2 leaf area m^{-2} ground area)

The light extinction coefficient k depends on the inclination of the sun and the leaf (or green area) angle distribution. A (hypothetical) canopy with solely vertical leaves has a k close to zero, whereas a canopy with only horizontal leaves has a k equal to one (Campbell and Norman, 1987).

If light is the only limiting resource, that is, if the availability of water and nutrients is ample, the light conversion efficiency ε_{s} for a certain crop species is fairly constant at its maximum. However,

ε_s reduces considerable if light limitation decreases, that is, if another resource than light also becomes limiting (Azam-Ali et al., 1994; see also Kho, 2000a).

Equations 4.7 and 4.8 show that increased k and LAI increases total light capture fS_0. If the conversion efficiency ε_s does not alter (i.e., if the limitations of water and nutrients do not change), this results in increased biomass production. LAI can be increased in an AF system by a denser green canopy cover or quicker development of that cover. Total biomass production is also increased by a greater integration period in Equation 4.7, which can be extended by a longer period that the ground is covered by a green canopy (e.g., in the case of temporal complementary).

More specifically, in a mixture of a tree and a crop species, light capture may be described by stratifying the canopy into horizontal layers. If these layers are numbered from top to bottom, the fraction of radiation captured in canopy layer h equals:

$$f_h = 1 - \mathrm{e}^{(-k_{h,\mathrm{tree}} \cdot \mathrm{LAI}_{h,\mathrm{tree}} - k_{h,\mathrm{crop}} \cdot \mathrm{LAI}_{h,\mathrm{crop}})}, \tag{4.9}$$

where $k_{h,\mathrm{tree}}$, $k_{h,\mathrm{crop}}$, $\mathrm{LAI}_{h,\mathrm{tree}}$, and $\mathrm{LAI}_{h,\mathrm{crop}}$ are the extinction coefficients and leaf area indices in layer h of the tree and the crop. The radiation passing layer h, reaching layer $h + 1$ equals

$$S_h = S_{(h-1)} \cdot (1 - f_h) \tag{4.10}$$

Combining Equations 4.7, 4.9, and 4.10, crop dry matter production in an AF system can thus be described by:

$$W_{\mathrm{crop}} = \varepsilon_{\mathrm{s,crop}} \int \sum_h f_{h,\mathrm{crop}} S_{(h-1)} \mathrm{d}t, \tag{4.11}$$

where

$\varepsilon_{\mathrm{s,crop}}$ is the conversion efficiency of the crop
$f_{h,\mathrm{crop}}$ is the fraction of radiation captured by the crop in canopy layer h

As a result of root competition, trees may reduce the availability of water and/or nutrients to the crop and therefore decrease light limitation and $\varepsilon_{\mathrm{s,crop}}$. However, the processes involved are not yet properly understood and mostly an empirical estimate of the conversion efficiency is used in Equation 4.11.

4.4.2 WATER

Unlike light, water and nutrients can be stored in the system. Competition for light can be only direct, that is, it is instantaneous. If the resource is not captured it is lost. Competition for water and nutrients can be direct and indirect (Kropff and van Laar, 1993). The competition is direct if the resource is limiting; the plant with the highest demand and the best access to the limiting resource (the deepest rooting system and the highest root length density) has an advantage. Competition can also be indirect: if the resource is in ample supply, all species can meet their resource requirements, but the amount of resource in the soil will be reduced. This affects the growing situation later in the season, when the resource might become limiting.

Such as biomass production as function of light capture, biomass production as function of transpiration (water uptake or water capture) can be described by the product of a conversion efficiency and the amount of captured resource (Ong et al., 1996):

$$W = \sum W_{\mathrm{actual}} = \varepsilon_{\mathrm{w}} \sum E_{\mathrm{actual}}, \tag{4.12}$$

where

ε_w is the conversion efficiency of transpired water into biomass (g mm^{-1} transpired H_2O)

ΣW_{actual} and ΣE_{actual} are the actual biomass production and actual transpiration accumulated over the growing season

If water is ample, actual transpiration equals potential transpiration. The stomata are fully open and biomass production is on its potential level as described by Equations 4.7 and 4.11. If water is limiting, stomata tend to close, increasing the diffusive resistances for water vapor movement (from stomatal cavities to the ambient air) but also for carbon dioxide movement (from the ambient air to the stomatal cavities). The relative reduction in growth rate is then more or less proportional to the relative reduction in transpiration rate (Kropff and van Laar, 1993):

$$\frac{W_{\text{actual}}}{W_{\text{potential}}} = \frac{E_{\text{actual}}}{E_{\text{potential}}}. \tag{4.13}$$

Combining Equations 4.12 and 4.13 shows that the conversion efficiency equals

$$\varepsilon_w = \frac{W_{\text{potential}}}{E_{\text{potential}}}, \tag{4.14}$$

where

$W_{\text{potential}}$ is the biomass production when water is not limiting (Equation 4.7 for sole crops and Equation 4.11 for a crop in an AF system)

$E_{\text{potential}}$ is the potential transpiration that is determined by the energy balance

4.4.2.1 Water Demand

The (latent heat loss used for) transpiration of a crop can be approximated by a modified version of the Penman–Monteith equation (Wallace, 1996):

$$\lambda E_{\text{crop}} = \frac{\Delta f_{\text{crop}} R_n + \rho c_p \frac{D}{r_{\text{a,crop}}}}{\Delta + \gamma\left(1 + \frac{r_{\text{s,crop}}}{r_{\text{a,crop}}}\right)}, \tag{4.15}$$

where

E_{crop} is the (potential) transpiration by the crop (kg m^{-2} s^{-1})

λ is the latent heat of vaporization of water (2.454×10^6 J kg^{-1})

Δ is the slope of saturated vapor pressure curve at air temperature (see Equation 4.17)

f_{crop} is the fraction of net radiation absorbed by the crop canopy (cf. Equations 4.8, 4.9, and 4.11)

R_n is the above canopy net radiation (W m^{-2})

ρ is the density of the air (1.204 kg m^{-3})

c_p is the specific heat capacity of air (1010 J kg^{-1} K^{-1})

D is the saturation vapor deficit (see Equation 4.16)

γ is the psychrometric constant (0.0662 kPa K^{-1})

$r_{\text{s,crop}}$ is the surface (or stomatal) resistance (about 60 s m^{-1} for most types of well-watered vegetation, that is, potential transpiration; Monteith, 1991)

$r_{\text{a,crop}}$ is the aerodynamic boundary layer resistance (roughly from <10 to >100 s m^{-1}; Penning de Vries et al., 1989) strongly dependent on wind speed and canopy height and architecture

Similar equations can be used for transpiration by the tree component and for evaporation from the bare soil (only with different f, r_s, and r_a).

The saturation vapor deficit D (kPa) equals

$$D = e_s - e_a, \tag{4.16}$$

where

e_a is the actual vapor pressure (kPa)
e_s is the saturated vapor pressure at air temperature

It can be approximated with the empirical function (Goudriaan, 1982)

$$e_s = 0.611 \cdot e^{\left(17.4\frac{T}{T+239}\right)}, \tag{4.17}$$

where

T is the air temperature in °C
Δ is the slope of this curve at air temperature (i.e., $\mathrm{d}e_s/\mathrm{d}T$)

The relative humidity equals $\mathrm{RH} = 100\%\ e_a/e_s$, so D can be calculated as: $D = (1-\mathrm{RH}/100)\ e_s$.

The tree component may have the following effects on the microclimate of the crop component in AF systems:

1. Reduction of f_{crop} and thus of the net radiation absorbed by the crop canopy
2. Reduction of air temperature, of the saturation vapor deficit D and of the slope of the saturation vapor pressure curve Δ
3. Reduction of wind speed and (thus) increased aerodynamic boundary layer resistance $r_{\mathrm{a,crop}}$

These effects result in lower potential transpiration (Equation 4.15) by the crop. On the other hand, by interactions via light and nutrients, trees may increase or decrease the potential production of the crop at ample water supply $W_{\mathrm{potential}}$, so that the net effect on the transpiration efficiency ε_w (see Equation 4.14) is not straightforward.

4.4.2.2 Water Supply

If water is sufficient, water uptake follows (potential) transpiration. If there is a shortage of water, stomata tend to close through loss of turgor, so (actual) transpiration follows water uptake (Penning de Vries et al., 1989). Given a certain rooted depth, water uptake is the result of the water demanding force on the one hand (i.e., potential transpiration; Equation 4.15) and soil moisture suction on the other. The latter depends on the water content in the soil and soil characteristics, notably texture, bulk density, and organic matter content. These soil characteristics determine the water retention (or pF) curve, the relation between volumetric water content θ (e.g., $\mathrm{cm}^3\ \mathrm{H_2O}\ \mathrm{cm}^{-3}$ soil), and the pF, that is, the logarithm (base 10) of the absolute soil moisture suction. Water available to the plant is roughly between field capacity (pF = 2.0, suction 100 cm) and permanent wilting point (pF = 4.2, suction 16,000 cm). At field capacity plants do not suffer from water stress and uptake is determined by demand (Equation 4.15). At permanent wilting point and beyond, plants cannot extract soil water. Trees may reduce the bulk density and increase the organic matter content, resulting in an increased capacity to store water between these two limits (i.e., the available water capacity).

The water content of the soil depends on the soil water balance:

$$P - I_p = R + E_{\mathrm{actual}} + E_{\mathrm{soil}} + D_p + \delta\theta, \tag{4.18}$$

where
P is the gross precipitation
I_p is amount intercepted by the canopy
R is the runoff
E_{actual} is the actual transpiration
E_{soil} is the evaporation directly from the soil
D_p is the drainage to below the root layers
$\delta\theta$ is the increase in soil water storage in the root layers (all in, e.g., mm day^{-1})

Regarding an AF system, all terms except P can be split up into a tree component and a crop component (Wallace, 1996).

Trees influence the water balance in the crop root layer by

1. Reducing the net input of rainfall by interception by the tree canopy. Wallace (1996) estimated annual interception loss in AF systems between 3% and 10% of rainfall.
2. Reduction of the runoff by reduced surface crusting and better soil hydraulic conductivity resulting from increased plant residues into the soil, and by reducing the surface slope in case of, for example, contour hedgerows.
3. Uptake of water through transpiration by the tree itself (Equation 4.15 with different f, r_s, and r_a).
4. Reduction of evaporation from the soil through increased shading. In most sole crop systems, a considerable fraction of the rain is lost by soil evaporation, mainly because the ground is not fully covered during the season.
5. Reduction of drainage by increasing available water capacity.

Below the crop root layer, trees may root in a deeper layer. Rooted depth (d) can be defined as the depth from which the plant effectively extracts water (i.e., a density of at least 0.10 cm root length cm^{-3} of soil volume; Penning de Vries et al., 1989). If trees root deeper than the crop, they satisfy a part of their water demand from this deeper layer. The demand on each layer, provided that water is ample and that the layers are equally moist, has been modeled proportional to the thickness of the layer relative to the rooted depth (Penning de Vries et al., 1989). The determinants concerning indirect competition are thus (1) from the demand side the share in absorbed radiation (f_{tree} and f_{crop}; see Equation 4.15) and (2) from the supply side the relative rooted depths (d_{crop}/d_{tree}). If water becomes limiting, root length density (l, in cm root length cm^{-3} soil) is a major factor determining uptake (Gregory, 1994). Thus, in case of direct competition, the share in the total root system, that is, the relative root length density ($l_{tree}/(l_{tree} + l_{crop})$) in common soil layers becomes increasingly important too. Outside the cropping season, trees may utilize remaining soil water.

4.4.3 NITROGEN AND OTHER NUTRIENTS

The mechanistic understanding of processes related to availability, uptake, and use of nutrients is not at the detailed level as that of light and water. On the other hand, the timescale to be considered must be longer than that related to light and water. For, concerning light, the interaction only consists of a competition that is direct. Concerning water, the interaction consists of direct and indirect competitions as well as of a direct beneficial effect through improved microclimate and an indirect beneficial effect through improved soil physical properties. But concerning nutrients, the interaction consists of direct and indirect competitions as well as of direct and indirect beneficial effects, of which important ones are through nutrient recycling and reduced erosion. Unlike water, all nutrients taken up are stored (incorporated) in the plant tissue. A (temporarily) shortage in uptake does not immediately implies a reduction in growth, because the shortage can be compensated by a decrease of the nutrient content. Compared with other nutrients, nitrogen takes a special place because (1) it can be fixed from the atmosphere, (2) without biological fixation and without inorganic fertilizers, it only becomes available by mineralization of organic material, and (3) it leaches easily to depths below the crop root layer.

Concerning a specific nutrient, biomass production can again be described by the product of a conversion efficiency ε_N (kg dm kg^{-1} nutrient taken up) and captured nutrient ΣN_{actual} (kg/ha).

$$W = \varepsilon_N \sum N_{\text{actual}}. \tag{4.19}$$

If the nutrient is the only limiting resource, the plant makes maximum use of it (i.e., the nutrient is on its minimum concentration). The maximum conversion efficiency for a cereal is (Van Duivenbooden, 1995)

$$\varepsilon_{N,\max} = \frac{100}{\text{HI} \cdot N_{\text{min,grain}} + (1 - \text{HI}) \cdot N_{\text{min,straw}}}, \tag{4.20}$$

where

HI is the harvest index (grain dry matter/total aboveground dry matter)

$N_{\text{min,grain}}$ and $N_{\text{min,straw}}$ are the minimum nutrient contents (%) of grain and straw, respectively

If the nutrient is in ample supply, higher uptake does not lead to increased production, but to an increased nutrient content. Based on a review of hundreds of fertilizer experiments, Van Duivenbooden (1995) reports minimum and maximum N, P, and K contents of five major cereals (Table 4.5).

Trees may reduce the conversion efficiency by reducing the limitation of the nutrient (i.e., by increasing the availability of the nutrient in question or by decreasing the availability of other resources through competition or both; Kho, 2000a).

Palm (1995) has shown that in general (4 t ha^{-1}) tree prunings can easily meet N and Ca requirements of 2 t of maize (plus 3 t straw), just the crop Mg demand, hardly the K demand and not the crop P demand (compare Tables 4.5 and 4.6). So, on P-deficient soils, external sources of P are in general needed (Buresh and Tian, 1998).

TABLE 4.5
Minimum and Maximum N, P, and K Concentrations and Conversion Efficiencies of Five Major Cereals (Harvest Index)

	Millet (0.26)		Sorghum (0.27)		Maize (0.42)		Rice (0.44)		Wheat (0.41)	
	Min	Max	Min	Max	Min	Max	Min	Max	Min	Max
N grain (%)	1.47	2.35	1.26	2.02	1.21	1.87	0.97	1.36	1.62	2.65
N straw (%)	0.38	1.07	0.39	0.94	0.48	0.91	0.44	0.82	0.30	0.69
N $\varepsilon_{\text{conv}}$ (kg/kg)	71	151	81	160	76	127	95	149	67	119
P grain (%)	0.24	0.37	0.18	0.34	0.21	0.40	0.10	0.27	0.25	0.49
P straw (%)	0.05	0.13	0.03	0.12	0.03	0.14	0.05	0.19	0.03	0.08
P $\varepsilon_{\text{conv}}$ (kg/kg)	769	2000	833	3333	714	3333	526	2000	1250	3333
K grain (%)	0.39	0.63	0.25	0.46	0.20	0.53	0.22	0.54	0.33	0.66
K straw (%)	1.27	2.01	0.57	1.61	0.68	1.88	1.18	2.70	1.06	1.92
K $\varepsilon_{\text{conv}}$ (kg/kg)	50	79	62	175	53	147	37	85	52	94

Source: Adapted from Van Duivenbooden, N., *Land Use Systems Analysis as a Tool in Land Use Planning, with Special Reference to North and West African Agro-ecosystems*, Wageningen Agricultural University, Doctoral Thesis, Wageningen, 1995.

TABLE 4.6
Nutrients (kg) Contained in 4000 kg of Leaf Material

Species	N	P	K	Ca	Mg
Leucaena leucocephala	154	8	84	52	13
Erythrina poeppigina	132	7	46	61	—
Inga edulis (fertile soils)	142	11	40	45	6
I. edulis (infertile soils)	127	9	50	30	7
Senna siamea	105	6	44	110	7
Dactyladenia barteri	60	4	31	40	8
Grevillea robusta	52	2	24	60	7

Source: From Palm, C.A., *Agroforestry Syst.*, 30, 105, 1995. With permission.

Nutrients reach plant roots through mass flow and diffusion, so nutrient uptake is closely related to the water regime. In humid areas (when infiltration exceeds evapotranspiration) and for mobile nutrients such as nitrogen, rooting depth (d) and synchronization of supply (from fertilizer or mineralization) with demand are the important aspects of nutrient uptake (Van Noordwijk and De Willigen, 1991). Consequently, the relative rooted depths ($d_{\text{crop}}/d_{\text{tree}}$) determine competition. In dry soils and for poorly mobile nutrients such as phosphorus, root length density (l), root hairs, mycorrhiza and synlocalization of roots, and the nutrient source are major factors determining uptake. Consequently, the relative root length density ($l_{\text{tree}}/(l_{\text{tree}} + l_{\text{crop}})$) determines competition. As a result, the idea of a "safety net" of tree roots under the crop root layer, capturing leached nutrients, is thus probably of importance for nitrate in humid climates, but not for phosphorus or in semiarid regions (see also Breman and Kessler, 1995).

Leguminous tree species may add nitrogen to the system through atmospheric nitrogen fixation.

If %N in the organic material is <1.74 as in fresh legumes, net mineralization will occur immediately. If %N is >1.74 as in senesced leaves and many nonlegumes, net N immobilization will occur. Field trials showed that only 10%–20% of the added N in prunings is recovered by the first crop (Palm, 1995). Most of the N in prunings add to the buildup of soil organic matter and may benefit subsequent crops. The nutrient benefit to crops is thus more through the long-term buildup of soil organic matter rather than through direct release from decomposition.

If the tree prunings are not added to, but recycled within the system and crop products are harvested, there is a net loss of nutrients from the system. To sustain productivity in the long term, external nutrient inputs (except possibly nitrogen inputs in case of leguminous tree species) are essential to offset those losses.

4.4.4 IMPLICATIONS FOR SPECIES CHOICE

Farmers' common sense says that two species form a good combination if one "fits in" the space not occupied by the other. Insights gained with the resource capture approach support and refine this opinion.

Equations 4.9 through 4.11 and 4.15 show that concerning light and water competition (and thus in light- or water-limiting environments) especially tree species with many vertical leaves and a dense erect canopy shape (i.e., a small extinction coefficient k_{tree}) may be suitable in simultaneous AF systems. These tree species combine a high leaf biomass with a low fraction of intercepted and absorbed radiation, that is, a large potential supply of tree prunings with low light and water competition (cf. Van Noordwijk, 1996). In light-limiting environments, C3 crops are more appropriate than C4 crops (Kho, 2000b).

Tree–crop combinations with a large rooted depth of the tree relative to that of the crop will minimize indirect and direct competition for water and nutrients (especially nitrogen). This does not only mean that in water- or nitrogen-limiting environments deep-rooted tree species should be considered for AF systems, but also shallow-rooting crop species (provided that these are adapted to the agroecological zone). In nitrogen-limiting environments nonleguminous crops will give a higher yield advantage than leguminous crops (Kho, 2000b).

A small tree root density (cm root length cm^{-3} soil) relative to that of the crop in common soil layers will minimize direct competition (especially for phosphorus). So, in water- or phosphorus-limiting environments, tree species with a low root length density and crop species with a high root length density should be considered. Van Noordwijk and Brouwer (1991) give a review of root length densities in different soil layers for various crops.

4.5 RESOURCE BALANCE APPROACH

The goal of tree–environment–crop interaction research is to determine which particular AF technology likely realizes a certain benefit in a specific situation (soil, climate, and topography). On the basis of the other approaches, this section starts with discussing the desirable properties of a general tree–environment–crop interaction model to reach this goal. After that, such a model is constructed and its use is illustrated.

4.5.1 Desirable Properties of a General Tree–Environment–Crop Interaction Model

As any model, a tree–environment–crop interaction model has two requirements that push the model to opposite directions. On the one hand, it should give predictive understanding, that is, explain as much as possible of the total variation imposed by the goal. This may lead to complex models (Section 4.4). On the other hand it should be simple, that is, be plain in structure and parsimonious with factors and parameters involved (Section 4.3).

The net result *I* of a particular AF technology (e.g., the well-studied alley-cropping technology; see Sanchez, 1995; Kho, 2000b) may vary in different soils and climates from strongly negative to strongly positive. So, *I* must be a function of tree effects interacting with environmental effects (i.e., crop production is the result of a three-way interaction between the trees, the environment, and the crop). Although the scientists were undoubtedly aware of this, Equations 4.3 through 4.5 (Section 4.3) approach *I* as a two-way (tree–crop) interaction. On the left-hand side of these equations there is a measure on crop performance (*I*), but on the right-hand side there are only tree effects expressed as percentage of sole crop. Environmental effects (interacting with tree effects) on crop performance do not have an explicit place in these equations, but must be taken into account when converting the tree effects to percentage crop yields. For a more comprehensive understanding, it is desirable if on the right-hand side of the equation both tree effects and environmental effects would explicitly appear. Because we are looking at an interaction (i.e., a combined effect) we do not expect a sum of tree effects and of environmental effects, but a product of these. As discussed in Section 4.3 another desirable property is that the factors are not interdependent, but can be estimated independent from each other.

In the resource capture approach, dry matter production is modeled as the product of the amount of captured resource and the conversion efficiency of the specific resource. Often, a positive linear relation between dry matter production and resource capture was found, suggesting conservativeness of the conversion efficiency (Ong et al., 1996). The premise is that dry matter production is the "dependent" variable and the capture of the resource in consideration the "independent" variable. However, because increased resource capture is both a cause and a consequence of increased production, Kho (2000a) argued that the relation between capture and production is a correlation, not a causal relation. For, in the same system, the captures of other resources are also positively related to production. Resource captures are thus confounded with each other that strictly speaking invalidates the premise.

The conversion efficiency will be at its maximum if the resource is the only limiting resource in the specific environment (cf. Equation 4.20) and will decrease as the degree of limitation decreases. That is, the conversion efficiency decreases if the availability of the resource increases or the availability of other resources decrease or both (Kho, 2000a). The processes involved are not yet clearly understood. By postulating that the resource in consideration is the only limiting resource (according to the law of the minimum) and that all other resources are non-limiting, the models of Section 4.4 can be used with constant conversion efficiencies that were empirically determined. However, because in most environments crops respond to changed availability of several resources (De Wit, 1992), this postulation is a theoretical idealization that in reality is seldom true (Kho, 2000a). For an AF system this is quite relevant, because the trees simultaneously alter the availability of several resources. Each altered resource will thus cause crop response and changed conversion efficiencies. So limitation (and tree–environment–crop interactions) does not involve only one resource that, if saturated, is replaced by another resource. Limitation (as well as tree–environment–crop interactions) simultaneously involves several resources, each resource affecting conversion efficiencies and production in its own degree depending on the balance of available resources in the environment.

A tree–environment–crop interaction model thus should simultaneously consider all resources that may be altered by the tree and that might cause crop response.

4.5.2 GENERAL TREE–ENVIRONMENT–CROP INTERACTION MODEL

4.5.2.1 Characterization of an Environment

Kho (2000a) showed that for a particular species the specific environment can be effectively characterized by the degree of limitation of resources. For resource *i* (light, water, nitrogen, or another nutrient) it is defined as

$$L_i = \frac{\partial W}{\partial A_i} \frac{A_i}{W}, \tag{4.21}$$

where

W is dry matter production

A_i is the availability of resource i

$\partial W/\partial A_i$ is the partial derivative of the production function to A_i (i.e., the response in W resulting from a small change in availability A_i, keeping the availability of all other resources constant)

L_i is dimensionless and independent from the units used to measure production or availability. L_i is positive and between zero and one. If L_i equals zero, the resource is not limiting at all for the given crop and if L_i equals one, the resource is the only limiting resource. Totaling over all resources, these coefficients most likely add up to one (Kho, 2000a). Methods to estimate L_i for a sole crop are discussed in Kho (2000a). Because trees alter resource availabilities, they most likely will alter the limitations for the crop in the AF system too. A method to estimate L_i at the tree–crop interface (for quantification of Equation 4.24) is given by Kho et al. (2001).

Note that the L_i may differ with crop species (e.g., the nitrogen limitation for a leguminous crop is likely to be lower than that for a nonleguminous crop).

4.5.2.2 Characterization of an AF Technology

Trees have positive and negative effects on the availability of a resource *i* (except light) to the crop (Section 4.4). The sum of the positive and the negative effects is the net tree effect on availability of the resource and equals the difference between the availability of that resource to the crop in the AF technology and the availability to the sole crop. By expressing this difference relative to the availability to the sole crop, it becomes dimensionless, independent of the units used to measure availability and less sensitive to environmental effects (Kho, 2000b):

$$T_i = \frac{A_{i;\mathrm{AF}} - A_{i;\mathrm{S}}}{A_{i;\mathrm{S}}}, \tag{4.22}$$

where

T_i is the relative net tree effect on availability of resource i

$A_{i;\mathrm{AF}}$ is availability of resource i to the crop in the AF system

$A_{i;\mathrm{S}}$ is availability of resource i to the sole crop

T_i may be negative, zero, or positive. In one AF technology a T_i for a specific resource may be negative, but in another AF technology it may be zero or positive. For example, for the resource light T_i is negative in a simultaneous AF technology and zero in a sequential AF technology. The T_i are mainly determined by tree–crop arrangement in time and space, tree species, tree age, and management. They are thus characteristics of the particular AF technology at a specific stage (i.e., with particular canopy architecture, particular rooted depths, and particular root length densities). In a way, much AF research aims to find tree species, tree–crop arrangements, and management that result in many large, positive T_i. Because the trees are dominant, these effects are most likely the same for different crop species in the same AF technology.

Note that the T_i may change in time as the trees grow. Young trees with a superficial and thus competitive rooting system will likely have lower (or more negative) T_i for water and nutrients than older trees that may have a more developed canopy and thus a more negative T_i for light.

4.5.2.3 Combining Environmental and Tree Effects

Within the temperature range that a crop species can grow and reproduce (roughly from 0°C to 35°C for temperate species and from 10°C to 45°C for tropical species; Ong and Monteith, 1985), crop dry matter production (W) in a specific environment is a function of resource availabilities:

$$W = f(A_1, A_2, \ldots, A_n) \tag{4.23}$$

where

A_i is the availability of resource i

n is the number of all resources

Apart from allelopathy and from effects on pests and diseases, trees do not influence crops by modifying the production function (Equation 4.23), but by their influence on the availability of resources to the crop (cf. Section 4.4). Let z denote the tree density (number ha^{-1}) of a particular AF technology. Then according to the chain rule:

$$\frac{\mathrm{d}W}{\mathrm{d}z} = \sum_{i=1}^{n} \frac{\partial W}{\partial A_i} \cdot \frac{\mathrm{d}A_i}{\mathrm{d}z}.$$

Multiplying both sides by dz, and expressing the differentials relative to the sole crop value (i.e., dividing both sides by the production of the sole crop W_s and multiplying the right-hand side by $A_{i;\mathrm{S}}/A_{i;\mathrm{S}}$) gives

$$\frac{\mathrm{d}W}{W_\mathrm{S}} = \sum_{i=1}^{n} \frac{\partial W}{\partial A_i} \frac{A_{i;\mathrm{S}}}{W_\mathrm{S}} \cdot \frac{dA_i}{A_{i;\mathrm{S}}}.$$

By approximating the differentials with differences, we get the tree–environment–crop interaction equation:

$$\frac{W_{\mathrm{AF}} - W_{\mathrm{S}}}{W_{\mathrm{S}}} = \sum_{i=1}^{n} \frac{\partial W}{\partial A_i} \frac{A_{i;\mathrm{S}}}{W_{\mathrm{S}}} \cdot \frac{A_{i;\mathrm{AF}} - A_{i;\mathrm{S}}}{A_{i;\mathrm{S}}},$$

where W_{AF} is the production of the crop in the AF system. Substituting Equations 4.2, 4.21, and 4.22 yields

$$I = \sum_{i=1}^{n} L_i T_i. \tag{4.24}$$

Therefore, each (negative or positive) relative net tree effect on the availability of a resource (T_i) is weighted by the (positive) limitation of that resource for the particular crop in the specific tree–crop environment (L_i). Adding these products for all limiting resources gives the overall interaction I (Figure 4.1). The equation thus states that if a resource is non-limiting for a given crop ($L_i = 0$), a tree effect on that resource has no effect on production of that crop. In addition, the equation states

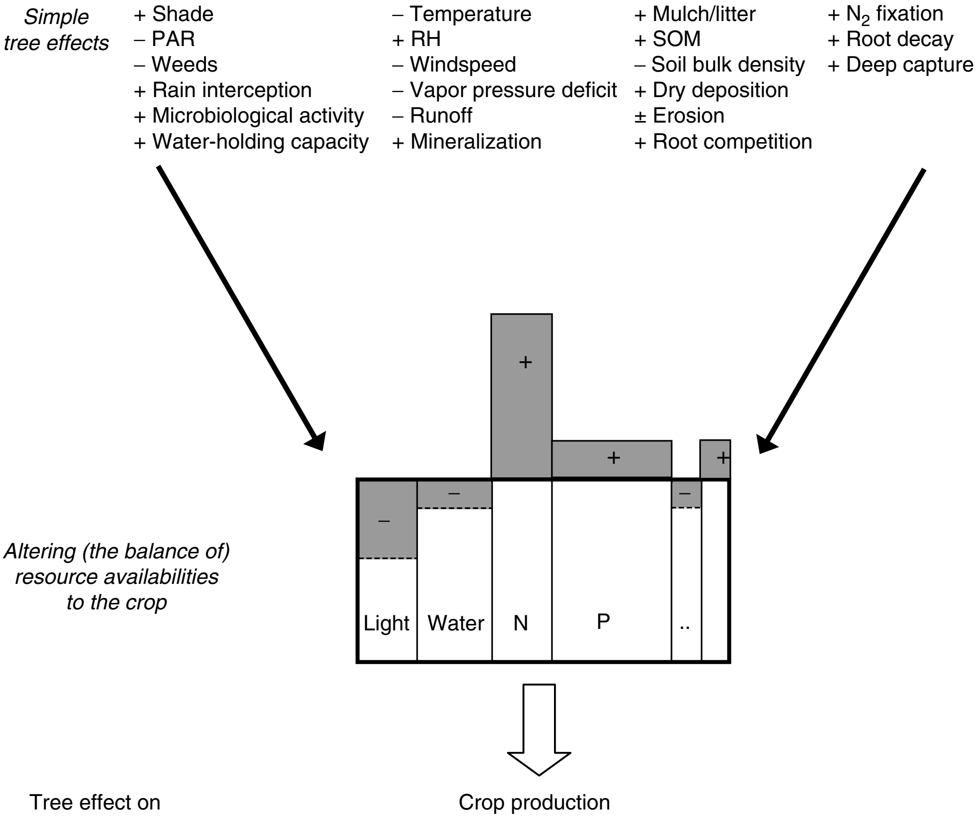


FIGURE 4.1 Trees influence crop production through altering (the balance of) resource availabilities to the crop. The height of each shaded area relative to the height of the rectangle represents the relative net tree effect on availability of the resource (T_i). The width of each shaded area relative to the total width represents the limitation of the resource in the tree–crop interface (L_i). The sum of positive and negative shaded surfaces relative to the total surface of the rectangle represents the overall tree effect I expressed as fraction of sole crop production. (From Kho, R.M., B. Yacouba, M. Yayé, B. Katkoré, A. Moussa, A. Iktam and A. Mayaki, *Agroforestry Syst*., 52, 219, 2001. With permission from Springer.)

that the more limiting ($0 < L_i < 1$) a resource, the greater the influence on crop production of the tree effect on that resource. If, for example, in a certain environment water is ample, a net tree effect on water availability to the crop will not count. But if water starts to become limiting, it becomes important and the more water is limiting, the larger the contribution.

Note that Equation 4.24 does not consider all possible tree effects on crop production. Allelopathy and tree effects on pests and diseases fall outside its scope. Because the net tree effects on availability of resources may change as the trees grow, quantification of Equation 4.24 gives a snapshot at a certain tree stage.

4.5.3 TOWARD A PREDICTIVE UNDERSTANDING OF AF SYSTEMS

Kho (2000b) showed that for the alley-cropping technology the T_i are most likely negative for light, water, and phosphorus (so, for these resources competition outweighs beneficial effects), and that it is positive for nitrogen (beneficial effects outweighs competition). Suppose that the only limiting resources are light, water, nitrogen, and phosphorus and that all other resources are in ample supply (Figure 4.2). Consequently, in a (sub-)humid climate on nitrogen-deficient soils, the alley-cropping effect is most likely positive (Figure 4.2a). Because of the high limitation of nitrogen, the positive nitrogen effect receives a high weight. In the same climate, but on acid soils, phosphorus will be more limiting, giving a high weight to the negative phosphorus effect, resulting in a negative overall effect

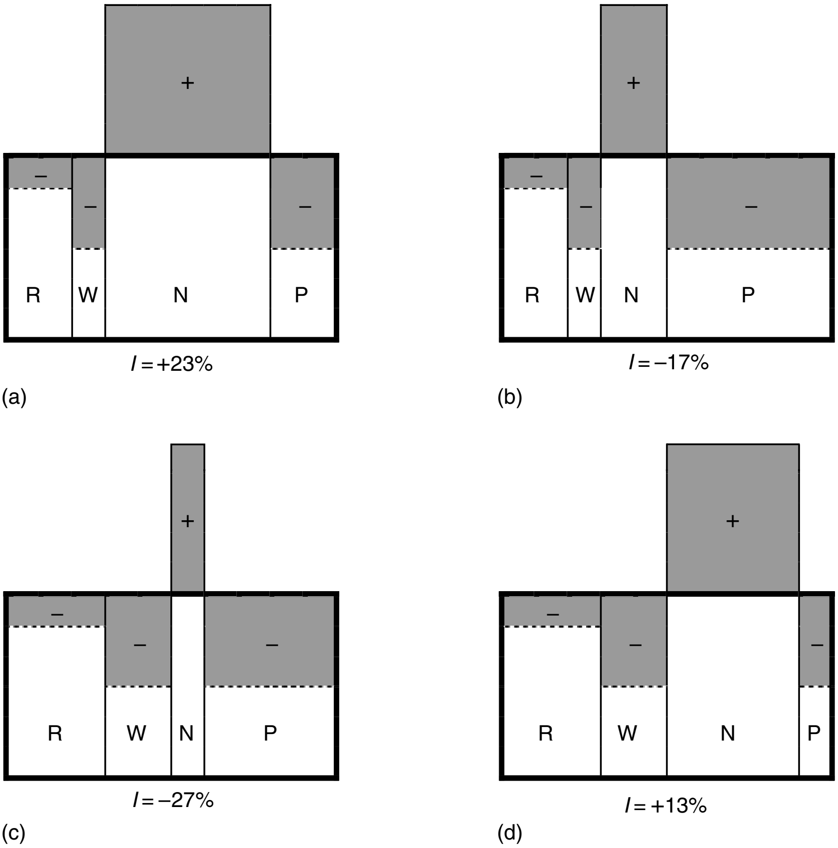
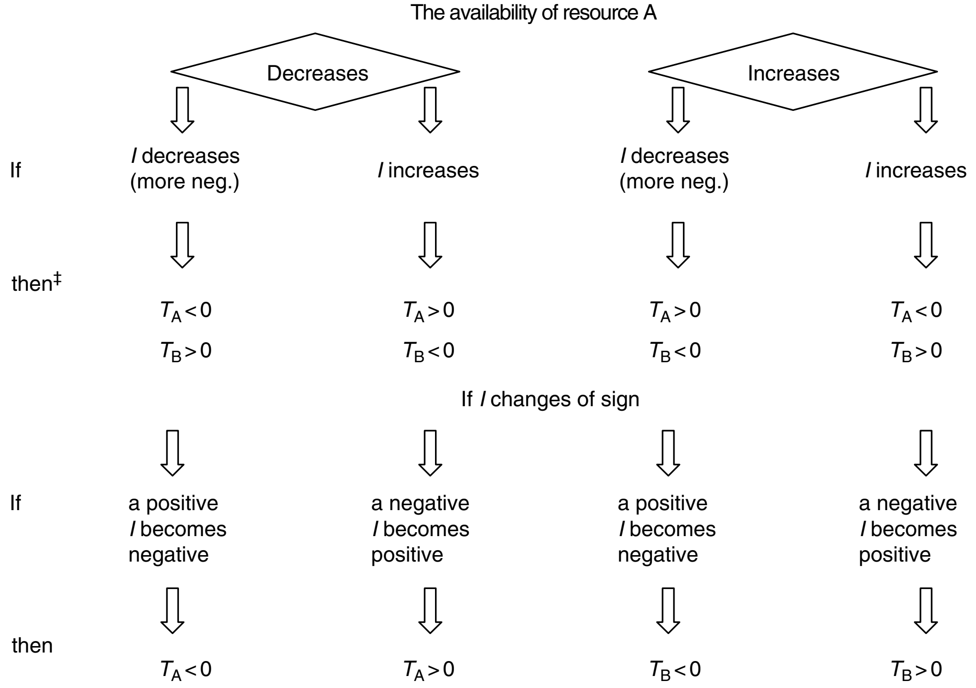


FIGURE 4.2 Possible tree effect balances of an alley-cropping technology in a humid climate (a) in nitrogen-deficient soils, (b) in acid (phosphorus-deficient) soils, (c) in nitrogen-deficient soils with nitrogen fertilizer, and (d) in acid soils with phosphorus fertilizer. The relative net tree effects on availability of each resource (T_i) remain equal; only the environment (i.e., resource limitations L_i) changes explaining the different overall effects (I).

(Figure 4.2b). When in the first case nitrogen fertilizer is added, nitrogen limitation is decreased and the overall alley cropping effect decreases (Figure 4.2c). When on the acid soils phosphorus fertilizer is added, the phosphorus limitation is reduced which may result in a positive overall effect (compared with the sole crop with phosphorus fertilizer; Figure 4.2d). Therefore, knowledge about the signs of the T_i of a particular AF technology (without knowing the exact size) already enables to determine whether in a specific environment a production benefit is likely to be realized.

Knowledge about the signs of the T_i also helps to determine whether a certain management option is appropriate. Management options can be translated into effects on the availability of resources. Management options that increase the availability of resources on which the AF technology has a negative net effect are appropriate. The management option decreases the weight to the negative T_i and increases other tree effects (cf. Figure 4.2b and d). Increasing the availability of resources on which the AF technology has a positive net effect decreases the effectiveness of the AF technology (cf. Figure 4.2a and c). For example, for the alley-cropping technology, phosphorus fertilization, water-conserving tillage and weeding of grasses and other densely rooting weeds are likely appropriate to increase I (compared with the sole crop with the same management). External inputs of organic or inorganic nitrogen probably reduces I.

Knowledge about the signs of the T_i is thus crucial for a predictive understanding. Fortunately, a lot of information about these signs is already present, hidden in the AF literature. By analyzing the direction of the change of I, responding to a change in the availability of a resource (all other factors constant), the sign of the net tree effect on this resource can be deduced (Kho, 2000b; Figure 4.3).



T_A refers to the net tree effect on the changed resource; T_B to that of another limiting resource.
‡ Both statements may be true. However, if I is negative, the statement with the negative T-value is most meaningful; if I is positive, the one with the positive T-value is most meaningful.
If the overall interaction I changes of sign, then certainty about one net tree effect is given.

FIGURE 4.3 Diagram to derive the sign of net tree effects on availability of a resource (other factors equal). (From Kho, R.M., *Agric. Ecosyst. Environ.*, 80, 87, 2000b. With permission from Elsevier.)

TABLE 4.7
A 2^3 Factorial Design (8 plots) to Determine the Sign of the Net Tree Effect of Two Resources

Pair No.	Resource 1	Resource 2	Sole Crop (without Trees)	AF Technology (with Trees)	I
1	Without addition	Without	S_{00}	AF_{00}	I_{00}
2		With	S_{01}	AF_{01}	I_{01}
3	With addition	Without	S_{10}	AF_{10}	I_{10}
4		With	S_{11}	AF_{11}	I_{11}

Note: I is calculated for each pair and is analyzed as a 2^2 factorial design.

Example 1. Suppose that for a certain AF technology the overall interaction I equals −30% on a nitrogen-rich soil and that the overall interaction equals +5% in the same climate, but on a nitrogen-deficient soil. So with decreased nitrogen availability, I increased. According to the diagram in Figure 4.3, the net tree effect of this AF technology on nitrogen availability may be positive ($T_A > 0$) and the net tree effect on another resource may be negative ($T_B < 0$). Because the overall interaction was negative, the last statement is probably true. However, by going to the other soil, I changed of sign, so that we know for certain that the net tree effect on nitrogen availability is positive.

Example 2. Suppose that a certain AF technology has an I equal to −5% in a season with good rains and that the same AF technology has an I equal to −25% in a season with poor rains. So with decreased water availability, the overall interaction decreased. According to the diagram, the net tree effect on water availability may be negative and the net tree effect on another resource may be positive. Because I was negative, the first statement (negative effect on water availability) is most likely true.

Analysis of existing literature and datasets provides indications about the signs of net tree effects on availability of resources. Because differences in availability of a resource can be confounded with other factors the indications should be viewed as hypotheses. Empirical evidence should come from randomized field experiments, which can be simple (fractional) factorial designs. An example is given in Table 4.7 in which two resources are investigated. The resources under investigation must be limiting in the environment. The main effects (on I) can be tested for significance and analyzed with the diagram of Figure 4.3.

Field experiments to fully quantify Equation 4.24 require a more complex design and analysis (Kho et al., 2001).

4.6 CONCLUSIONS

Separating simple tree effects (e.g., the mulch, the light competition, or the root competition effect) leads to insights that are only valid in the specific local situation. The insights do not help to determine whether the AF technology would likely realize a yield advantage in another situation with other soil and climate. A fundamental cause is that the three-way interaction between trees, the environment, and the crop is approached as a two-way (tree–crop) interaction. Environmental effects on crop production that interact with tree effects are not explicitly considered, but must be taken into account implicitly.

The resource capture approach postulates that one resource is "major limiting" and the processes that determine the capture of this resource in AF systems are quantified and modeled. To predict dry matter production, the amount of captured resource is multiplied by a conversion (or utilization) efficiency that is mostly empirically determined. Many key component processes about the influence of other ("minor limiting") resources and of tree effects on the conversion efficiencies are not yet properly understood. It is thus not yet clear how to use the models to predict the yield advantage of an AF technology in another situation in which the resource is not "major limiting." The resource capture approach gives insight in the processes involved. This insight is important for species choice and for optimization of tree–crop arrangements (i.e., for design and development of AF technologies).

The resource balance approach takes the view that the whole balance of available resources determines crop production and that trees influence crop production through altering (this balance of) resource availabilities to the crop. This balance can be quantified by the degree of limitation of resources that serve as weights to the net tree effects on resource availability. The main problem is that of most environments the degree of limitation of resources is not yet accurately known. However, without full quantification, the approach already helps to predict whether a particular AF technology will likely realize a yield advantage in another situation. It also helps to predict the suitability of alternative management options.

The resource capture and the resource balance approaches are complementary. The "reductionistic" resource capture approach gives a profound mechanistic understanding of resource flows in AF systems, but is still difficult to use for predicting the production over a wide range of environments. The more "holistic" resource balance approach is static, but gives a broad insight in the relative importance of the resources in a specific situation and a predictive understanding comprising different situations.

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5 Weeds, Diseases, Insect Pests, and Tri-Trophic Interactions in Tropical Agroforestry

G. Sileshi, Götz Schroth, Meka R. Rao, and H. Girma

CONTENTS

5.1 INTRODUCTION

Under the International Plant Protection Convention, a pest is defined as any species, strain, or biotype of plant, animal, or pathogenic agent injurious to plants or plant products (ISPM, 2006). The coverage of this definition includes weeds and other species that have indirect effects on plants. This definition also applies to the protection of wild flora that contribute to the conservation of biological diversity. Unless otherwise stated, throughout this chapter the term "pest" refers to weedy plants and parasitic higher plants, plant pathogenic organisms (viruses, bacteria, mycoplasma, fungi), plant parasitic or pathogenic nematodes, arthropods (herbivorous mites and insects), and vertebrate pests (herbivorous birds and mammals) that affect trees and associated crops in agroforestry.

Weeds may be classified as ruderals (annual or biennial plants that primarily infest waste places), argestals (annual or biennial weeds of cultivated lands), and environmental weeds (invasive alien species). Weeds compete with trees and crops for water, light, and nutrients. Many weed species also serve as alternative hosts of plant pathogenic organisms and nematodes. Exotic tree species used in agroforestry can also become invasive and affect ecosystem functions and biodiversity. According to a recent estimate (Richardson, 1998), out of over 2000 species used in agroforestry, some 25 species (1%) are invasive. These include *Acacia* (8 spp.), *Prosopis* (3 spp.), *Casuarina* (2 spp.), *Leucaena leucocephala*, and *Sesbania bispinosa*. It must be noted here that not all alien species are invasive, and not all invasive species may be economically important. Transformer species—a subset of invasive plants that change the character, condition, form, or nature of a natural ecosystem over a substantial area—have profound effects on ecosystem functions and biodiversity and are invasive (Richardson, 1998).

A disease can be defined as any physiological disturbance of the normal functioning of a plant as a result of a detrimental interaction between the pathogen, the environment, and the host (Agrios, 1988). Diseases affect the production and utilization of trees and crops by reducing the health of the plant and directly reducing yield, quality, or storage life. Plant parasitic nematodes mostly affect plants by inhibiting root growth, and hence overall plant development, and this usually results in poor crop performance or complete failure. Many plant parasitic nematodes also interact with other microorganisms such as viruses, bacteria, and fungi in the development of disease complexes (Kleynhans et al., 1996). Herbivorous mites and insects can physically feed on various parts of the tree, crop, or both, and also transmit diseases.

In the tropics, weeds, diseases, and insect pests are estimated to account for 13%, 13%, and 20% of losses, respectively (Oerke et al., 1994). Weed control takes over 50% of the total labor needed to produce a crop. Pests have been cited as one of the factors diminishing the benefits from tropical agroforestry (Mchowa and Ngugi, 1994; Karachi, 1995; Rao et al., 2000). Unless the biological constraints imposed by pests are removed, the potential benefits of agroforestry in terms of increased capture and efficient use of resources cannot be translated into economic benefits (Ong and Rao, 2001). If the current enthusiasm of farmers for testing and eventually adopting the various agroforestry practices is to be sustained, it is essential to know how this practice affects pest populations and their natural enemies.

Although the relevance of pest interactions with agroforestry practices has been recognized many years ago (Huxley and Greenland, 1989), very few detailed studies of their influence on tree–crop interactions exist. There seems to be more focus on population ecology of selected pest species at the expense of ecosystem ecology. In fact, there exist certain general misconceptions, which hold that trees have no or fewer pests and that diversity based on trees reduces pest problems in agroforestry (Desaeger et al., 2004). This has hindered progress in the understanding of tri-trophic interactions in agroforestry. Even in the more recent books on agroforestry (Schroth and Sinclair, 2003; Nair et al., 2004; van Noordwijk et al., 2004), there is little, if any, mention of the effects of tree–crop interactions on pests and their natural enemies. In the recent reviews, Day and Murphy (1998) and Rao et al. (2000) dealt mainly with insect pests affecting agroforestry trees and their management. Schroth and coworkers (2000) dealt with insect pests and diseases in agroforestry systems of the humid tropics. The review by Gallagher et al. (1999) and Ong and Rao (2001) focused on managing tree–crop interactions in relation to weeds. Desaeger et al. (2004) dealt with nematodes and other soil-borne pathogens. The review on the effect of trees on abundance of natural enemies (Dix et al., 1995) focused on agroforestry systems of the temperate zone.

Though complex interactions are known to occur between various categories of pests (e.g., weeds, pathogens, nematodes, insects, etc.), the nature of such interactions is poorly understood and little quantified in tropical agroforestry (Hitimana and McKinlay, 1998). This work is the first attempt to draw together information on the different categories of pests and natural enemies, and apply the knowledge to the challenges of pest management in tropical agroforestry. In this chapter, an extensive review of literature pertinent to tree–crop interactions and pest risks in agroforestry was conducted. In view of the vast number of tree and crop species used in agroforestry and numerous pest species, a complete treatment of the subject matter is beyond the scope of this chapter. Only a selection of the most widely used agroforestry systems are given here as examples, and typical cases are examined. The objective is to analyze the factors that influence pest incidence in the light of existing ecological hypotheses. In the discussions, more emphasis has been on information generated after the recent reviews by Day and Murphy (1998), Rao et al. (2000), and Schroth et al. (2000). This is intended to fill the gaps in knowledge and complement the existing reviews.

5.2 PARTITIONING THE COMPLEXITY OF PEST INTERACTIONS

In agroforestry systems, plants have close relations with abiotic and biotic components in the community. According to Ong et al. (2004), the net effect of one plant component on another can be expressed as:

$$I = F + C + M + P + L + A,$$

where

I is the overall interaction
F is effects on chemical, physical, and biological soil fertility
C is competition for light, water, and nutrients
M is effect on microclimate
P is effect on pests
L is soil conservation
A is allelopathic effects

Many of these effects are interdependent and cannot be experimentally estimated independently of one another. This means that when studying the effect of pests in agroforestry, we cannot ignore the effects inter alia of soil fertility, competition, or microclimate.

Pests of an agroforestry system are essentially the pests of its components (the crops and woody perennials), and their dynamics is governed by the complexity and degree of interaction between the crop, tree, and the composition of other plant communities such as weeds. Direct interactions between trees and crops for growth resources may exercise a strong influence on pests and natural enemies of either or both components of the system (Table 5.1). In the following discussion, the manner in which each component affects the other in terms of pest populations is briefly summarized. A simplified model of potential interactions between the plant community, herbivores, pathogens, and natural enemies in a simultaneous agroforestry practice is presented in Figure 5.1.

5.2.1 INTERACTIONS BETWEEN THE PLANT COMMUNITY, HERBIVORES, AND THEIR NATURAL ENEMIES

The plant community (or producers), including the trees, crops, and weeds, constitute the first trophic level. Each plant species may be attacked by a wide range of herbivores (i.e., primary consumers), which constitute the second trophic level. Herbivorous species in turn are attacked by natural enemies (i.e., secondary consumers), which constitute the third trophic level. Natural enemies include predatory arthropods (e.g., insects, predaceous mites, spiders, scorpions, centipedes, etc.) and vertebrates (e.g., insectivorous birds and mammals), parasitic insects (i.e., parasitoids), and pathogenic bacteria, viruses, fungi, protozoa, and nematodes, which play a significant role in the population dynamics of pests of agroforestry (Sileshi et al., 2001).

The interactions that occur between the plants, herbivores, and their natural enemies are called tri-trophic interactions. The plant community may affect these interactions in a variety of ways, as depicted in Figure 5.1 and Table 5.1. For instance, trees through shading or their physical presence may directly influence the migration, host location, and feeding of insect pests of the crop in

TABLE 5.1
Summary of Tree–Crop Interactions and Their Consequences on Pests and Diseases in Major Groups of Agroforestry Systems

	Process	**Possible Effects**
Sequential systems	Tree canopy shading/smothering the understory vegetation	Reduction of annual and perennial weeds
	Tree/shrub species may stimulate germination of parasitic weed *Striga*	Weed seed-bank depleted
		Striga population and its seed-bank are reduced
	Trees producing allelopathic chemicals	Reduction of weed populations
	Tree species profusely producing seed and volunteer seedlings	Tree species becomes an environmental weeds
		Increase costs of control
	Tree in fallow or boundary planting harboring pests	Increased pests damage in adjacent crop fields
	Increases the pool of available soil nutrients, especially inorganic N	Increased crop vigor to withstand some pests
		Increased vigor inducing susceptibility to other pests
	Tree fallows breaking the cycles of insect and pathogens	Reduction in insect, disease and nematode damage on subsequent crops
	Trees serving as alternative hosts to insects, nematodes and pathogens	Increased pest damage on subsequent crops
	Mulches increasing soil humidity and lowers soil temperature	Increased soil-borne disease populations
	Trees serving as refuge and food source for natural enemies	Reduction of pest problems in adjacent crop fields
Simultaneous systems	Trees dominating crops by competition for growth resources	Reduced vigor inducing susceptibility to pests attack
	Trees serving as refuge and food source for natural enemies	Reduction of pest problems in adjacent crop fields
	Trees lines act as mechanical barriers for the spread insect pests, vectors and pathogens	Reduction of pest colonization
	Trees improving microclimate in harsh environments	Increased crop vigor
		Buildup of pests and pathogens
	Trees serving as alternate hosts to crop pests and disease vectors	Increased pest damage on crops
	Tree prunings used as mulch	Reduction of shade sensitive weeds
	Tree and crop sharing the same pest	Increase in pest problems
	Tree canopy and leaf litter keeping the ground covered for most part of the year	Buildup of some disease

addition to acting as a refuge for natural enemies. Trees can also influence pest incidence by acting as alternative hosts of a crop pest or vector of a pathogen. Trees, through their indirect effects on the nutrition of the crop, may also influence demographic parameters of crop pests such as natality, longevity, and mortality. This in turn may trigger changes in the migration, host location, feeding, and demographic patterns of natural enemies. Trees may also cause shading and reduce air circulation, leading to high humidity and an increase in disease incidence. A detailed knowledge of tri-trophic interactions associated with a given pest or pest complex is required if refuge for natural enemies is to be conserved or established.

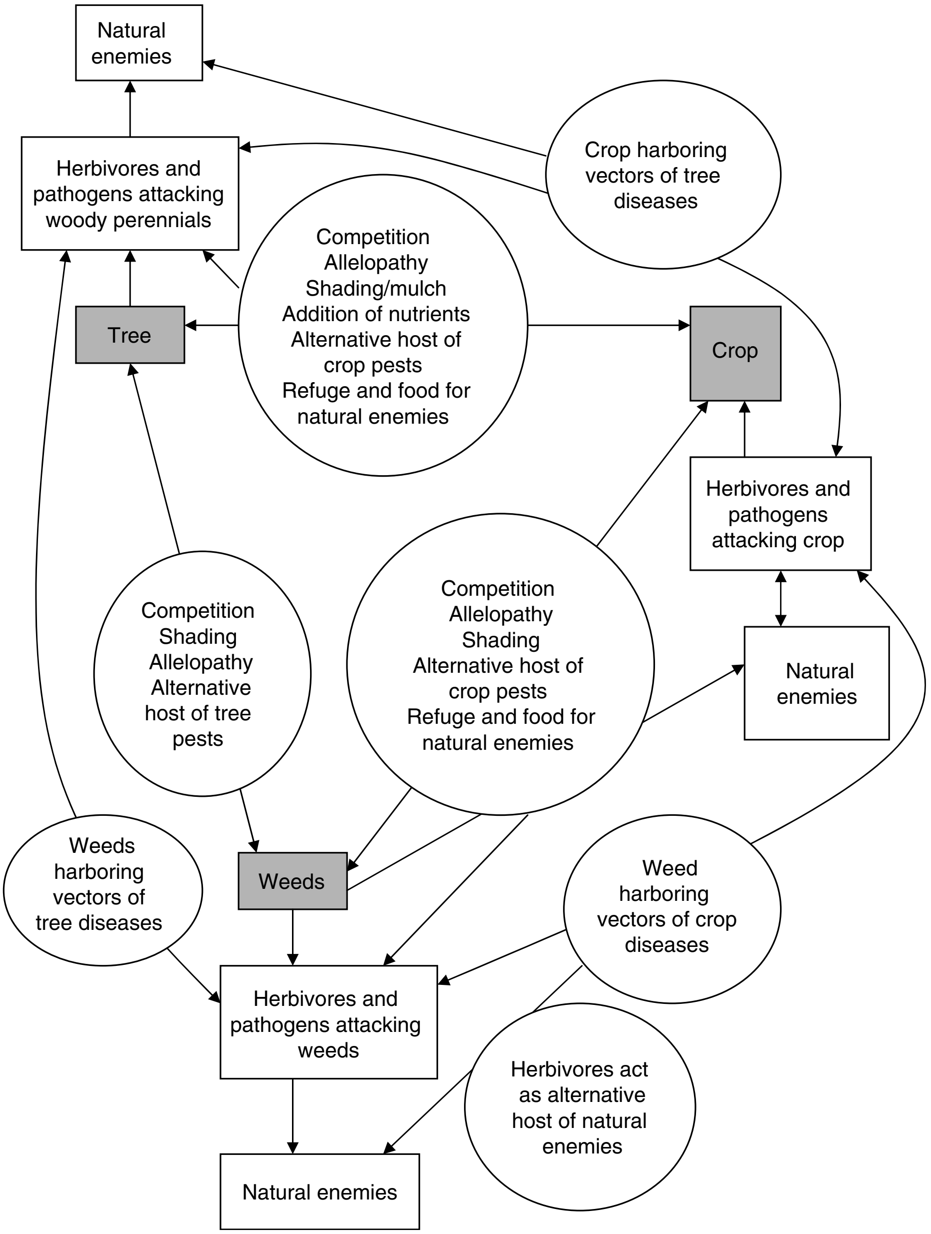

FIGURE 5.1 Potential interactions between the plant community, herbivores, pathogens, and natural enemies in a simultaneous agroforestry practice.

Weeds, in addition to competing with the tree and crop components, may also act as alternative hosts of pests of the tree or crop components. For instance, in western Kenya, *Striga hermonthica*, a parasitic weed of cereals, is a good host for root-knot nematodes, which attack agroforestry species such as *Sesbania sesban* and *Tephrosia vogelii* (Desaeger et al., 2004). Cultivated ground cover plants and weeds (e.g., in orchards) can increase the heterogeneity of the habitat, alter the quality and quantity of bioresources, and regulate ecological niches of various species in the community. Such plants can provide a variety of resources for predators and parasitoids, including shelter, food, and information on the location of their herbivorous prey (Bugg and Waddington, 1994; Liang and

Huang, 1994). Liang and Huang (1994) reported that the weed *Ageratum conyzoides*, growing in citrus orchards, plays an important role in stabilizing populations of the predatory mites (*Ambleyseius* spp.), which are effective natural enemies of the citrus red mite (*Panonychus citri*). Understory vegetation can also sustain significantly higher generalist predators such as lady beetles, ground beetles, hover flies, mirid bugs, and lacewings in orchards than clean-weeded orchards (Bugg and Waddington, 1994). Many aphids that colonize weeds can play an important role as reservoirs of polyphagous natural enemies such as lady beetles, hover flies, and lacewings.

5.2.2 Interactions between Herbivores and Plant Pathogens

The manner in which herbivores interact with plant pathogenic organisms include (1) acting as vectors, (2) wounding agents, (3) host modifiers, (4) rhizosphere modifiers, and (5) resistance breakers (Agrios, 1988). Desaeger et al. (2004) provide specific examples of such interactions between nematodes and soil-borne pathogens. Homopterous insects, beetles, and mites vector viral, bacterial, and fungal diseases, which cause substantially greater losses than those caused by the direct feeding injury by the insects. For instance, the green peach aphid (*Myzus persicae*) is known to be a vector of more than 180 virus diseases. The cotton aphid (*Aphis gossypii*) transmits more than 80 kinds of virus diseases. The black citrus aphid (*Toxoptera citricidus*) is a vector of virus diseases of coffee, citrus tristeza virus, citrus infectious mottling virus, and little leaf and lemon-ribbing virus of lemon (Michaud, 1998; EPPO, 2006). Some xylem fluid-feeding leafhoppers also transmit the bacterial plant pathogen *Xylella fastidiosa*, which induces diseases of grapevines (e.g., Pierce's disease) and citrus (citrus variegated chlorosis), and also other diseases of coffee and stone fruits. Citrus-variegated chlorosis transmitted by the glassy-winged sharpshooter (*Homalodisca coagulata*) has now expanded throughout many citrus-growing areas of South America (Redak et al., 2004).

One of the classic examples of a disease vectored by beetles is the Dutch elm disease, a vascular-wilt fungus, *Ophiostoma (Ceratocystis) ulmi*, carried from an infected tree to a healthy one by bark beetles of the genus *Scolytus* (Agrios, 1988). Recently, the weevil *Pissodes nemorensis* has been reported as a vector and wounding agent of the pitch canker fungus (*Fusarium circinatum*) and *Diplodia pinea* causing dieback on pines (*Pinus* species) (Gebeyehu and Wingfield, 2003). The bean beetle *Ootheca mutabilis*, which attacks *Sesbania sesban*, also transmits cowpea mosaic virus, one of the commonest viral diseases of cowpea reducing yields by up to 95% (van Kammen et al., 2001). Arthropods that transmit plant diseases may vector plant pathogens to and from the tree, crop, and weed hosts in agroforestry (Figure 5.1).

5.2.3 Interactions among Herbivores

Interactions also occur among herbivores in the form of competition and mutualism. Competition is defined as the interaction between individuals, brought about by a shared requirement for a resource in limited supply, and leading to a reduction in the survivorship, growth, and reproduction of the competing individuals (Speight et al., 1999). Generally, competition can occur among individuals of the same species (intraspecific) or members of different species (interspecific). Damage by one herbivore species could influence populations of a second species through changes in plant quality, even if the herbivores lived at different times of the year. West (1985) demonstrated that spring defoliation by caterpillars of two Lepidoptera, *Operophthera brumata* (Geometridae) and *Tortrix viridana* (Tortricidae), on oak leaves can reduce leaf nitrogen content, which adversely affects the survival of the Lepidopteran leaf-miner *Phyllonorycter* (Gracillaridae) and aphids later in the season.

Mutualism is a type of symbiosis in which two or more organisms from different species live in close proximity to one another and rely on one another for nutrients, protection, or other life functions. For example, many ants are known to tend homopterous pests such as aphids, mealy bugs, and scale insects, where the ants protect these insects from predation and parasitism. In turn, the ants get honey dew from their hosts. On the other hand, ants are predators and may well have a positive effect as biocontrol agents. In shade coffee production systems Vandermeer and coworkers (2002) demonstrated that ants (*Azteca* sp.) can not only have potential as pests through their positive effect on scale insects, but also have potential as biological control agents through their effect on other herbivores.

5.3 INTERACTIONS IN SELECTED AGROFORESTRY PRACTICES

Section 5.3.1 presents the characteristics of the various agroforestry practices as they affect the occurrence and development of weeds, insect pests, and diseases. Agroforestry systems were broadly grouped into sequential (rotational) and simultaneous systems (Rao et al., 1998). The presentation was structured from the simplest to the more complex tree–crop associations to facilitate comprehension of the interactions.

5.3.1 Sequential Agroforestry Practices

5.3.1.1 Rotational Woodlots and Improved Fallows

In the rotational woodlot system, food crops are intercropped with leguminous trees during the first 2–3 years. Then the trees are left to grow, harvested in about the fifth year, and food crops are replanted (Otsyina et al., 1996). The food crops grown following the tree harvest are expected to benefit from improved soil conditions by the woodlot species. Improved fallows, on the other hand, consist of deliberately planted species—usually legumes with the primary purpose of fixing nitrogen as part of a crop–fallow rotation (Mafongoya et al., 1998; Sanchez, 1999). The legumes can be planted as either single species or mixed stands. Compared with single-species fallows, mixed-species fallows are believed to increase the biodiversity and sustainability of the fallow system, provide insurance against failure, produce multiple products, improve utilization of available plant growth resources, and reduce buildup of pests (Gathumbi, 2000; Sileshi and Mafongoya, 2002).

Rao et al. (1998) recognized three distinct phases based on the major soil changes that occur in the rotation of tree fallows by crops. These changes may directly or indirectly affect the populations of weeds, pathogens, and insect pests affecting the subsequent crop (Schroth et al., 2000; Sileshi and Mafongoya, 2002, 2003). One of the significant impacts of these changes in vegetation cover is on the parasitic weeds (*Striga* spp.), which are widespread in most parts of sub-Saharan Africa and cause annual cereal yield losses estimated between $7 and 13 billion (Annon, 1997). In two separate studies conducted in eastern Zambia (Sileshi et al., 2006), rotational fallows of *Sesbania sesban* significantly reduced incidence of *Striga asiatica* on subsequent maize compared with continuously cropped monoculture maize, or that grown after a traditional bush fallow. This effect of the *Sesbania sesban* fallow persisted through three consecutive cropping cycles. Similarly in Kenya, *S. sesban* reduced the number of *Striga hermonthica* seeds in the soil by 34%, whereas in monoculture maize plots the *Striga* populations increased over the same period by 11% (ICRAF, 1993). The effect of *Sesbania sesban* on *Striga* was due to the combined effects of *S. sesban* causing suicidal germination of *Striga hermonthica* (i.e., a "trap crop" effect) and improving soil inorganic N, which is known to be detrimental to *Striga* (Gacheru and Rao, 1998).

Tree fallows also reduce the incidence of weeds in general including the perennial grasses such as spear grass (*Imperata cylindrica*) (Garrity, 1997). In Sri Lanka, weed populations were lower by 42% and 54% in maize planted in improved fallow of *Crotalaria juncea* and *Tithonia diversifolia* than in a natural fallow (Sangakkara et al., 2004). In Nigeria, 3 years of planted fallows of *Dactyladenia barteri* caused 36% decrease in the weed seed-bank relative to the cropped field, whereas the same duration of bush fallow increased the weed seed-bank by 31% (Akobundu and Ekeleme, 2002). Studies in Zambia (Sileshi and Mafongoya, 2003; Sileshi et al., 2006) have demonstrated that some legume fallows can reduce the infestation of maize by arable weeds.

In one study (Sileshi and Mafongoya, 2003), total weed biomass in maize grown after a natural fallow was six times higher than that grown after pure *Sesbania sesban* and pigeon pea fallows. The weed biomass was correlated negatively with leaf litter indicating that the reduction is due to smothering of the weeds through initial suppression of aboveground weed growth, and the thick mulch layer formed by the leaf litter from the fallow trees subsequently depleting the weed seed-bank (Sileshi and Mafongoya, 2003). Many fallow species release a wide range of compounds, commonly referred to as allelochemicals, which can inhibit weed seed germination or reduce weed vigor. Legume cover-crop residues in the course of decomposition release volatile organic compounds with potential herbicidal properties (Gallagher et al., 1999).

Rotational fallows have also been shown to affect plant-parasitic nematodes that attack crops. Some fallow species (e.g., *Sesbania*, pigeon pea, *Tephrosia*, and *Acacia*) are hosts for plant parasitic nematodes such as *Meloidogyne* and *Pratylenchus* spp. (Page and Bridge, 1993; Duponnois et al., 1999; Desaeger and Rao, 2000). With the introduction of *S. sesban* for soil fertility improvement in the tobacco-growing areas of southern Africa, the root-knot nematode problem became serious (Karachi, 1995; Shirima et al., 2000). In Tanzania, *Meloidogyne* infection was consistently higher when tobacco was planted after a 2-year *S. sesban* fallow compared with the crop rotated with a 2-year natural fallow (Shirima et al., 2000). In a study conducted in western Kenya, *Meloidogyne* infestation caused 52%–87% yield reduction in beans (*Phaseolus vulgaris*) planted after *S. sesban* (Desaeger and Rao, 2000). A *Crotalaria agatiflora* cover-crop increased root-lesion nematode (*Pratylenchus zeae*) populations to levels that could limit maize growth, whereas it decreased *Meloidogyne incognita* and *M. javanica* populations during the same time (Desaeger and Rao, 2000). In another study (Desaeger and Rao, 2001), bean crop that followed mixed-species fallows of *S. sesban* + *Tephrosia vogelii* had increased root-knot nematode damage compared with bean grown after pure fallows of the respective species. On the contrary, bean crops that followed *S. sesban* + *Crotalaria grahamiana* and *T. vogelii* + *C. grahamiana* did not experience yield losses. In a separate study conducted in the same area in western Kenya (Kandji et al., 2003), beans grew poorly when planted after *T. vogelii* and *C. grahamiana* because of high incidence of *Meloidogyne* spp. in the first cropping cycle. In the second and third cropping seasons, while the population of *Meloidogyne* spp. decreased, spiral nematode (*Scutellonema* spp.) populations increased, which caused heavy losses of beans and maize planted after the legume fallows (Kandji et al., 2003). Studies by Kandji and coworkers (2001) found a positive correlation of *Scutellonema* populations with exchangeable bases in the soil. *Pratylenchus* populations were positively correlated with bulk density, whereas *Meloidogyne* populations were correlated with clay, potassium, and organic carbon content of the soil. On the other hand, *Paratrichordorus* and *Xiphinema* populations were correlated with calcium and soil bulk density (Kandji et al., 2001).

Rotational fallows also have significant effects on the incidence of insect pests of crop plants. According to Rao et al. (2000), chaffer grubs, which destroy maize seedlings, increased in maize planted after *Sesbania sesban* fallows in Kenya. Snout beetles (*Diaecoderus* sp.) that breed on *S. sesban*, pigeon pea, *C. grahamiana*, and *T. vogelii* during the fallow phase attacked maize planted after fallows with these plant species in eastern Zambia (Sileshi and Mafongoya, 2003). In an experiment involving pure fallows and mixtures of these legume species, the density of snout beetles was significantly higher in maize planted after *S. sesban* + *C. grahamiana* compared with maize planted after natural grass fallow. The population of beetles was significantly positively correlated with the amount of nitrate and total inorganic nitrogen content of the soil and cumulative litter fall under fallow species (Sileshi and Mafongoya, 2003). Besides *S. sesban* being an alternative host of the beetle (Sileshi et al., 2000), its mixture with other legumes appeared to offer a favorable environment for the survival of the beetles during the fallow phase.

In the same study in eastern Zambia, Sileshi and Mafongoya (2003) recorded lower termite damage (% lodged plants) on maize planted after *T. vogelii* + pigeon pea, *S. sesban* + pigeon pea, and pure *S. sesban* than on maize grown after natural fallow. Monoculture maize grown after the natural fallow had about 11 and 5 times more termite damage compared with maize grown after *T. vogelii* + pigeon pea and *S. sesban* + pigeon pea, respectively. The higher termite damage recorded in the natural fallow was apparently due to stress caused by weed competition. In another study conducted at four sites in eastern Zambia, Sileshi and coworkers (2005) found no difference between treatments in termite damage on maize plants after *T. vogelii*, *Tephrosia candida*, *S. sesban*, and *Crotalaria pawlonia*, a traditional grass fallow, monoculture maize grown with and without fertilizer. Though the differences were not statistically significant, maize planted after *Tephrosia candida* fallows had consistently lower termite damage than fully fertilized monoculture maize at three out of the four sites. In western Kenya, incidence and damage due to groundnut hopper (*Hilda patruelis*) increased on farms where *C. grahamiana* was planted as a rotational fallow compared with new sites (Girma, 2002). The abundance of natural enemies and tri-trophic interactions in rotational woodlots and improved fallows has not been studied. Rotational systems at the landscape level may create a mosaic of fallowed and cropped plots and how such a situation affects pests needs to be evaluated.

5.3.2 SIMULTANEOUS AGROFORESTRY PRACTICES

5.3.2.1 Trees on Cropland

Rao et al. (1998) recognized three distinct categories of trees on cropland—scattered trees, boundary planting, and intercropping of annual crops between widely spaced rows of trees. Scattered trees in cropland, often known as ‘‘parklands,’’ are widespread traditional practices in the semiarid tropics. The best known ones are those involving *Faidherbia (Acacia) albida*, *Parkia biglobosa*, *Vitellaria paradoxa*, *Azadirachta indica* in West Africa, and mango, *Melia volkensii*, *Adansonia digitata*, *Parinari curatellifolia*, *Acacia* spp. in the semiarid parts of eastern and southern Africa. Trees in these systems are rarely planted but are derived from natural regeneration and are protected by farmers. In such a setup, a pest may be shared between the tree and the associated crop or the adjacent vegetation and the resultant interactions may assume considerable significance. For instance, fruit flies (*Ceratitis* spp.) and false codling moth (*Cryptophlebia leucotreta*) are one such group of pests with a wide host range (De Meyer, 1998). The marula fly (*Ceratitis cosyra*) and false codling moths attack fruits of *Uapaca kirkiana* and *P. curatellifolia* as well as commercial fruits including mango, guava, avocado, peach, and citrus (Sileshi, unpublished data).

Trees in boundary planting and intercropping systems are deliberately planted and managed. Boundary planting involves trees on farm and field boundaries, soil conservation structures, and terrace risers. Intercropping systems use widely spaced rows of fast-growing trees such as *Cedrela odorata*, *S. sesban*, and *Grevillea robusta* in banana and bean fields. The management of trees used as windbreaks around orchards and surrounding trees and bushes has also a significant effect on the populations of pest organisms and natural enemies. The effect of trees on cropland on pests has been reviewed by Rao et al. (2000) and Schroth et al. (2000). However, systematic studies investigating the effect of trees on cropland on tri-trophic interactions are virtually lacking.

5.3.2.2 Mixed Intercropping

Mixed intercropping involves relay intercropping and coppicing legume fallows. In the context of using leguminous trees for soil fertility replenishment, relay intercropping has been found to be more appropriate than rotational fallows in areas characterized by high population density and land scarcity, where farmers cannot forgo crops for the tree–fallow phase. A typical situation is that of southern Malawi, where trees or shrubs such as pigeon pea, *Tephrosia* spp., and *S. sesban* are planted between rows or within the rows of an already established maize crop (Phiri et al., 1999).

Coppicing tree fallows are another variant of mixed intercropping combining the elements of rotational fallow (the fallow phase) and intercropping (the resprouting phase) (Sileshi and

Mafongoya, 2006). Tree species that resprout when cut at fallow termination are called coppicing species. The legume species used in coppicing fallows include *Acacia* spp., *Gliricidia sepium*, *Leucaena* spp., *Calliandra calothyrsus*, *Senna siamea,* and *Flemingia macrophylla*. Pure stands of these species are normally planted at a spacing of 1 × 1 m and the fallows are left to grow for 2–3 years. At the end of the fallows, the trees are cut, and the leaves and twigs are incorporated into the soil with a hand hoe. Every time the stumps resprout, the coppice biomass is cut and incorporated into the soil. A cereal crop, often maize, is planted on the ridges between the tree stumps.

Like the short-duration fallow species, legumes grown in mixed intercropping have a significant impact on witch weeds. The incidence of *Striga asiatica* was monitored (Sileshi et al., 2006) in 1995–1997 cropping seasons in coppicing fallows established in 1991 and 1992 at Msekera in eastern Zambia. The density of *S. asiatica* weeds was lower in maize grown in the coppicing fallows of *Senna siamea*, *Flemingia congesta*, and *L. leucocephala* than in monoculture maize, whereas maize grown in those of *C. calothyrsus* and *G. sepium* did not differ from monoculture maize.

Legume trees grown in mixed intercropping can also influence insect pest populations. In a study in Malawi, Sileshi et al. (2000) found higher densities of the bean beetle (*Ootheca benningseni*) in farms where *Sesbania sesban* was relay cropped with legumes such as cowpea (*Vigna unguiculata*), bean, soybean (*Glycine max*), and bambara groundnut (*V. subterranea*). In another study in Zambia, the beetle density and damage was higher in farms where *S. sesban* was planted next to cowpea and Hyacinth bean (*Dolichos lablab*). The beetle caused 100% defoliation of both *S. sesban* and the other legumes (Sileshi et al., 2000).

Sileshi and coworkers (2005) monitored termite damage on maize for 2 years in an experiment established in 1992 (described earlier) and a second experiment established in 1997 at Msekera. In the experiment established in 1992, maize grown in the traditional fallow and *Senna siamea* had significantly higher percentage of lodged plants than fully fertilized monoculture maize during the 2001–2002 cropping season. The damage to maize grown in *C. calothyrsus*, *Gliricidia sepium,* and *F. macrophylla* did not differ from that in monoculture maize. On the contrary, during the 2002–2003 cropping season, fully fertilized monoculture maize had significantly more damaged plants than maize grown in the different fallows except *F. macrophylla*. In this experiment, total inorganic nitrogen, soil water at planting, and coppice biomass applied during the season accounted for 59% of the variance in the percentage of lodged maize plants. In the experiment established in 1997, the percentage of lodged plants was significantly higher in fully fertilized monoculture maize grown continuously without fertilizer than in maize grown in *Acacia anguistissima* fallows in the 2001–2002 cropping season, whereas in the 2002–2003 cropping season, no difference was noted among treatments. The percentage of lodged maize plants was significantly correlated with preseason inorganic nitrogen (Sileshi et al., 2005). Hardly did any study investigate the effect of mixed intercropping on natural enemies.

5.3.2.3 Alley Cropping

Alley cropping (also called hedgerow intercropping) involves continuous cultivation of annual crops within hedgerows formed by leguminous trees and shrubs. The legumes are periodically pruned and their biomass is applied either as mulch or incorporated into the soil to improve soil fertility (Kang, 1993).

Trees in alley-cropping arrangements can have significant effects on the incidence of weeds, diseases, and insect pests. Studies in Kenya (Jama et al., 1991; Jama and Getahun, 1992) showed 42%–98% reduction in weed biomass in maize and green gram (*Phaseolus aureus*) alley cropped with *Faidherbia (Acacia) albida* and *L. leucocephala* compared with the respective monocrops. In Costa Rica, Rippin et al. (1994) reported a 52% and 28% reduction in weed biomass in maize grown between *Erythrina poeppigiana* and *G. sepium* hedgerows, respectively. One of the most important aspects of alley cropping is control of problematic weeds such as speargrass (*Imperata cylindrica*)

(Garrity, 1997). On Alfisols in Nigeria, hedgerows of *L. leucocephala* and *G. sepium* reduced the population of speargrass by 51%–67%, aboveground biomass by 78%–81%, and belowground rhizomes by 90%–96% compared with a speargrass bush fallow (Anoka et al., 1991). Similarly, on Ultisols in Indonesia, hedgerows of *G. sepium* reduced speargrass infestation (ICRAF, 1996). However, hedgerow species show striking differences in their ability to control weeds. For instance, *G. sepium* was better than *L. leucocephala* in suppressing speargrass on tropical Alfisols in Nigeria (Anoka et al., 1991). On the contrary, Yamoah et al. (1986) reported that *S. siamea* controlled weeds better than *G. sepium* and *Flemingia macrophylla* in Nigeria. In Peru, *Inga* achieved greater weed control than *Leucaena* or *Erythrina* (Salazar et al., 1993). These differences have been suggested to be due to differences in canopy spread among hedgerow species, the amount of biomass they produce, and the decomposition rate of the biomass (Rao et al., 1998).

Alley cropping may affect the development of crop diseases positively or negatively. Studies by Yamoah and Burleigh (1990) in Rwanda suggested that alley cropping with *Sesbania sesban* slowed down the development of maize rust (*Puccinia sorghi*). The proportion of infected leaves per plant, number of uredinia per plant, and area under disease progress curve in monocrop maize were significantly greater than in alley-cropped maize. Rust development on maize in middle rows was also significantly greater than that in the rows bordering *S. sesban* hedges (Yamoah and Burleigh, 1990). In Côte d'Ivoire, *G. sepium* hedgerows reduced severe virus infestation and incidence of late leaf spot (*Phaeoisariopsis personata*) and rust (*Puccinia arachidis*) of alley-cropped groundnut (Schroth et al., 1995a). Mulch with *G. sepium* foliage also reduced the incidence of late leaf spot and rust when applied to a monocrop groundnut. In Kenya, however, the incidence and severity of angular leaf spot (*Phaeoisariopsis griseolal*) and anthracnose (*Colletotrichum lindemuthianum*) on beans were higher in *L. leucocephala* alleys than in monocropped beans (Koech and Whitbread, 2000). The incidence and severity of both these diseases increased as the alley width decreased from 8 to 2 m. The disease incidence in this study was related to microclimate change, whereas in the previous study a suppressive effect of tree mulch on groundnut diseases was the cause. However, Schroth et al. (1995a) found an increase in groundnut disease in those parts of alleys that were the most shaded by trees. In Philippines, the incidence of blast (*Pyricularia oryzae*) and its damage on rice was higher in alley cropping than in a monocropped control (Maclean et al., 1992).

Hedgerows of trees were reported to affect pests of different alley crops differently. In a study that evaluated the effects of alley cropping on the abundance of insect pests of beans and maize in semiarid Kenya, Girma et al. (2000) recorded higher bean fly (*Ophiomyia* spp.) infestation on beans in the presence of *G. sepium*, *Grevillea robusta*, *Senna siamea*, *Senna spectabilis*, *Flemingia congesta*, *Croton megalocarpus*, *Morus alba*, *Calliandra calothyrsus,* and *Lantana camara* hedgerows than in their absence. In contrast, maize in the hedgerows experienced significantly lower stalk borer (*Busseola fusca* and *Chilo* spp.) and aphid (*Rhophalosiphum maidis*) infestations than monocrop maize. Aphid (*Aphis fabae*) infestation of beans, however, did not differ between treatments (Girma et al., 2000). In another study conducted at two sites in Kenya (Mtwapa and Amoyo), the abundance of adult, larval, and pupal stages of stem borers, defoliation, stem damage, and plant mortality due to maize stem borers (*Chilo partellus*, *Chilo orichalcociliellus*, and *Sesamia calamistis*) was significantly lower in *L. leucocephala* alley cropping than in a maize monocrop (Ogol et al., 1999). There were also significantly fewer stem borer eggs in unweeded maize–*Leucaena* alley cropping than in the weeded plots.

Not only do trees in alley cropping affect weeds, diseases, and insect pest, but also vertebrate pests. In Nigeria, it was difficult to establish annual crops such as maize closer to *L. leucocephala* and *Gliricidia sepium* trees than away from them because of increased damage to seedlings by birds and rodents. In Côte d'Ivoire, rodents also fed preferentially on maize and groundnut seeds sown close to the hedgerows. At harvest, the number of plants in the first crop row from the trees was reduced by 25% and 20% for maize and groundnut, respectively (Schroth et al., 1995b). In Côte d'Ivoire and elsewhere, birds and mice hiding in the foliage of the *G. sepium* hedgerows were observed to feed on the maturing rice grains (Schroth et al., 1995b).

The effect of trees on natural enemies and tri-trophic interactions has been studied more systematically in alley cropping than in the agroforestry practices discussed earlier. In the study by Girma et al. (2000), the population of ladybird beetles closely followed their prey (aphids). Activity of wasps was significantly greater close to hedgerows than away from them. Spider abundance during the maize season was 77% greater in the presence of hedgerows than in their absence, but catches during other seasons were similar between the two cropping systems. In an experiment conducted at two sites (Mtwapa and Amoyo) in Kenya, mean rates of parasitism on maize stem borer eggs, larvae, and pupae were not affected by alley cropping of maize with *L. leucocephala* at Mtwapa, whereas parasitism was significantly higher in maize monocrop than in alley cropping at Amoyo (Ogol et al., 1998). Predation of stem borer eggs was significantly higher in monocrop maize than in alley-cropping plots. There were no differences in predation between unweeded alley cropping and clear weeded plots.

5.3.2.4 Multistrata Agroforestry Systems

Multistrata agroforestry systems with tree crops comprise a variety of land use systems ranging from plantations of commercial crops under shade trees to highly diversified multistorey tree-based homegardens. Multistrata agroforestry systems may also include plantations of such crops as coffee (*Coffea* spp.), cacao (*Theobroma cacao*), or tea (*Camellia sinensis*) with various shade tree species (Beer et al., 1998). In many of these systems, coffee and cocoa are grown under a canopy of shade trees that may be remnants of the original forest or have been deliberately planted.

Tropical homegardens are the most complex of the multistrata agroforestry practices (Fernandes and Nair, 1986). In the homegardens, intensive mixed intercropping is practiced throughout the year. This involves the integration of several trees with food, cash crops, and livestock simultaneously on the same unit of land. On an average-sized farm (0.2–1.2 ha) over a hundred different plant species can be found, making this system highly integrated. The spatial arrangement of components is irregular and appears very haphazard with trees or shrubs and food crops intimately mixed. Vertically, however, 2–4 relatively distinct canopy layers can be recognized (Fernandes and Nair, 1986).

Whether a particular interaction in multistrata systems is detrimental or beneficial in terms of pest and disease incidence is largely dependent on complex factors, including management practices, the pest species, the climate, soil, and so on. Correct pruning and avoidance of heavy shade can provide some control of the many coffee diseases such as coffee berry disease (CBD), or insect pests such as Antestia bugs (*Antestisopsis* spp.), which are common pests of Arabica coffee throughout sub-Saharan Africa. *Anthestia lineaticollis* caused less damage where coffee is properly shaded, whereas capsid bugs (*Lycidocoris mimeticus*) and the coffee berry borer (*Hypotenemus hampei*) populations are favored by dense shade in coffee (Beer et al., 1998). In the Brazilian Amazon, rice, bean, and maize experienced higher pest infestations when these crops were intercropped with trees than in their respective pure crops. According to Fazolin and Estrela (1999), pest infestations depended on the tree species. A detailed overview of shade effects on crop pests is provided by Schroth et al. (2000).

Several studies have shown that trees in multistrata agroforestry can influence the abundance of natural enemies. Moderate shade favored the parasitic wasp *Cephalonomia stephanoideri* and the entomopathogenic fungus *Beauveria bassiana*, which control the coffee berry borer (Beer et al., 1998). Coconut planted in cocoa provided nest sites for the predatory ants *Dolichoderus* and *Oecophylla*, which reduced *Helopeltis* damage to cocoa (Way and Khoo, 1990). Klein et al. (2002) found an increased predator–prey ratio in more diverse traditional agroforestry systems compared with intensified systems in Indonesia. In the Guatemalan farms, Greenberg et al. (1997b) found a 30% increase in bird abundance and 15% more species in shaded than in sun coffee plantations. Greenberg et al. (1997a) found even greater increases in these values in Mexico, where shade tree canopies were more structurally and floristically diverse and less well pruned. Birds reduced the abundance of large arthropods by at least 64%–80% (Greenberg et al., 2000).

5.4 ECOLOGICAL HYPOTHESES REGARDING INTERACTIONS

From the review in the earlier section it is clear that tree–crop associations can increase, reduce, or have no effect on pest loads in agroforestry systems. This agrees with studies on mixtures of annual crops, especially intercrops (Risch et al., 1983). This shows that the response of herbivores to vegetation diversity is highly dependent on both host plant and pest species as well as management regimes (Table 5.1). An understanding of the causes for reduction in pest load in intercrops has received considerable attention. The pattern of pest incidence in agroforestry practices apparently results from a variety of causes (Table 5.1, Figure 5.1), and some of these do not have parallels in annual intercrops. In the course of this chapter, we noted that most of the studies focused on attempting to detect differences between a monoculture and an agroforestry system, which are two unrelated land use practices. In most of the studies where differences were reported, no further attempts were made to identify the underlying mechanisms that led to such differences. We found few cases (Ogol et al., 1998, 1999; Koech and Whitbread, 2000; Rao et al., 2000) where attempts were made to relate the biophysical changes that result from tree–crop interactions and their effects on pests in relation to ecological hypotheses.

Recent agroforestry literature has placed considerable emphasis on the effect of plant diversity on agroforestry pest management (Vandermeer and Perfecto, 1998; Rao et al., 2000; Schroth et al., 2000) with the optimism that structural heterogeneity and genetic diversity in agroecosystems regulate pest populations. However, the question remains as to how much diversity of plant species is required to achieve the desired pest control. There are several hypotheses describing the possible roles of increasing biodiversity in ecosystem function (Lawton, 1994), and each hypothesis can be illustrated by showing the effect of increasing species richness on the rate of an ecosystem process such as decomposition, predation, parasitism, and so on.

First, the redundant species hypothesis suggests that ecosystem processes benefit from an increase in biodiversity up to a threshold level beyond which there is no influence of further increase in species diversity. In contrast, the rivet hypothesis suggests that each species plays a significant role in affecting the ecosystem process; even a small decrease in diversity will result in a decrease in the rate of an ecosystem process. According to this hypothesis, various forms of the function between the ecosystem process and diversity are possible, but all assume that each species has a unique contribution to that process. Third, the idiosyncratic response hypothesis suggests that increasing biodiversity affects ecosystem functions in an unpredictable way because of the complex and varied roles of individual species. Finally, the null hypothesis is that ecosystem function is insensitive to species deletion or addition (Lawton, 1994).

The plant species diversity in agroforestry systems ranges from as few as two to over 100 species, and rules on the effect of diversity on pests and natural enemies, if they exist, are unlikely to apply in the same way to all systems. The experimental data available from simpler studies comparing single- and two-species mixtures show variability in the responses of individual pests. For instance, *Sesbania sesban* + *Tephrosia vogelii* increased root-knot nematodes on bean in Kenya (Desaeger and Rao, 2001), whereas the same treatment increased the incidence of snout beetles on maize in Zambia (Sileshi and Mafongoya, 2003). Similarly, while *S. sesban* + *Crotalaria grahamiana* reduced root-knot nematodes in bean in Kenya (Desaeger and Rao, 2001), the same treatment increased snout beetle incidence on maize in Zambia (Sileshi and Mafongoya, 2003) compared with pure fallows of the respective species. Although multistrata agroforests are regarded as the most diverse of all agroecosystems, the number of clearly documented cases of reduction in pest damage or increase in natural enemies is limited. There is also clear lack of experimental data to support any of the hypotheses mentioned earlier. In the following discussion, we examine the implications of tree–crop interactions on pests in the light of six other ecological hypotheses.

5.4.1 PLANT STRESS HYPOTHESIS

According to the plant stress hypothesis (White, 1984), plants that are under physiological stress represent higher quality of food for insect herbivores than those growing under optimal conditions and are likely to be more prone to pest attacks. The mechanism underlying this hypothesis is that some plants respond to stress with increases in soluble nitrogen and free amino acids in their tissues. However, not all plants respond to stress in this way, and other mechanisms have been suggested. Stress-induced changes in leaf size, leaf toughness, plant architecture, resin production, and plant physiology have also been associated with increased susceptibility to insect attack. Although water deficit is a common cause of stress in plants, factors such as browsing and excessive exposure to sun, hail damage, damage by other insects, root disturbance, and nutrient deficiency can all alter the susceptibility to insect herbivores (Speight et al., 1999; Gebeyehu and Wingfield, 2003). Drought-induced stress has been one of the well-documented cases inducing insect pest outbreaks (Mattson and Haack, 1987). Termites often attack plants stressed by drought (Logan et al., 1990). The pine weevil *Pissodes nemorensis* has been associated with trees that are stressed by hail damage and poor species-site matching (Gebeyehu and Wingfield, 2003). Stress may also be induced by damage due to insects, nematodes, plant pathogens, or weed competition. Plants damaged by one type of herbivore may also be more suitable for another. For instance, pine trees damaged by the wood wasp (*Sirex noctilio*) provide attractive breeding material for the weevil *P. nemorensis* (Gebeyehu and Wingfield, 2003). Another wood wasp (*S. giga*) depends on the fungus *Amylosteruem* spp., and acts as its vector from one pine tree to another to breakdown the host's heartwood so that its larvae can develop on the stressed host. Even routine management practices can induce stress and promote pest attack. For instance, in Kenya, pruning of *S. siamea* invoked a significant increase in attack by stem-boring larvae (Opondo-Mbai, 1995), with a resultant decline and eventual death of the plants.

In simultaneous agroforestry systems, competition between trees and crops for limited resources could increase their stress level. In semiarid environments, competition for water and nutrients dominates tree–crop interactions (Rao et al., 1998), which may affect the growth and susceptibility of crops to insects. Poor-quality tree litter in the course of decomposition may immobilize nutrients, especially nitrogen, in the soil (Mafongoya et al., 1998), and this may increase damage by insects. For example, nitrogen levels in the soil influence the level of termite damage on maize (Sileshi et al., 2005). Greater termite attack due to water stress is another major cause for high mortality of maize and tree seedlings. Agroforestry species may reduce stress indirectly by their ability to reduce weed infestation. Sileshi and Mafongoya (2003) demonstrated that termite damage on maize grown after agroforestry was lower compared with those after a natural fallow due mainly to reduction in stress caused by weed competition in the agroforestry plots. Some tree species used in agroforestry are also known to inhibit crop growth underneath their canopies due to allelopathic effects of root exudates and or litter decomposition products (Bhatt et al., 1997). The plant stress hypothesis has been a subject of considerable controversy (Speight et al., 1999).

5.4.2 PLANT VIGOR HYPOTHESIS

The plant vigor hypothesis (Price, 1991) contends that insect herbivores perform better on vigorous, not stressed plants. The plant vigor hypothesis has been supported by several cases of insect–plant associations (Speight et al., 1999). Despite the prominence of soil fertility studies in agroforestry research, there is little mention in the literature of agroforestry effects on crop health via nutrient availability. Improved soil structure and root development, and biological nitrogen fixation by legume trees significantly improve crop nutrition. On nutrient-deficient sites, the additional nitrogen, phosphorus, and potassium supply from leguminous biomass may markedly improve crop vigor (Schroth et al., 2000). For instance, in alley cropping, coppicing fallows and systems with perennial crops and leguminous shade trees, large quantities of nitrogen-rich biomass may be applied to the crops. Studies show that nitrogen is the major determinant of insect community structure (West, 1985; Speight et al., 1999). The plant vigor hypothesis points to the valid concern that nitrogen may reduce crop resistance against insects and diseases when supplied in excess.

5.4.3 Carbon–Nutrient Balance Hypothesis

According to this hypothesis, carbon-rich defensive compounds such as tannins and terpens should occur in greater concentrations in low-nutrient or high-light environments (Bryant et al., 1983). Individual plants growing under low-nutrient conditions have high carbon/nutrient ratios. Carbon-rich secondary metabolites act as sinks for excess carbon, and pathways that generate carbon-rich defenses are favored. Under high-nutrient conditions, however, carbon–nutrient ratios are lower, and pathways associated with growth and reproduction are favored over defense. Nitrogen availability is easiest to increase through agroforestry measures, and its increased supply from nitrogen-rich biomass may increase crop susceptibility. This has been demonstrated in a rice-blast pathosystem in alley cropping (Maclean et al., 1992). High nitrogen supply is also known to increase the infestation of obligate parasites such as rust fungi (*Puccinia* spp.). In addition to nitrogen, potassium is another nutrient through which agroforestry practices are most likely to affect crop health. A high potassium supply generally improves the resistance of plants to fungi, bacteria (Marschner, 1995), and nematodes up to the level required for optimum plant growth. However, woody biomass may contain high concentrations of potassium, which becomes readily available on decomposition. Mulching with prunings from legume trees in alley cropping has also been found to improve the potassium nutrition of maize (Schroth et al., 1995b). High nitrogen and potassium supply favors attack of field crops by insect pests, mainly because of the increased content of amino acids in the plant (Marschner, 1995). This points out to the fact that, in addition to increasing plant vigor, the high nutrient availability in agroforestry could reduce carbon–nutrient ratios and hence the plant's defense system. The carbon–nutrient balance hypothesis has provided a framework for much valuable research on environmental-based variation in plant defense, yet it remains controversial. Nonetheless, a significant number of studies support, at least partially, this hypothesis (Speight et al., 1999).

5.4.4 Natural Enemies Hypothesis

The natural enemies hypothesis posits that vegetation diversity increases both population size and impact of predators and parasitoids that regulate herbivorous arthropod pests (Root, 1973). Many studies of polyculture systems have supported the natural enemies hypothesis, whereas others have reported neutral or even negative responses (Letourneau, 1987; Ogol et al., 1998). Only few studies have assessed the effect of agroforestry practices on interactions in relation to the natural enemies hypothesis.

Ogol et al. (1998) evaluated the natural enemies hypothesis in a maize–*L. leucocephala* hedgerow intercropping at two sites in Kenya. In this study, the rates of egg, larval, and pupal parasitism of maize stem bores contradicted the natural enemies hypothesis at one site, whereas parasitism showed a neutral response to plant diversity at the other site. In the same study, egg predation rates contradicted the natural enemies hypothesis, whereas pathogen-associated mortality of stem borer larvae exhibited a neutral response to plant diversity (Ogol et al., 1998). The examples above indicate that the notion that vegetation diversity in agroforestry increases abundance of natural enemies is clearly not tenable. In fact, there is no rule of thumb or general theory that globally predicts population size or activity of natural enemies in diverse agroecosystems, and thus each system must be evaluated individually.

5.4.5 Resource Concentration Hypothesis

The resource concentration hypothesis (Root, 1973), also called disruptive-crop hypothesis (Vandermeer, 1989), may operate when a pest (1) is less likely to find its host plant because of some kind of chemical or physical confusion imposed by a second species and (2) after finding a host plant, it is more likely to leave that patch because of the presence of nonhost plants. The disruptive species may also exert its influence indirectly by creating an unfavorable microclimate for the pest, or by affecting the quality of the host plants making them less desirable to the pest compared with individuals in monoculture. The resource concentration hypothesis is largely applicable to specialist herbivores (Vandermeer, 1989). However, it is equally applicable to the incidence of diseases.

Trees being the taller component in agroforestry may act as physical barriers to the dispersal and colonization of a crop by both herbivores and natural enemies or have a biological role in repelling pests because of their unfavorable morphological features. Upper story trees may camouflage the understory host crops and prevent pests from recognizing them from a distance. In such systems, herbivores such as aphids are more likely to be affected because of their relatively poor efficiency in locating their host plants and their inability to survive for long without feeding. As the number of plant species increases, the number of aphid species is also known to decrease (Dixon et al., 1987).

Ogol and coworkers (1998) provide a direct support for this hypothesis from a maize–*L. leucocephala* alley-cropping study in Kenya. In their study, colonization by maize stem borers (*Chilo* spp.), which are relatively specialist herbivores, was lower in alley cropping compared with a monocrop maize. Host location was probably affected by the presence of both *Leucaena* and weeds, which reduced the borers' ability to locate their hosts. The weeds and hedges also acted as a mechanical barrier to the dispersal of the young larvae of maize stalk borers. This is evidenced by *Chilo* egg batches deposited on weeds (Ogol et al., 1998).

Like the other hypotheses, the resource concentration hypothesis has been a subject of considerable controversy, and may not adequately explain some of the population variations. For instance, Rhainds and English-Loeb (2003) experimentally manipulated attributes of patches with strawberry plants to assess the impact of patch size and host density on the abundance of tarnished plant bug and fruit damage. The density of nymphs increased with patch size and host density for some but not all generations of plant bug, providing partial support for the resource concentration hypothesis. The validity of the resource concentration hypothesis needs to be tested in the more species-rich tropical homegardens.

5.4.6 MICROCLIMATE HYPOTHESIS

The microclimate hypothesis (Koech and Whitbread, 2000) is based on the observation that agroforestry practices affect the microclimate around the crop. The microclimate changes caused by trees in tree–crop associations include shading of the understory crops, increased relative humidity, reduced air and soil temperatures, and decreased wind speed (Schroth et al., 1995a; Koech and Whitbread, 2000). The canopy cover also affects the microclimate of the understory (Perfecto and Vandermeer, 1996). These changes may have negative, positive, or neutral impacts on weeds, pathogens, insect, and their natural enemies (Sileshi, 1997). For instance, the complementary effect of shading in the alleys, mulch from prunings, and potential allelopathy from hedgerow species reduce weed populations in alley cropping (Kang, 1993; Rao et al., 1998). Shading by *Gliricidia sepium* and *L. leucocephala* caused 31% and 25% rhizome mortality in speargrass, respectively (Anoka et al., 1991). Hedgerow shading and mulch may lead to shifts over time in the composition of weed species (Anoka et al., 1991; Ong and Rao, 2001). In Nigeria, *G. sepium* and *L. leucocephala* caused a shift from speargrass to other weeds such as *Rottboellia, Hippocratea, Chromolaena, Talinum,* and *Euclasta* (Anoka et al., 1991).

Changes in light, temperature, and relative humidity and leaf wetness have been cited as causes for increased incidence and severity of anthracnose and angular leaf spot on beans in rows adjacent to *L. leucocephala* hedges (Koech and Whitbread, 2000), and leaf spot and rust on groundnut in *G. sepium* alley cropping (Schroth et al., 1995a). Similarly, increased populations of *Cicadulina* sp., which is the vector of maize streak virus, under trees in Burkina Faso was due to the change in microclimate (Traoré and Quedraogo, 1997). Similarly, the shading due to the canopy appears to be a major factor influencing the arthropod community in shade-coffee (Perfecto and Vandermeer 1996; Greenberg et al., 2000).

5.5 SUMMARY AND CONCLUSIONS

Tropical agroforestry practices range from the short-duration improved fallows where a single legume tree or shrub species is rotated with a crop to the most diverse multistrata homegardens. In simultaneous systems such as alley cropping and coppicing fallows, the contrasts between the component species in their physical dimensions, their life span, and their physiological responses may lead to complex interactions between the tree and the crop species. Therefore, the tri-trophic interactions occurring here are expected to be more complex than those in sequential tree–crop fallow systems. The arrangement and the management of trees in relation to crops within an agroforestry technology has a bearing on microclimatic factors. Both bottom-up effects of the abiotic environment and top-down effects of herbivores and their natural enemies can modify pest incidence in the system. Pest problems may increase in some systems that either induce stress (plant stress hypothesis) or improve crop vigor (crop vigor hypothesis and carbon–nutrient balance hypothesis).

None of the proposed hypotheses really explain all the possible mechanisms by which pest risks will increase or decrease in agroforestry. It is well to remember that all of the hypotheses have been convincingly demonstrated in one system or another, and it is not a question of one being generally right and the other wrong. Some degree of advocacy seems to have also evolved concerning them (Vandermeer, 1989; Speight et al., 1999). However, this does not necessarily mean that they are adequate to explain those mechanisms that reduce or increase pest load in all systems. For instance, where trees are dispersed in cropland, the influence of trees on pests is probably limited to the crop under the tree canopy (Rao et al., 2000). In alley cropping or boundary planting, the interaction between trees and crops is mostly confined to the tree–crop interface, so trees are unlikely to influence pests on crop plants several meters away from them as indicated by the various examples (Yamoah and Burleigh, 1990; Jama et al., 1991; Koech and Whitbread, 2000). This calls for reexamination of the various hypotheses in the different systems.

In some cases, joint operation of two hypotheses is a clear possibility. It must also be borne in mind that it is difficult to reconcile some of the hypotheses, for instance, the plant stress hypothesis and the plant vigor hypothesis, into a single theory to explain patterns of insect attack. Some of the contradictions arise from the fact that there are many different kinds of stress, many idiosyncratic responses by plants, and equally diverse responses by insects. The biggest challenge now is to understand these idiosyncrasies and apply them to design of agroforestry systems. If agroforestry is to thrive well as an applied ecological science offering strategies for sustainable utilization of natural resources, pest management should be based on applying ecological principles and practical decision-making tools.

The extent of pest damage in any of the systems may be determined by the interactions (1) between the plant community and the herbivore or pathogen, (2) between the herbivores and their natural enemies, and (3) among components (tree, crop, soil, and environment) of the system. The consequences of these interactions may have a positive, negative, or neutral effect on pests. An understanding of these interactive effects on pests and their natural enemies at different spatial and temporal scales is essential. This will help in designing more robust agroforestry practices that lower pest problems. It is an ecological maxim that diversity is closely related to stability (Risch et al., 1983). However, simply increasing diversity will not necessarily increase the stability of all ecosystems. Ewel (1999) pointed out from the experience of his constructed mimics that diversity cannot be counted on to afford protection from herbivores, and some times can have the opposite effect. For employing plant diversity strategically in agroforestry design, more is required than simply adding more plant species to a species-poor system. As argued by Ewel (1999), any addition of species into existing systems must also be based on recognition of the existing biophysical conditions and less so on the structural and functional dynamics of ideal native vegetation or man-made models. In short, design of innovative agroforestry practices that reduce pest management should be based on ecological principles.

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6 Ecologically Based Pest Management in Agroforestry Systems

Miguel A. Altieri and Clara I. Nicholls

CONTENTS

6.1 INTRODUCTION

Agroforestry is an intensive land-management system that combines trees and shrubs with crops and livestock in time and space on a landscape level to achieve optimum benefits from biological interactions between soils, plants, and arthropods. Agroforestry systems (AFS) aim at balancing ecosystem demands to sustain diversity and productivity, while meeting multiple-use and sustained-yield needs of agriculture (Nair, 1993; Sanchez, 1995). Indigenous farmers in the developing world who usually understand land-use interactions in their local ecosystems often apply the systems successfully. Examples include the multistoried coffee- and cacao-based agroforests in Latin America and the complex homegardens in Asia. Many of the benefits of AFS are derived from the increased diversity of these systems compared with corresponding monocultures of crops or trees. Although little research has been conducted on pest interactions within AFS, agroforestry has been assumed to reduce pest outbreaks usually associated with monocultures. Although the effects of various agroforestry designs on pest populations can be of a varied nature (microclimatic, nutritional, natural enemies, etc.), regulating factors do not act in isolation from each other.

The few reviews on pest management in agroforestry (Rao et al., 2000; Schroth et al., 2000) stipulate that the high plant diversity associated with AFS provide some level of protection from pest and disease outbreaks. To explain such regulation, these authors use the same theories advanced by agroecologists to explain lower pest levels in annual polycultural agroecosystems (Andow, 1991; Altieri and Nicholls, 2004). Some authors caution that the use of high plant diversity as a strategy to reduce pest and disease risks in AFS meets considerable technical difficulties as the design and management of complex systems is cumbersome. Similar to orchard situations, AFS can be considered semipermanent, relatively undisturbed systems, with no fallow or crop rotation, thus exhibiting particular biological situations affecting insects. Insect populations are more stable in complex AFS because a diverse and more permanent habitat can maintain an adequate population of the pest and its enemies at critical times (van den Bosch and Telford, 1964). For most entomologists, the relative permanency of AFS affords the opportunity of manipulating the components of the habitat to the benefits of ecologically sound pest management practices (Prokopy, 1994). Such practices include the manipulation of ground cover vegetation and of shade tress to either directly stress arthropod pests or enhance their mortality through biological control.

This chapter focuses on the effects of vegetationally diverse AFS on the ecology of insect pests, concentrating more specifically on the actual or potential mechanisms underlying pest reduction in AFS and provides key information to design ecologically based pest management systems in AFS.

6.2 BIODIVERSITY, BIOTIC INTERACTIONS, AND IDEAS FOR PEST MANAGEMENT

The biodiversity components of AFS can be classified in relation to the role they play in the functioning of AFS. According to this, biodiversity can be grouped as follows (Swift and Anderson, 1993):

1. *Productive biota*: crops, trees, and animals chosen by farmers that play a determining role in the diversity and complexity of the agroecosystem
2. *Resource biota*: organisms that contribute to productivity through pollination, biological control, decomposition, and so on
3. *Destructive biota*: weeds, insect pests, microbial pathogens, and so on, which farmers aim at reducing through cultural management

Two distinct components of biodiversity can be recognized in AFS (Vandermeer and Perfecto, 1995). The first component, planned biodiversity, includes the crops and livestock, purposely included in AFS by the farmer, and which varies depending on the management inputs and crop spatial or temporal arrangements (Hart, 1980). The second component, associated biodiversity, includes all soil flora and fauna, herbivores, carnivores, decomposers, and so on, which colonize the agroecosystem from surrounding environments and that will thrive in the agroecosystem depending on its management and structure. The relationship of both types of biodiversity components is illustrated in Figure 6.1. Planned biodiversity has a direct function, as illustrated by the bold arrow connecting the planned biodiversity box with the ecosystem function box. Associated biodiversity also has a function, but it is mediated through planned biodiversity. Thus, planned biodiversity also has an indirect function, illustrated by the dotted arrow in Figure 6.1, which is realized through its influence on the associated biodiversity. For example, the trees in an AFS create shade, which makes it possible to grow only sun-intolerant crops. So, the direct function of this second species (the trees) is to create shade. Yet, along with the trees, wasps might come to seek out the nectar in the tree's flowers. These wasps may in turn be the natural parasitoids of pests that normally attack crops. The wasps are part of the associated biodiversity. The trees then create shade (direct function) and attract wasps (indirect function) (Vandermeer and Perfecto, 1995).

Complementary interactions between the various biodiversity components can also be of a multiple nature. Some of these interactions can be used to induce positive and direct effects on the biological control of specific crop pests, soil fertility regeneration, and enhancement and soil conservation. The exploitation of these interactions in real situations involves agroforestry design and management and requires an understanding of the numerous relationships between soils, microorganisms, plants, insect herbivores, and natural enemies.

According to agroecological theory (Altieri, 1995), the optimal behavior of AFS depends on the level of interactions between the various biotic and abiotic components. By assembling a functional biodiversity, it is possible to initiate synergisms that subsidize AFS processes by providing

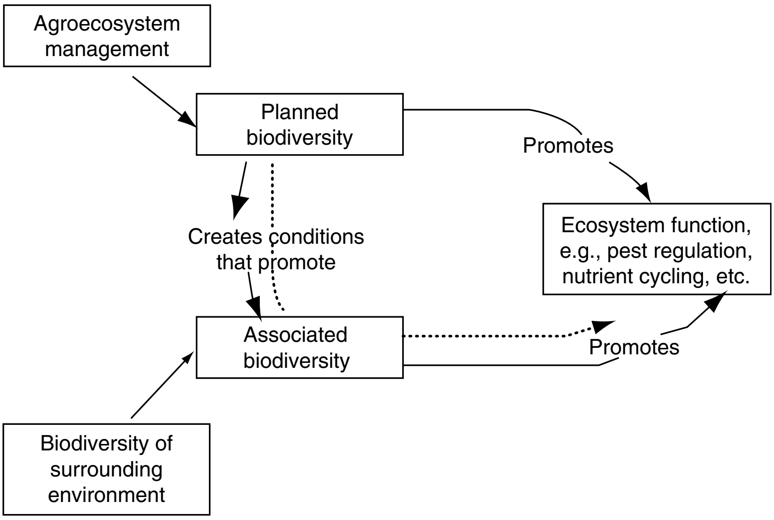

FIGURE 6.1 Types of biodiversity and their role in pest regulation in agroforestry systems.

ecological services such as the activation of soil biology, the recycling of nutrients, the enhancement of beneficial arthropods and antagonists, and so on (Gliessman, 1999), all important in determining the sustainability of agroecosystems.

The experimental evidence suggests that biodiversity can be used for improved pest management in agroecosystems (Andow, 1991; Altieri and Nicholls, 2004). Several studies have shown that it is possible to stabilize the insect communities of agroecosystems by designing and constructing vegetational architectures that support populations of natural enemies or have direct deterrent effects on pest herbivores (Gurr et al., 2004).

The key is to identify the type of biodiversity that is desirable to maintain and enhance in order to carry out ecological services, and then to determine the best practices that encourage the desired biodiversity components (Figure 6.2). There are many agricultural practices and designs that have the potential to enhance functional biodiversity, and others that negatively affect it. Although many of these strategies apply to agricultural systems, the idea is to apply the best management practices to enhance or regenerate the kind of biodiversity that can subsidize the sustainability of AFS by providing ecological services such as biological pest control, nutrient cycling, water and soil conservation, and so on. The role of agroecologists should be to encourage those agricultural practices that increase the abundance and diversity of aboveground and belowground organisms, which in turn provide key ecological services to AFS. Shelterbelts, cover crops, and shade trees are among the best practices to stimulate synergy in AFS.

Thus, a key strategy of agroecology is to exploit the complementarity and synergy that result from the various combinations of crops, trees, and animals in AFS featuring novel spatial and temporal arrangements. In real situations, the exploitation of these interactions involves agroecosystem design and management and requires an understanding of the numerous relationships among soils, microorganisms, plants, insect herbivores, and associated natural enemies.

6.3 ECOLOGICAL CONSEQUENCES OF BIODIVERSITY REDUCTION IN AFS: A CASE STUDY FROM PORTUGAL

One way to appreciate the key ecological role of biodiversity in AFS is to study systems in which biodiversity levels are reduced in traditional agroecosystems such as in the case of centuries-old vineyard agroforests in the Vinho Verde Region of northwest Portugal (Altieri and Nicholls, 2002).

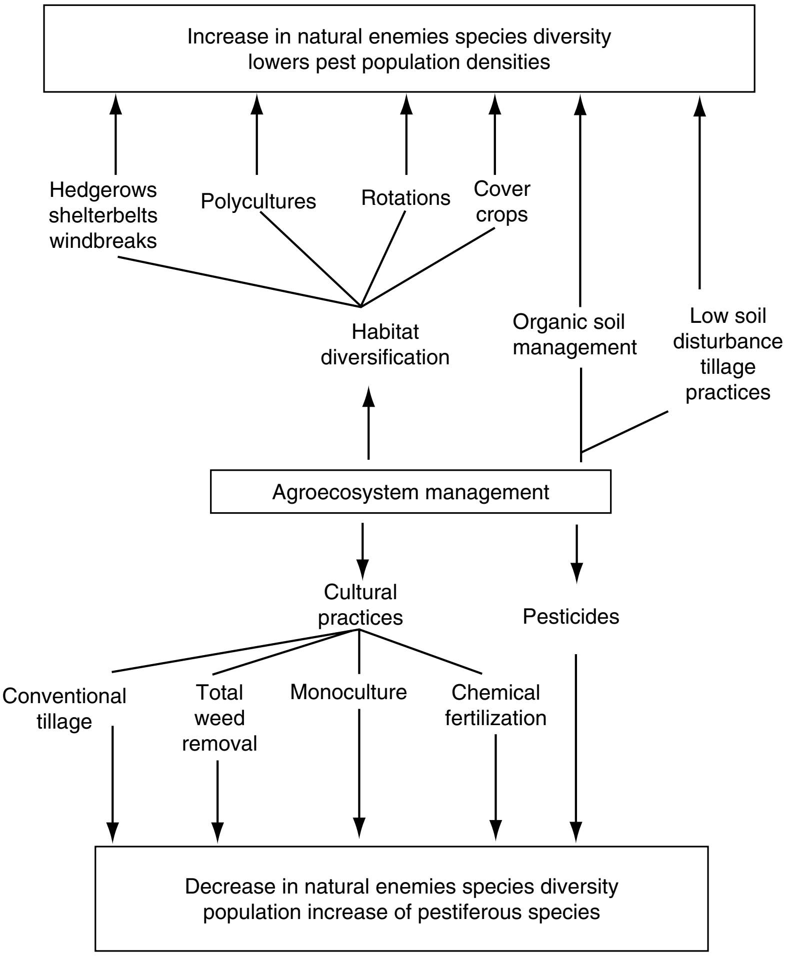

FIGURE 6.2 Assortment of agricultural practices that enhance beneficial biodiversity in agroforestry systems.

Traditionally, vines are grown on host trees circumscribing small fields diversified with crops, vegetables, and forage for animals. In these systems, arbor style diversified vines integrated into cropping systems modify the environment of associated understory plants, influencing their growth, pest susceptibility, and yields. The greatest modification for crops apparently results from the interception of wind and some solar radiation, but for vines growing in vertical structures there are clear microclimatic effects. There are a number of traditional agroforestry patterns, all of which represent an ingenious response to land constraints by allowing vertical agriculture:

1. Association of vines and trees dispersed within fields. This simple system consists of a tree with 4–8 vines planted around the base. The vines ascend and follow the branches.
2. ‘‘Festoon’’ system in which younger cross-branches of the vines join together every year from the nearest trees planted along field margins.
3. ‘‘Arjoado’’ system is a form of festoon, but with vertical wires attached to the wire that runs between the trees. In addition to planting vines against the tree trunks, several vines can be planted in the intervening area.

In these systems, preferred host trees are Portuguese Oak (*Quercus lusitanica*), elm (*Ulmus* sp.), poplar (*Populus* sp.), and wild cherry (*Prunus* sp.). The trees tolerate heavy trimming, have deep roots, grow fast, and are long lived. Most yield products such as wood, bark, and fruits. Many trees provide additional benefits such as altering the microclimate (interception of winds and lower evaporation rates) and protecting vines from winter frosts of the valley bottom. Trees can also reduce dispersion of weed seeds, insects, and pathogen inocula by forming a physical barrier.

The centers of the fields are available for grain (mostly maize, *Zea mays*), legumes, and vegetables. Normal crop rotations include oat grain (*Holcus lanatus*), rye grain (*Lolium multiflorum*), and the legumes *Ornithopus sativa* and *Trifolium incarnatum*, all used as fodder. Some fields are left fallow for the growth of volunteer legumes (mostly species of *Ulex* and *Spartium*) used for "cattle beds" in the stalls. On mixing with urine and feces of the cattle, the semi-decomposed materials of the beds are worked into the soil of the farms as organic amendment.

In northern Portugal, vineyards are affected by various pathogens, insects, and mites. Among insect pests, the tortricid moth, *Lobesia botrana*, is the most persistent one. Of the three generations of this lepidopteran, the two first generations are of greatest economic significance. Leafhoppers are also present (especially *Empoasca vitis*, cigarrinha verde), puncturing leaves and eventually causing leaves to fade, dry up, and fall off the vine. Downy mildew (*Plasmopara viticolor*), powdery mildew (*Uncinula necator*), and bunchrot (*Botrytis cinerea*) are the most prevalent fungal pathogens of grapes in the area. Most of these insects and fungi reach, only sporadically, epidemic proportions in traditional agroforests.

During the past 10 years, major economic policy-induced changes have occurred in the Vinho Verde wine industry. Farmers are encouraged to plant varieties that produce better-quality white wines and move away from agroforestry-based vineyards to the "cordao" monoculture system characterized by short, vertical trellises for easy mechanization. Although the systems reduce labor costs and may enhance profit levels, the cordao involves less-intensive land use. The modern system is totally integrated into the market, and little importance is given to production of crops and wine for home consumption. In addition, the intensification of grape production changes the diversity and microclimate of the vineyard, creating new environmental conditions that may favor some pests.

During 1997–1999 growing seasons, field surveys were conducted in a few selected fields to elucidate levels of insect species diversity and the population trends of pest insects (the leafhopper *E. vitis* and the lepidoptera *L. botrana*) and associated natural enemies, and the resulting degree of pest damage in two dominant vineyard systems (vineyards under traditional management—arjoado system and vineyards in the process of modernization under monoculture-cordao system) (Altieri and Nicholls, 2002).

In both years (1997 and 1999), the number of insect species and the total number of individuals collected per plot was greater in AFS than in monocultures. The number of predator and parasite species was substantially greater in the traditional diversified arjoado systems than in the cordao monocultures. Main predator species included various species of Coccinellidae (*Stethorum punctilum* and others), Syrphidae, *Chrysoperla carnea*, *Orius* spp., and others. Parasitoids belonged predominantly to the family Ichneumonidae, although we detected parasitism of *L. botrana* eggs by naturally occurring *Trichogramma* spp. parasitic wasps.

In the arjoado systems, higher insect biodiversity is probably the result of increased spatial heterogeneity and complexity of the agroforests. The presence of a diversity of crops and also of some weeds in the "arjoado" increased the amount of food resources (flowers, extra floral nectarines, and alternate prey), which may explain the greater abundance and diversity of natural enemies. In contrast, the lack of insect biodiversity in mechanized systems was probably due to the lack of plant diversity, and to the higher load of insecticides (mainly organophosphates and carbamates) that cordao systems receive.

Abundance monitoring of herbivores was difficult in the monoculture systems as insecticide applications prevented pest population buildup. However, delayed spraying in one modernized farm in 1999 allowed us to compare densities of *L. botrana* and *E. vitis* nymphs between this vineyard monoculture and a neighboring traditional vineyard. As observed in Figure 6.3, densities of leafhopper nymphs tended to be substantially lower from early June to mid-September on leaves

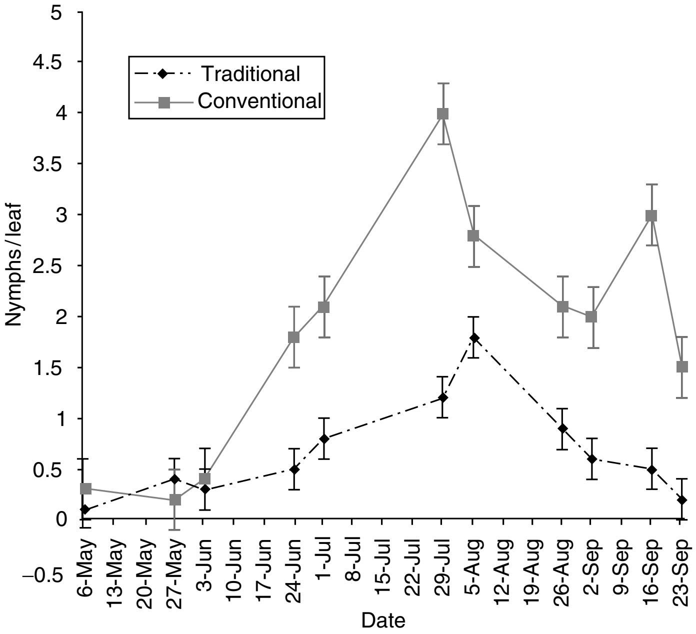

FIGURE 6.3 Nymphal densities of *Empoasca vitis* in modern and traditional vineyards in northwestern Portugal (1999).

in AFS than in the cordao monoculture. Similarly, from late June to mid-July, larval densities of *L. botrana* were higher in monocultures than those in the traditional system (Figure 6.4), which corresponded with a higher proportion of vine inflorescences infested by *L. botrana* larvae in monocultures than in the vine agroforest.

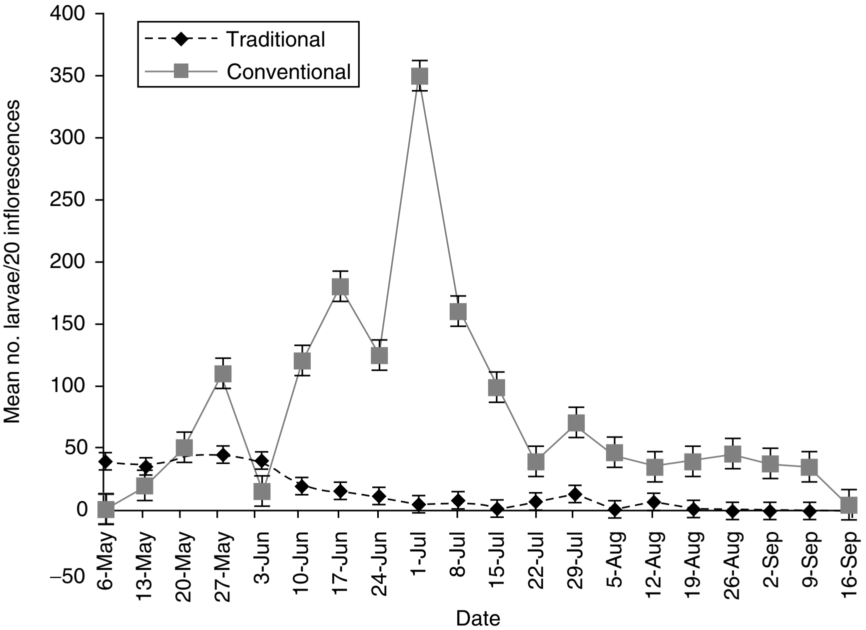

FIGURE 6.4 Infestation of grapes by *Lobesia botrana* in traditional and modern vineyards in northwestern Portugal (1999).

The field data suggest that AFS exhibit higher levels of insect biodiversity possibly linked to the higher vegetational complexity of such systems, they are less dependent on external inputs (chemical pesticides), and tend to have fewer insect pest and disease problems than unsprayed modern vineyard monocultures.

Although the shift toward cordao monoculture potentially represents a more labor-saving and profitable system, at the same time it can be a risky specialization in production. In the few vineyards where we were able to compare through systematic sampling, our findings suggest that promoted modern technological schemes may be ecologically unsound. Vineyards converted to monocultures exhibited larger numbers of leafhopper and lepidopteran pests, than more diversified traditional adjacent systems featuring the same grape varieties. The strategy of yield maximization with pest control left primarily to pesticides has increased grape production by 20%–35%, but at the expense of higher vulnerability of the vineyards and possible environmental risks.

There is strength in the diversity of traditional vineyards, and it should not be reduced by extensive monoculture, especially when consequences of doing so may result in serious ecological and social problems. Instead, modernization should be guided by agroecological principles, principles whose source are the very traditional systems that modernity is destroying. As rural change occurs in Portugal, given EEC policy-driven agricultural modernization trends, knowledge of traditional management practices and the ecological rationale behind them is gradually being lost.

6.4 EFFECTS OF TREES IN AGROFORESTRY ON INSECT PESTS AND ASSOCIATED NATURAL ENEMIES

The deliberate association of trees with agronomic crops can result in insect management benefits because of the structural complexity and permanence of trees and to their modification of microclimates and plant apparency within the production area. Individual plants in annual cropping systems are usually highly synchronized in their phenology and short lived. In such systems, the lack of temporal continuity is a problem for natural enemies because prey availability is limited to short periods of time and refugia and other resources, such as pollen, nectar, and neutral insects, are not consistently available. The addition of trees of variable phenologies or diverse age structure through staggered planting can provide refuge and a more constant nutritional supply to natural enemies because resource availability through time is increased (Rao et al., 2000). Trees can also provide alternate hosts to natural enemies, as in the case of the planting of prune trees adjacent to grape vineyards to support overwintering populations of the parasitoid *Anagrus epos*, which later migrate into adjacent vineyards and regulate populations of the grape leafhopper (Murphy et al., 1996).

6.4.1 TREE SHADE EFFECTS

Shade from trees may markedly reduce pest density in understory intercrops. Hedgerows or windbreaks of trees have a dramatic influence on microclimate; almost all microclimate variables (heat input, wind speed, soil desiccation, and temperature) are modified downwind of a hedgerow. Tall intercrops or thick groundcovers can also alter the reflectivity, temperature, and evapotranspiration of shaded plants or at the soil surface, which in turn could affect insects that colonize according to "background" color or those that are adapted to specific microclimatological ranges (Cromartie, 1991). Both immature and adult insect growth rates, feeding rates, and survival can be markedly affected by changes in moisture and temperature (Perrin, 1977).

The effect of shade on pests and diseases in agroforestry has been studied quite intensively in cocoa and coffee systems undergoing transformation from traditionally shaded crop species to management in unshaded conditions. In cocoa plantations, insufficient overhead shade favors the development of numerous herbivorous insect species, including thrips (*Selenothrips rubrocinctus*) and mirids (*Sahlbergella, Distantiella*, and so on). Even in shaded plantations, these insects concentrate at spots where the shade trees have been destroyed, for example, by wind (Beer et al., 1997). Bigger (1981) found an increase in the numbers of Lepidoptera, Homoptera, Orthoptera, and the mirid *Sahlbergella singularis* and a decrease in the number of Diptera and parasitic Hymenoptera from the shaded toward the unshaded part of a cocoa plantation in Ghana.

In coffee, the effect of shade on insect pests is less clear than that in cocoa, as the leaf miner (*Leucoptera meyricki*) is reduced by shade, whereas the coffee berry borer (*Hypothenemus hampei*) may increase under shade. Similarly, unshaded tea suffers more from attack by thrips and mites, such as the red spider mite (*Oligonychus coffeae*) and the pink mite (*Acaphylla theae*), whereas heavily shaded and moist plantations are more damaged by mirids (*Helopeltis* spp.) (Guharay et al., 2000).

In Central America, coffee berry borer appears to perform equally well in open sun and managed shade, but naturally occurring *Beauveria bassiana* (an entomopathogenic fungus) multiplies and spreads more quickly with greater humidity, therefore entomopathogenic fungus applications should coincide with peaks in rainfall (Guharay et al., 2000). After a study of how the microclimate created by multi-strata shade management affected herbivores, diseases, weeds, and yields in Central America coffee plantations, Staver et al. (2001) defined the conditions for minimum expression of the pest complex. For a low elevation dry coffee zone, shade should be managed between 35% and 65%, as shade promotes leaf retention in the dry season and reduces *Cercospora coffeicola*, weeds and *Planococcus citri* (Figure 6.5).

Obviously, the optimum shade conditions for pest suppression differ with climate, altitude, and soils. The selections of tree species and associations, density and spatial arrangements as well as shade management regimes are critical considerations for shade strata design.

The complete elimination of shade trees can have an enormous impact on the diversity and density of arthropods, especially ants. Studying the ant community in a gradient of coffee plantations going from systems with high density of shade to shadeless plantations, Perfecto (1995) reported a significant decrease in ant diversity. Although there exists a relationship between ant diversity and pest control, research suggests that a diverse ant community can offer more safeguards against pest outbreaks than a community dominated by just a few species. In Colombia, preliminary reports point to lower levels of the coffee borer, the main coffee pest in the region, in shaded coffee plantations. There is an indication that a nondominant small ant species is responsible for the

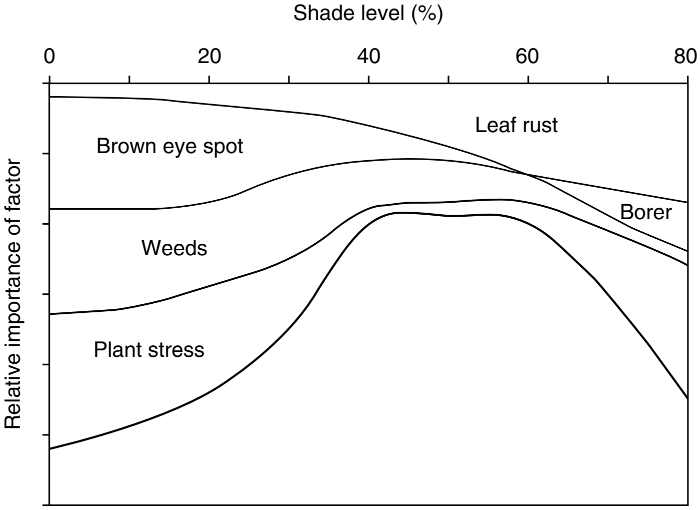

FIGURE 6.5 Conceptual graph depicting the relative importance of yield-reducing factors in a low, dry coffee zone in Nicaragua. Effects are shown to be additive with the effect of each successive pest represented by the area between the lines. The lowest line indicates the accumulated potential for yield reduction at different shade levels. Since the *y*-axis is negative, the range of least yield reduction is 35%–65%.

control. Apparently, this species does not live in unshaded plantations. In cocoa, ant species that flourish under shaded conditions have been very successful in controlling various pests. One of the most obvious consequences of pruning or shade elimination, with regard to the ant community, is the change in microclimatic conditions. In particular, microclimate becomes more variable with more extreme levels of humidity and temperature, which in turn promotes changes in the composition of the ant community (Perfecto and Vandermeer, 1996).

6.4.2 CROP ATTRACTIVENESS

Chemical cues used by herbivores to locate host plants may be altered in an AFS. Trees may exhibit a markedly different chemical profile than annual herbaceous plants intercropped in the system, masking or lessening the impact of the chemical profile produced by the annual crop. Several studies have demonstrated olfactory deterrence as a factor in decreasing arthropod abundance (Risch, 1981). The attractiveness of a plant species for the pests of another species can be usefully employed in agroforestry associations in the form of trap crops that concentrate the pests or disease vectors, a place where they cause less damage or can be more easily neutralized (e.g., by spraying or collecting). Such trap crops are an interesting option when they attract pests from the primary crop within the field (local attraction), but not when they attract pests from areas outside the field (regional attraction). Nascimento et al. (1986) demonstrated the strong attraction of the *Citrus* pest *Cratosomus flavofasciatus* by the small tree *Cordia verbenacea* in Bahia, Brazil, and recommended the inclusion of this tree at distances of 100–150 m in *Citrus* orchards. They speculated that pests of several other fruit crops could similarly be trapped by this tree species.

In certain AFS, such as alley cropping, which usually include leguminous shade trees, relatively large quantities of N-rich biomass are applied to crops via branch trimmings left on the soil surface. In cases of luxury additions of N, this may result in reduced pest resistance of the crops. The reproduction and abundance of several insect pests, especially Homoptera, are stimulated by high concentration of free nitrogen in the crop's foliage resulting from N fertilization (Altieri and Nicholls, 2003).

6.4.3 COVER CROP EFFECTS

The manipulation of ground cover vegetation in tropical plantations can significantly affect tree growth by altering nutrient availability, soil physics, and moisture, and the prevalence of weeds, plant pathogens, and insect pests and associated natural enemies (Haynes, 1980). A number of entomological studies conducted in these systems indicate that plantations with rich floral undergrowth exhibit a significantly lower incidence of insect pests than clean cultivated orchards, mainly because of an increased abundance and efficiency of predators and parasitoids, or other effects related to habitat changes. In the Solomon Islands, O'Connor (1950) recommended the use of a cover crop in coconut groves to improve the biological control of coreid pests by the ant *Oecophylla smaragdina subnitida*. In Ghana, coconut gave light shade to cocoa and supported, without apparent crop loss, high populations of *O. longinoda,* keeping the cocoa crop free from cocoa capsids (Leston, 1973).

Wood (1971) reported that in Malaysian oil palm (*Elaeis guineensis*) plantations, heavy ground cover, irrespective of type, reduced damage to young trees caused by rhinoceros beetle (*Oryctes rhinoceros*). The mode of action is not certain, but it appears that the ground cover impedes flight of the adult beetles or restricts their movement on the ground. Economic control of this pest was possible by simply encouraging the growth of weeds between the trees.

6.4.4 PLANT DIVERSITY AND NATURAL ENEMIES

In Kenyan studies assessing the effects of nine hedgerow species on the abundance of major insect pests of beans and maize, and associated predatory or parasitic anthropods, Girma et al. (2000)

found that beanfly (*Ophiomyia* spp.) infestation was significantly higher in the presence of hedgerows (35%) than in their absence (25%). Hedgerows did not influence aphid (*Aphis fabae*) infestation of beans. In contrast, maize associated with hedgerows experienced significantly lower stalk borer (*Busseola fusca* and *Chilo* spp.) and aphid (*Rhophalosiphum maidis*) infestations than pure maize, the margin of difference being 13% and 11%, respectively, for the two pests. Ladybird beetles closely followed their prey, aphids, with significantly higher catches in sole cropped plants than in hedgerow plots and away from hedgerows. Activity of wasps was significantly greater, close to hedgerows than away from them. Spider catches during maize season were 77% greater in the presence of hedgerows than in their absence, but catches during other seasons were similar between the two cropping systems.

In one of the few studies of the influence of temperate agroforestry practices on beneficial arthropods, Peng et al. (1993) confirmed the increase in insect diversity and improved natural enemy abundance in an alley-cropping system over that of a monoculture crop system. Their study examined arthropod diversity in control plots sown to peas (*Pisum sativum* var. *sotara*) versus peas intercropped with four tree species (walnut, ash, sycamore, and cherry) and hazel bushes. They found greater arthropod abundance in the alley-cropped plots than in the control plots, and natural enemies were more abundant in the tree lines and alleys than in the controls. The authors attributed the increase in natural enemies to the greater availability of overwintering sites and shelter in AFS. In subsequent work, Stamps et al. (2002) examined the effects of two forages (alfalfa and smooth bromegrass) on the growth, nut production, and arthropod communities of alley-cropped eastern black walnut, *Juglans nigra*. They found no differences in tree growth among alleyway treatments. The first season's nut yield was greater from trees with vegetation-free alleyways; otherwise, nut production did not differ among the treatments. Arthropods were more numerous and diverse in alley-cropped alfalfa than in alley-cropped bromegrass or in the vegetation-free controls. Alley-cropped bromegrass supported a more diverse population of arthropods than did the vegetation-free control.

In Turkey, Akbulut et al. (2003) found that beneficial arthropods reached significantly higher numbers in maize, bean, and zucchini grown between alleys of hybrid poplar than in monocultures. Trees provided a more favorable habitat for beneficial insects, and therefore AFS contributed to increased arthropod biodiversity. Stamps and Linit (1997) argue that agroforestry holds promise for increasing insect diversity and reducing pest problems because the combination of trees and crops provides greater niche diversity and complexity in both time and space than the polyculture of annual crops.

6.5 ECOLOGICAL PRINCIPLES FOR DESIGN

As traditional farmers have done, natural successional communities can be used as models for agroecosystem design because they offer several traits of potential value to agriculture: (1) high resistance to pest invasion and attack, (2) high retention of soil nutrients, (3) enhanced agrobiodiversity, and (4) reasonable productivity (Ewel, 1999). As stated by Gliessman (1998), a major challenge in the tropics is to design agroecosystems that, on the one hand, take advantage of some of the beneficial attributes of the early stages of succession yet, on the other hand, incorporate some of the advantages gained by allowing the system to reach the later stages of succession. Only one desirable ecological characteristic of agroecosystems—high net primary productivity—occurs in the early stages of development, an important reason to create more permanent agroecosystems through the inclusion of perennials. The application of the following principles can lead to the design of more mature, complex, and pest-stable AFS:

1. *Increasing species diversity* as this promotes fuller use of resources (nutrients, radiation, water, etc.), protection from pests, and compensatory growth. Many researchers have highlighted the importance of various spatial and temporal plant combinations to facilitate

complementary resource use or to provide intercrop advantage such as in the case of legumes facilitating the growth of cereals by supplying it with extra nitrogen. Compensatory growth is another desirable trait as if one species succumbs to pests, weather, or harvest, another species fills the void maintaining full use of available resources. Crop mixtures also minimize risks, especially by creating the sort of vegetative texture that controls specialist pests.

2. *Enhance longevity* through the addition of perennials that contain a thick canopy thus providing continual cover that protects the soil. Constant leaf fall builds organic matter and allows uninterrupted nutrient circulation. Dense, deep-root systems of long-lived woody plants are an effective mechanism for nutrient capture offsetting the negative losses through leaching.
3. *Impose a fallow* to restore soil fertility through biomass accumulation and biological activation, and to reduce agricultural pest populations as life cycles are interrupted with a rotation of fallow vegetation and crops.
4. *Enhance additions of organic matter* by including legumes, biomass producing plants, and incorporating animals. Accumulation of both "active" and "slow fraction" organic matter is the key for activating soil biology, improving soil structure and macroporosity, and elevating the nutrient status of soils.
5. *Increase landscape diversity* by having in place a mosaic of agroecosystems' representative of various stages of succession. Risk of complete failure is spread among, as well as within, the various farming systems. Improved pest control is also linked to spatial heterogeneity at the landscape level.

6.6 NEED FOR FURTHER RESEARCH

Knowledge of the complex interactions among trees, crops, and their associated fauna is necessary to determine the viability of a particular agroforestry practice. The effects of agroforestry designs and technologies on pests and diseases can be divided into biological (species-related) and physical effects of components (e.g., microclimate). The former is highly specific for certain plant–pest or plant–disease combinations and have to be studied on a case-by-case basis. The latter is easier to generalize, but even they depend on the regional climatic conditions. On the basis of results from intercropping studies, agroforesters expect that AFS may provide opportunities to noticeably increase arthropod diversity and lower pest populations compared with the polyculture of annual crops or trees by themselves (Schroth et al., 2000). However, more work is needed in specific areas of research such as studies of the differences in arthropod populations between agroforestry and traditional agronomic systems, research into the specific mechanisms behind enhancement of pest management with agroforestry practices, and basic research into the life histories of target pests and potential natural enemies. An understanding of what aspects of trees modify pest populations—shelter, food, or host resources for natural enemies, temporal continuity, microclimate alteration, or apparency—should help in determining future agroforestry design practices (Rao et al., 2000).

Well-designed agroforestry techniques can reduce crop stress by providing the right amount of shade, reducing temperature extremes, sheltering off strong winds, and improving soil fertility, thereby improving the tolerance of crops against pest and disease damage, while influencing the developmental conditions for pest and disease organisms and their natural enemies. Poorly designed systems, on the other hand, may increase the susceptibility of crops to pests.

It is important to realize that the majority (75%) of agroforests are located in developing countries managed by traditional farmers who cultivate a few hectares of land. They rely on low-energy, labor-intensive production methods and few agrochemicals (Altieri, 1995). These resource-poor farmers have practiced agroforestry for centuries: they used trees for fences and pest control, as well as for food, fodder, construction materials, and fuel (Altieri and Farrell, 1984; Greathead, 1988). These small farmers cannot afford high-input technologies or expensive agroforestry designs. The key challenge is to maintain a highly diverse farm with woodlands, forests, and herbaceous edges, as in this way, allow traditional agroforesters to regulate pest populations by providing food and habitat for birds, spiders, parasites, and other natural enemies of pests.

Although small farmers may lack the research tools used by scientists in industrial countries, traditional agroforesters do have valuable knowledge to contribute toward the design of sustainable AFS. They have developed practical systems for identifying damaging stages of pests, understanding their biologies, and applying management techniques to suppress their populations. This knowledge can be tapped through participatory research schemes whereby farmers and researchers engage in a true collaborative partnership.

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7 A Case Study on the Potential of Contour Hedgerow Intercropping for Sustainable Crop Production on Sloping Highlands in Humid and Subhumid Zones of Sri Lanka

W.A.J.M. De Costa, A.G. Chandrapala, P. Surenthran, and L.G.N. Dharmasiri

CONTENTS

7.1 INTRODUCTION

Managed food production systems started with the early civilizations that saw humankind change its lifestyle from hunting and gathering of food to agriculture. This transformation was accompanied by a reduction in the number of plant species on which man depended for essential nutrition. Although the hunting and gathering stage was characterized by man's dependence on food from a wide range of trees, shrubs, and herbaceous plant species, a limited number of mainly herbaceous annual plant species emerged as staple foods in different parts of the world with the advent of managed crop husbandry. Intensification of crop production during the last century with inorganic fertilizers and agrochemicals resulted in large areas of natural vegetation, with a predominance of trees and shrubs, being replaced by monocultures of mainly annual crops. By the latter part of the twentieth century, several of these monocultures had developed problems of sustainability such as land degradation and declining yields. It was in this scenario that agroforestry, which is the deliberate mixing of trees with agricultural crops, emerged as a distinct scientific discipline (Bene et al., 1977) by the beginning of 1990s, as a set of practices (Nair, 1990, 1993; Sanchez, 1995) that ensures sustainable food production while conserving and replenishing the natural environmental resource base.

Mixtures of trees and annual crops have always been part of traditional cropping systems in most parts of the world, especially in the tropical climatic zone. Multilayered homegardens (Nair and Sreedharan, 1986; Jacob and Alles, 1987; Gillespie et al., 1993; Jensen, 1993; Kumar and

Nair, 2004), shifting cultivation (Nye and Greenland, 1960; Robinson and McKean, 1992; Palm et al., 1996), and traditional parkland savannah systems (Vandenbeldt, 1990) are notable examples of agroforestry systems or practices that have survived among rural communities of the tropics. However, during the recent reemergence of agroforestry to ensure sustainability of monocropping systems, the challenge has been to devise and introduce ecologically sustainable and socially acceptable ways of incorporating trees and shrubs into existing monocultures. Contour hedgerow intercropping (CHI) is one such agroforestry system that has been introduced during the last two decades to annual and perennial monocultures that are grown on sloping terrain.

7.2 CONTOUR HEDGEROW INTERCROPPING—ITS PERCEIVED ADVANTAGES AND POSSIBLE DISADVANTAGES

CHI involves incorporation of tree hedges along contours into existing monocropping systems on sloping lands (Figure 7.1), whose sustainability is severely threatened because of high rates of nutrient losses due to soil erosion, leaching, and nutrient export as crop yield. CHI aims to address these specific problems of sustainability of annual and perennial monocultures on sloping lands. Incorporation of tree hedges is expected to increase soil fertility on a sloping land because of many processes. The presence of hedgerows acts as a biological barrier to trap eroding soil within the field itself (Garrity, 1996; Craswell et al., 1997). Most of the tree species selected to be used as hedgerows are fast growing, leguminous species that can be pruned regularly. These prunings can be incorporated into the soil as mulch. When this mulch is decomposed gradually, it adds nutrients and organic matter to the soil (De Costa and Atapattu, 2001), thus improving both the chemical and

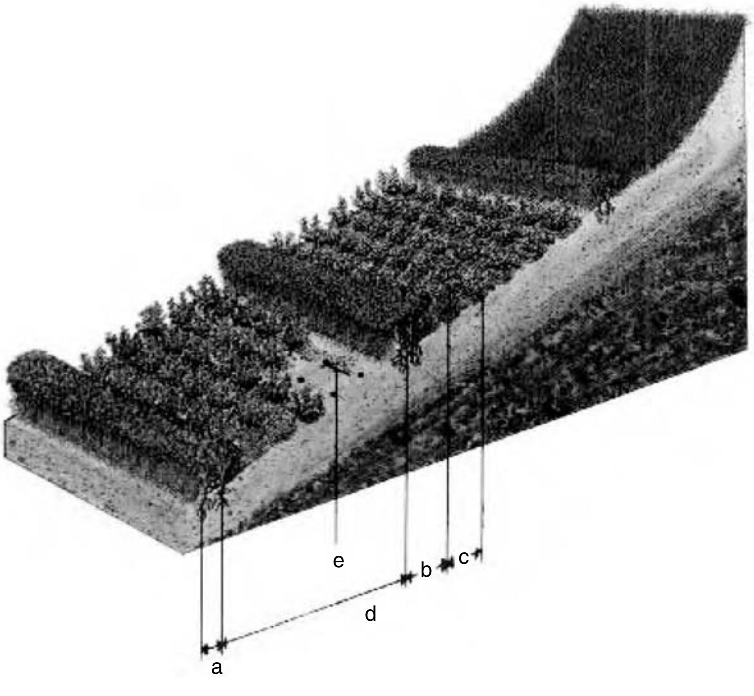

FIGURE 7.1 Schematic diagram of contour hedgerow intercropping with tea on sloping terrain. (a) Distance between double contour hedgerows (0.50 cm); (b) Distance between double hedgerows and the nearest tea row (0.61 m); (c) Interrow spacing for tea (1.22 m); (d) Interrow spacing for double hedgerows (6–8 m depending on the slope); (e) Intrarow spacing for tea (0.61 m).

the physical aspects of soil fertility. Thus, CHI is expected to help increase and sustain the yield production capacities of annual and perennial crops grown on sloping and undulating terrain.

Despite these benefits of contour hedgerows, incorporation of them into an existing agricultural crop could have potential negative effects on the crop as well (Sanchez, 1995). The principal negative effect is the competition by tree hedges with the crop for essential growth resources such as water, nutrients, and light (Cannell et al., 1996; Ong, 1996). If the tree hedges capture part of the resources that would otherwise be available to the crop, it could lead to a reduction of crop yields under CHI as compared with yields of crops growing without tree hedges. Therefore, the capacity of CHI to ensure sustainability of crop yields would be determined by the net balance between the positive and the negative effects of contour hedgerows on their associated agricultural crop. Hence, it is crucially important to identify the specific tree–crop interactions (TCIs) (Ong, 1996) present in CHIs and quantify their magnitudes to determine the ecological and economic sustainability, which would in turn determine its social acceptability and adoption by farmers. Such an in-depth analysis of TCIs in CHI is also important in view of the negative response of the farmers to alley cropping (Sanchez, 1995; Rao et al., 1998), which was one of the "flagship" systems that were promoted during the initial reemergence of agroforestry in the 1980s as a means of achieving sustainable crop production.

This chapter describes a series of long-term experiments in which the positive and the negative interactions of contour hedgerows on several annual crops (i.e., maize, mung bean, and cowpea) and a specific perennial crop (i.e., tea) have been investigated. The experiments covered CHIs on steep sloping highlands of the humid (annual rainfall >2000 mm) zone and on gently undulating uplands of the subhumid (annual rainfall 800–1000 mm) zone of Sri Lanka. Selection of an appropriate tree species for hedgerows is crucial to the success of CHI. Because of the inherent differences between different tree species (Huxley, 1996), their positive and negative interactions on the agricultural crop would also differ in magnitude. The experimental programs referred to herein have quantified the interspecies variation of TCIs of a range of potential hedgerow tree species and two grass species.

7.3 ABOUT THE STUDY SITE AND AREA

Sri Lanka is a tropical (7–10°N in latitude and 79–82°E in longitude) island in the Indian Ocean with a total land extent of 65,000 km^2. Historical evidence of human settlement and irrigated agriculture on the island dates back to more than 2000 years (Ray, 1959).

7.3.1 CLIMATE

A major part of Sri Lanka has a humid, tropical climate, with two rainy seasons per year brought about by the South-West (SW) monsoon (May to August) and North-East (NE) monsoon (November to January). In addition, there are two short "inter-monsoonal" rainy periods brought about by convectional rains in September and April. The island is divided into three major climatic zones (Figure 7.2) depending on the amount and within-year distribution of rainfall. The "wet" zone (~35% of the land area), located in the South-West, receives rainfall from both monsoons and the two inter-monsoons and consequently has a total annual rainfall exceeding 2000 mm. Therefore, crops can be grown almost throughout the year without supplementary irrigation. The "dry" zone (~55% of the land area), which spans the North, North-West, East, and South-East, has a major rainy season from the NE monsoon, but only a minor season with the SW monsoon. Hence, the total annual rainfall varies between 800 and 1400 mm (depending on the location), and its distribution is largely nonuniform in a bimodal pattern. A notable feature is the prolonged rain-free period between June and September. Therefore, only short-duration and rain-fed annual crops can be grown during this minor rainy season. The "intermediate" zone is a narrow strip (~10% of the land area) between the wet and dry zones, with a climate that is intermediate between the two adjacent zones.

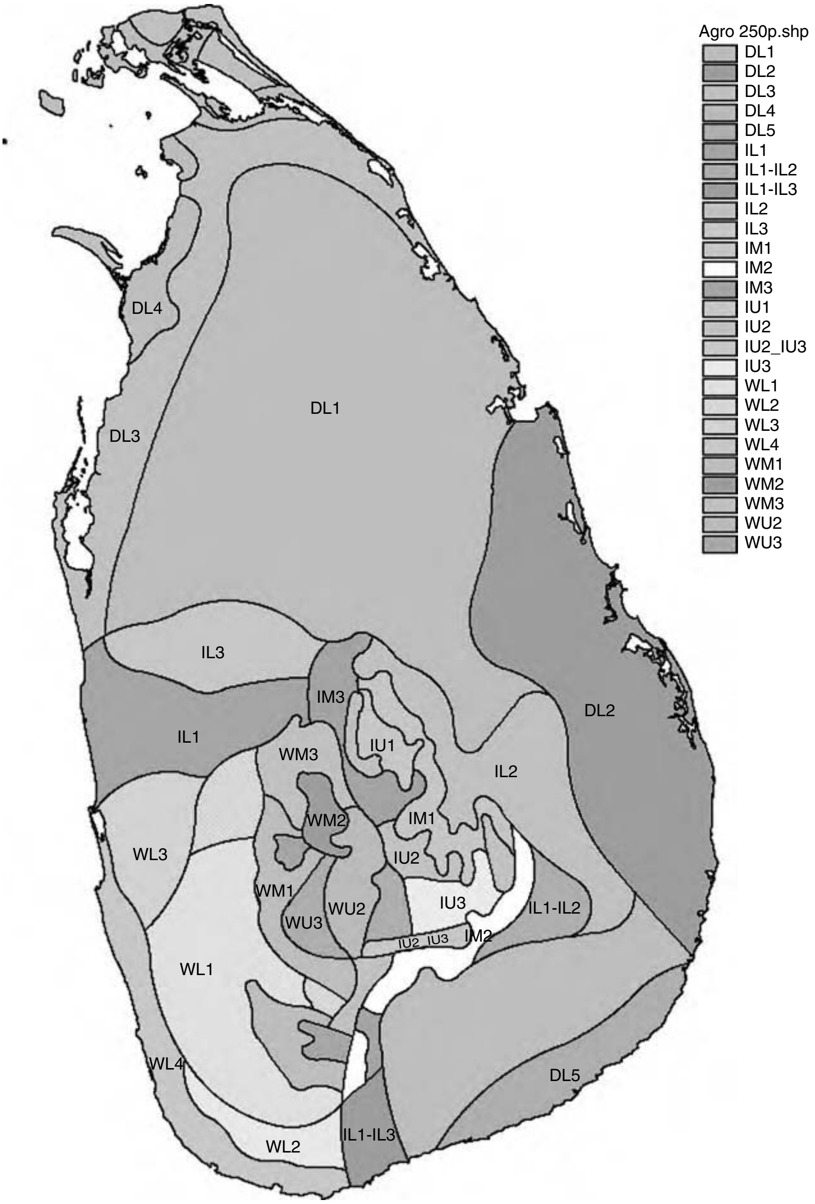

FIGURE 7.2 Agroecological regions and zones of Sri Lanka. Those beginning with W, I, and D are within wet, intermediate, and dry zones, respectively.

The boundaries between the three zones are not distinct with the climate and characteristic vegetation changing gradually from one to the other.

7.3.2 TOPOGRAPHY

In addition to the rainfall, topography plays a significant role in determining the climate of Sri Lanka, especially in its central region. While the whole of dry zone and the coastal belt of the wet zone is flat to gently undulating terrain, the central region (which includes parts of wet and intermediate zones) largely contains hilly terrain with slopes of varying degrees interspersed with valleys. The elevation increases up to 2200 m above sea level in the central hills, with the air temperature declining along with it from around 28°C to 34°C in the plains down to around 12°C–18°C in the hills. A major part of the central hills is located in the wet zone and receives annual rainfalls in excess of 2500 mm.

7.3.3 SOIL

Because of the variation in climate and temperature, a wide range of soil types can be observed in Sri Lanka, with seven out of the nine major tropical soil types being present (Spaargaren, 1994; Panabokke, 1996; Dassanayake and Hettiarachchi, 1999). However, the two major soil types are Ultisols (Great Group—Rhodudults), classified as Haplic Alisol by FAO/UNESCO and Typic Hapludult by USDA (Anonymous, 1975), which is predominant in the wet zone and the Alfisols (Great Group—Rhodustalfs; Order—Alfisols; Suborder—Ustalfs according to USDA), which dominates the dry zone. In the local classification, the former is known as "Red-Yellow Podzolic (RYP)" and the latter as "Reddish-Brown Earth (RBE)." Because of the high rainfalls, both soil types are highly leached, the RYP being more so, with the top (0–15 cm) soil cation exchange capacity (CEC) and pH levels ranging, respectively, from 5 to 10 milliequivalents/100 g soil (meq/100 g) and 4.0–5.5 in RYP and 15–20 meq/100g and 6.0–7.5 in RBE. The soil organic matter (SOM) contents are very low (<1%) in RBE because of the higher soil temperatures in the dry zone (which promotes faster litter decomposition) making the soil C:N ratio around 10. In contrast, depending on the vegetation present and the land use practices, SOM in RYP can be higher (up to 5%–6%) with greater C:N ratios. The soil texture in both major soil types is sandy, clay loam.

7.3.4 NATURAL VEGETATION, AGRICULTURE, AND AGROFORESTRY

Because of its well-distributed and high rainfall, the wet zone has rich natural vegetation, including moist, tropical rainforests and multilayered homegardens (Jacob and Alles, 1987). The plant species density and the tree density are high in both these vegetation systems. The island's three major perennial agricultural crops, that is, tea, rubber, and coconut, are also concentrated in the wet zone along with rice, which is the major annual crop grown as well as the staple diet. However, all these natural vegetation and agricultural systems are threatened by increasing population pressure and land degradation due to high soil erosion, which is especially high in the central hills.

Natural vegetation in the dry zone is characterized by dry, deciduous, and evergreen forests, shrub forests, and savanna-type vegetation. The dry-zone agriculture is predominantly rice (in the major rainy season) and a short-duration annual crop (in the minor rainy season) rotation. The dry zone has a well-connected network of tanks, built during ancient times and restored during the twentieth century, which collects a major portion of rain water of the NW monsoon. These are supplemented by a network of reservoirs and canals built in the 1980s to collect part of the rainfall in wet and intermediate zones and divert to the dry zone. Hence, in areas where irrigation water is available, a rice–rice cropping system is practiced. Rice is grown in low-land paddies under puddled anaerobic soil conditions. In addition to these, many dry-zone farmers have been practicing shifting cultivation. Despite a government ban, it is still practiced by farmers with very low incomes. A majority of dry-zone homesteads contains homegardens with woody perennials, multipurpose shrubs, annual crops, and medicinal plants. However, tree density is much lower than in the multilayered homegardens of the wet zone and vertical stratification of vegetation is much less pronounced. The Jaffna peninsula located at the Northern tip of the dry zone is characterized by a semiarid climate (600 mm yr^{-1} of rainfall), a calcareous soil, and the absence of any rivers. However, intensely managed annual cropping consisting of a variety of annual crops is practiced with lift irrigation of groundwater.

Natural vegetation of the intermediate zone is intermediate between the dense rainforests or homegardens of the wet zone and the dry, deciduous or savannah of the dry zone. More importantly, agriculture of the intermediate zone has a rich diversity, with the tea and coconut being major perennials in the relatively wetter and drier subzones, respectively. In addition, the intermediate zone has an optimum climate for many spice crops such as pepper (*Piper nigrum*), nutmeg (*Myristica fragrans*), clove (*Syzygium aromaticum*), and cardamom (*Elettaria cardamomum*). These perennials and spices are a core component in the multilayered homegardens of this zone. There are considerable extents of plantation forestry, primarily for timber, in the intermediate (*Swietenia macrophylla* and *Artocarpus heterophyllus*) and dry (*Tectona grandis*) zones.

In addition to the above, the natural vegetation and agriculture of the central hills are of critical importance to the topic discussed in this chapter. Tea cultivation on contours is the predominant land use in the sloping lands of the central hills. Because of the requirement of partial shade for obtaining maximum tea yields, the tea plantations contain several "shade tree" species (e.g., *Erythrina lithosperma*, *Gliricidia sepium*, *Albizzia molucana*, *Acacia melanoxylon*, and *Grevillea robusta*), which are planted at specified distances as square plantings among the tea rows. In areas where wind speeds are high, tea plantations also contain wind breaks and shelter belts. Some plantation forestry with *Pinus* and *Eucalyptus* is present in the hill crests and valleys. Apart from tea, cultivation of a variety of tropical and temperate vegetables (e.g., carrot, beet, leeks, cabbages, potatoes, beans, etc.) in intensely managed mixtures on man-made terraces is the other major cropping system practiced in the central hills.

A comprehensive description of the variety of agroforestry systems that are practiced in Sri Lanka is given by Ranasinghe and Newman (1993). Therefore, a detailed description is not attempted here. Most of the important ones have already been mentioned.

7.4 CONTOUR HEDGEROW INTERCROPPING SYSTEMS (CHIs) INTRODUCED IN SRI LANKA

This chapter presents results on work carried out on TCIs in different CHIs in the wet (humid) and intermediate (subhumid) zones of Sri Lanka. The humid zone systems include the CHI involving tea and six different hedgerow tree species (Case Study 1) and CHI involving maize and contour hedges of *Gliricidia sepium* (Case Study 2). The subhumid zone systems include the contour hedgerow systems involving mung bean (*Vigna radiata*), cowpea (*V. unguiculata*), and hedgerows of different multipurpose tree and shrub species (Case Study 3). A comparison of tree or shrub hedges with grass hedges carried out as part of Case Studies 2 and 3 is presented at the end.

7.5 CASE STUDY 1: TREE–CROP INTERACTIONS IN CHIs INVOLVING TEA AND DIFFERENT HEDGEROW SPECIES IN THE HUMID (WET) ZONE

7.5.1 BACKGROUND

Tea (*Camellia sinensis* (L.) O. Kuntz) is the major perennial crop grown in Sri Lanka. An unpruned tea plant can grow up to be a tree of medium (5–10 m) height. However, in commercial plantations, tea is pruned and maintained as a bush at a height of ~1 m with a flat canopy of leaves, which is called the "plucking table." The tender shoots, consisting of leaves and buds, that arise from the plucking table are harvested at specific intervals and are used for manufacturing tea, which is used as a beverage all over the world. Maintaining a high tea yield per unit land area is extremely vital for the economy of Sri Lanka because of the high earnings that are brought in by tea exports.

The cooler climate found in the central hills of Sri Lanka is ideal for production of high-quality tea. Therefore, a significant proportion of tea plantations are located on the sloping lands, with rows of tea bushes established along contours at an average distance of 1.22 m between rows. The foliage canopy of a well-managed tea crop covers the soil almost completely and therefore allows very little soil erosion even under high intensity rainfall. However, gaps in the canopy arising because of death or inadequate growth of tea bushes can open the soil to the direct impact of rainfall and subsequent erosion. This has occurred in many of the tea plantations in the central highlands. Gradual soil erosion and export of nutrients as plucked tea leaves have resulted in gradual decline of soil fertility in tea plantations, most of which are more than 100 years old. This gradual decline in soil nutrient pool has been coupled with declining SOM and consequent decrease of CEC. Hence, increasing amounts of inorganic fertilizer are needed to maintain higher tea yields. However, lower nutrient retention capacity (due to low SOM and CEC) of the soil and high-intensity rainfall have caused a high amount of applied inorganic fertilizers to be leached or carried away with runoff water before being absorbed by tea roots. All these processes have resulted in gradual, long-term decline of tea yields in many regions of the central highlands of Sri Lanka. Therefore, effective interventions were needed to arrest the decline of soil fertility and tea yields. During the early 1990s, incorporation of multipurpose tree hedges as contour hedgerows at specified distances (on average at 6–8 m) between tea rows was proposed as a means to achieve this.

7.5.2 System Description and Tree–Crop Interactions Involved

Incorporation of hedgerows in between tea rows creates a physical barrier to retain the eroding soil within the field. In addition, the hedgerow trees could be pruned at regular intervals to prevent excessive shading of the tea crop and thereby provide a supply of mulch or green manure material to be applied between tea rows. The decomposition of mulch material gradually releases its nutrients to be absorbed by tea roots. With the absorption of nutrients from deeper soil layers by hedgerow root systems, this creates a more closed nutrient cycle than a monoculture of tea and helps arrest the decline of soil fertility. However, against these beneficial effects of tree hedges on tea, their potential negative impacts also have to be considered. Resource competition between hedgerows and tea is a critical aspect in this regard. Tree roots could compete with tea for nutrients and water, whereas interception of light by the taller canopies of hedgerows could reduce the radiation energy available for photosynthesis and biomass production in tea. Therefore, whether incorporation of contour hedgerows increases or decreases tea yield depends on the balance between these positive and negative interactions between hedgerows and tea.

Another aspect relevant to the resource competition is the selection of specific tree species to be used as contour hedgerows. As tree species differ in their root and shoot growth, both the capacity to improve soil fertility through erosion control and mulch production and the competition exerted on tea would vary for different tree species. Therefore, there is a need to screen a wide range of potential tree species that could be used in CHI with tea. Results on the TCIs for a range of hedgerow species are also presented in the following sections.

7.5.3 Impact of Contour Hedgerows on Long-Term Tea Yields

Results of three long-term experiments carried out in CHIs involving tea growing on sloping terrain (30%–35% slope) in the humid (2500 mm yr^{-1} rainfall; 20.5°C annual average temperature), central highlands (945 m above sea level) of Sri Lanka showed that most of the hedgerow species (spaced at 8–9 m) exerted significant resource competition on tea and consequently reduced tea yields in comparison with a monoculture of tea (De Costa and Surenthran, 2005). Figure 7.3 shows the variation of tea yields over one complete pruning cycle (i.e., the period between two successive prunings) of 36 months (expressed in terms of harvested leaf dry weight) under CHIs involving six different hedgerow species (i.e., *Calliandra calothyrsus* Meissner, *Senna (Cassia) spectabilis* (DC.)

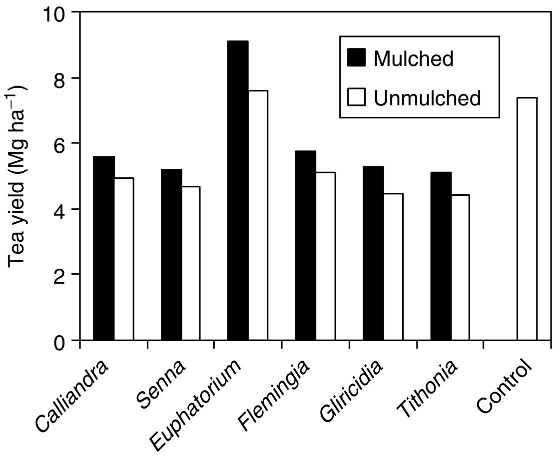

FIGURE 7.3 Mature tea yields (in terms of leaf dry weight and summed over a 36 month period) in CHIs with different tree species under mulched and unmulched conditions. (From De Costa, W.A.J.M. and P. Surenthran, *Agroforest. Syst.*, 63, 199, 2005. With permission.)

H. Irwin and Barneby, *Euphatorium innulifolium* (R.M. King and H. Rob.) H.B.K., *Flemingia congesta* Aiton F., *G. sepium* (Jacq.) (Kunth.) Walp., and *Tithonia diversifolia* (Hemsl.) A. Gray) and two mulching treatments, that is, mulched (hedgerow prunings at 4 month intervals added to the tea plots) and unmulched. Tea yields in all unmulched CHIs were lower by up to 40% relative to sole-cropped tea. Adding hedgerow prunings as mulch significantly increased tea yields in all CHIs, with yield responses to mulches ranging from 11% to 20%. Interestingly, the mulched CHI under *E. innulifolium* showed a 23% greater increase in tea yield above the monocrop yield, thus showing that in this CHI, the positive effects of contour hedgerows had exceeded the negative effects of resource competition. However, in CHIs involving the rest of the hedgerow species, yield reductions due to competition outweighed the positive effects.

The contention that it was the competition for resources that reduced tea yields in the majority of the CHIs was supported by several observations. First, yields of individual tea rows increased with increasing distance from hedgerows (Figure 7.4) indicating that competition was greater closer to the hedges. Second, the tea yield reductions were greater during periods when either soil moisture (Figure 7.5) or incident solar radiation (Figure 7.6) were limiting, when resource competition is expected to be greater. Interestingly, it could be noted that during periods when soil moisture or incident solar radiation was not limiting, tea yields of several CHIs were greater than the sole crop (SC) control.

These results were obtained on well-established (i.e., mature) tea crops that were in their third pruning cycle (i.e., 7–9 years after planting). To test the hypothesis that tea would be more susceptible to competition from hedgerows when it was at a younger stage, a parallel long-term experiment was carried out on recently established tea that was on its first pruning cycle. Continuous yield measurements over the first pruning cycle showed that yield reductions relative to the SC control were greater (Figure 7.7) than the respective reductions in CHIs with mature tea (Figure 7.3).

A third parallel experiment was carried out on mature tea to separate the aboveground and belowground components of resource competition from hedgerows. In this experiment, a 1 m deep trench was cut between hedgerows and the nearest tea row to prevent hedgerow roots from extending into the tea plots. Trenching increased tea yields in CHIs under all hedgerow species (Figure 7.8) thus confirming that belowground competition was reduced by trenching. However, it also showed that yield reductions were not completely eliminated by trenching. This was probably because hedgerow roots may have grown deeper than 1 m and hence would have extended into tea

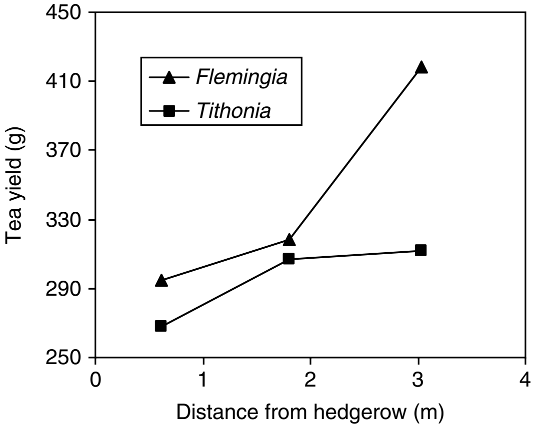

FIGURE 7.4 Variation of tea yield (g per 6 m of row) in two CHIs with increasing distance from hedgerows.

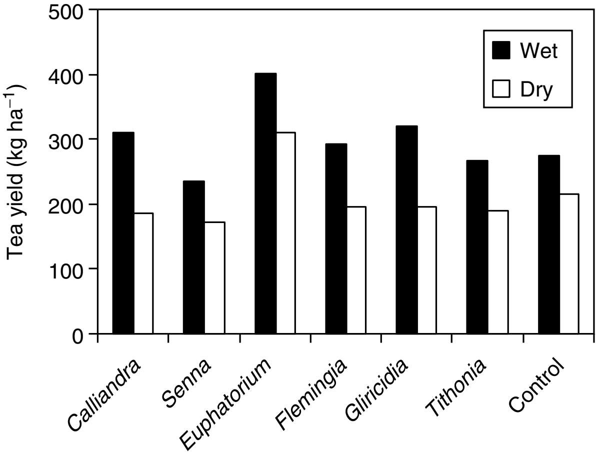

FIGURE 7.5 Monthly tea yields in CHIs during 2 months with adequate soil moisture (wet) and with a shortage of soil moisture (dry).

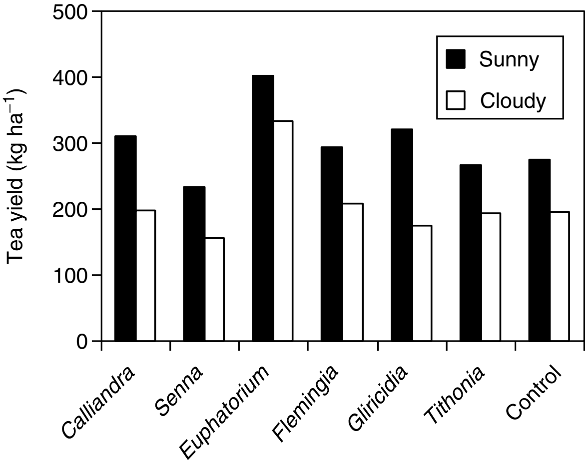

FIGURE 7.6 Monthly tea yields in CHIs during 2 months having sunny and cloudy conditions.

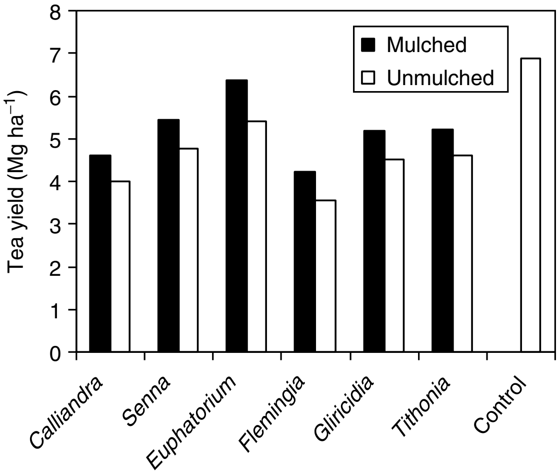

FIGURE 7.7 Young tea yields (in terms of leaf dry weight and summed over a 36 month period) in CHIs with different tree species under mulched and unmulched conditions.

plots despite the trench. In addition, the aboveground competition from hedgerows also contributed to the observed yield reductions. As argued by Huxley (1999), in many agroforestry experiments involving either alley cropping or CHI, aboveground competition could be more responsible for the observed crop yield variations because hedgerow roots may probably have invaded the whole experimental plot.

7.5.4 RELATIONSHIP BETWEEN HEDGEROW BIOMASS PRODUCTION AND TEA YIELD VARIATION IN CHIS

A point underscored by all results presented earlier is that there was significant variation between different hedgerow species in the degree of resource competition exerted on tea. This was only to be expected because of the inherent variation in the growth rates of different tree species. Simultaneous

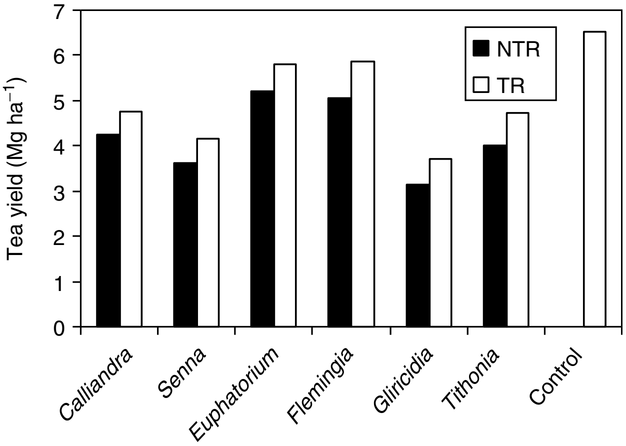

FIGURE 7.8 Response of mature tea yields in CHIs to removal of belowground competition by trenching. NTR, without trenching; TR, with trenching.

measurement of biomass of prunings during the 36 month period when tea yields were measured showed a significant negative linear relationship between tea yield of CHIs and the respective pruned biomass of hedgerows (Figure 7.9). This indicated that those hedgerow species that have higher growth rates and consequently greater biomass of prunings exerted greater competition on tea and reduced its yield to a greater extent and vice versa. However, slow-growing species that produce a lower biomass of prunings also add fewer nutrients to the soil and therefore have a lower impact on soil fertility improvement. Therefore, an ideal hedgerow species would be one that would have an intermediate rate of growth so that it is able to add significant amounts of nutrients to the soil while exerting as less competition as possible on tea. In this experimental program, *E. innulifolium*, which had an intermediate level of biomass production, came close to being such a species in this particular agroforestry system.

In this regard, it is important to note the central agroforestry hypothesis as stated by Cannell et al. (1996) that "for incorporation of trees to be beneficial to the crop, the trees should be able to capture resources that are not available to the crop." Therefore, an ideal hedgerow species in this specific agroforestry system should either be able to capture nutrients and water in deeper layers of the soil profile or solar radiation levels at higher intensities that cannot be used by the tea crop. It is important to note that tea is usually established by rooted stem cuttings and therefore does not have a tap root. Measurements in the present and other experiments have shown that a substantial proportion of the tea root system is confined to the upper 30–45 cm of the soil profile. Therefore, a hedgerow root system that has an adequate amount of active root length at depths below 30–45 cm should be able to absorb nutrients (and water), transfer it to biomass during biosynthesis, and recycle it back to the topsoil through prunings without exerting significant competition on tea. Likewise, it is relevant to note that because tea had originated as an understory plant in the tropical rainforests of Burma, a certain degree of shade is essential for a monoculture of tea to give its maximum yield. Mohotti et al. (2000) have shown that the photosynthetic apparatus of tea leaves experiences photoinhibition damage at higher light intensities. Therefore, hedgerows species that can capture and utilize higher light intensities should be able to synthesize higher levels of biomass without impacting negatively on tea yields.

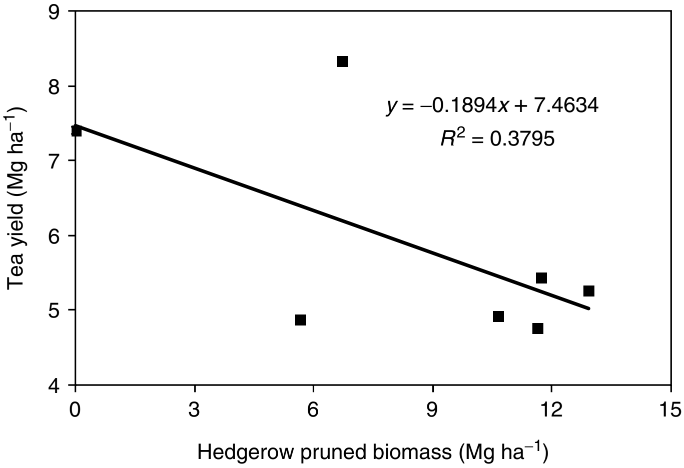

FIGURE 7.9 Relationship between tea yield under different hedgerow species and hedgerow pruned biomass over a 36 month period. The data point with zero hedgerow biomass represents the yield of sole-cropped tea. Yields of CHI plots are the means of the respective plots with and without mulching of hedgerow prunings.

7.5.5 VARIATION OF ABOVEGROUND AND BELOWGROUND HEDGEROW CHARACTERS AND THEIR ROLE IN TEA YIELD DETERMINATION IN CHIs

Different hedgerow species showed significant variation in terms of their aboveground characters such as canopy lateral spread, height, and "potential canopy volume" above the pruning height (calculated for a 10 m length of each hedgerow as the product between canopy height above 0.45 m and twice the lateral spread to take into account both sides of the hedgerow) (Table 7.1) as well as belowground characters such as vertical distribution of root length density (Figure 7.10).

The interplay between these different hedgerow characters determined the overall resource competition exerted by a given hedgerow species. The lower lateral spread, height, and volume of hedgerow canopies combined with a lower root length density in the topsoil layer could have been responsible for the lower competition exerted by *Euphatorium* (Table 7.1; Figure 7.10). The greater mulching effect of *Euphatorium* (Figure 7.3) could have also been responsible for the higher tea yields with *Euphatorium*. In contrast, *Tithonia* hedgerows had greater lateral canopy spread, height, and volume along with greater root length densities at all soil depths. All these hedgerow characteristics would have acted to intensify resource competition that was responsible for tea with *Tithonia* showing the lowest yields among the hedgerow intercrops (Figure 7.3). Although *Calliandra* and *Senna* had lower root length densities than *Euphatorium*, both those species had greater lateral spread, height, and volume in their canopies, thus reducing tea yields due to competition for light. On the other hand, despite having slightly smaller canopies than *Euphatorium*, hedgerows of *Flemingia* and *Gliricidia* had greater root length densities in the topsoil layer where competition for water and nutrients are likely to be greatest. These specific characteristics could have been responsible for the lower tea yields shown in these four CHIs relative to sole-cropped tea.

7.5.6 DECOMPOSITION CHARACTERISTICS OF HEDGEROW PRUNINGS

The rate of decomposition of hedgerow prunings that are added to the soil as mulch is central to the release of nutrients to be absorbed by tea roots and therefore to determine the response of tea to mulching. Hence, decomposition and nutrient release characteristics of different hedgerow species

TABLE 7.1
Lateral Canopy Spread, Canopy Height and Potential Canopy Volume of Different Contour Hedgerow Species at 4 Months after Pruning

Hedgerow Species	Lateral Canopy Spread from Mid-Point of Hedgerow (m)	Canopy Height (m)	Potential Canopy Volume above Pruning Height of a 10 m Hedgerow (m^3)
Calliandra calothyrsus	1.33 b	1.54 b	32.08 b
Senna spectabilis	1.12 c	1.59 b	26.26 c
Euphatorium innulifolium	0.97 c	1.36 c	18.02 d
Flemingia congesta	0.73 d	1.48 b	15.94 e
Gliricidia sepium	0.96 c	1.10 d	14.22 e
Tithonia diversifolia	1.51 a	1.75 a	43.92 a
CV (%)	10.15	7.13	6.25

Source: From De Costa, W.A.J.M. and P. Surenthran, *Agroforest. Syst.*, 63, 199, 2005. With permission.

Note: Potential canopy volume above pruning height for a 10 m length of each hedgerow was calculated as the product between height above 0.45 m and twice the lateral spread. Each value is a mean of measurements over five pruning cycles in three replicate plots with each containing two hedgerows. Means vertically connected with the same letter are not significantly different at $p = 0.05$.

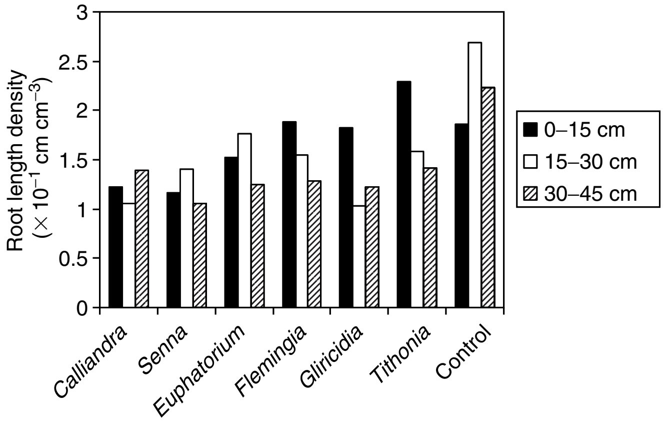

FIGURE 7.10 Root length density at different soil depths of different hedgerow tree species used in contour hedgerow intercrops with tea. (From De Costa, W.A.J.M. and P. Surenthran, *Agroforest. Syst.*, 63, 199, 2005. With permission.)

were studied using the litter bag technique (De Costa and Atapattu, 2001). Decomposition of prunings was characterized by the single exponential decay function (Wieder and Lang, 1982):

$$\frac{W_t}{W_0} = \mathrm{e}^{-kt}, \tag{7.1}$$

where W_0 and W_t are litter dry weights at the beginning and after time t (in weeks), respectively. The decomposition constant (k) was estimated as the slope of the linear regression between $\log_e(W_t/W_0)$ and t. The turnover time was calculated as

$$T_f = \frac{[\log_e(1 - W_t/W_0)]}{k}, \tag{7.2}$$

where T_f is the time (in weeks) required to decompose f fraction of the initial litter dry weight. Therefore, half-life ($T_{0.50}$) was given as $0.693/k$.

Table 7.2 shows the decomposition constants and $T_{0.50}$ values for leaf and stem prunings of different hedgerow species. For leaf prunings, significantly faster decomposition rates were shown by *Senna*, *Gliricidia*, and *Tithonia*. *Flemingia* and *Calliandra* leaves showed much slower decomposition rates, whereas *Euphatorium* showed an intermediate rate. For stem prunings, the fastest decomposition was shown by *Euphatorium*. Stem decomposition was slower than leaves in all species. Slower release of nutrients may probably be an advantage in the central highlands of Sri Lanka where potential leaching losses are high because of the high rainfall (2500 mm yr^{-1}). Therefore, for this agroclimatic zone, species having lower to intermediate decomposition rates such as *Flemingia*, *Calliandra*, and *Euphatorium* could be more suitable than *Gliricidia*, *Senna*, and *Tithonia*, which have faster decomposition rates.

7.5.7 AMOUNTS OF NUTRIENTS ADDED THROUGH HEDGEROW PRUNINGS

Although the rate of decomposition is an important characteristic in determining suitable tree species for CHI, the amounts of nutrients added to the soil through prunings is an even more important character in determining the overall success of a specific CHI. The tree species differed significantly in the biomass of prunings produced (Table 7.3) and in their nutrient contents (Table 7.4). Accordingly, the different hedgerow species added different amounts of major nutrients to the

TABLE 7.2

Decomposition Constant (*k*) and Half-Life ($T_{0.50}$) for Dry Weight Loss of Prunings from Different Tree Species Growing as Contour Hedgerows in Tea Plantations in the Central Highlands of Sri Lanka

Species	Part	k (wk^{-1})	Std. Error	Adj. R^2	$T_{0.50}$ (wk)
Calliandra	Leaf	0.0509	0.0078	0.91	13.61
Senna	Leaf	0.1650	0.0101	0.99	4.20
Euphatorium	Leaf	0.1062	0.0152	0.92	6.53
Flemingia	Leaf	0.0335	0.0069	0.85	20.69
Gliricidia	Leaf	0.1618	0.0260	0.90	4.28
Tithonia	Leaf	0.1419	0.0381	0.76	4.88
Calliandra	Stem	0.0388	0.0089	0.78	17.86
Senna	Stem	0.0413	0.0043	0.95	16.78
Euphatorium	Stem	0.0633	0.0060	0.96	10.95
Flemingia	Stem	0.0225	0.0046	0.82	30.81
Gliricidia	Stem	0.0587	0.0079	0.92	11.81
Tithonia	Stem	0.0549	0.0080	0.90	12.63

Source: From De Costa, W.A.J.M. and A.M.L.K. Atapattu, *Agroforest. Syst.*, 51, 201, 2001. With permission.

Note: k was estimated as the slope of the linear regression between $\log_e (W_t/W_0)$ and t in Equation 7.1. $T_{0.50}$ was estimated as $0.693/k$ as given in Equation 7.2.

soil (Table 7.3). *Senna*, *Tithonia*, and *Calliandra* provided significantly greater N than the rest. *Calliandra* provided the highest P, while *Tithonia* and *Calliandra* provided significantly greater K and Ca than the rest.

A comparison of the major plant nutrients added to the soil through hedgerow prunings with the recommended rates of inorganic fertilizer for tea showed that several hedgerow species supplied significant proportions of the recommended amounts of major nutrients. For example, *Senna*, *Tithonia*, and *Calliandra* provided 65%–74% of the total annual N requirement (i.e., 220 kg N ha^{-1} yr^{-1}) and also provided more than the total annual K requirement (i.e., 110 kg K ha^{-1} yr^{-1}).

TABLE 7.3

Total Pruned Biomass and Total Amounts of Nitrogen, Phosphorus, Potassium, and Calcium Added to the Soil through Prunings of Different Hedgerow Tree Species Used in Contour Hedgerow Intercrops with Tea in the Sloping Highlands of Sri Lanka

Species	Total Pruned Biomass (kg ha^{-1} yr^{-1})	Nitrogen (kg ha^{-1} yr^{-1})	Phosphorus (kg ha^{-1} yr^{-1})	Potassium (kg ha^{-1} yr^{-1})	Calcium (kg ha^{-1} yr^{-1})
Calliandra	4.31 a	144 b	19.4 a	180 b	32 b
Senna	3.54 a	162 a	9.9 c	119 c	26 c
Euphatorium	2.24 b	69 d	6.2 d	96 d	18 d
Flemingia	3.90 a	124 c	8.9 c	104 d	18 d
Gliricidia	1.89 b	64 d	5.9 d	89 d	20 c
Tithonia	3.88 a	158 a	11.1 b	216 a	54 a
CV (%)	6.99	7.49	8.04	7.38	8.80

Source: From De Costa, W.A.J.M., P. Surenthran, and K.B. Attanayake, *Agroforest. Syst.*, 63, 211, 2005. With permission.

Note: Means connected vertically by the same letter are not significantly different at $p = 0.05$.

TABLE 7.4
Nutrient Concentrations of Leaves and Stems of Prunings of Different Hedgerow Tree Species

	Nitrogen (mg g^{-1})		Phosphorus (ppm)		Potassium (ppm)		Calcium (ppm)	
Species	**Leaves**	**Stems**	**Leaves**	**Stems**	**Leaves**	**Stems**	**Leaves**	**Stems**
Calliandra	37.75 b	28.27 b	1807 d	7500 a	12944 d	74197 b	3244 e	12331 b
Senna	47.36 a	43.72 a	2182 c	3600 c	18383 c	53278 c	7080 d	8005 c
Euphatorium	48.24 a	18.13 c	2499 b	2967 c	23323 b	56544 c	10649 b	6098 d
Flemingia	32.69 b	30.04 b	2723 a	1840 e	13393 d	45942 d	2753 f	7094 cd
Gliricidia	36.20 b	31.20 b	1830 d	5182 b	19381 c	91739 a	7728 c	14809 a
Tithonia	45.43 a	37.97 a	2402 b	3258 d	25669 a	78486 b	16324 a	12269 b
CV (%)	9.62	10.63	8.40	9.53	9.08	6.76	10.45	6.82

Note: Means connected vertically by the same lowercase letter are not significantly different at $p = 0.05$.

Therefore, the addition of inorganic fertilizer, especially in N and P, can be reduced with the incorporation of hedgerows in a tea plantation. This could cause a significant increase in profit margins as inorganic fertilizer constitutes a major fraction of the cost of production of tea (Sivapalan et al., 1986). It was also significant to note that none of the hedgerow species tested provided an adequate proportion of the recommended rate of P (i.e., 37 kg P ha^{-1} yr^{-1}), with only *Calliandra* providing 52% of the requirement.

A correlation analysis showed that total amounts of all these nutrients added to the soil had strong positive correlations (r^2 values of 0.89, 0.74, 0.72, and 0.51 for N, P, K, and Ca, respectively) with the total biomass of prunings. However, there were no consistent, significant correlations with the respective nutrient concentrations in leaf or stem prunings. Therefore, total amounts of nutrients added to the soil from prunings of respective hedgerow species were determined more by the hedgerow biomass production (Table 7.3) levels than by nutrient concentrations of prunings (Table 7.4). For example, although *Euphatorium* leaves had the highest leaf nitrogen concentration, the total amount of N added to the soil from *Euphatorium* was low because of the lower biomass of its prunings. In contrast, *Senna* leaves had a slightly lower leaf N concentration than that of *Euphatorium*, but added the highest amount of N to the soil because of its greater biomass of prunings.

7.5.8 CHANGES IN SOIL FERTILITY WITH INCORPORATION OF CONTOUR HEDGEROWS

Soil fertility is a very broad term that encompasses not only the plant nutrients that are held in the soil but also the soil physical properties that facilitate or impede root growth and determine the nutrient and water-holding capacities of the soil. The impact of incorporating contour hedgerows on chemical soil fertility in the short term is determined by the balance between the amount of nutrients added by hedgerow prunings and that absorbed competitively by hedgerows for their growth. In addition, the soil nutrient content in a tea plantation at a given point of time would be determined by the amount of nutrients extracted by tea and that exported from the cropping system as tea leaf yield. These aspects were also measured in the present study at 4 month intervals during the second and third years of the pruning cycle.

7.5.8.1 Changes in Soil Nutrient Contents

Topsoil nitrogen content showed significant ($p < 0.05$) variation between different hedgerow intercrops and mulching treatments (Table 7.5). When prunings were added as a mulch, topsoil N levels of all CHIs except that with *Calliandra* were greater than the control. In contrast, under unmulched conditions, all CHIs except that with *Tithonia* had lower topsoil N than the control. Mulching increased the topsoil N in all CHIs.

TABLE 7.5

Total Nitrogen and Available Phosphorus of Topsoil of Contour Hedgerow Intercrops Involving Tea and Different Tree Species under Mulched and Unmulched Conditions

	Total Nitrogen (mg/100 g Soil)		Available Phosphorus (ppm)	
Species	Mulched	Unmulched	Mulched	Unmulched
Calliandra	298 c	266 ab	43.28 bc	31.36 bc*
Senna	315 bc	261 ab*	26.94 d*	16.07 e*
Euphatorium	339 ab*	260 ab*	45.40 ab	37.54 ab
Flemingia	323 bc*	235 b*	32.04 cd*	25.64 cd*
Gliricidia	307 c	261 ab*	30.52 d*	21.70 de*
Tithonia	356 a*	318 a	55.54 a*	43.37 a
Mean	323 A	267 B	38.95 A	29.28 B
Control	306		43.84	
CV (%)	10.09	12.42	8.56	8.51

Source: From De Costa, W.A.J.M., P. Surenthran and K.B. Attanayake, *Agroforest. Syst.*, 63, 211, 2005. With permission.

Note: Means connected vertically by the same lowercase letter are not significantly different at $p = 0.05$. Means connected horizontally by the same uppercase letter are not significantly different at $p = 0.05$. Significant ($p = 0.05$) differences from the control treatment are shown by*.

Similar to N, the topsoil available P also showed significant variation between different hedgerow tree species and mulching (Table 7.5). However, except in the mulched CHI with *Tithonia*, the topsoil P of the rest under both mulched and unmulched conditions was lower than that of the SC control. However, the topsoil P of all CHIs responded positively to mulching. The topsoil concentrations of specific nutrient cations such as exchangeable potassium, magnesium, and calcium (Table 7.6) showed broadly similar patterns of variation. All three cations showed

TABLE 7.6

Exchangeable Potassium, Magnesium, and Calcium of Topsoil of Contour Hedgerow Intercrops Involving Tea and Different Tree Species under Mulched and Unmulched Conditions

	Exchangeable Potassium (ppm)		Exchangeable Magnesium (ppm)		Exchangeable Calcium (ppm)	
Species	Mulched	Unmulched	Mulched	Unmulched	Mulched	Unmulched
Calliandra	0.280 c*	0.140 b*	10.30 c	5.15 b*	1.79 bc*	1.22 c
Senna	0.183 d*	0.123 c*	20.90 b*	7.32 b*	1.48 bc*	0.68 d*
Euphatorium	0.378 a*	0.193 a*	42.22 a	16.41 a	3.83 a*	3.00 a*
Flemingia	0.137 e*	0.107 d*	10.60 c*	5.78 b*	1.26 c*	0.68 d*
Gliricidia	0.193 d*	0.095 d*	11.51 c*	7.19 b*	1.14	0.68 d*
Tithonia	0.323 b*	0.148 b*	9.91 c*	6.65 b	2.68 ab*	1.82 b*
Mean	0.249 A	0.134 B	17.57 A	29.28 B	2.03 A	1.35 B
Control	0.230		11.85		1.08	
CV (%)	10.23	11.22	8.92	10.25	10.54	11.22

Source: From De Costa, W.A.J.M., P. Surenthran and K.B. Attanayake, *Agroforest. Syst.*, 63, 211, 2005. With permission.

Note: Means connected vertically by the same lowercase letter are not significantly different at $p = 0.05$. Means connected horizontally by the same uppercase letter are not significantly different at $p = 0.05$. Significant ($p = 0.05$) differences from the control treatment are shown by*.

significant variation between different hedgerow species and mulching treatments. In the majority of unmulched CHIs, the exchangeable K, Mg, and Ca concentrations were lower than those in the SC control. In all the mulched CHIs, the soil-exchangeable Ca ion concentration was greater than that of the control. However, in the cases of K and Mg, some of the mulched CHIs showed increases, whereas the rest showed decreases relative to the control (Table 7.6). Mulching substantially increased the topsoil concentrations of all three exchangeable cations.

The changes describes earlier in the soil nutrient contents indicate that incorporation of contour hedgerows could increase the depletion of major nutrient pools in the soil, and especially those of nitrogen and phosphorus. This was probably due to competitive absorption of nutrient by hedgerows for their growth and biomass production. However, the positive response to mulching by hedgerow prunings in all nutrients tested in this study showed that the hedgerows were able to recycle part of the nutrients that they absorb from the soil. However, during the 8 year period since the establishment of hedgerows in this particular tea plantation, the positive effect of nutrient recycling has been outweighed by the negative effect of competitively extracting soil nutrients by hedgerows.

7.5.8.2 Changes in Nutrient Contents of Harvested Tea Leaves

Examination of leaf nutrient contents of harvested leaves of tea growing with different hedgerow species (Table 7.7) provides valuable insights into the impact of possible nutrient competition from hedgerows on the tea crop. Mulched tea with all hedgerow species except *Tithonia* had greater leaf N than sole-cropped tea. In contrast, leaf N of unmulched tea in CHIs showed both increases and decreases relative to the control. All CHIs showed increases of tea leaf N contents in response to mulching.

Similar to leaf N, mulched tea under a majority of hedgerow species had greater leaf P contents than the SC control (Table 7.7). In contrast, in all unmulched CHIs except in tea under *Gliricidia*, leaf P contents were lower than those of the control. However, all CHIs showed increases in tea leaf P due to mulching. Mulched tea in CHIs showed both increases and decreases in leaf K relative to the control (Table 7.7). In contrast, all unmulched CHIs had lower tea leaf K than the control. However, mulching increased the tea leaf K content in all CHIs. In contrast to the other nutrients

TABLE 7.7

Nitrogen, Phosphorus, Potassium, and Calcium Concentrations of Harvested Leaves of Tea in Contour Hedgerow Intercrops with Different Tree Species under Mulched (M) and Unmulched (UM) Conditions

	Nitrogen (mg g^{-1})		Phosphorus ($\times 10^3$ ppm)		Potassium ($\times 10^3$ ppm)		Calcium ($\times 10^3$ ppm)	
Species	**M**	**UM**	**M**	**UM**	**M**	**UM**	**M**	**UM**
Calliandra	30 d	27b	3.1 b*	2.8 b	32 b*	30 a*	3.0 d*	2.6 c*
Senna	42 a*	29 c	2.6 f*	2.3 e*	30 b*	28 b*	3.0 d*	2.4 d*
Euphatorium	35 c*	34 a*	2.7 e*	2.5 d*	32 b*	31 a	3.5 c*	3.2 b*
Flemingia	34 c*	30 d	3.0 c*	2.5 d*	20 e*	16 c*	3.5 c*	3.1 b*
Gliricidia	38 b*	31 d*	3.3 a*	2.9 a*	34 a*	31 a	3.7 b*	3.5 a*
Tithonia	28 d	26 b*	2.9 d*	2.7 c*	27 d*	26 b*	4.0 a*	3.5 a*
Mean	34 A	29 B	3.0 A	2.6 B	29 A	27 B	3.4 A	3.1 B
Control	29		2.8		31		2.4	
CV (%)	10.27	7.04	7.50	8.85	10.89	11.55	9.94	6.82

Note: Means connected vertically by the same lowercase letter are not significantly different at $p = 0.05$. Means connected horizontally by the same uppercase letter are not significantly different at $p = 0.05$. Significant ($p = 0.05$) differences from the control treatment are shown by*.

measured, tea leaf Ca content showed substantial increases above the SC control in CHIs under both mulched and unmulched conditions (Table 7.7). Similar to all other nutrients measured, mulching increased the tea leaf Ca in all hedgerow intercrops.

Observations in Table 7.7 showed that the majority of mulched hedgerow intercrops had significantly higher leaf nutrient contents than the sole tea crops that were growing without hedgerows. On the other hand, the majority of unmulched tea crops had lower nutrient contents (with the exception of Ca) in harvested leaves. This indicated that when recycled nutrients through addition of hedgerow prunings are absent, competition from hedgerows for nutrients has a significant impact on the tea crop and decreases its leaf nutrient content. While competition would be present in mulched CHIs as well, it does not have a significant impact on tea leaf nutrient contents because recycled nutrients are available through decomposition of added prunings. These results agree with findings from CHIs growing elsewhere under comparable conditions.

In agreement with the observations of the present study, Szott et al. (1991) also observed a lower soil nutrient status in CHIs growing on acid-infertile soils of the humid tropics. In addition, there are several reports of significant competition for nutrients between tree hedgerows and crops in sites receiving as high rainfall as the present site (Fernandes et al., 1993; Rao et al., 1998). Significant competition for nitrogen has been demonstrated using ^{15}N in hedgerow intercrops involving maize with black walnut and red oak (Jose et al., 2000) and between sorghum and *Acacia saligna* (Lehmann et al., 2002).

7.5.8.3 Changes in Soil Physical Properties

The variations described earlier in soil and plant nutrient contents indicated that in the short run (i.e., during the first 6–8 years) incorporating hedgerows into a tea crop could decrease its nutrient pool. However, measurements of soil physical properties and pH in the different CHIs provide cause for optimism in the long run. The topsoil (0–15 cm) bulk density (BD), organic matter content (SOM), and CEC showed significant variation between CHIs having different hedgerow species and between mulching treatments (Tables 7.8 and 7.9).

TABLE 7.8
Bulk Density and Organic Matter Content of Topsoil of Contour Hedgerow Intercrops Involving Tea and Different Tree Species under Mulched and Unmulched Conditions

	Bulk Density (g cm^{-3})		Organic Matter (g/100 g Soil)	
Species	Mulched	Unmulched	Mulched	Unmulched
Calliandra	0.996 ab*	1.135 a	6.38 b*	5.43 b
Senna	1.020 ab*	1.089 a*	6.37 b*	5.15 c*
Euphatorium	0.987 ab*	1.082 a*	7.05 a*	5.46 b
Flemingia	0.934 b*	1.079 a*	6.54 b*	5.01 c*
Gliricidia	1.116 a	1.165 a*	3.99 c*	3.75 d*
Tithonia	1.018 ab*	1.115 a	7.07 a*	5.70 a*
Mean	1.012 A	1.111 B	6.23 A	5.08 B
Control	1.117		5.32	
CV (%)	6.75	9.48	8.83	7.09

Note: Means connected vertically by the same lowercase letter are not significantly different at $p = 0.05$. Means connected horizontally by the same uppercase letter are not significantly different at $p = 0.05$. Significant ($p = 0.05$) differences from the control treatment are shown by*.

TABLE 7.9

Cation Exchange Capacity (CEC) and pH of Topsoil of Contour Hedgerow Intercrops Involving Tea and Different Tree Species under Mulched and Unmulched Conditions

	CEC (meq/100 g Soil)		pH	
Species	**Mulched**	**Unmulched**	**Mulched**	**Unmulched**
Calliandra	27.24 bc*	25.27 bc*	4.70 b	4.70 b
Senna	22.97 c*	22.99 c*	5.04 b	4.85 b
Euphatorium	47.54 a*	36.36 ab*	6.04 a*	6.03 a*
Flemingia	32.38 b*	29.91 b*	4.95 b	5.02 b
Gliricidia	47.42 a*	39.80 a*	5.03 b	5.25 b
Tithonia	26.34 bc*	26.59 bc*	5.07 b	4.86 b
Mean	33.98 A	30.15 B	5.14 A	5.08 B
Control	18.51		5.07	
CV (%)	12.66	12.95	13.16	11.82

Note: Means connected vertically by the same lowercase letter are not significantly different at $p = 0.05$. Means connected horizontally by the same uppercase letter are not significantly different at $p = 0.05$. Significant ($p = 0.05$) differences from the control treatment are shown by*.

All CHIs that received their prunings as mulch had lower BDs than the sole tea crops (Table 7.8). Mulching decreased BD in all CHIs. Even when hedgerow prunings were not added as mulch, the topsoil BD of the majority of CHIs was lower than that of the SC control. SOM showed significant increases above the SC control in the majority of CHIs that received mulching (Table 7.8). In contrast to the mulched CHIs, SOM of unmulched CHIs was either lower or only slightly higher than the control. Mulching significantly increased SOM in CHIs under all tree species. All CHIs had significantly greater CEC than the SC control under both mulched and unmulched conditions (Table 7.9). In the majority of CHIs, mulching increased CEC.

These results clearly showed that incorporation of tree hedges improved some of the key soil physical properties related to long-term soil fertility (Eswaran et al., 1993; Weischet and Caviedes, 1993). These included reduced topsoil BD, increased SOM, and CEC. The observed reduction of BD in the CHIs as compared with the SC control agreed with observations of Yamoah et al. (1986), Agus et al. (1997), and Samsuzzaman et al. (1999). This could be achieved by the spreading of hedgerow roots that loosen the soil and bind the soil particles to form larger aggregates (Mapa and Gunasena, 1995). In agreement with the observations of the present work, Hulugalle and Kang (1990) also observed that mulching decreased the topsoil BD. This could be due to increased activity of soil microorganisms that are involved in decomposition of added mulches (Oades, 1984; Woomer et al., 1994). The increased SOM, microbial by-products, and secretions from roots and microbes would all help to bind soil particles to form larger aggregates and thereby loosen the soil and decrease its BD. Increased topsoil CEC in the hedgerow intercrops of the present study is supported by the observations of Hulugalle and Ndi (1994).

The greater SOM in the mulched CHIs was obviously the result of addition of hedgerow prunings. Decomposition of these prunings would increase SOM. Moreover, turnover of fine roots of hedgerows would also increase SOM in both mulched and unmulched hedgerow intercrops (Sanchez, 1995). Variation of SOMs in the different hedgerow intercrops was probably due to the different levels of prunings added from different tree species and due to variation in the size of their root systems and their rates of turnover. Accordingly, the mulched CHIs with *T. diversifolia* that had significantly higher levels of biomass production (Table 7.3) and root length density (Figure 7.10) showed the highest increase in SOM (Table 7.8). Conversely, the hedgerow intercrop with *G. sepium*, which was the only mulched HI to show a reduction in SOM relative to the control, had the lowest level of hedgerow biomass production and moderate root length density. Similar to the observations of the present work, Kang et al. (1999) also observed increases in SOM in hedgerow intercrops as compared with SCs.

The observed increases in some key soil physical properties which determine long-term soil fertility (Eswaran et al., 1993; Weischet and Caviedes, 1993) and the possibility that these improvements may continue with time, indicate that incorporation of contour hedgerows has the capability to regenerate soil fertility in tea plantations on sloping highlands in Sri Lanka, especially when tree prunings are added as mulch. Further evidence for long-term sustainability and possible regeneration of soil fertility is shown when the amounts of nutrients removed by tea crops are compared with those removed by sole tea crops.

7.5.9 NUTRIENT REMOVAL AS HARVESTED TEA YIELD

Total amounts of some of the major nutrients (i.e., N, P, K, and Ca) that were removed as tea yield are shown in Table 7.10. Total amounts of all nutrients removed differed significantly between different CHIs and mulching treatments. The total amounts of N removed from the majority of mulched CHIs were lower than the corresponding amount removed from the sole tea crop. Out of the unmulched CHIs, only those under *Euphatorium* had a greater amount of N removed than the control, whereas the rest had lower amounts of total N removed. Because of greater tea yields in mulched CHIs (Figure 7.3), the total N removed from all mulched CHIs were substantially greater than those from the respective unmulched CHIs.

The total P removed as tea yield from all CHIs except the mulched CHI under *Euphatorium* was lower than that from the SC control (Table 7.10). Similarly, except for the mulched and unmulched CHIs under *Euphatorium*, the total amounts of K removed from CHIs were lower than those from the control. In contrast to other nutrients, the total amounts of Ca removed from a majority of mulched CHIs were higher than those removed from the control. However, among unmulched

TABLE 7.10
Total Amounts of Major Nutrients Removed as Yield of Tea in Contour Hedgerow Intercrops with Different Tree Species under Mulched (M) and Unmulched (UM) Conditions

	Nitrogen ($kg\ ha^{-1}\ yr^{-1}$)		Phosphorus ($kg\ ha^{-1}\ yr^{-1}$)		Potassium ($kg\ ha^{-1}\ yr^{-1}$)		Calcium ($kg\ ha^{-1}\ yr^{-1}$)	
Species	**M**	**UM**	**M**	**UM**	**M**	**UM**	**M**	**UM**
Calliandra calothyrsus	55 c*	45 b*	5.8 b*	4.6 b*	59 b*	50 b*	5.6 d	4.3 c*
Senna spectabilis	72 b	45 b*	4.6 c*	3.6 c*	52 c*	43 bc*	5.2 d*	3.8 d*
Euphatorium innulifolium	105 a*	85 a*	8.3 a*	6.3 a	97 a*	78 a	10.6 c*	8.1 b*
Flemingia congesta	66 b	51 b*	5.8 b*	4.2 b*	39 e*	28 d*	6.7 c*	5.2 b*
Gliricidia sepium	67 b	46 b*	5.9 b*	4.3 b*	61 b*	46 b*	6.6 b*	5.3 a*
Tithonia diversifolia	47 c*	38 b*	4.9 bc*	4.1 bc*	46 d*	39 cd*	6.8 a*	5.2 a*
Mean	69 A	52 B	5.87 A	4.5 B	59 A	47 B	6.9 A	5.3 B
Control	69		6.9		75		5.9	
CV (%)	6.63	7.63	10.97	8.34	7.54	6.14	10.80	6.05

Source: From De Costa, W.A.J.M., P. Surenthran and K.B. Attanayake, *Agroforest. Syst.*, 63, 211, 2005. With permission.

Note: Means connected vertically by the same lowercase letter are not significantly different at $p = 0.05$. Means connected horizontally by the same uppercase letter are not significantly different at $p = 0.05$. Significant ($p = 0.05$) differences from the control treatment are shown by*.

CHIs, all except that with *Euphatorium* had lower amounts of Ca removed than the control. The total amounts of P, K, and Ca removed from all mulched CHIs were substantially greater than those from the respective unmulched CHIs (Table 7.10). A correlation analysis showed that the total amounts of all above nutrients removed were significantly ($p < 0.0001$) positively correlated with both tea yield and the respective nutrient concentrations in the tea leaves.

7.5.10 IMPLICATIONS OF EXPERIMENTAL RESULTS FOR SUSTAINABILITY OF CHI IN TEA PLANTATIONS ON SLOPING HIGHLANDS IN SRI LANKA

The variations described earlier of soil physical and chemical properties showed that CHI without mulching is not a sustainable land management system for tea plantations in the present environment. After a detailed quantification of the soil nitrogen balance of hedgerow intercrops, Livesley et al. (2002) also concluded that removal of prunings would lead to long-term reduction of soil nutrients due to nutrient mining by tree roots. Although the removal of nutrients as tea yields (Table 7.10) was lower even in unmulched CHIs than in the control, when the nutrients removed by hedgerow prunings are also taken into account, the system has a net negative nutrient balance. Such a system would require increasing amounts of inorganic fertilizer to sustain tea yields even at the present levels. However, increased fertilizer application could increase hedgerow biomass production (as hedgerows competitively absorb more nutrients) and intensify competition on tea. On the other hand, increase of CEC and SOM (under some hedgerow species only) in the topsoil could decrease the loss of nutrients through leaching even in unmulched CHIs. Moreover, with time, increased topsoil pH (under some hedgerow species only) could make the growing conditions more favorable for tea in the CHI plots. Therefore, in the long run, these processes might shift the nutrient balance of unmulched CHIs toward the positive direction and make them sustainable.

In contrast to unmulched CHIs, the amounts of major nutrients (i.e., N, P, K) removed by all mulched CHIs except that with *Euphatorium* were lower than the respective amounts removed by the SC control (Table 7.10). Therefore, in the long run, the soil nutrient pool and consequently the tea yields are likely to increase. This ensures sustainability of this system. Although the higher nutrient availability in the soil may increase competitive absorption of nutrients by hedgerows, those nutrients would come back to the soil as mulch (i.e., nutrient recycling).

Results of the present study have also made it clear that addition of hedgerow prunings as mulch makes a significant impact on the overall nutrient balance and the long-term sustainability of incorporating contour hedgerows into tea plantations. In the majority of mulched CHIs, most of the essential nutrients were higher than those in SCs (Tables 7.5 through 7.7). However, phosphorus was an important exception. This is not surprising as phosphorus has been identified as the most limiting nutrient in CHIs on acid, infertile soils such as those of the present experiment (Palm, 1995; Rao et al., 1998). In contrast to observations of the present study, Samsuzzaman et al. (1999) observed that soil N, P, and organic C of a CHI on an acidic soil did not increase even with the addition of prunings and inorganic fertilizer. On the other hand, in agreement with the present study, Kang et al. (1999) observed increased topsoil organic C, K, and reduced pH in hedgerow intercrops, while Ikerra et al. (1999) observed increased topsoil N. However, Kang et al.'s (1999) observations of increased topsoil P and reduced Mg and Ca in hedgerow intercrops were contrary to the observations of the present study.

Despite their positive nutrient balance, the present tea yields in all mulched CHIs except those with *Euphatorium* were significantly lower than in the SC control (Figure 7.3). This shows that at this particular stage, the competition for other resources such as light and nutrients still outweighed the positive nutrient balance of mulched CHIs with the exception of that with *Euphatorium*. However, it is highly likely that in the long run, the mulched CHIs would minimize their nutrient losses (i.e., due to erosion and leaching) due to increased SOM and CEC and greater mulch production. This could probably push the tea yields of the rest of the mulched CHIs up and above that of the SC control, which would most probably decline gradually.

7.6 CASE STUDY 2: TREE–CROP INTERACTIONS IN CHIs INVOLVING MAIZE AND HEDGEROWS OF *G. SEPIUM* IN THE HUMID (WET) ZONE

7.6.1 BACKGROUND

This study also focuses on sloping lands of the humid, central highlands of Sri Lanka. While Case Study 1 described a CHI with a perennial crop (i.e., tea), the present study involves a CHI with a seasonal annual crop, maize (*Zea mays* L.). Over the years, a considerable area of sloping lands grown with tea had been converted to seasonal annual crops or abandoned because long-term, continuous degradation had made the soil too poor to support a crop. In lands converted to annual cropping, a variety of crops such as maize, legumes, and fresh vegetables are grown. These are normally sown at the beginning of the rainy season, leading to significant soil erosion because of land clearing and soil loosening during periods of intensive rainfall. Therefore, CHI had been introduced in these lands to arrest soil erosion, regenerate fertility, and sustain crop production. Results of investigations of TCIs of this CHI involving annual crops are presented in this case study.

7.6.2 SYSTEM DESCRIPTION AND TREE–CROP INTERACTIONS INVOLVED

The long-term (1997–2003) experiment consisted of a CHI with *G. sepium* in contour hedgerows spaced at 6 m and maize at 0.5 × 0.5 m in between hedgerows. Each CHI plot was 10 m (along the contour) × 20 m (along the slope). Each hedgerow involved two rows (0.5 m in between) of closely planted *Gliricidia* stem cuttings. The experimental land had a 35% slope and the soil was a mountain Regosol (Panabokke, 1996). The site had an average rainfall of 2000 mm yr^{-1}. The four experimental treatments were designed to estimate the competition (C) and fertility (F) effects of the TCI equation proposed by Ong (1996):

$$\text{TCI} = C + F. \tag{7.3}$$

There were two hedgerow intercrops with and without *Gliricidia* prunings added as mulch (H_m and H_n, respectively) and two SCs of maize with (S_m) and without (S_n) prunings as mulch. Hence, C and F could be estimated as:

$$F = Y_{S_\text{m}} - Y_{S_\text{o}}, \tag{7.4}$$

$$C_2 = Y_{H_\text{o}} - Y_{S_\text{o}}, \tag{7.5}$$

where Y denotes the maize yield in each of the four treatments. All estimates were standardized as a percentage of the yield of S_o (i.e., SC without added mulch).

The four treatments were replicated twice in a randomized complete block design and the 200 m^2 plots were separated from each other by a 1 m deep drain that reached down to the hard gravel layer. Hence, there was no possibility of hedgerow roots growing laterally into adjoining SC plots. Hedgerows were pruned at a height of 0.75 m just before sowing of maize and the prunings were applied as mulch in H_m and S_m treatments.

7.6.3 YIELD VARIATION OF MAIZE ACROSS A TRANSECT BETWEEN TWO CONTOUR HEDGEROWS

In the CHIs, variation of row-wise maize yields across a transect of the alley between two adjacent *Gliricidia* hedgerows showed that maize yields were reduced in rows closer to the hedgerows on both sides (Figure 7.11), thus indicating that hedgerows exerted significant resource competition. Row-wise yields had a single peak in the middle of the plot, which showed that competition decreased gradually with increasing distance from hedgerows. Interestingly, row-wise yields of the SC also showed a somewhat similar pattern, but one that was more skewed to the downslope

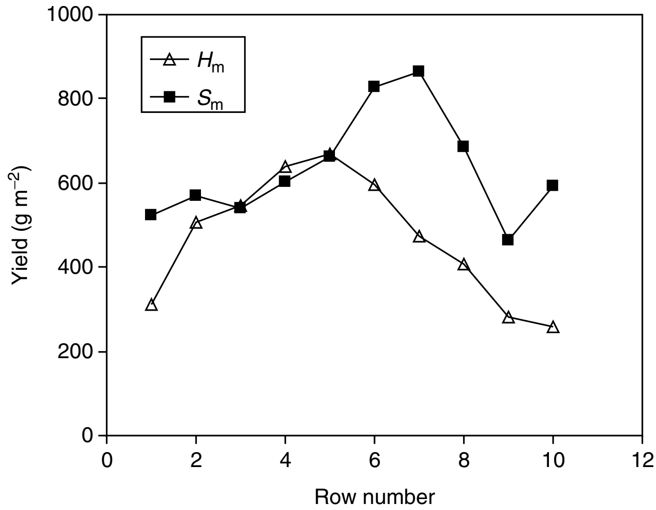

FIGURE 7.11 Variation of row-wise maize yields (in terms of cob fresh weights) across a transect of the alley between two adjacent *Gliricidia* hedgerows. Row numbers start from the top of the slope and rows were spaced at 0.5 m.

side. However, except for the third, fourth, and fifth rows from the top of the plot, all other rows of the CHI had lower yields than the corresponding rows of the SC, thus confirming the presence of competition from hedgerows.

SCs with mulch added (S_m), where the fertility effect (F) was present but competition effect (C) was absent, showed the highest cob and grain yield (Y) and total biomass (W) in three seasons over a 3 year period (Figure 7.12). In contrast, hedgerow intercrops without added mulch (H_n), where F was absent but C was present, showed the lowest Y and W. Hedgerow intercrops with added mulch (H_m), where both F and C were present, had slightly greater Y and W than SCs without mulch (S_n), where both F and C were absent. In the first season, crops were grown without any inorganic fertilizer to get an idea of the initial fertility of the soil. This was the reason for the extremely low yields in this season. In the subsequent seasons, inorganic fertilizers were added in dosages recommended for maize (Anonymous, 1990).

Figure 7.13 shows the overall TCI and its components estimated on the basis of Y and W. The fertility effect (F), estimated based on both Y and W, was positive in all three seasons, indicating that S_m always had a greater yield than S_n. As expected, competition effect (C), was negative in 1999 and 2002, indicating lower Y and W in H_n than in S_n. The year 2000 was an exception where estimates of C on the basis of W and Y showed a discrepancy. In spite of this slight irregularity, the overall TCI was above zero in all seasons indicating that F always outweighed C and that this particular CHI was biologically sustainable.

The biomass of prunings produced by hedgerows showed a slight increasing trend with time (Figure 7.14). However, there was no significant difference between the mulched and the unmulched hedgerows.

7.6.4 CONTROL OF SOIL EROSION, SURFACE RUNOFF, AND NUTRIENT LOSS BY CONTOUR HEDGEROWS

The extent of soil erosion and surface runoff was measured continuously using erosion plots (Wiersum, 1991). There was a system of collection pits for eroding soil. Volume of runoff water was measured by a tipping bucket devise. Amount of soil loss as suspended particles in runoff water was measured by sampling runoff water. There was substantial reduction of soil erosion

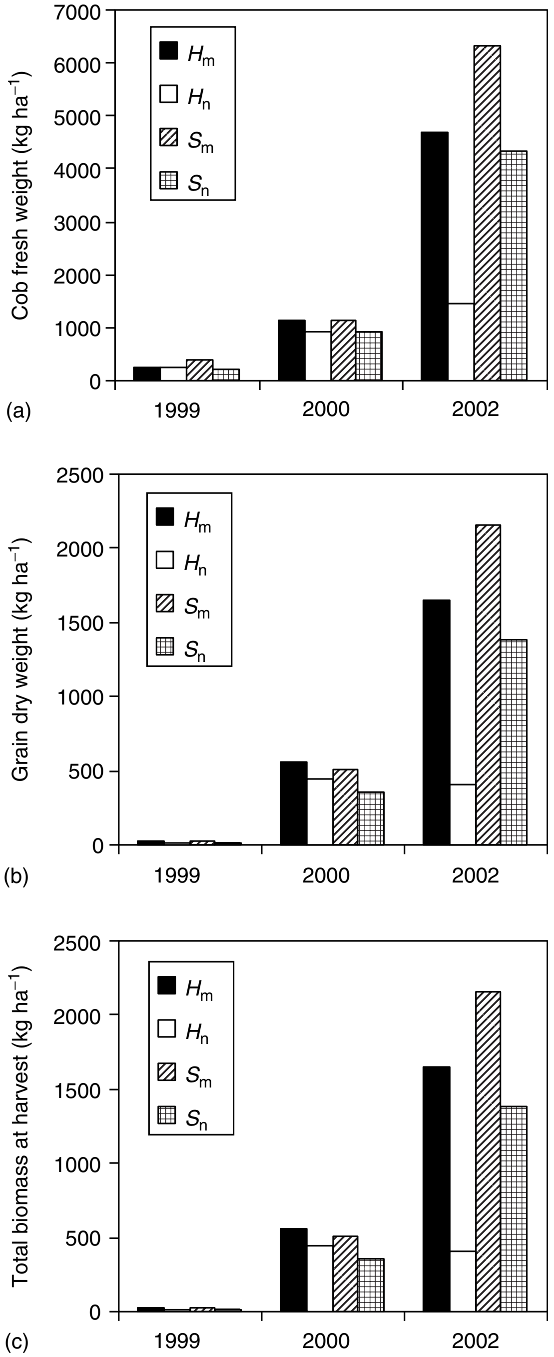

FIGURE 7.12 Variation of maize yield (a, b) and total biomass at harvest (c) in mulched and unmulched sole crops (S_m and S_n) and CHIs (H_m and H_n) with *Gliricidia sepium* on sloping terrain in the humid zone of Sri Lanka.

and surface runoff in CHIs as compared with SC (Figure 7.15). Over a 1 year period with a well-distributed rainfall of 2034.9 mm, total soil loss through erosion was 9.22 and 60.77 t ha^{-1} yr^{-1} in CHI and SC, respectively. In both systems, ~70% of soil erosion occurred during the cropping period, especially at land preparation and early crop growth when the soil was loose and

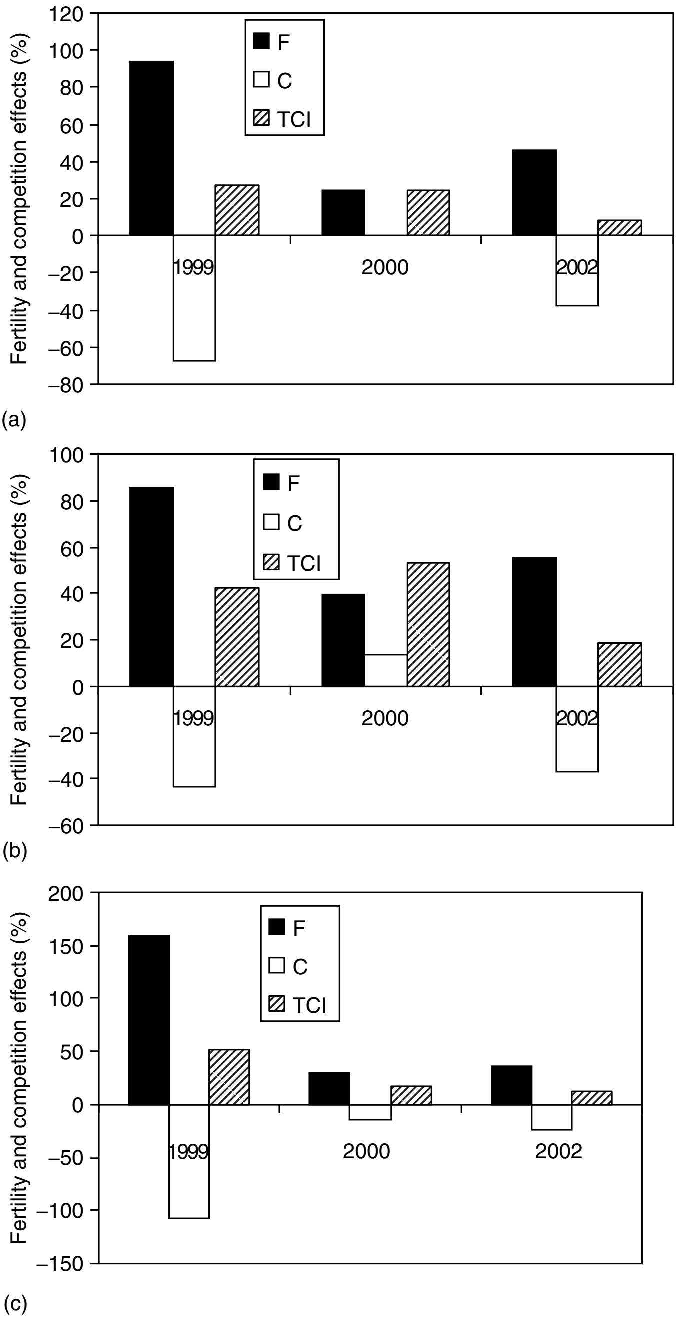

FIGURE 7.13 Overall tree–crop interaction (TCI) and its components (Fertility, *F*, and Competition, *C*, effects) estimated on the basis of cob fresh weight (a), grain dry weight (b), and total biomass at harvest (c) of maize.

exposed to direct rainfall. During the fallow period, the ratio of soil erosion between CHI and SC was around a constant value of 0.16. This increased to 0.25 during the cropping period.

During the same 1 year period, total surface runoff was 189 and 821 mm yr^{-1} in CHI and SC, respectively. The percentages of rainfall lost as surface runoff were 9.29% and 40.35% for CHI and SC, respectively. In both systems, the percentage of total runoff occurring during the fallow period (52%) was slightly higher than that during the cropping period (48%).

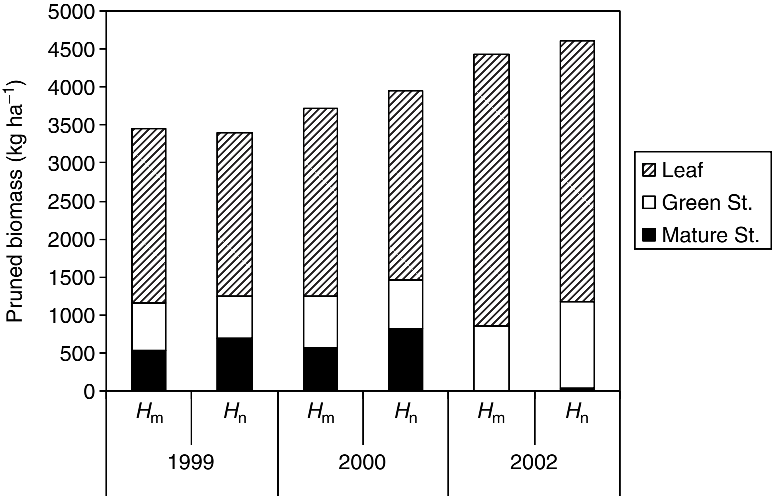

FIGURE 7.14 Variation of biomass prunings in mulched (H_m) and unmulched (H_n) hedgerows of *Gliricidia sepium* in a CHI with maize on sloping terrain in the humid zone of Sri Lanka.

Losses of major soil nutrients were substantially higher in SC than in CHI (Table 7.11). This was because of the greater amount of soil loss and the slightly higher topsoil nutrient concentrations in the SC plots. In terms of absolute amounts, the greatest nutrient loss occurred in total soil N, whereas the lowest was in available P.

7.6.5 VARIATION OF SOIL AND PLANT NUTRIENTS

Maize in both the mulched CHI and SC had significantly greater P and K contents than the respective unmulched treatments (Figure 7.16). This was clearly because of the fertility effect of mulching, which made available a greater pool of nutrients for maize to absorb. The lowest P and K were shown in H_n again showing the competition effect of hedgerows, which decreased the nutrients available to maize. In contrast to P and K, plant N contents did not differ significantly between the four treatments.

At the end of the cropping season, topsoil (0–10 cm depth) total N showed reductions (relative to the levels at the beginning of the season) in all treatments except S_m (Figure 7.17). The highest reduction was in H_m probably because the greater biomass production in this treatment absorbed a greater amount of N from the soil. Topsoil available P and pH showed increases in all treatments, with increases in the two mulched treatments being substantially greater than those in the two unmulched treatments. Exchangeable K showed reductions at the end of the season in all four treatments.

7.7 CASE STUDY 3: TREE–CROP INTERACTIONS IN CHIs INVOLVING MUNG BEAN (*V. RADIATA*) AND DIFFERENT HEDGEROW SPECIES IN THE SUBHUMID (INTERMEDIATE) ZONE

7.7.1 BACKGROUND

Like Case Study 2, this study also investigated CHIs involving an annual crop (i.e., mung bean) grown on sloping terrain. However, the site of the present study was located in the subhumid intermediate zone, which received less rainfall than the humid wet zone. It had a distinctly bimodal rainfall pattern with significant dry periods in between. The soil was also different, being a moderately, well-drained sandy clay loam belonging to Rhodudults (known as Reddish-Brown Latosolic according to the local classification by Panabokke (1996)). Moreover, the soil pH was around 6.5,

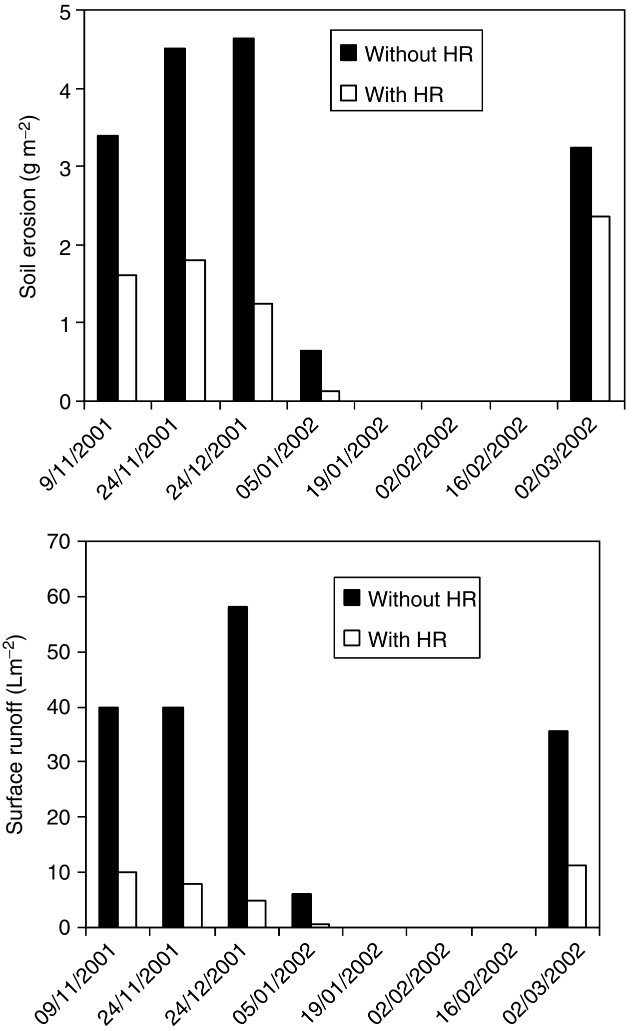

FIGURE 7.15 Variation of soil erosion and surface runoff during a growing season of maize on sloping terrain with (with HR) and without (without HR) contour hedgerows of *Gliricidia sepium* in the humid zone of Sri Lanka. Maize was planted on 15 November 2001 and harvested on 25 February 2002.

TABLE 7.11

Loss of Major Nutrients from Topsoil (0–10 cm) through Erosion in Contour Hedgerow Intercrops (CHI) and Sole Crops (SC) of Maize on Sloping Terrain in the Humid Zone of Sri Lanka

		Initial Topsoil Nutrient Content			**Nutrient Loss (kg ha^{-1} yr^{-1})**		
	Soil Loss (t ha^{-1}yr^{-1})	**Tot. N (%)**	**Av. P (mg/100 g)**	**Exch. K (mg/100 g)**	**Tot. N**	**Av. P (×10^{-3})**	**Exch. K**
CHI	9.22	0.159	0.267	20.72	14.64	24.59	1.91
C	60.77	0.179	0.338	23.71	108.66	205.58	14.41

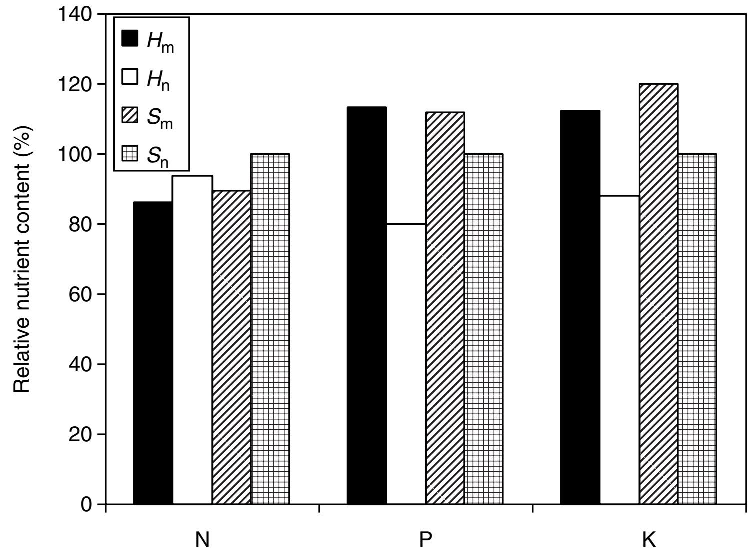

FIGURE 7.16 Nutrient contents of maize (expressed as % of S_n) in mulched and unmulched sole crops (S_m and S_n) and CHIs (H_m and H_n) with *Gliricidia sepium* on sloping terrain in the humid zone of Sri Lanka.

which was higher than that in sites of Case Studies 1 (~4.0) and 2 (~5.0). The site of Case Study 3 represented lands that have been either continuously cultivated with seasonal annual crops or homegardens converted to annual cropping. Gradual erosion and loss of soil fertility are major problems in these lands as well and hence CHI has been a recommended option to regenerate soil fertility and sustain crop production. Another distinct feature of the climatic zone represented by this study site is the practice of animal husbandry (cattle and goat) by a considerable portion of its subsistence farmers. Therefore, the hedgerows could also serve as a source of fodder for farm animals.

7.7.2 SYSTEM DESCRIPTION AND TREE–CROP INTERACTIONS INVOLVED

The experimental treatments consisted of six CHIs and a SC control. The intercrop treatments consisted of mung bean (*V. radiata* (L.) Wilczek) with six different tree species namely, *Calliandra calothyrsus*, *Desmodium ransonii*, *F. congesta*, *G. sepium*, *S. spectabilis*, and *T. diversifolia*. The control treatment was a SC of mung bean. The tree species had been established 5 years before the commencement of the experiment as double hedgerows (15 cm apart) along contours in a land with an even slope of 10%. The distance between two adjacent double hedgerows was 4–5 m. Mung bean was established in the alleys between hedgerows at an interrow spacing of 30 cm and an intra-row spacing of 8 cm.

Each experimental plot consisted of a hedgerow length of 5 m and its associated mung bean plot area spanning the whole width of the alley. To eliminate effects of the hedgerow at the far end of the experimental plot, a trench was cut to a depth of 1 m. Each treatment had three replicate plots in a randomized complete block design. All hedgerows were pruned to a height of 0.5 m about 1 week before sowing of the annual crop. The prunings were deposited between the double hedgerows (i.e., *not* in the alley). Thereafter, the hedges were allowed to grow freely during the cropping season. Annual crops were grown without any addition of chemical fertilizers or organic manure.

7.7.3 VEGETATIVE GROWTH AND SEED YIELD OF MUNG BEAN UNDER DIFFERENT HEDGEROW SPECIES

Growth and yield of mung bean, measured across transects in the alleys between contour hedgerows, showed increases with increasing distance from hedgerows in all CHIs (Table 7.12;

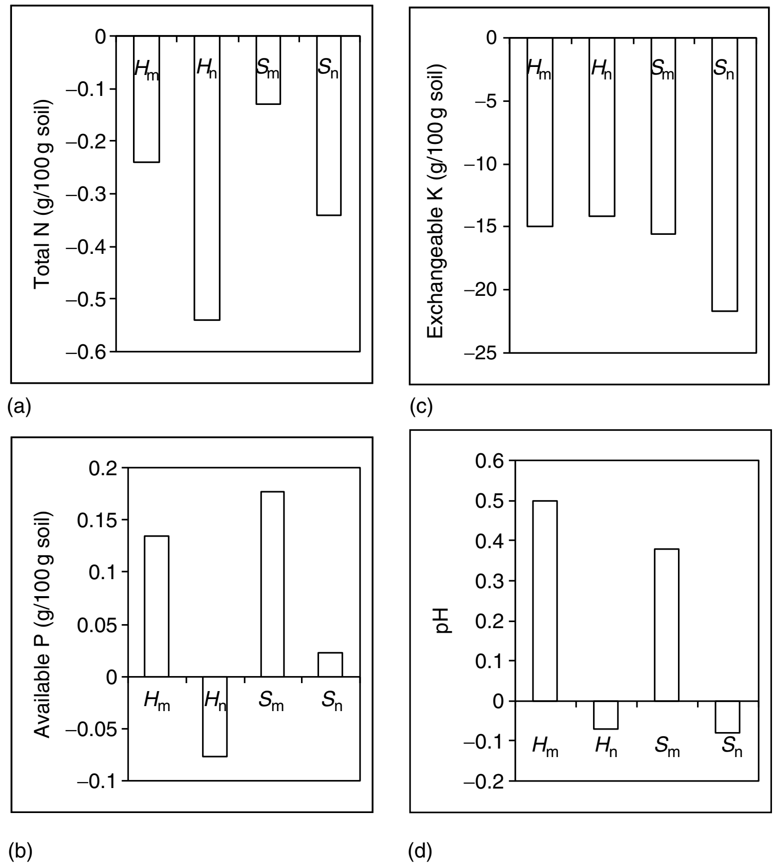

FIGURE 7.17 Changes in major soil nutrients (a, b, c) and soil pH (d) after a cropping season of maize in mulched and unmulched sole crops (S_m and S_n) and CHIs (H_m and H_n) with *Gliricidia sepium* on sloping terrain in the humid zone of Sri Lanka.

TABLE 7.12

Variation of Leaf Area and Total Biomass of Mung Bean under Different Hedgerow Tree Species at Different Distances (cm) from the Hedgerow in the Subhumid Zone of Sri Lanka

	Leaf Area (cm^2 plant^{-1})			Total Biomass (g plant^{-1})		
Tree Species	**30**	**150**	**300**	**30**	**150**	**300**
Calliandra	87	352	253	0.39	0.77	1.04
Senna	124	574	373	0.32	2.24	1.77
Desmodium	239	499	621	1.01	2.06	1.55
Flemingia	93	232	356	0.42	0.72	1.39
Gliricidia	174	459	445	0.59	2.30	1.43
Tithonia	138	325	432	0.43	1.47	2.16
Control	392	426	537	1.28	1.35	1.99
LSD$_{0.05}$	71	82	76	0.24	0.46	0.32
CV (%)	9	10	6	15	9	8

Source: From De Costa, W.A.J.M. and A.G. Chandrapala, *J. Agron. Crop Sci.*, 184, 43, 2000. With permission.

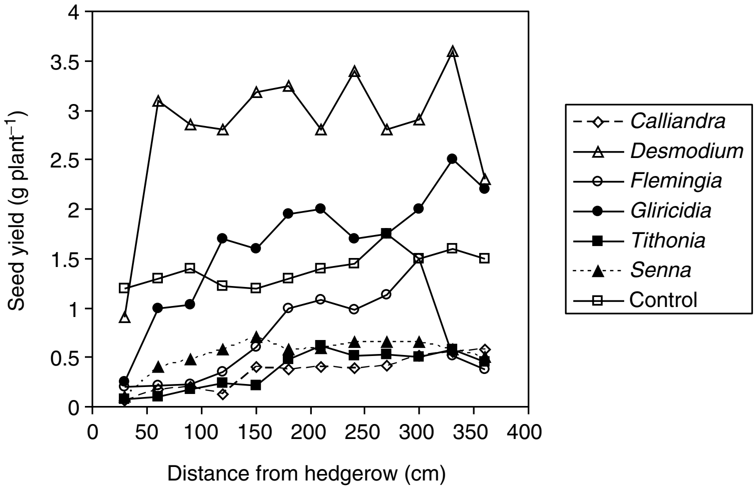

FIGURE 7.18 Variation of seed yield of mung bean with increasing distance from hedgerows in CHIs with different hedgerow species in the subhumid zone of Sri Lanka. (From De Costa, W.A.J.M. and A.G. Chandrapala, *J. Agron. Crop Sci.*, 184, 43, 2000. With permission.)

Figure 7.18). In mung bean growing under all hedgerow species, both leaf area (*L*) and total dry weight (*W*) at a 30 cm distance from the hedgerow were substantially lower than the corresponding values at 150 and 300 cm. In contrast, there was no such substantial decline of *L* and *W* in the control treatment. At 30 cm from the hedgerow, both *L* and *W* of all mung bean crops were lower than those of the control. On the other hand, at 150 and 300 cm, *L* and *W* of mung bean under some hedgerow species were similar to or higher than the corresponding values in the control. Mung bean growing with *Desmodium*, *Senna*, and *Gliricidia* had greater *L* and *W* at 150 cm from the hedgerow. In addition, mung bean under *Tithonia* also had greater *W* than the control. Vegetative growth of mung bean under *Calliandra* and *Flemingia* did not reach that of the control despite increases of *L* and *W* with increasing distance from the hedgerow.

There were differences in seed yield both between different tree species and in comparison with the yield of the SC control (Figure 7.18). Mung bean grown in combination with all tree species showed yield decreases of varying magnitudes at distances closer to the hedgerows. In contrast, there was no such yield variation across the control plot. Following the increase of yield with increasing distance from the hedgerow, there was another yield decline around 300–330 cm. This was probably because of the second hedgerow that is spaced at a distance of 4 m from the first. At 30 cm from the hedgerow, mung bean yields of all CHIs were lower than the SC yield. On the other hand, at 150 cm, yields under *Desmodium* and *Gliricidia* were greater than that of the control. However, despite the gradual yield increase with increasing distance from the hedgerow, intercropped mung bean yields under the rest of the tree species at 150 cm were still lower than the corresponding SC yield. The yield increases away from hedgerows were most prominent in mung bean with *Desmodium* and *Gliricidia*. In the yield variation pattern with distance under each tree species, a minimum distance (D_{min}) at which the yield reached a value which was not significantly different from its maximum (Y_{max}) could be identified. Under *Desmodium*, a near Y_{max} yield was achieved around a D_{min} of 60 cm, whereas with *Gliricidia* it was achieved around 120 cm. In the treatment group where Y_{max} was always less than that of control, D_{min} was achieved around 180 cm (*Flemingia*), 150 cm (*Senna* and *Calliandra*), and 210 cm (*Tithonia*). In identifying Y_{max}, the values at 300–330 cm distances were disregarded because of the possible influence from the adjacent hedgerow.

The different tree species used as hedgerows showed significant variation in the biomass of prunings produced (Table 7.13) during the 90 day life span of the associated mung bean crop.

TABLE 7.13

Biomass Production of Different Hedgerow Tree Species during the Life Span of the Mung Bean Crop in the Subhumid Zone of Sri Lanka

Tree Species	Fresh Biomass (t ha^{-1})	Dry Biomass (t ha^{-1})
Calliandra	5.00	1.44
Desmodium	3.93	0.98
Flemingia	1.26	0.33
Gliricidia	10.38	2.21
Tithonia	14.57	2.32
Senna	11.89	3.48
$LSD_{0.05}$	3.07	0.62
CV (%)	21.56	19.08

Source: From De Costa, W.A.J.M. and A.G. Chandrapala, *J. Agron. Crop Sci.*, 184, 43, 2000. With permission.

Tithonia, *Senna*, and *Gliricidia* produced significantly higher biomass (in terms of both fresh and dry weights) of prunings than the rest. Biomass of prunings was lowest in *Flemingia* and *Desmodium*, with *Calliandra* showing intermediate levels.

This case study also showed that hedgerows exerted significant competition on the associated crop (mung bean in this study), leading to significant reductions in both vegetative growth and yield. As in Case Study 1, different tree species differed in the magnitude of competition exerted. This was shown in terms of both the difference between Y_{max} of the respective CHIs and the SC and the distance from hedgerow at which Y_{max} was achieved (i.e., D_{min}). Among the tree species examined in the present study, *Desmodium* and *Gliricidia* had a positive influence on the associated mung bean. This was probably because the beneficial effects of these tree species outweighed the negative effects of competition exerted by them. Such beneficial effects could be the improvement of soil physical and chemical properties through addition of tree prunings as mulch (Kang et al., 1990; De Costa et al., 2005, and shown earlier in Case Study 1) and amelioration of the aboveground and belowground environmental conditions because of the presence of hedgerows (De Costa and Chandrapala, 2000, and shown later in this chapter). One possible reason for the significant net positive effect of *Desmodium* could be its low aboveground biomass production (Table 7.16). Hence, the demand for capture of resources would be lower and thereby the competition on the annual crop would be lower. However, results on other tree species showed that the aboveground biomass production was not the only criterion that determines the overall competition exerted on the annual crop. Most probably, variations in lateral extension and depth of the tree root systems among the different tree species (van Noordwijk et al., 1996) could have been responsible for the observed yield variations in mung bean. For example, van Noordwijk and Purnomosidhi (1995) showed that regular pruning of the shoot to reduce competition for aboveground resources could enhance the competition for belowground resources via stimulation of the production of fine roots in the topsoil where majority of roots of the annual crop are also present. Therefore, it is important to separate the aboveground and belowground competition exerted by the different tree species and also relate them to species characteristics.

7.7.4 SEPARATION OF ABOVEGROUND AND BELOWGROUND COMPETITION OF CONTOUR HEDGEROWS

In a separate experiment at the same site, aboveground (C_A) and belowground (C_B) competition exerted by different hedgerows species were quantified by cutting a 1.5 m deep trench between the hedgerow and the nearest mung bean row. In a split-plot design where hedgerow species were the main plots, each CHI plot had two subplots, trenched (TR) and nontrenched (NTR). The trench prevented lateral growth of hedgerow roots into mung bean plots and thereby prevented belowground competition. C_A and C_B were estimated on the basis of mung bean pod yields (Y) using the following equations:

$$C_A = Y_S - YH_{TR}, \tag{7.6}$$

$$C_B = YH_{TR} - YH_{NTR}, \tag{7.7}$$

where

Y_S is sole mung bean yield

YH_{TR} and YH_{NTR} are the respective mung bean yields in CHIs with and without trenching

These absolute estimates of competition were also expressed as percentages of total competition (i.e., $Y_S - YH_{NTR}$).

As in the previous experiment, there were significant reductions in leaf area index (L), total biomass (W), and seed yield (Y) of mung bean growing in both TR and NTR CHIs. With trenching, both L and W increased under all species, with the highest increases being shown in mung bean with *Calliandra* and *Tithonia* (data not shown). On the other hand, the positive responses of L and W to trenching were lower in mung bean crops with *Gliricidia* and *Desmodium*. Within each CHI, mung bean grown with trenching had greater Y than those grown without trenching (Table 7.14). The greatest positive response to trenching in terms of Y was shown in mung bean with *Calliandra,* whereas mung bean with *Senna* and *Tithonia* showed the lowest positive response.

Table 7.14 also shows the estimates of C_A and C_B based on yield variation between TR and NTR treatments with SC yield as the control. The highest C_A, in both absolute and percentage terms, was shown by *Senna*, whereas *Gliricidia* showed the lowest C_A. In absolute terms, C_B was highest in mung bean with *Calliandra* and lowest with *Senna*. On the other hand, when considered as a percentage of the total yield reduction relative to the control, the percentage C_B was highest in mung bean with *Gliricidia*. When the total yield reduction due to hedges was partitioned because of C_A and C_B, only *Gliricidia* and *Calliandra* showed percentage C_B levels that exceeded 25%. C_A was the dominant component in all hedgerow species except *Gliricidia* where the two components were approximately equal.

TABLE 7.14

Variation of Mung Bean Yield under Different Hedgerow Tree Species and Trenching Treatments along with Estimates of Shoot and Root Competition

	Pod Dry Weight (kg ha^{-1})		**Competition (Absolute)[a] (kg ha^{-1})**		**Competition (Percentage of Total)[a] (%)**	
Tree Species	**TR**	**NTR**	**Shoot**	**Root**	**Shoot**	**Root**
Control	806.3					
Cassia	142.5	105.3	663.8	37.2	94.7	5.3
Calliandra	455.3	265.7	351.0	189.6	64.9	35.1
Gliricidia	686.8	563.3	119.5	123.5	49.2	50.8
Flemingia	310.6	202.1	495.7	108.5	82.0	18.0
Tithonia	361.8	317.8	444.5	44.0	91.0	9.0
Desmodium	330.2	201.0	476.1	129.2	78.7	21.3
Mean	441.9	351.6				
SE (df = 6)	57.66					

[a] See text for explanation.

Interestingly, the main finding of this experiment was the dominance of aboveground competition (C_A) over belowground competition (C_B) in this environment. This contrasted with the findings of Singh et al. (1989), Rao et al. (1990, 1991), and Ong et al. (1991) who observed greater C_B. However, dominance of C_B could vary with the availability of belowground resources such as soil nutrients (Fernandes et al., 1993) and moisture (Govindarajan et al., 1996; McIntyre et al., 1997). It is also possible that root systems of hedgerow species and mung bean may have occupied complementary zones in the soil profile which would have minimized the competition for absorption of water and nutrients. Such a situation has been observed by Huxley et al. (1994) in a *Grevillea robusta* × maize hedgerow intercropping system. Observed variation in C_B between tree species could be due to variation in the extent and depth of their root systems (Ruhigwa et al., 1992; van Noordwijk and Purnomosidhi, 1995). Teubig (1996), who studied the morphology of hegderow root systems in the present site, observed that *Gliricidia* had a shallow and spreading root system, whereas the other tree species had deeper and less-spreading root systems. This may explain the greater percentage C_B by *Gliricidia*. The dominance of C_A in the present experiment means that appropriate hedgerow management practices should be formulated to minimize C_A. These could include pruning of hedgerows in the middle of the cropping season instead of waiting until the harvesting of mung bean.

7.7.5 VARIATION OF PLANT AND SOIL NUTRIENTS IN CHIs WITH DIFFERENT TREE SPECIES

Analysis of major plant nutrients in mung bean showed that hedgerows exerted significant competition for nutrients and that trenching reduced it. Trenching significantly ($p < 0.001$) increased N content in mung bean grown with all tree species (Table 7.15). P and K contents of mung bean in all CHIs were significantly lower than that of the control in both trenching treatments. Removal of belowground competition by trenching significantly increased the K content of mung bean with all hedgerow species as compared with NTR. On the other hand, although mung bean under the TR had higher P than the corresponding NTR except in the crop with *Desmodium*, the difference was not significant ($p = 0.05$).

In the first season of experimentation (without trenching), both soil total N and available P (in depth layers 0–10 and 20–40 cm) showed depletions at the end of the season as compared with the beginning (Table 7.16). Interestingly, although the soil N depletions in a majority of CHIs were

TABLE 7.15

Major Nutrient Contents of Mung Bean Grown with Different Hedgerow Tree Species with (TR) and without (NTR) Trenching in the Subhumid Zone of Sri Lanka

	Nitrogen (%)		Phosphorus (mg/100 g Dry Weight)		Potassium (mg/100 g Dry Weight)	
Tree Species	**TR**	**NTR**	**TR**	**NTR**	**TR**	**NTR**
Control	1.059		187.33		75.22	
Cassia	0.752	0.613	124.44	112.22	41.00	33.33
Calliandra	1.117	1.011	137.22	122.78	52.00	44.89
Gliricidia	1.126	1.023	142.78	128.33	67.44	59.78
Flemingia	0.957	0.938	124.44	105.56	52.78	46.00
Tithonia	1.074	0.994	117.78	107.78	56.89	42.67
Desmodium	1.086	1.018	114.44	125.56	46.44	42.11
Mean	1.024	0.951	135.49	127.08	55.97	49.14
SE (df = 18)	0.029		8.21		3.28	

Note: Each value is the mean of measurements at three distances from hedgerows (30, 180, and 360 cm) in three replicate plots of each species × trenching treatment combination.

TABLE 7.16
Soil Nutrient Contents in Plots between Hedgerows at Sowing (S) and Harvesting (H) of Mung Bean in CHIs in the Subhumid Zone of Sri Lanka

	Available Nitrogen (mg/100 g Soil)		Available Phosphorus (mg/100 g Soil)	
Tree Species	**S**	**H**	**S**	**H**
Calliandra	5.72	3.27	28.19	18.44
	2.45		*9.75*	
Desmodium	7.47	3.03	21.37	12.78
	4.44		*8.59*	
Flemingia	5.72	2.92	27.38	16.95
	2.80		*10.43*	
Gliricidia	6.07	2.10	42.24	22.22
	3.97		*20.02*	
Tithonia	5.78	1.52	39.83	27.07
	4.26		*12.76*	
Senna	6.30	2.80	31.80	16.70
	3.50		*15.10*	
Control	6.07	3.38	53.88	21.23
	2.69		*32.65*	
$LSD_{0.05}$	0.72	0.53	7.60	4.82
CV (%)	15.74	16.54	13.73	13.35

Source: From De Costa, W.A.J.M. and A.G. Chandrapala, *J. Agron. Crop Sci.*, 184, 145, 2000. With permission.

Note: The amounts of N and P removed during the season are given in italics. Each value is averaged over two soil depths (0–10 cm and 20–40 cm) and three distances from the hedgerows (15, 165, and 315 cm).

greater than those of the SC control, P depletions in all CHIs were lower than those in the control. It is possible that the presence of hedgerow roots in CHIs initiate mechanisms that increase the availability of P in the soil either through reduced fixation or increased release (e.g., through turnover of hedgerow roots). During the second season, except in the CHI with *Gliricidia*, soil P in all CHIs showed increases by the end of the season (Table 7.17), whereas that of the control showed a depletion. Soil available K also showed enrichment in all CHIs and the control. Trenching increased enrichment or reduced depletion in all nutrients. This again indicates that despite their significant competition for nutrients, contour hedgerows have the ability to build up the soil nutrient pool over a longer time scale.

7.7.6 ENVIRONMENTAL INTERACTIONS

The possible competition from contour hedgerows for radiation and soil water were also investigated, especially because of the lower rainfall and the longer dry periods that prevail in this climatic zone. The fraction of incoming radiation intercepted was measured at the top of the canopy and the ground level, at 15 and 200 cm distances from hedgerows by tube solarimeters at different times of the crops' life cycle. Likewise, topsoil (0–10 cm) and subsoil (20–40 cm) water contents (SWC) were measured at 2 week intervals by gravimetric sampling.

7.7.6.1 Radiation Interception by Hedgerow Canopies

At 15 cm from hedgerows, a majority of CHIs intercepted a greater fraction of incoming radiation (F_R) than the SC control (Figure 7.19a). It is highly likely that at 15 cm from hedges a greater

TABLE 7.17

Changes in Soil Nutrient Contents and pH during the Cropping Season of Mung Bean Grown with Different Hedgerow Tree Species with (TR) and without (NTR) Trenching in CHIs in the Subhumid Zone of Sri Lanka

	Total N (%) ($\times 10^{-3}$)		Available P (mg/100 g)		Available K (mg/100 g)	
Species	**TR**	**NTR**	**TR**	**NTR**	**TR**	**NTR**
Control	−10.0		−0.942		+16.14	
Senna	−0.4	−10.8	+0.343	+0.265	+22.75	+22.16
Calliandra	−1.9	−32.1	+0.424	+0.300	+24.53	+23.49
Gliricidia	−76.9	−80.1	−1.116	−1.380	+21.50	+19.61
Flemingia	+4.4	+1.0	+0.283	+0.164	+25.14	+23.10
Tithonia	−6.9	−20.4	+0.390	+0.252	+24.06	+21.33
Desmodium	+9.9	−15.2	+0.291	+0.265	+23.49	+21.27

Note: − indicates nutrient depletion; + indicates nutrient enrichment.

fraction of F_R would constitute radiation absorbed by canopies of hedgerows as crop growth is depressed (as shown earlier) near hedges. The CHI with *Tithonia*, which had the "most laterally spreading canopy," had significantly greater F_R at 15 cm than the rest of the species from 6 weeks after sowing (WAS) onwards. In general, F_R at 200 cm (Figure 7.19b) from hedgerows was lower

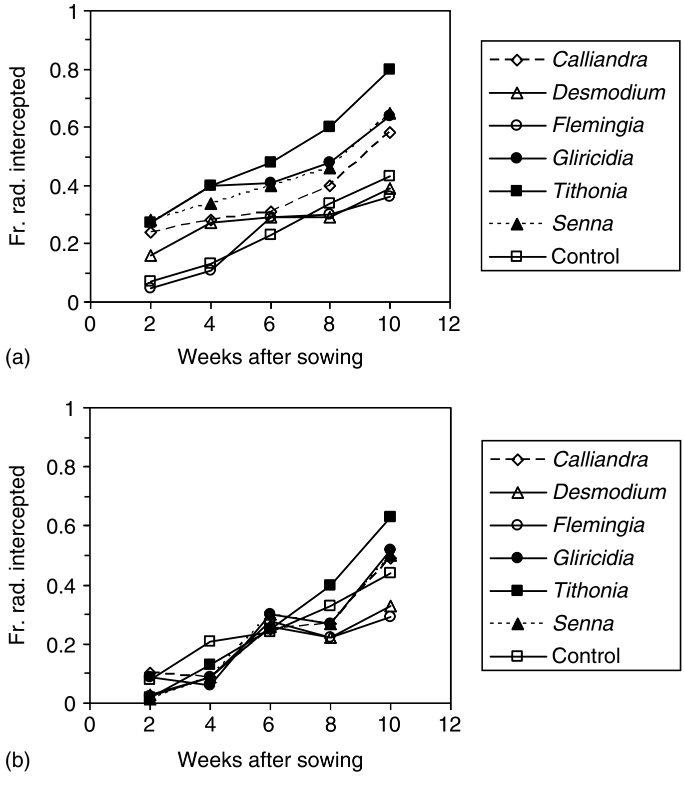

FIGURE 7.19 Variation of the fraction of incoming radiation intercepted at 15 cm (a) and 200 cm (b) from hedgerows in CHIs with mung bean and different hedgerow species in the subhumid zone of Sri Lanka.

than that at 15 cm, because it largely indicates the radiation intercepted by the crop only. During the first 4 weeks, SCs had significantly greater or equal F_R at 200 cm than all CHIs. This was because of greater initial crop growth in the SCs that did not face competition for light from hedgerows. At 6 WAS, F_R of all CHIs did not differ significantly from the SC. From then onwards, the CHI with *Tithonia* had significantly greater F_R at 200 cm than all other crops including the control. This was because of the greater canopy volume of *Tithonia*. In contrast, CHIs with hedgerow species such as *Flemingia* and *Desmodium*, which had the "least laterally spreading" canopies, had significantly lower F_R at 200 cm than the rest.

These results on radiation interception showed that most hedgerow species, especially those such as *Tithonia*, exert significant competition on the associated annual crop for light as well, particularly at distances closer to the hedges. Corlett et al. (1992) also observed greater shading of annual crops when canopies of hedges were allowed to spread by less frequent pruning.

7.7.6.2 Variation of Soil Water Content in CHIs

Topsoil (0–10 cm) and subsoil (20–40 cm) water contents at 6 WAS in different CHIs under TR and NTR conditions are shown in Table 7.18. Except in CHIs with *Tithonia*, SWC of CHIs was lower than that of the control, probably due to the greater extraction of soil water by hedgerows. The spreading canopy of *Tithonia* probably allowed soil moisture conservation, especially in the topsoil, through shading. Trenching increased the SWC in both topsoil and subsoil in all CHIs because of the lower spread of hedgerow roots into the area occupied by mung bean.

Table 7.19 shows the variation of SWC with increasing distance from hedgerows throughout the season. During this particular season, crops were planted on stored soil moisture. There was a substantial rainfall (201 mm) during the 2 week period between 3 and 5 WAS followed by a relatively dry period until season's end. It could be observed that during periods when the soil was not at full or near saturation (i.e., 2, 8, and 10 WAS), SWC closer to the hedgerows (i.e., at 15 cm distance) was significantly greater than that at 200 cm. This clearly indicated that there was soil moisture conservation closer to the hedgerows due to shading by their canopies. This could have been achieved by reducing transpiration from the crop surface and direct evaporation from the soil surface.

7.7.6.3 Variation of Soil Temperature in CHIs

Prevention of the development of excessively high soil temperatures, especially during periods of dry weather and at the crop establishment stage, is crucial in ensuring good seed germination,

TABLE 7.18

Variation of Topsoil (0–10 cm) and Subsoil (20–40 cm) Water Contents at 6 Weeks after Sowing under Trenched (TR) and Nontrenched (NTR) Conditions in Different CHIs in the Subhumid Zone of Sri Lanka

	SWC at 0–10 cm (% Dry Weight)		SWC at 20–40 cm (% Dry Weight)	
Species	**TR**	**NTR**	**TR**	**NTR**
Calliandra	16.5	15.5	15.3	15.0
Desmodium	22.0	19.7	19.8	17.7
Flemingia	17.5	16.7	15.5	15.0
Gliricidia	22.0	18.0	18.3	17.0
Tithonia	27.5	25.5	24.0	21.5
Senna	17.5	16.7	18.0	16.5
Control	23.0	23.0	23.5	23.5
$LSD_{0.05}$	0.42	0.38	0.33	0.44

TABLE 7.19

Seasonal Variation of Soil Water Contents (% Dry Weight Basis) at Different Distances from Hedgerows in Topsoil (0–10 cm depth) and Subsoil (20–40 cm) Layers for Treatments with (TR) and without (NTR) Trenching in CHIs with Mung Bean in the Subhumid Zone of Sri Lanka

Weeks after Sowing	Distance from Hedge (cm)	Topsoil		Subsoil	
		TR	NTR	TR	NTR
2	15	19.31	16.21†	16.84	14.11†
	200	13.89*	13.64*$^{,\,ns}$	14.98$^{\mathrm{ns}}$	14.25$^{\mathrm{ns},\,ns}$
4	15	26.41	24.53†	26.02	24.69†
	200	27.43*	26.24*$^{,\,\dagger}$	27.03$^{\mathrm{ns}}$	26.09$^{\mathrm{ns},\,ns}$
6	15	21.31	20.12†	18.89	18.13^{ns}
	200	21.48$^{\mathrm{ns}}$	20.12$^{\mathrm{ns},\,\dagger}$	19.54$^{\mathrm{ns}}$	19.08$^{\mathrm{ns},\,ns}$
8	15	10.08	9.96^{ns}	14.47	12.93†
	200	8.51*	8.60*$^{,\,ns}$	11.52*	11.09*$^{,\,ns}$
10	15	8.74	8.00^{ns}	10.17	8.39†
	200	7.42*	6.89*$^{,\,ns}$	8.06*	7.46*$^{,\,ns}$

Note: Each value is the mean of six tree species (plus control) and three replicates. Significance (or otherwise) of mean comparisons between distances within each trenching treatment is shown by* (or $^{\mathrm{ns}}$). Significance (or otherwise) of mean comparisons between trenching treatments within each distance is shown by† (or ^{ns}).

seedling growth, subsequent crop growth, and yield formation. The role of contour hedgerows in controlling soil temperature (at depths of 3 and 10 cm) was investigated by measuring it at different distances (15, 165, and 315 cm) from hedgerows. At 35 days after sowing when both the crop and the hedgerow canopies had developed, the soil temperature of CHIs was significantly lower than that of the SC control (Table 7.20). Mean soil temperature (averaged across different hedgerow tree species and depths) showed a significant increase with increasing distance from the hedgerow

TABLE 7.20

Variation of Mean Soil Temperature at 35 Days after Sowing in CHIs with Mung Bean and Different Hedgerow Species in the Subhumid Zone of Sri Lanka

Species	Mean Soil Temperature (°C)
Calliandra	28.1
Desmodium	27.8
Flemingia	27.0
Gliricidia	28.0
Tithonia	27.2
Senna	29.0
Control	31.0
$\mathrm{LSD}_{0.05}$	0.22

Note: Each value is the mean of two soil depths (3 and 10 cm) and three distances (15, 165, and 315 cm) from hedgerows.

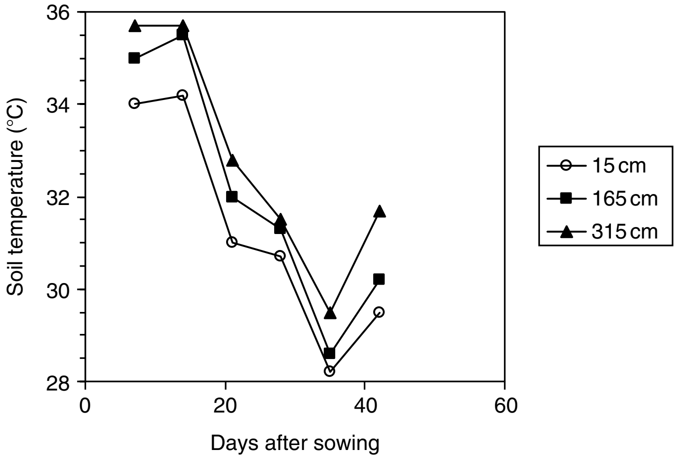

FIGURE 7.20 Variation of mean soil temperature (averaged across two depths, 3 and 10 cm, and seven CHI treatments) with time in CHIs with mung bean and different hedgerow species in the subhumid zone of Sri Lanka. (From De Costa, W.A.J.M. and A.G. Chandrapala, *J. Agron. Crop Sci.*, 184, 145, 2000. With permission.)

(Figure 7.20). Especially, the soil temperature near the hedgerow (i.e., at 15 cm) was significantly lower than that at distances of 165 and 315 cm. This showed that shading by the hedgerow canopies plays an important role in controlling soil temperature during dry periods.

7.8 TREE HEDGES VERSUS GRASS HEDGES

The results described earlier from all three case studies showed that contour hedgerows of tree or shrub species exerted significant resource competition on the associated annual (Case Studies 2 and 3) or perennial (Case Study 1) crops. It could be expected that contour hedgerows of grasses would exert less competition because of their possibly shallower root systems and lower biomass production than trees or shrubs. This hypothesis was tested in two experiments, one in the humid (wet) zone and the other in the subhumid (intermediate) zone.

The first was in the experiment described in Case Study 2, which started in 1997 with contour hedgerows of *Gliricidia sepium* and maize. In 2001, two additional treatments, that is, maize intercropped with contour hedgerows (spaced at exactly similar distances as those of *G. sepium*) of *Pennisetum purpureum* with (G_m) and without (G_n) grass prunings added as mulch, were included in this experiment. This enabled comparison of the impact of grass hedges with tree hedges of *Gliricidia*.

The second was at a site ~2 km away from that of Case Study 3, located within the same climatic zone and having the same soil and terrain characteristics. This experiment consisted of 4 year old, 4 m spaced, double contour hedgerows of *Vetiveria zizanioides* (called the Vetiver grass), *T. diversifolia,* and *G. sepium* with cowpea (*Vigna unguiculata*) grown in between. This experiment also had two subplot treatments as "mulched" (hedgerow prunings added to the plots as mulch) and "unmulched" (prunings not added) to determine the effect of mulching on subsequent crop growth and yield.

7.8.1 COMPARISON OF TREE HEDGES WITH GRASS HEDGES IN THE HUMID ZONE

In the humid zone, CHIs with mulched grass (*P. purpureum*) hedges (G_m) performed slightly better than the mulched tree (*G. sepium*) hedges (H_m), giving an 8% higher (in terms of cob dry weight) or 18% higher (in terms of cob fresh weight) maize yield than H_m (Table 7.21). Under unmulched conditions, maize under both grass (G_n) and tree (H_n) hedges showed yield reductions as compared

TABLE 7.21

Variation of Maize Yield in CHIs with Hedgerows of *Pennisetum purpureum* (*G*), *Gliricidia sepium* (*H*), and as Sole Crops (S) under Mulched (m) and Unmulched (n) Conditions on Sloping Terrain in the Humid Zone of Sri Lanka

Treatment	Cob Fresh Weight (kg ha^{-1})	Grain Dry Weight (kg ha^{-1})	Total Dry Weight at Harvest (kg ha^{-1})
G_m	5553.0	1773.0	2738.9
G_n	3834.5	1158.4	1954.7
H_m	4688.2	1644.3	2768.7
H_n	1474.3	409.7	964.0
S_m	6327.6	2156.5	3355.9
S_n	4332.6	1386.0	2479.2
$LSD_{0.05}$	394.7	106.2	122.6
CV (%)	11.9	10.3	12.7

with the control (i.e., unmulched SC, S_n), indicating that grass hedges also exerted significant resource competition on maize. However, as hypothesized at the beginning of the experiment, competition from grass hedges was much less than that from tree hedges, and under unmulched conditions, maize in G_n performed much better than that in H_n (Table 7.21).

Variation of maize yield across a transect between two adjacent hedgerows (Figure 7.21) showed that yield reductions closer to the hedgerows were much lower under grass hedges, especially under unmulched conditions (Figure 7.21a), than under tree hedges. Under unmulched conditions, while the middle rows of maize under *Pennisetum* achieved yields comparable with the SC yields, maize under *Gliricidia* showed significant yield reductions as compared with SCs even in the middle rows. However, under mulched conditions (Figure 7.21b), maize under *Gliricidia* showed much improved yields, especially in the middle rows.

To see whether the lower competition from grass hedges was possibly due to lower root growth than in *Gliricidia*, a detailed root sampling was carried out using soil cores before sowing of the maize crop in one season. Core samples were taken on both the upslope and downslope sides of the hedges, at 15 cm distances away from hedges up to 60 cm and at 15 cm depth intervals down to 60 cm. Surprisingly, the results revealed that at almost all depths and distances, grass hedges had greater root length densities than tree hedges (Table 7.22). This meant that the greater resource competition of *Gliricidia* hedges could be more due to its greater aboveground competition. Moreover, the grass hedges produced a significantly greater amount of pruned biomass than *Gliricidia* hedges (Table 7.23). This was because a greater amount of total biomass of the grass hedge was above the pruning height of 0.75 m. In contrast, *Gliricidia* hedges contained a significant proportion of its standing biomass below the pruning height. Therefore, *Gliricidia* hedges had to capture a greater amount of resources to produce a unit weight of pruned biomass than grass hedges. Conversely, a grass hedge would produce a greater amount of pruned biomass, which can subsequently be added to the soil, per unit of resources captured than a *Gliricidia* hedge.

7.8.2 COMPARISON OF TREE HEDGES WITH GRASS HEDGES IN THE SUBHUMID ZONE

In the subhumid zone also, grass hedges were superior to the tree hedges, with cowpea under *Vetiveria* yielding significantly greater than those under the two tree hedges (Table 7.24). The yield advantage under grass hedges was more pronounced under unmulched conditions. Crop growth at 50% flowering, as indicated by leaf area index and total dry weight, was only slightly better under grass hedges than under tree hedges. It is most likely that superiority of grass hedges over tree

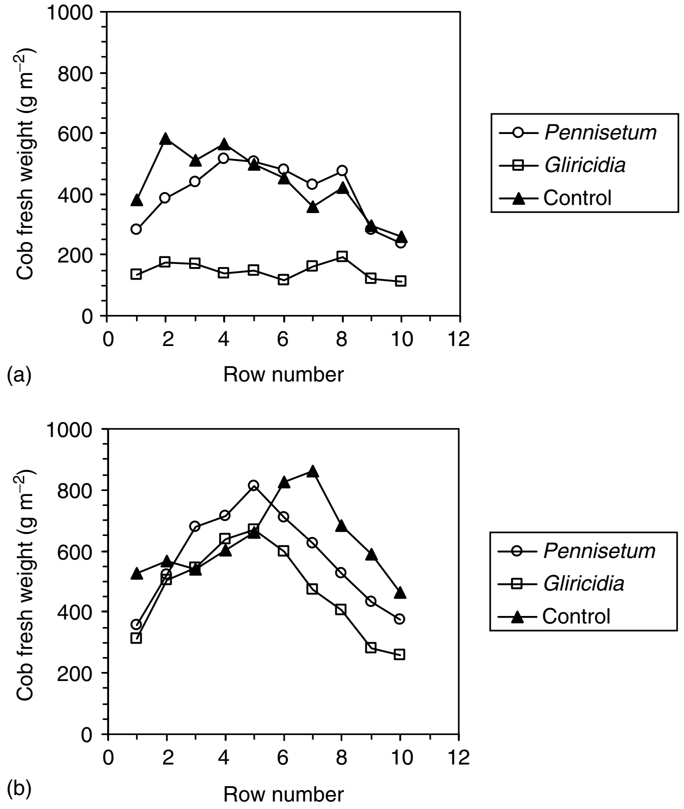

FIGURE 7.21 Variation of maize yield across transects between contour hedgerows of *Pennisetum purpureum* and *Gliricidia sepium* under unmulched (a) and mulched (b) conditions on sloping terrain in the humid zone of Sri Lanka. Row numbers start from the upslope side.

TABLE 7.22

Variation of Root Length Density (×10^3 cm cm^{-3}) of Contour Hedgerows of *Gliricidia sepium* and *Pennisetum purpureum* with Soil Depth and Distance from Hedgerows on the Upslope and Downslope Sides on Sloping Terrain in the Humid Zone of Sri Lanka

	Distance from Hedgerow on the Upslope Side (cm)					Distance from Hedgerow on the Downslope Side (cm)				
Depth (cm)	**60**	**45**	**30**	**15**	**5**	**5**	**15**	**30**	**45**	**60**
Gliricidia sepium										
15	177	97	236	138	191	127	144	126	144	98
30	123	113	85	138	219	93	50	69	118	73
45	62	67	73	135	92	48	35	74	65	45
60	48	78	72	120	100	107	55	32	32	46
Pennisetum purpureum										
15	190	309	256	401	347	282	274	246	274	93
30	157	223	144	337	283	103	177	156	95	94
45	218	131	198	191	173	107	102	90	114	61
60	116	128	49	110	133	74	61	82	100	32

TABLE 7.23

Biomass of Prunings Produced during a 100 Day Cropping Season by Contour Hedgerows of *Gliricidia sepium* (*H*) and *Pennisetum purpureum* (*G*) under Mulched (m) and Unmulched (n) Conditions on Sloping Terrain in the Humid Zone of Sri Lanka

	Leaf Dry Weight (mg ha^{-1})	Stem Dry Weight (mg ha^{-1})	Total Dry Weight (mg ha^{-1})
G_m	4.22	2.73	6.95
G_n	3.92	2.60	6.52
H_m	0.93	0.84	1.77
H_n	0.74	0.78	1.52
$LSD_{0.05}$	0.37	0.41	0.57
CV (%)	13.19	8.62	10.82

hedges emerged during the latter part of the crops' life cycle. This is understandable as the competition from tree hedges would have intensified during this period with increased growth of hedgerow canopies and root systems. Both growth and yield of cowpea responded positively to mulching. The highest yield response was to mulches of *Gliricidia*, while the highest in terms of growth was to mulches of *Vetiveria*.

Yield variation across transects (Figure 7.22) showed that although cowpea under grass hedges suffered yield reductions closer to hedgerows, the middle rows yielded either better than or similar to the SC control. Cowpea under grass hedges performed better than that under *Tithonia* in all rows across the transect and also better than that under *Gliricidia* in a majority of rows. Both whole plot and row-wise data on growth and yield showed that *Tithonia* exerted the highest competition followed by *Gliricidia* and *Vetiveria*. This is directly related to their respective biomass of prunings (Table 7.25), with *Tithonia* producing a substantially larger quantity. It would be interesting to

TABLE 7.24

Variation of Pod Dry Weight (*Y*), Total Plant Dry Weight (*W*), and Leaf Area Index (*L*) at 50% Flowering of Cowpea in CHIs with Tree and Grass Hedgerows under Mulched (M) and Unmulched (UM) in the Subhumid Zone of Sri Lanka

	Y (g m^{-2})		*W* (g m^{-2})		*L*	
Species	**M**	**UM**	**M**	**UM**	**M**	**UM**
Gliricidia	688 b*	587 c*	147 a*	105 b	3.00 a*	2.26 a*
Tithonia	367 c*	351 b*	159 a*	136 a*	2.38 b*	1.87 b
Vetiveria	791 a	756 a*	161 a*	118 b*	3.19 a*	1.76 b
Mean	615 A	565 B	156 A	120 B	2.86 A	1.96 B
Control	814		106		1.79	
CV (%)	7.22	6.91	5.42	8.01	5.87	7.34

Note: Means connected vertically by the same lowercase letter are not significantly different at $p = 0.05$. Means connected horizontally by the same uppercase letter are not significantly different at $p = 0.05$. Significant ($p = 0.05$) differences from the control treatment are shown by*.

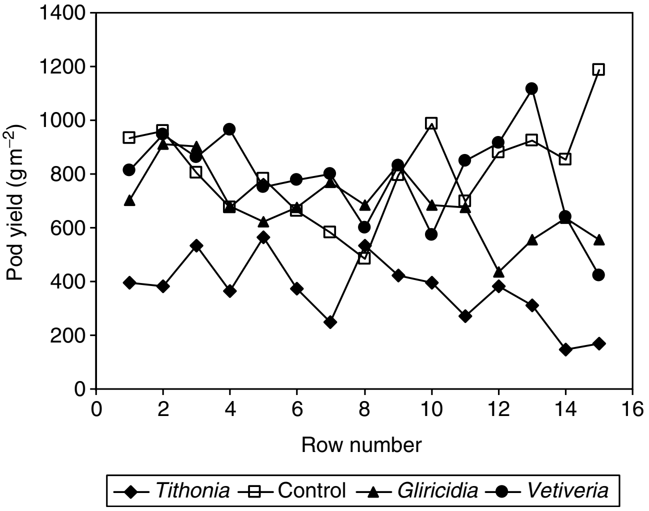

FIGURE 7.22 Variation of cowpea yield across transects between contour hedgerows of different tree and grass species under mulched conditions in the subhumid zone of Sri Lanka. Row numbers start from the upslope side.

follow the yield variation of cowpea under *Tithonia* to see whether the addition of such a large quantity of mulch and therefore nutrients would increase cowpea yields up to those of the other CHIs and the SC. It can be noted that unlike in the humid zone, in the subhumid zone with less soil water available, grass hedges produced significantly less-pruned biomass than tree hedges.

7.9 DISCUSSION AND SYNOPSIS

As mentioned earlier in the introductions to case studies, CHI has been introduced in Sri Lanka to cropping systems where considerable soil erosion and loss of fertility have already taken place. It is an imperative need that crop production is sustained in these lands. In the case of tea plantations, sustainable tea production is vital to the entire economy and a considerable fraction of the population that is dependent on the tea industry. In the case of lands where seasonal annual crops are grown, a large number of subsistence farmer families depend on sustained crop production for their livelihood. A synopsis of results of all three case studies show that CHI as a means of ensuring sustainable crop production in these lands presents the growers with a "double-edged sword."

TABLE 7.25

Biomass of Prunings Produced during an 80 Day Cropping Season by Contour Hedgerows of Different Tree and Grass Species in the Subhumid Zone of Sri Lanka

	Fresh Weight (Mg ha^{-1})	Dry Weight (Mg ha^{-1})
Gliricidia sepium	27.92	7.05
Tithonia diversifolia	68.79	20.12
Vetiveria zizanioides	6.97	2.59
$LSD_{0.05}$	2.43	1.1
CV (%)	11.60	7.66

If used improperly (e.g., too closely spaced hedgerows, export of hedgerow prunings from the system without using it as mulch, too infrequent pruning of hedgerows, etc.), it could result in accelerated decreases of crop yields because of excessive resource competition. On the other hand, if properly practiced, CHI has demonstrated, in these case studies, its capability of regenerating soil fertility on a longer time scale and sustaining crop yields in these highly fragile and degraded lands.

Results of these studies leaves us in no doubt that almost all species used in contour hedgerows, whether tree, shrub, or grass, compete with the crop for essential growth resources. However, results also show that it is possible to select species with lower resource competition. They also demonstrate that it is possible to maximize the positive effects of contour hedgerows to an extent that the negative effects of competition are outweighed. First, properly established and maintained contour hedgerows reduce soil erosion and surface runoff significantly and thereby slow down the whole process that leads to final land degradation. Second, by adding hedgerow prunings as mulch, the processes of regenerating soil fertility are set in motion. Evidence that this rebuilding process has started was shown in the present studies, with observations of reduced soil BD, increased SOM and CEC, and increased nutrient recycling. However, it should be realized that regeneration of soil fertility, which has been lost over a period of several decades, through CHI is a very slow process. During these intervening years, the system has to be managed with patience and adequate care with proper maintenance of hedgerows, filling of any gaps that occur due to tree death, application of hedgerow prunings as mulch to the crop, and minimum disturbance of soil during crop establishment. During this period of rebuilding soil fertility, yield reductions due to competition could be minimized by practices such as trenching and judicious application of inorganic fertilizer. As shown in the case of tea, cost of fertilizer can be reduced by taking into account the nutrients provided by hedgerow prunings.

These studies also demonstrated the possibility that grass hedges could exert less competition than tree or shrub hedges and thereby provide higher yields in the associated crop. However, the capacity of grass hedges for long-term regeneration of soil fertility have to be studied before deciding whether to replace tree hedges with grass hedges. Perhaps, grass hedges could be recommended for lands that are only slightly degraded or have been rehabilitated for a considerable period with tree hedges.

In addition to the agronomic aspects, socio-economic aspects of CHI have to be given due attention to ensure its continued adoption by farmers in these fragile ecosystems. Demonstration plots, extension, and training programs coupled with moderate incentive packages and close monitoring of progress during the period of regenerating soil fertility are of crucial importance in this regard.

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Part II

The Belowground Ecology

8 Belowground Interactions in Tree–Crop Agroforestry: Need for a New Approach

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CONTENTS

8.1 INTRODUCTION

During the past 30 years, agroforestry has been developed and promoted as a means to combat rural poverty and increase food security while conserving the natural resource base in the tropics. Accordingly, most agroforestry research has focused on practices involving trees and staple food crops such as maize (*Zea mays*), rice (*Oryza sativa*), beans (*Phaseolus* spp., *Vigna* spp.), and cassava (*Manihot esculenta*). In a tropical climate, annual crop production faces certain difficulties, such as rapid soil organic matter loss on soil exposure and tillage with corresponding deterioration of the soil structure; nutrient leaching especially at the onset of the rainy season when crop root systems are still poorly developed; and a risk of soil erosion in mountainous areas. Agroforestry practices that specifically address these problems have been developed, including mulch-production systems, contour hedgerows, and improved fallow systems (Young, 1997). Correspondingly, the most influential concepts and theories in agroforestry research explicitly or implicitly address systems with annual crops. They include the synchrony hypothesis, according to which nutrient use is more efficient if nutrient sources are applied and managed so that temporal patterns of nutrient release and uptake coincide (Heal et al., 1997), and the safety-net hypothesis, which postulates deep-rooting trees that capture leached nutrients from the soil beneath shallow-rooted (annual) crops (van Noordwijk et al., 1996).

When studying associations of trees and annual crops, agroforestry researchers and practitioners soon noticed the crucial importance of below-ground interactions. Especially in dry climates and on nutrient-poor and shallow soils, the presence of trees in crop fields implies strong trade-offs between beneficial effects of trees on soils and competition with crops for soil resources. Motivated by this insight, a considerable number of studies addressed the problem of root competition between trees and annual crops, focusing on the identification of incompetitive tree species (Schroth and Lehmann, 1995; Ong et al., 1999), effects of tillage, biomass application (Schroth et al., 1995), root pruning (Korwar and Radder, 1994), and shoot pruning (van Noordwijk and Purnomosidhi, 1995) among other factors. Failure to identify satisfactory solutions to the soil improvement versus competition trade-off led to a certain shift in the focus of agroforestry research from tree–crop associations to improved fallow systems, where trees are grown in rotation with annual food crops, during the past 10 years (Sanchez, 1995).

While tropical farmers usually depend on annual crops for their subsistence, they often depend on tree crops for monetary income. Tree crops such as coffee (*Coffea* sp.), cocoa (*Theobroma cacao*), and rubber (*Hevea brasiliensis*) constitute the backbone of rural economies in large parts of the humid tropics. These crops are often grown by tropical smallholders together with other planted and spontaneous trees in highly diversified systems, which have recently drawn much research attention for their potential to provide environmental benefits such as biodiversity conservation, watershed protection, and maintenance of carbon stocks (Michon and de Foresta, 1999). However, attempts to conserve, promote, and intensify such diversified tree crop-based systems because of their desirable environmental and socioeconomic properties are facing a critical lack of knowledge of above- and especially belowground interactions within such systems. Researchers are confronted with questions such as: How will cocoa interact with shade trees in a drier climate? Can cloned rubber trees, if introduced into rubber agroforests, support the same level of competition from spontaneous forest regrowth as the traditional seedling rubber? Will fruit trees when introduced into extensively managed rubber or cocoa agroforests for their diversification, tolerate these conditions and be productive? How will the substitution of traditional legume shade trees by fast-growing timber trees impact on coffee production under different pedoclimatic situations (Beer et al., 1998)? There is presently only a small amount of information from research in tree crop agroforestry systems on which answers to such questions could be based (Schroth et al., 2001; Ong et al., 2003). It is therefore tempting to rely on the much broader knowledge base from agroforestry systems with annual crops when addressing such problems. But is it justified?

In this chapter, we review some aspects of the belowground ecology of agroforestry systems with tree crops. We start by pointing out the fundamental differences between agroforestry systems based on annual crops and those based on tree crops. We then introduce some common types of tree crop agroforestry systems and point out the implications for the management of belowground interactions. Subsequently, we review root interactions in systems with fast-growing timber trees and introduce the concept of self-organization of interacting root systems. Finally, we present some recent results about the possibility to manipulate tree root distribution with biological means and show potential applications in tree crop agroforestry.

8.2 ANNUAL CROP AND TREE CROP AGROFORESTRY—TWO DIFFERENT STORIES

Since much more research has been carried out on the belowground ecology of agroforestry systems with annual crops than those with tree crops (van Noordwijk et al., 1996; Schroth, 1999), it is logical to ask whether insights gained in such research can be transferred from one type of system to the other. However, there are profound differences between these types of systems, which would have to be taken into account in such an approach (Table 8.1):

1. While temporal dynamics of the soil occupation by root systems are important in systems both with annual and with tree crops, they occur on different temporal scales and require different management interventions. Systems with annual crops are characterized by a pronounced *seasonal variability* of root development (Schroth and Zech, 1995a), hence the importance of synchronizing nutrient supply with demand. In tree crop-based systems, in contrast, this variability and thus the importance of synchrony are much lower and may even be negligible in climates with no pronounced seasonality (Schaller et al., 2003). Seasonal differences in root growth and activity may, however, also occur in tree crops as a consequence of weather and phenological rhythms and may provide opportunities for

TABLE 8.1
Important Differences between the Root Ecologies of Annual Crop (ac) and Tree Crop (tc) Based Land Use Systems

Characteristic	Tree Crop-Based Practices	Annual Crop-Based Practices	Consequences for Belowground Interactions and Management
Temporal variability of soil occupation by roots	Over several years	Seasonal	Synchrony of resource availability and demand important for ac but less so for tc; successional sequence of crops with different growth rates important strategy for establishing tc
Spatial variability of soil occupation by roots	High	Low	Synlocation of resource availability and demand important in tc but less so in ac; intercropping important strategy for ensuring soil occupation during early tc development
Soil disturbance by tillage	No	Usually yes	Disturbance of root systems of weeds and other vegetation creates competition-free space for ac at the beginning of the cropping season; root systems of tc develop undisturbed over many years
Shade tolerance	Yes for some important species	No	Shade trees usually intimately mixed with tc such as coffee, cocoa, and tea so that root interactions are intensive; intimate mixtures of ac with trees usually unfavorable and spatially zoned or rotational designs more common
Nutrient and water uptake	Gradual, distributed over large part of the year and large soil volume	Temporally fast, spatially concentrated	Once established, tc probably less affected by root competition than ac; wide spacing necessary for tc if no access to deep water reserves during dry season
Rooting depth	Often deep	Often shallow	Tc often not dependent on other trees for deep nutrient uptake and less sensitive to competition in the topsoil than ac

targeted fertilizer application if peaks of root activity of one species coincide with troughs of root activity of another species (Muñoz and Beer, 2001). In contrast to systems based on annual crops, tree crop-based agroforestry systems are characterized by a pronounced *successional dynamic* of soil occupation by root systems. This results from the progressive establishment of young tree crops over several years; these are therefore commonly associated initially with annual and short-living perennial crops to make use of the space and soil resources not yet exploited by tree roots (e.g., Budelman and Zander, 1990; Gouyon et al., 1993).

2. Whereas the short-term *temporal variability* of the soil occupation by root systems is much smaller in systems based on tree crops than in those based on annual crops, its *spatial variability* is much greater. Annual crops may colonize the whole topsoil with a dense network of roots within weeks or months of their germination, whereas tree crops may need several years for a more or less homogeneous occupation of the available soil volume, or may not reach this situation at all. For example, in a 15 year old oil palm (*Elaeis guineensis*) plantation in Amazonia, the palms, which were planted at 9 m by 9 m triangular spacing (143 trees ha^{-1}) as recommended for monocultures, had not developed sufficiently extensive root systems to prevent nitrate leaching halfway between neighboring palms, whereas the nitrate in the soil close to the palms was effectively taken up

(Schroth et al., 2000). This pronounced spatial variability of root activity makes *synlocation* of nutrient sources with zones of high root activity a much more important strategy in agroforestry systems with tree crops than it is in systems with annual crops (i.e., placement of mulch and fertilizer close to the trees). This spatial variability also stresses again the interest of associating crop species with different growth and production cycles during the establishment phase of tree crop systems to make optimum use of available soil resources, minimize nutrient losses, and generate early returns to investments (Schroth et al., 2001).

3. Regular soil disturbance through tillage is a characteristic of most annual cropping systems. In agroforestry systems, it temporally creates a competition-free space for the germinating crops in the topsoil by destroying tree roots. Although it is not known for how long this effect lasts, it certainly gives a temporary advantage to the crop roots (Schroth et al., 1995). In agroforestry systems with tree crops, soil disturbance is neither possible (because of the damage to the tree crop roots) nor desirable. Root interactions between system components therefore develop more under conditions of a dynamic equilibrium, which may, however, be periodically disturbed by the dieback of root systems induced by shoot pruning of certain trees (Nygren and Campos, 1995; Schroth and Zech, 1995b).
4. Annual crops are generally shade sensitive, and systems where annual crops are intimately associated with trees are therefore usually less desirable than systems characterized by a certain degree of spatial zoning, such as the planting of trees and crops in alternating strips, or even the temporal separation of trees and crops, as in fallow rotations. These designs also limit the intensity of belowground interactions. In contrast, in systems with shade-tolerant crops such as coffee, cocoa, and tea (*Camellia sinensis*), the creation of a certain degree of shade and a pest- and disease-suppressive, protected microclimate are often the very reasons for the presence of trees, which are therefore intimately intermingled with the crops (Beer et al., 1998; Guharay et al., 1999). Alternative designs such as box plots, where the trees are planted on the boundary of blocks of tree crop plants have, however, also been tested and may be an option under conditions where more intimate mixtures would lead to root competition during the dry season (Foster and Wood, 1963).
5. Nutrient uptake by annual crops occurs relatively rapidly during a short time interval, whereas that of tree crops is more evenly distributed over a longer interval and occurs from a larger soil volume. It is therefore likely that tree crops, once they are established, are less sensitive to root competition for nutrients and water than annual crops, although tree crop seedlings are usually very sensitive to competition. Tree crops which have no access to deep water sources may be sensitive to competition for water in the topsoil and require wide spacing to avoid drought stress during the dry season. Annual crops can then be interspersed with the tree crops during the rainy season to make use of surplus water (Daniel et al., 1996).
6. Although the root systems of annual crops often remain relatively shallow during the limited time available for their development, tree crops can have very deep root systems. For example, tea bushes in deep soils in East Africa may have roots to 5–6 m depth and may wilt later than associated shade trees, suggesting that the tea plants have access to deeper water resources than the trees (Willey, 1975). Similarly, arabica coffee can root more than 4.5 m deep (Webster and Wilson, 1980) and had a more homogeneous vertical root distribution in the top 40 cm of an Andisol in Costa Rica than *Eucalyptus deglupta* trees which were associated with them (Schaller et al., 2003). This suggests that tree crops may often not depend on trees for intercepting leached nutrients and recycling nutrients from the subsoil, and may also be less sensitive to competition from other species in the topsoil than annual crops.

In summary, the possibility of transferring research results on belowground interactions from agroforestry systems with annual crops to systems with tree crops is limited due to a number of fundamental differences between these system types. Furthermore, even the application of well-established principles and basic hypotheses from annual crop agroforestry, such as the synchrony and safety-net hypotheses, have only restricted applicability in tree crop-based agroforestry systems.

8.3 BELOWGROUND INTERACTIONS IN DIFFERENT TYPES OF TREE CROP AGROFORESTRY

There are many different types of agroforestry systems with tree crops. Some are very simple systems, consisting of one tree crop species such as coffee and one shade tree species, whereas others may contain a variety of planted tree crops and spontaneously grown trees. In addition, the intensity of management, including weeding and fertilization, differs widely between different tree crop-based systems. Species composition and management are likely to be important factors determining belowground interactions in a system and how they are perceived and managed by farmers.

1. *Shaded tree crop systems* usually consist of only one shade-tolerant tree crop species, such as coffee, cocoa, or tea, which dominates the system economically and forms most of its understory (Beer et al., 1998). The diversity of the shade canopy may vary from very low, in the case of a single planted-tree species, to high, if the canopy is formed by trees retained from the previous forest canopy, a traditional practice in cocoa systems in West Africa and Brazil (de Rouw, 1987; Johns, 1999). The shade canopy may contain economically valuable tree species, which may be either planted or spontaneous. As the understory tree crops are the economic backbone of the system, the other species are selected, thinned, pruned, and otherwise managed to their benefit. This is most obvious for leguminous service trees with little or no economic value, such as *Erythrina* spp. and *Inga* spp., which are widely used as coffee and cocoa shade in Latin America, but can also be observed with timber trees. A farm survey in San Isidro, Costa Rica, showed that, despite a poorly developed market and low value of its wood, farmers preferred *E. deglupta* as shade tree compared to other timber trees because of its fast growth (and thus shade establishment), the light shade cast by the small leaves which is considered ideal for coffee, and the reduced pruning requirements compared with the traditional leguminous shade tree species (Tavares et al., 1999). The shade cast by other timber species such as *Terminalia* spp. and *Gmelina arborea* was considered by farmers as too dense for coffee. This suggests that the selection of timber tree species as coffee shade by the farmers was more influenced by considerations related to the production of coffee than to the production of timber. To which extent belowground interactions are taken into consideration in the selection of shade tree species by farmers is less clear but they are most likely to influence farmers' decisions on whether and which shade trees to retain in dry regions (Jiménez and Alfaro, 1999; see below). Although farmers will generally tend to design and manage their shade canopy in a way considered most beneficial for the understory tree crop, certain very valuable shade species may be tolerated in a plantation even if they are severe competitors with the understory crop, as is the case with oil palm as cocoa shade in West Africa (de Rouw, 1987). This situation leads over to the following type of tree crop agroforestry system.
2. *Homegardens* are composed of a number of valuable tree species of different size and growth form (Fernandes and Nair, 1986; Torquebiau, 1992). The different species are arranged according to their respective needs within a mosaic of niches which they both create and to which they respond. As these are intensively managed systems, it is likely

that aboveground interactions between species are closely observed and managed if required, for example, through pruning of branches if shading becomes too intensive, although this is not well documented. Almost nothing is known about the perception and management of belowground interactions in such systems. The age of a system at which belowground interactions between neighboring tree crop plants start depends on their initial spacing and lateral root development. With increasing age, root interactions should become more intensive and may complicate the management of a system. Kummerow and Ribeiro (1982) found that in a mixed plantation of cocoa and rubber in Bahia, Brazil, the fine root mass of rubber in the top 15 cm of soil directly under the cocoa trees was twice as high as that of cocoa, and suggested that this strong intermingling of root systems made targeted fertilizer application to the cocoa trees difficult. If negative belowground interactions occur and are perceived as such in homegardens, their mitigation is much more complicated than in shaded tree crop systems, because instead of manipulating or even removing all other species at the benefit of a single, economically dominant tree crop, the requirements of several species have to be balanced against each other, taking their respective present and future economic value into account. Although we have virtually no information about root interactions and their management in homegardens, it is likely that the application of manure, household refuse, and other nutrient sources plays an important role in the mitigation of nutrient competition between species.

3. *Agroforests* are composed of a single or several planted tree crop species and a large amount of spontaneous forest regrowth, which is a consequence of extensive management. Agroforests are essentially economically enriched secondary forests (Michon and de Foresta, 1999). To which degree they are ecologically impoverished compared with spontaneous secondary forests is not well known and is likely to depend on the specific site and management history. In Indonesian rubber agroforests, rubber trees may reach a sufficient size for tapping with a delay of several years compared to clean-weeded plantations (Gouyon et al., 1993), suggesting competition between young rubber trees and forest regrowth. Similar practices exist in the Amazon (Schroth et al., 2003). Similarly, tree crops such as cupuaçu (*Theobroma grandiflorum*) may start to produce fruit with a delay of several years if they are planted into an annual crop but are then not further weeded after the annual crop has been harvested until the trees start flowering, which is a common practice in the Amazon (Sousa et al., 1999). In these cases, negative interactions between tree crops and forest regrowth are tolerated because, within the planning horizon of the farmer, the effort for reducing them by more intensive weeding is not justified by the expected gain from earlier maturing of the tree crops. Also, tree shade may reduce the growth of more noxious species such as *Imperata* grass (Williams et al., 2001). Agroforests have obviously developed under conditions where agricultural activities were more limited by the availability of labor than by that of land, and this has important consequences for the conservation of these relatively diverse systems under conditions of increased population pressure.

These brief accounts testify for the wide range of conditions under which tree crops are grown in tropical agroforestry systems. These influence the intensity with which interactions between system components are managed and the options which exist for their management. These options range from intensive management of the whole system at the benefit of a single, economically dominant species, through the balancing of requirements and tolerances of different valuable species, to a partial or sometimes even total tolerance of (negative) interactions under socioeconomic conditions that favor extensive management.

8.4 ROOT INTERACTIONS BETWEEN TREES: COMPETITION AND SELF-ORGANIZATION

As mentioned earlier, root interactions between trees and tree crops within a mixed plantation should become more intensive, but also more diffuse as trees and their root systems become larger and the youngest roots grow further away from the individual plants (Schroth, 1999). It is not possible to make efficient use of a piece of crop land without permitting a certain amount of root interaction (and competition) between the plants. However, besides being unavoidable, root competition is not a purely negative phenomenon in mixed species plantations but also has important regulatory functions with respect to the spatial and temporal exploration of the soil volume by the root systems (Schroth et al., 2001). If root systems compete with each other in the topsoil, they may grow deeper and make better use of subsoil water and nutrient resources, thereby also acquiring greater resistance to drought (Eastham et al., 1990). Where deep- and shallow-rooted plants are associated with each other, the shallow-rooted species may only profit from nutrients recycled by the deep-rooted ones if they compete effectively with them in the topsoil for nutrients released from litter, whereas this competition may be the very incentive for the other species to form deep roots (Schroth et al., 2001). In other words, nutrient uptake from the subsoil by deep-rooted trees may actually only occur and be beneficial for associated shallow-rooted crops if there is root competition in the topsoil. Similarly, nitrogen-fixing tree species may only continue fixing nitrogen over many years if the nitrogen released by them is effectively removed by nonfixing species and does not accumulate in the soil. Interactions between plant species could also accentuate differences in phenology, including temporal dynamics of root growth and activity, thus extending phases of high water and nutrient uptake over a longer time period and making more efficient use of water and nutrients in the soil (Schroth et al., 2001).

The regulatory effect of competition on the spatial exploration of the soil by root systems in mixed plantations is well illustrated by a coffee plantation shaded by *E. deglupta* trees in a farmer's field in Costa Rica, which was studied by Schaller et al. (2003) to find out what allowed such a fast-growing and competitive tree species to be associated with coffee apparently without negative effects on coffee yields. Despite the high planting density of the coffee (5000 plants ha^{-1}), the soil was not homogeneously occupied by the coffee roots, which were mostly concentrated in the proximity of the coffee rows, whereas the tree roots spread preferentially in the interrow spaces (Figure 8.1). Through this small-scale partitioning of the soil, which was obviously the result of root interactions, spatial overlap between the root systems was reduced. Interestingly, the total root length density was very similar in all positions (2.5–2.7 cm cm^{-3}), suggesting that in a self-organizing process, without any specific management intervention, root interactions between the two species had led to a homogeneous exploration of the soil by the combined root systems. Together with the high nutrient and water availability in the soil at this site, the spatial division of the soil space enforced by the relatively competitive coffee roots was seen as explanation for the successful use of very fast-growing shade trees in coffee in Costa Rica, and possibly other tropical regions such as Indonesia where coffee is shaded by the very fast-growing *Paraserianthes falcataria* (Schaller et al., 2003). The observed rooting patterns also provided a scientific justification for the farmers' practice to apply fertilizer for the coffee along the coffee rows and not to broadcast between the rows, which had previously seemed questionable because of the close spacing of the coffee bushes.

Such self-organizing processes make it difficult if not impossible to predict rooting patterns in mixed-species associations from the known patterns of the individual species grown in isolation. They are not restricted to interactions between different tree species, but also occur if root systems of individuals of the same species interact. Such reactions result in reduced root overlap and increased exploration of soil parcels where rooting densities are still low, including in the subsoil (Atkinson et al., 1976; Eastham and Rose, 1990). They should increase the efficiency with which available soil

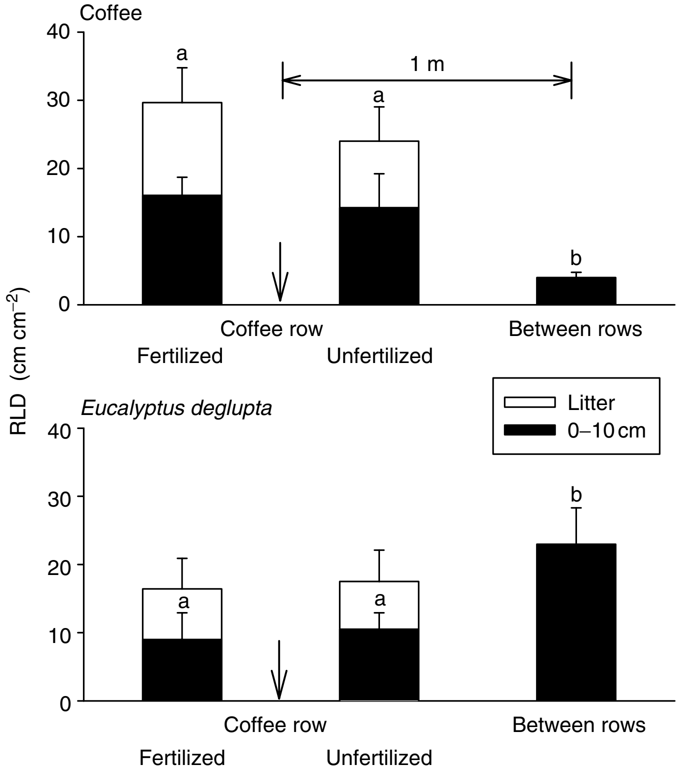


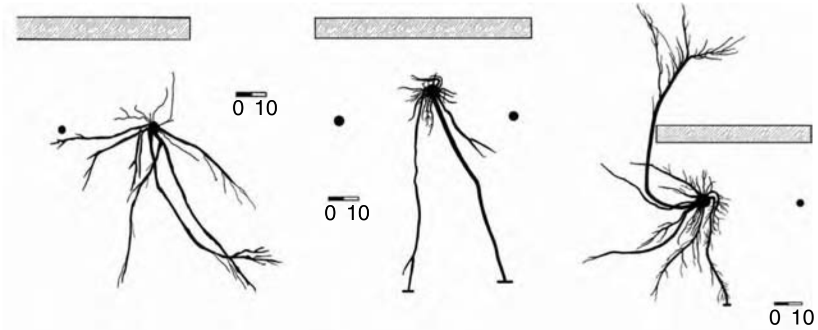
FIGURE 8.1 Fine root length density (RLD; cm cm^{-2} ground area; d < 2 mm) of coffee (*Coffea arabica*) and *Eucalyptus deglupta* shade trees at different positions between rows of coffee spaced 2 m at Juan Viñas, Costa Rica (means and standard errors). Different letters indicate significant differences at $p < 0.05$. (From Schaller, M., G. Schroth, J. Beer and F. Jiménez, *Forest Ecol. Manag*., 175, 205, 2003. With permission from Elsevier.)

resources are used, reduce nutrient losses by leaching, and delay the occurrence of growth and yield losses caused by competition.

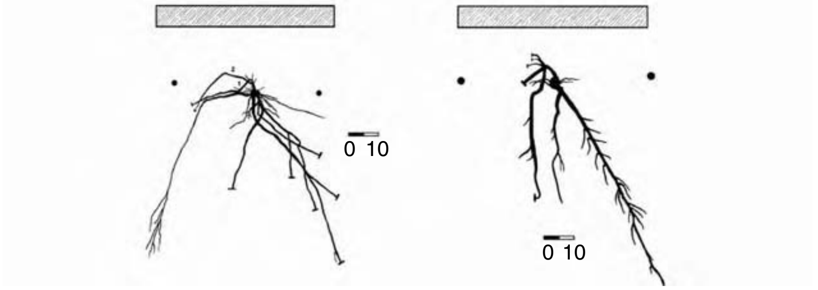
However, competition will occur if the combined requirements for water and nutrients of associated plants exceed the amounts available in the soil for prolonged periods of time. In the central valley of Costa Rica, where the dry season is longer than at the study site of Schaller et al. (2003), Jiménez and Alfaro (1999) measured lower soil water contents and observed symptoms of drought stress of coffee shaded by *E. deglupta* in comparison to unshaded coffee and coffee shaded by the traditional *Erythrina poeppigiana* trees, suggesting that the aforementioned processes of complementary exploration of the soil were no longer sufficient to protect the coffee plants from tree root competition and that other, less fast-growing and competitive shade species were needed.

8.5 MANIPULATING TREE ROOT DISTRIBUTION BY BIOLOGICAL ROOT PRUNING

On sites with a pronounced dry season or infertile soils, or where very fast-growing and competitive tree species are used in association with tree crops, it may be advantageous to design systems in a way that root interactions between trees and crops are reduced. As discussed earlier, the options for this are fewer in systems with tree crops than in systems with annual crops, because the tree shade is often considered necessary for the tree crops and disturbance of the tree root systems through tillage is not an option. Root systems of trees may respond to the presence of competing root systems of herbaceous plants with reduced lateral and increased vertical extension, as shown in a classic study by Yocum (1937) for apple trees (*Malus domestica*) associated with maize (see also Schroth, 1999). Grass strips planted on the contour are a recommended soil conservation measure in sloping areas, such as those widely used for coffee plantations in Central America. Schaller (2001) hypothesized that these strips could simultaneously be used for manipulating the lateral root spread of timber tree species planted in rows on the contour, instead of evenly distributed in a plantation, and bordered on both the upper and lower sides by strips of grasses. In a screening experiment, strips of five different grass species were planted on one side along rows of *Cordia alliodora* seedlings to identify the most promising species for subsequent field experimentation. At the age of 8 months, when the trees in the control treatment without grasses were 2.6 m high, all five grass species had caused pronounced deformations of the tree root systems, with the most-pronounced reactions caused by guinea grass (*Panicum maximum*), brachiaria (*Brachiaria brizantha*), and sugar cane (*Saccharum officinarum*), and less-pronounced reactions by vetiver (*Vetiveria zizanioides*) and citronella grass (*Cymbopogon nardus*, Figure 8.2). Tree roots growing in the direction of the grasses either remained much shorter and thinner than those growing in the opposite direction or they changed direction before reaching the grasses. Such abrupt changes in growth direction were rarely observed in the absence of grass strips. The avoidance reaction of the tree roots to the grass root systems resulted in their effective exclusion from the soil beyond the grass strips, suggesting an effect of "biological root pruning." Such effects were only rarely observed when seedlings of *E. deglupta* were exposed to the effect of the same grasses, presumably because of the faster growth and ability of their superficial roots to respond opportunistically to weak points within the grass barriers (Schaller, 2001).



Panicum maximum *Saccharum officinale* *Brachiaria brizantha*



Cymbopogon nardus *Vetiveria zizanioides*

FIGURE 8.2 Root system of 8 month old *Cordia alliodora* saplings as influenced by strips of different grass species, planted at 30 cm from the trees, in Turrialba, Costa Rica. *Arrows* indicate the direction of the tree line in the case of border trees. (Modified from Schaller, M., *Quantification and Management of Root Interactions between Fast-Growing Timber Tree Species and Coffee in Plantations in Central America*. Doctoral Thesis, University of Bayreuth, Bayreuth, 2001.)

While these results show a surprisingly strong effect of grass strips on *Cordia alliodora* seedlings, indicating a potential to manipulate the root distribution of trees at an early development stage with relatively simple means, they also make clear that no generalization for other tree species is allowed. Furthermore, it is not yet clear to what extent the ‘‘pruning’’ of the seedling root systems translates into an altered root architecture of older trees, what the consequences for root interactions within a system would be, and whether the costs incurred by planting the grass strips, including competitive effects on trees and neighboring crop rows, are outweighed by benefits arising from soil conservation and reduced root interactions between trees and crops.

8.6 CONCLUSIONS

Much less is known about belowground interactions in agroforestry systems based on tree crops than in systems based on annual crops. Because of the numerous fundamental differences between these types of agroforestry systems, it is difficult to apply research results obtained in one system type to the other system type. Rather, a whole new set of concepts may be necessary for a deeper analysis of belowground processes of agroforestry systems involving tree crops. Based on a differentiation of tree crop agroforestry systems according to their composition and management intensity, which has implications for the perception and management of above- and belowground interactions between system components, we propose the concept of self-organization of belowground interactions and offer an approach to the manipulation of tree root distribution and root interactions with simple, biological means. We see these in no way as definite results, but rather propose them as guides and inspirations for further research.

ACKNOWLEDGMENTS

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9 Tree–Grass Interactions and Water Use in Silvopastoral Systems in N.W. Patagonia

*Javier E. Gyenge, María Elena Fernández,
and Tomás M. Schlichter*

CONTENTS

9.1 INTRODUCTION

Balance between facilitation and competition interactions in plants changes with species characteristics and environmental conditions (Callaway and Walker, 1997; Holmgren et al., 1997). In natural ecosystems, such as savannas, shrublands, or salt marshes, facilitation effects have been reported as a frequent interaction, particularly in stressful environments (Belsky, 1994; Pugnaire and Luque, 2001; Bertness and Ewanchuk, 2002), as in dry years within a site (Frost and McDougald, 1989; Bertness and Ewanchuk, 2002). All these findings described for natural plant associations may suggest that the same balances in artificial agroecosystems, such as agroforestry systems, may be expected. However, Ong and Leakey (1999) have pointed out that agroforestry systems behave in a different way from savanna ecosystems in spite of being composed of both trees and grasses. These authors suggested that high density of trees in agroforests increases their negative effects over grasses or crops due to rainfall and radiation interception, and competition for soil water. Thus, negative effects may be stronger than beneficial ones, such as decrease in evaporative demand.

However, mimic ecological interaction patterns of natural ecosystems under certain conditions may be possible, for example, if the selected tree and forage species are complementary in soil water use (due to their different root distributions and phenology). Kho (2000a, 2000b) proposed that agroforestry technologies may be able to improve site productivity in temperate climates, in situations in which resources other than radiation are limiting (e.g., dry areas). In this case, it is also expected that species more vulnerable to water stress may take advantage of facilitation effects produced by the presence of other plants differently than stress-tolerant species. This could result in differences in the nature and strength of biological interactions even in the same site and under the same environmental conditions.

N.W. Patagonia, Argentina, has a Mediterranean-type climate, considering precipitation distribution, with wet–cold winters and dry–hot summers; thus, water is the most limiting resource for plant productivity (Jobbágy et al., 2002). The region is also characterized for its West–East precipitation gradient (differences of more than 1000 mm in 60 km, Jobbágy et al., 2002). To the west, there are forests dominated by *Nothofagus* spp. whereas the Patagonian steppe occupies the east portion of the gradient. The ecotone between both ecosystem types is occupied by open forests of the native conifer *Austrocedrus chilensis* as well as native grasslands. In this portion of the gradient (between 600 and 900 mm of mean annual precipitation) afforestation with exotic fast growing conifers appears as a promising productive activity (see below). The traditional economic activity in Patagonian steppe and the forest–steppe ecotone is sheep or cattle raising based on the use of natural grasslands. Presently the sustainability of these production systems is threatened by desertification, with its negative effects on pasture quality and quantity. During the last few years, forest plantations have been promoted through a subsidy policy in the country. In Patagonia, the exotic *Pinus ponderosa* Doug. ex Laws (ponderosa pine) is the most commonly planted species. Considering cultural, economic, and environmental aspects, silvopastoral systems may be an interesting alternative for small and medium landowners in semiarid Patagonia. In addition, from a scientific point of view, the introduction of this deep-rooting tree in these ecosystems could lead to new ecological interactions.

Based on this background, the development of silvopastoral systems including ponderosa pine and native forage species began to be studied at the end of the 1990s. In particular, two main goals were pursued: (1) to quantify water use of different land managements (natural grasslands and silvopastoral systems with different tree density) and (2) to understand tree–grass interactions (competition, facilitation, and the net balance) and their influence on forage growth.

9.2 STUDY SITE AND TRIAL DESCRIPTION

The study was carried out in silvopastoral plots installed in Estancia Lemú Cuyén, (40.3°S, 71.1°W), in Lanín National Park, Patagonia, Argentina. Average annual rainfall (period 1978–1999) is 684 ± 283.1 mm (with ~579 mm in fall–winter and 105 mm in spring–summer). Maximum and minimum annual average temperatures are 17.1°C ± 0.5 and 4°C ± 2.1, respectively.

The experiment included two ponderosa pine densities and an open grassland area (control). Five plots of 1600 m^2 each with 350 pruned pines ha^{-1} (350 P) and five plots with 500 pruned pines ha^{-1} (500 P) were installed in 1999 when trees were 15 years old (see Table 9.1 for tree canopy cover level in each treatment and growing season). Within the plots with trees, tussocks of two native grass species were measured: *Festuca pallescens* and *Stipa speciosa*, which were located in two situations, under (UC) and between tree crowns (half distance from two tree trunks, BTC). Both species differ in drought resistance, *F. pallescens* being the most vulnerable to water deficits, as was indicated by physiological measurements (Fernández et al., 2002; Fernández, 2003) and its natural spatial distribution in sites with better water balances than those occupied by *S. speciosa*.

TABLE 9.1
Tree Canopy Cover—Mean (Standard Deviation)—Measured with a Spherical Densitometer in Each Forested Treatment and Growing Season

	350 UC	350 BTC	500 UC	500 BTC
1999–2000	41.8 (9.02)	31.8 (11.4)	72.3 (6.9)	63.2 (4.5)
2000–2001	68.9 (11.5)	49.3 (4.7)	86.4 (6.7)	60.7 (3.4)
2001–2002	66.3 (8.0)	62.5 (9.5)	75.0 (3.1)	69.8 (3.1)

UC = under canopy; BTC = between tree crowns; 350/500 = number of trees ha^{-1}.

In addition, *F. pallescens* has a higher forage quality than *S. speciosa*, and it is preferred by sheep and cattle (Bonino et al., 1986). For this reason, and due to overgrazing, this last species is dominant in most grasslands in which ponderosa pines are commonly planted.

The study was carried out during three consecutive growing seasons (1999–2002). Environmental variables, such as soil water every 20 cm from the surface to 140 cm of soil depth, radiation level along the day, air temperature, relative humidity, and soil fertility, were studied comparing the different treatments. In addition, the response of both plant species to different microenvironmental conditions was analyzed. In this sense, water status (water potential at predawn and along the day) and individual plant growth (tiller production, leaf elongation, and new leaf production) was measured (see Fernández et al., 2002; Gyenge et al., 2002 for a more detailed description of the trial and measurements).

9.3 WATER USE OF DIFFERENT LAND MANAGEMENTS

Measurements of soil water content during the three studied growing seasons indicated that trees did not affect superficial water content compared to the grassland, except after small rainfall events (such as those falling during summer or early-autumn) (Gyenge et al., 2002). In those cases, rain interception in more dense treatments delayed soil water recharge. However, at the beginning of all growing seasons and almost all along them, there were no differences in soil water from the surface to 60 cm of soil depth between treatments (see Figure 9.1a with the example of results from 0 to 20 cm of soil depth). In contrast to these results, during the summer, that is, the dry period, less water was available in deep soil layers in forested plots compared to the grassland (see Figure 9.1b with results from 120 to 140 cm of soil depth). This indicates a differential water use of deep reserves by the trees compared to the native vegetation in the study site (Gyenge et al., 2002; Fernández, 2003).

The magnitude of deep-water depletion depended on climatic characteristics of the season. In a wetter season (such as 2000–2001), in which small rainfall events fall all along the growing season, trees extracted less water from deep layers than in a driest season (such as 1999–2000 or 2001–2002) (Figure 9.1b). This indicates that there is no complete niche separation in relation to soil water use between pines and grasses in these systems. In contrast, pines extracted water from shallow layers, as grasses did, and used deep reserves when shallow ones were depleted. This implicates competition for water resources between trees and grasses. However, at the same time, pines can decrease evaporative demand for understory plants growing under or between their crowns and can ascend water hydraulically (Fernández, 2003). The net balance of these negative–positive interactions is discussed below.

Evapotranspiration (EVT) in different treatments was estimated through water balances (with soil measurements until 140 cm of soil depth), and additionally, sap flow measurements (based on the method of Granier, 1987) were carried out in trees of silvopastoral plots. As was expected, EVT decreased during the growing season in correlation with soil-water depletion (Figure 9.2; Gyenge et al., 2002). However, sap flow measurements indicated that trees continued with similar transpiration rates during the whole growing season (Figure 9.3; Gyenge et al., 2003), indicating that they were extracting water from deeper layers than those measured. This points out the limitations of extracting conclusions of water use based on water balances and the need for additional methods such as sap-flow measurements. In this sense, based on water balances, mean EVT in the season 1999–2000 (period September–May) was 2.96 and 2.87 mm day^{-1} in silvopastoral treatments with 350 and 500 pines ha^{-1}, respectively, and 2.67 mm day^{-1} in the open grassland. For the period November–May of the same season, sap-flow measurements indicated that trees transpiration was 3.03 and 4.17 mm day^{-1} in treatments with 350 and 500 pines ha^{-1}, respectively. This means that tree transpiration was equal or even higher than the whole system EVT estimated from the water balance.

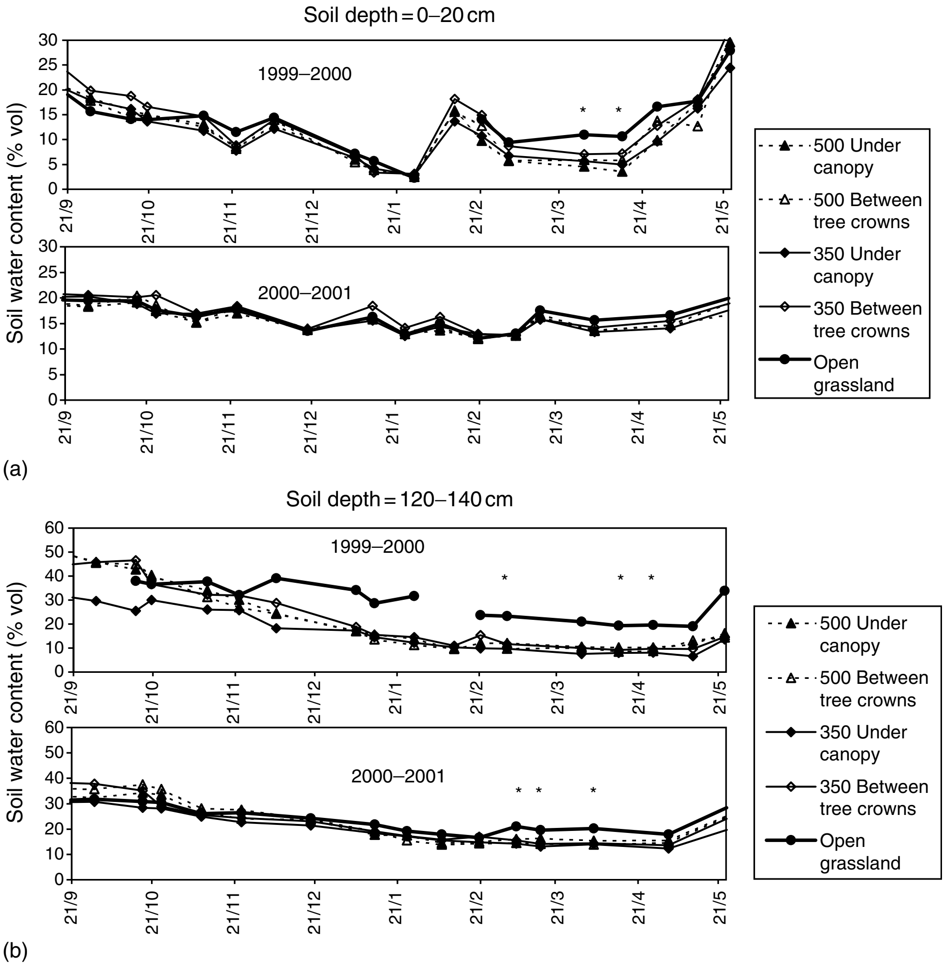

FIGURE 9.1 (a) Soil water content measured at 0–20 cm of soil depth during two growing seasons with a TDR equipment (Imko GmbH, Germany). (b) Soil water content measured at 120–140 cm of soil depth during two growing seasons. 350/500 = number of trees ha^{-1}. Significant differences between all forested plots and the open grassland are indicated with asterisks. (Data from Fernández, M.E., *Influencia del Componente Arbóreo Sobre Aspectos Fisiológicos Determinantes de la Productividad Herbácea en Sistemas Silvopastoriles de la Patagonia Argentina*, Doctoral Thesis, Universidad Nacional del Comahue, Bariloche, Argentina, 2003.)

9.4 TREE–GRASS INTERACTIONS IN SILVOPASTORAL SYSTEMS

Predawn water-potential measurements indicate neutral, positive, or even negative effects of the trees over the grasses (Figure 9.4), depending on soil water content and evaporative demand (Fernández et al., 2002; Fernández, 2003). In both grass species, in periods with high soil water content, the net effect over plant water status was neutral or positive, particularly in treatments with higher tree covers. This may be due to a similar soil water availability but a lower evaporative demand under trees than the grassland. On the other hand, when soil water content was low (less than 13% Vol) and evaporative demand was high, neutral to negative effects were detected in plants growing under trees compared to those in the grassland or BTC (Figure 9.4). This may result from root competition between trees and grasses for scarce water resources, and in the case of the position Under Crowns, a relatively high evaporative demand because of high radiation levels compared to position BTC (e.g., Fernández, 2003). This was due to the movement of shadows at these high

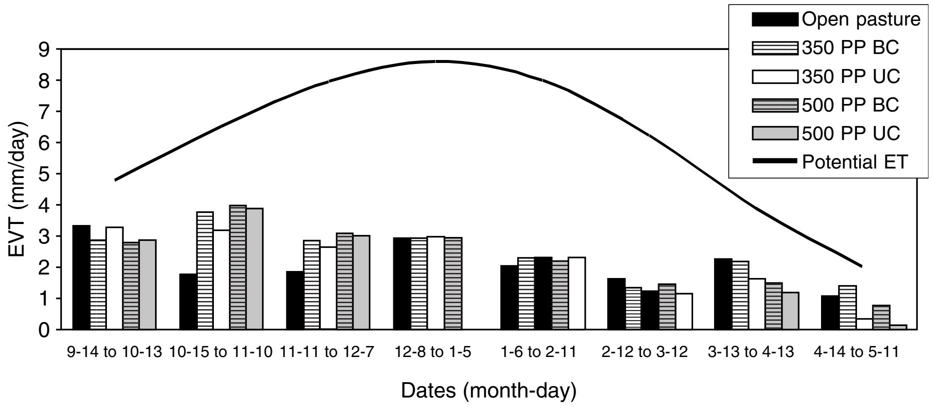

FIGURE 9.2 Mean EVT (in mm per day) of different treatments during the growing season 1999–2000 estimated from water balances. 350/500 PP = number of pines ha^{-1}; UC = under canopy; BTC = between tree crowns. Potential EVT (mm per day) for each period is also indicated. (Reprinted from Gyenge, J.E., M.E. Fernández, T.M. Schlichter and D. Dalla Salda, *Agroforest. Syst.,* 55, 47, 2002. With permission of Kluwer Academic Publishers.)

latitudes, which are displaced with respect to the object that produces them. However, considering the Integral of Water Potential over the whole growing season (Myers, 1988), trees in the more dense treatments showed a cumulative positive effect over grass water status (Table 9.2).

Contrary to similar results of both species in relation to water status, relative growth (evaluated through a Growth Index, which considers tiller and leaf production, see Fernández et al. (2002) for more details) showed a different pattern between both species. Growth of *S. speciosa* decreased as

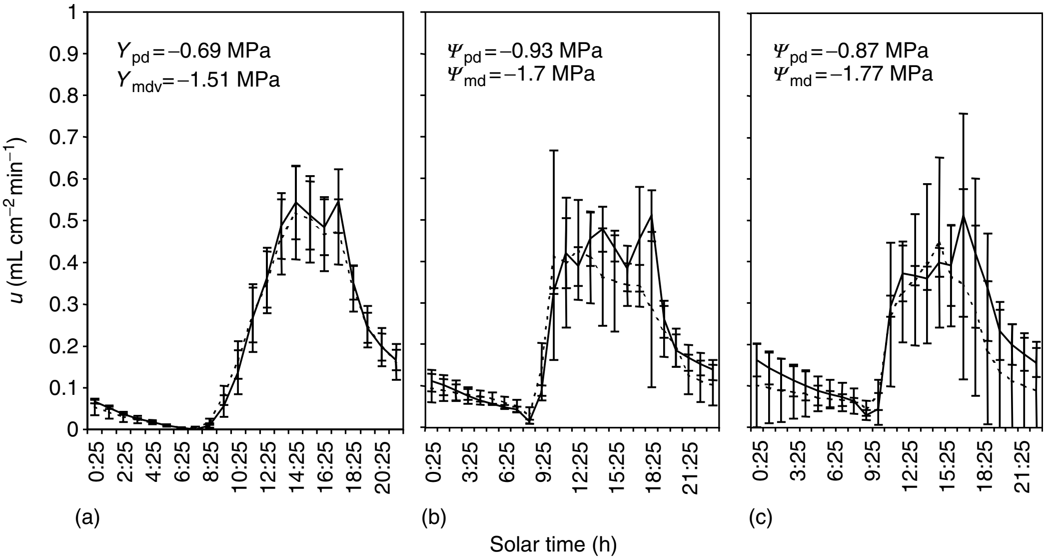

FIGURE 9.3 Sap-flow density (u ± S.D.) of *Pinus ponderosa* in three bright days during the season 1999–2000. (a) 23 November 1999, (b) 26 January 2000, and (c) 11 March 2000. *Filled lines* represent the average u of treatment with 500 trees ha^{-1}, and *dashed lines* represent trees from the treatment with 350 trees ha^{-1}. Mean predawn water potential (Ψ_{pd}) and midday water potential (Ψ_{md}) of trees in each treatment and date are also indicated. (Reprinted from Gyenge, J.E., M.E. Fernández and T.M. Schlichter, *Trees*, 17, 417, 2003. With permission of Springer-Verlag.)

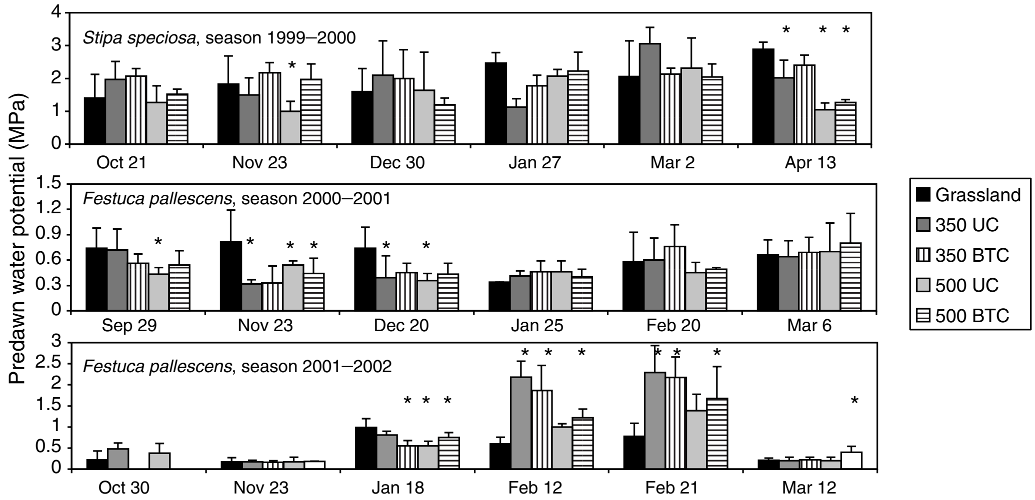

FIGURE 9.4 Predawn water potential (in MPa) of *Stipa speciosa* and *Festuca pallescens* tussocks growing in different treatments. Significant differences between plants of any forested treatment and those of the open grassland are indicated with asterisks. 350/500 = number of pinesha^{-1}; UC = under canopy; BTC = between tree crowns (Data from Fernández, M.E., *Influencia del Componente Arbóreo Sobre Aspectos Fisiológicos Determinantes de la Productividad Herbácea en Sistemas Silvopastoriles de la Patagonia Argentina*, Doctoral Thesis, Universidad Nacional del Comahue, Bariloche, Argentina, 2003.)

tree cover increased (Figure 9.5; Fernández et al., 2002). In contrast, growth of *F. pallescens* was similar in all treatments until relatively high tree-cover level (75%–80%) (Figure 9.5). In this species, growth was measured in two growing seasons contrasting in climate conditions: a wet season (2000–2001) and a dry one (2001–2002). The magnitude of growth was higher in the first wetter season (see maximum values in Figure 9.5), but a trend (not statistically significant) of a higher positive effect of trees over grass growth was detected in the driest season. In 2000–2001, mean growth of plants in the grassland was intermediate of that of plants in forested plots. However, mean growth of plants in the grassland was lower than in forested systems in the dry year.

TABLE 9.2

Integral of Predawn Water Potential along the Whole Growing Season (October–April, in MPa Days): Higher Values Indicate Higher Water Stress

	Stipa speciosa (1999–2000)	*Festuca pallescens* (2000–2001)
Open grassland	258.03	72.7
350 Under canopy	276.67	73.2
350 Between tree crowns	265.82	67.7
500 Under canopy	235.02	53.6
500 Between tree crowns	252.95	60.7

Note: 350/500 = number of pines ha^{-1}. Each number is the average of 3–4 plants.

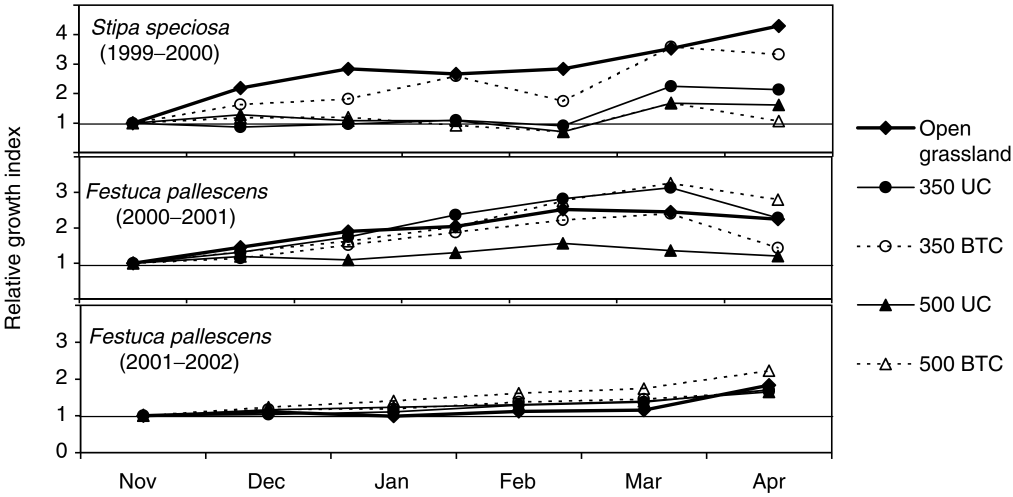

FIGURE 9.5 Relative growth index estimated for *Stipa speciosa* and *Festuca pallescens* tussocks growing in different treatments. 350/500 = number of pines ha^{-1}; UC = under canopy; BTC = between tree crowns. The only significant differences were observed between plants of *Stipa speciosa* growing in the grassland respect to those in forested treatments with 500 pines ha^{-1}, in January and March. (Data from Fernández, M.E., *Influencia del Componente Arbóreo Sobre Aspectos Fisiológicos Determinantes de la Productividad Herbácea en Sistemas Silvopastoriles de la Patagonia Argentina*, Doctoral Thesis, Universidad Nacional del Comahue, Bariloche, Argentina, 2003.)

These results agree with those of natural ecosystems in which facilitation effects are more intense under more stressful conditions (e.g., Bertness and Ewanchuk, 2002).

In the case of *S. speciosa*, facilitation or neutral effects over its water status were detected under trees (Gyenge et al., 2002). However, growth results indicate that the net balance of interactions was negative (Fernández et al., 2002). In this drought tolerant species, radiation had a higher relative limitation than water, thus competition for this resource was more important than any amelioration in water conditions under trees.

Considering results of *F. pallescens*, plant water status in the first wetter season showed that plants in all treatments were in the same good conditions. For this reason, net tree effects over grasses were neutral to positive, specially considering that grasses in forested plots have proportionally much less roots than in the open (Fernández et al., 2004). Growth values agreed with these results, that is, there were no differences between treatments, and in some forested treatments, mean values were even higher than in the open (but not statistically different). From these results, we can infer that in relatively wet summers, facilitative interactions are more important than competition for resources, resulting in a positive net balance. On the contrary, in a very dry summer, competition for soil water between trees and grasses appeared to be more important than any amelioration in environmental conditions under trees. These results support Ong and Leakey (1999) ideas about ecological interactions in agroforestry systems. In February 2002, plants growing in the treatment with lower tree density had more negative water potentials than plants in the open, and had even lower water potentials than plants growing in the densest treatment (Figure 9.4). In plots with 350 pines ha^{-1}, plants growing under tree crowns were those which experienced the highest water stress, probably due to high evaporative demand under a relatively low tree cover, and at the same time, high competition for soil water with tree roots. In the plots with 500 pines ha^{-1}, plants also experienced competition for soil water with trees, that is, they had water potentials lower than in the open grassland. However, they were exposed to lower evaporative demand due to shading than in plots with 350 trees ha^{-1}. Therefore, the net balance had a different result (less negative) than in lower tree densities. In this case, the nature of ecological interactions—their net balance—was the same, but its strength was different. This same response was also found in other plant associations depending on environmental or species characteristics (Bertness and Ewanchuk, 2002). On the other hand, considering only the wetter periods within the second growing season, results of water status agreed with those of the former year; plants in forested treatments showed equal or even better hydric conditions than in the open. Based on these results of plant water status, we can conclude that in dry seasons or periods the net balance of tree–grass interactions is negative, similar to what was described in other silvopastoral systems (e.g., De Montard et al., 1999), but opposite to what happens in natural ecosystems (e.g., Frost and McDougald, 1989; Callaway and Walker, 1997). However, growth results suggest a contrary conclusion: in the drier growing season, positive effects are also higher than negative ones, that is, silvopastoral systems based on the studied species in Patagonia behave as tree–grass associations in savannas.

How can we reconcile the opposite results of water status and growth in dry periods, considering also that *F. pallescens* is a species vulnerable to water deficits (Fernández, 2003)? One possibility is that, in spite of plants in the open grassland showing a better water status, high evaporative demand probably forced stomata to be closed early during the day, decreasing carbon (C) fixation. Stomatal conductance of this species is linearly related to relative humidity (RH) of the air for values below 50% (Fernández, 2003). Despite the fact that we did not find statistical differences in this environmental variable (measured 15 cm above plant canopies) between open and forested plots (Fernández, 2003), leaf temperature under direct radiation was probably higher in plants of the open grassland, thus decreasing the RH of the layer of air close to the leaf surface. In addition to this hypothesis, it is also possible that a higher C fixation in plants of the open (due to their better water status), could have been counterbalanced by high respiration losses by roots. As mentioned earlier, root:shoot ratios of *F. pallescens* plants of the open were significantly higher than in forested plots (Fernández et al., 2004), and therefore, respiration was expected to be higher. Moreover, root respiration rate of plants in the open could have been higher due to higher soil temperatures in the open than in shaded treatments (e.g., Kitzberger, 1995). High water potentials of plants in the open were probably maintained with a high C allocation to root production, while in shaded treatments, biomass allocation to aboveground structures was increased. These changes are expected to be a primary response to radiation decrease in forested plots, as was described for a great number of species growing under shade conditions (e.g., Allard et al., 1991; Cruz, 1997; Valladares et al., 2002). Biomass allocation changes could confer these plants a lower competitive capacity when water reserves are low, but also would imply less maintenance costs of belowground structures. Finally, it is important to note that *F. pallescens* has a typical bimodal aboveground growth pattern, with one growth peak in early spring and the other in autumn (Defossé et al., 1990), coinciding with periods of high water availability. For this reason, worse hydric conditions in the driest month do not necessarily have to imply a reduction in the overall seasonal growth. In spite of this being the common pattern in the field, this species is able to take advantage of rainfall events during the summer as was seen in the first growing season and also under irrigation conditions (Fernández, 2003).

In addition to better water status of *F. pallescens* plants in periods with high soil-water content and a different biomass allocation under trees, other morphological variables changed in plants growing under shade. Whole plant architecture (leaf angle distribution) as well as specific leaf area changed in a way that allow the plants better light capture in radiation-limited microenvironments (Fernández et al., 2004).

“Results from both studied species agree with the hypothesis that radiation being a more limiting resource than water in drought-tolerant species we can expect a different balance between facilitation–competition interactions in different species growing in the same environment.”

9.5 GENERAL CONCLUSIONS

Silvopastoral systems in N.W. Patagonia use more water than native grassland mainly due to deeper rooting systems of pines. These results agree with those of Schulze et al. (1996), which indicated that deep-water reserves are underutilized in native Patagonian ecosystems. Use of these water reserves can enhance ecosystem productivity, but at the same time could have negative impacts on the regional scale if they feed external economies, such as wetlands. This is an important point to be studied in the future.

Tree–grass interactions in silvopastoral systems based on *P. ponderosa* and native forage species depend on physiological characteristics of the grass species (i.e., drought tolerance). On the other hand, the strength of the balance in a particular association of plants depends on climatic conditions of the considered period. As a whole, our results indicate that ponderosa pine – *F. pallescens* constitute a viable species combination for the development of silvopastoral systems in Patagonia. However, studies oriented to evaluate grazing tolerance of *F. pallescens* under shade are needed to recommend this tree–grass association definitively.

As can be predicted based on considerations of Kho (2000b), results of this study indicate that in temperate ecosystems, such as those of Patagonia, development of silvopastoral systems is possible because water is a more limiting resource than radiation due to the precipitation regime. In this case, facilitation effects of trees over water status of grasses can compensate their interference for radiation. However, though facilitation for water was measured in both studied grass species, growth response differed between them. Thus, given the general conditions proposed by Kho (2000b), that is, a temperate climate with water deficits, not all species responded in a similar way to tree introduction. For this reason, knowledge of physiology and morphological plasticity of different species is crucial to predict the result of a particular agroforestry technology in a particular environment.

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10 Litter Dynamics in Plantation and Agroforestry Systems of the Tropics—A Review of Observations and Methods

B. Mohan Kumar

CONTENTS

10.1 INTRODUCTION

Establishing forest plantations to meet the ever-increasing demand for tree products has been a longstanding tradition in the tropics (Evans, 1982), albeit it gained momentum only after the Second World War. According to FAO (2001), the area under tropical forest plantations has increased at an estimated annual rate of 1.9 million ha reaching about 68 million ha in the late 1990s. Of this, India alone has about 32.58 million ha. Other important tropical countries with significant area under forest plantations areas are Indonesia (9.87 million ha), Brazil (4.98 million ha), Thailand (4.92 million ha), Vietnam (1.71 million ha), Venezuela (0.86 million ha), Myanmar (0.82 million ha), Bangladesh (0.63 million ha), Cuba (0.48 million ha), and Madagascar (0.35 million ha). The humid tropics are also characterized by diverse land use systems that integrate woody perennials with other life forms, called agroforestry. Although precise area estimates of agroforestry-type land use are not available, it probably covers a substantial part of the tropics (Nair, 1993). Overall, the man-made forests and agroforests are thought to ease pressure on the tropical forests, which are ‘‘our doomed warehouses of global biodiversity’’ (Ewel, 1999).

Although agroforestry is generally regarded as sustainable (see Kumar and Nair, 2004), fostering quick rotation plantations to resolve the chronic wood shortages faced by millions of people in the tropical regions has raised concerns about its sustainability (Nambiar, 1996; Vance, 2000). Loss of nutrients during the harvest, especially when rotations are short, may exceed the rate of replenishment by weathering of minerals and by atmospheric inputs (Kumar et al., 1998a) implying that site quality deterioration is almost a cliché (Goncalves et al., 1997). Furthermore, the global warming accelerates soil organic matter (SOM) oxidation, making degradation of nutrient-poor soils faster in the tropics (Walker and Steffen, 1997; Seneviratne, 2000). Consequently, there is a major uncertainty, that is, whether the tropical tree plantations and agroforests could be grown perpetually on the same site without serious risk to their vitality and productivity.

To be sustainable, a managed land use system should imitate the structure and functioning of natural ecosystems, which are the results of natural selection over long periods (Ewel, 1999). That is, the dynamics of litterfall, decomposition, and the subsequent bioelement release, which play a fundamental role in the stability of natural ecosystems (see reviews by Bray and Gorham, 1964; Singh and Gupta, 1977; Swift et al., 1979; Brown and Lugo, 1980; Vogt et al., 1986; Ewel et al., 1991; Facelli and Pickett, 1991; Caldentey et al., 2001) should be relevant to the man-made forests and agroforests too (Cuevas and Medina, 1988; Grigal and Vance, 2000). Although plant litter is an important source of ‘‘slow-release’’ nutrients, questions relating to organic matter turnover in the managed tropical land use systems did not receive adequate attention in the past. With the advent of ‘‘organic’’ farming practices, however, research on addition and decomposition of fresh agricultural wastes, green manure and litter may regain some of its past glory in the ‘‘prechemical’’ farming era (see reviews by Kumar and Goh, 2000; Palm et al., 2001).

Tropical forest plantations and agroecosystems also involve diverse kinds of trees, and their impact on the nutrient cycling process is probably variable. It is, therefore, essential to have a clear understanding of the tree species’ impacts on various aspects of SOM dynamics and nutrient cycling, including the effects of litter green manure additions on soil nutrient availability. In addition, small farmers with limited access to chemical fertilizers often remove detritus from the plantation or forest floor for use in their fields or homegardens (Byard et al., 1996; Russell et al., 1997). The impacts of such litter transfer on the nutrient dynamics of the plantation and the agroecosystems have been seldom addressed. Therefore, the current state of knowledge on litter dynamics of managed land use systems in the tropical region and their potentially important role in maintaining soil fertility are summarized here. In particular, variations in litterfall production and the factors affecting litter decomposition, will be analyzed. The need to have consistency in the methodology used for characterising litterfall and decay, and aspects relating to nutrient release from litter cannot be overstated. The paucity of information on nutrient release from litter and its synchrony with nutrient uptake by the associated crops is in part due to the inconsistent experimental approaches. So, the methodological aspects of characterising litterfall and decay rates will be addressed in this chapter.

10.2 LITTERFALL RATES IN TROPICAL PLANTATIONS PARALLEL SITE PRODUCTIVITY

Following from the inverse relationship between total detritus production and latitude of the region, which *inter alia* represents a productivity gradient (Bray and Gorham, 1964), litterfall is an important covariate of aboveground biomass production. To further gauge the nature of interrelationships between litterfall and productivity in plantations, published data on plantation or agroforest productivity and litterfall were examined. Figure 10.1 shows that total litterfall increased

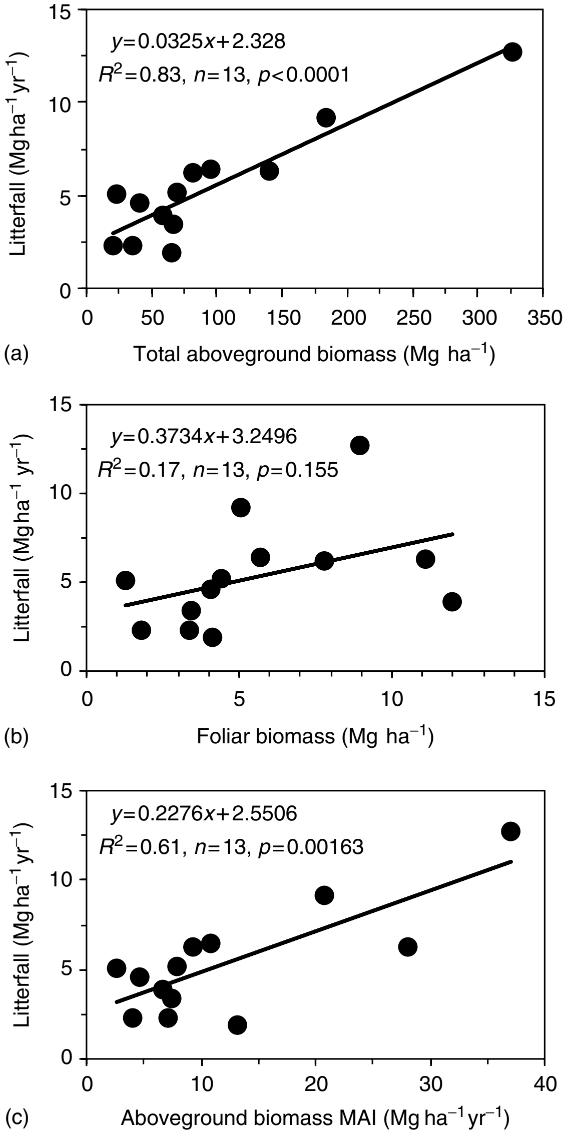

FIGURE 10.1 Relationships between mean annual litterfall and (a) total aboveground biomass yield, (b) foliar biomass, and (c) total aboveground mean annual increment of nine tree species of two age classes (8.8 and 5 years) and grown under two experimental protocols in Kerala, India. (Compiled from the biomass data presented in Kumar, B.M., S.J. George, V. Jamaludheen and T.K. Suresh, *Forest. Ecol. Manag.*, 112, 145, 1998a and from the litterfall data given in George, S.J. and B.M. Kumar, *Int. Tree Crops J.*, 9, 267, 1998 and Jamaludheen, V. and B.M. Kumar, *Forest Ecol. Manag.*, 115, 1, 1999.)

linearly with total aboveground biomass yield and biomass mean annual increment (MAI). Although this is consistent with the findings of Lugo (1992) and Parrotta (1999), due to the complex interactions among environmental factors, productivity, and biomass allocation patterns, and because the same environmental factors influence both productivity and litterfall, such relationships should not be considered as simple cause-and-effect.

A key question is whether there is a direct link between litterfall and the micrometeorological parameters. In this respect, Brown and Lugo (1980) obtained a significant quadratic relationship between annual litter production and the temperature to precipitation (T/P) ratio of the site, which paralleled the biomass–T/P curve. Furthermore, increasing atmospheric concentration of CO_2 (currently at 1.8 ppm per annum) due to anthropogenic emissions is likely to increase the litterfall rates. This is because plant biomass production and net terrestrial carbon storage may increase as atmospheric CO_2 concentrations increase (Amthor and Koch, 1996). However, little or no direct evidences are available in this respect (Kumar et al., 2005).

10.3 VARIATIONS IN LITTERFALL FLUXES IN TROPICAL PLANTATIONS

Although the general pattern of higher litterfall rates in the tropical latitudes hold good on large spatial scales, such a relationship is often masked by within-zone variations. As a result, stand-level differences in annual litterfall abound (range: 1.02–14.5 Mg ha^{-1} yr^{-1}; Table 10.1), and such variations generally reflect the underlying influence of stand age, basal area, species characteristics, and edaphic and climatic factors.

10.3.1 BASAL AREA AND STAND AGE

Basal area and age structure are recognized as major determinants of litterfall (Lugo, 1992), yet there is no consensus on that. For instance, Arunachalam et al. (1998a) noticed a strong correlation ($r = 0.93$, $p < 0.05$) between annual litter production and stand basal area in three regrowing forest stands on a shifting cultivation site in northeastern India. Many others (Kumar and Deepu, 1992; Parrotta, 1999; McDonald and Healy, 2000), however, thought that litterfall rates did not directly relate to stand basal area and density, especially in old-growth stands. Understandably, in young developing stands, annual litterfall rates increase as crown coverage increases (with age and stand basal area), and it plateaus out at about the same time as that of canopy closure. It then follows an asymptotic pattern similar to that of gross primary production and may decline in very old stands. It can thus be concluded that peak litterfall for a wide range of stands under steady-state conditions is independent of stand basal area and stand density. However, the rate at which this equilibrium is approached is not; and denser stands may reach this equilibrium faster than sparse stands.

10.3.2 SPECIES ATTRIBUTES

Species-related variations in quantity as well as periodicity of litterfall in managed tropical land use systems are paramount. For instance, mean annual litterfall of 49 tropical species ranged from 1.02 (*Eucalyptus tereticornis*) to 14.5 Mg ha^{-1} yr^{-1} (*Pinus caribaea*; Table 10.1). Some authors argue that evergreen versus deciduous habit and N-fixing ability of the tree species are major determinants in this respect, in addition to their biomass production potential (Bray and Gorham, 1964; Swamy and Proctor, 1994). Therefore, the question of whether evergreen trees produce more or less litter than deciduous tree species was examined using two experimental datasets (Cuevas and Lugo, 1998; Jamaludheen and Kumar, 1999) of 11 evergreen (range in litterfall: 3.9–14.3 Mg ha^{-1} yr^{-1}; Figure 10.2) and 7 deciduous tropical tree species (range 3.4–10.8 Mg ha^{-1} yr^{-1}). Surprisingly, the results of homoscedastic t-test comparing functional categories such as evergreen and deciduous species were not significant (t statistic $= 1.4703$; p $(T \leq t)$ one-tail $= 0.0804$), signifying that the differences among species within a functional category exceed the variations

TABLE 10.1
Total Litterfall and Standing Crop of Litter in Tree Plantations and Agroforestry Systems in the Tropics

Species or Agroforestry System	Location	Stand Age (years)	Litterfall (Mg ha^{-1} yr^{-1})	Source
Acacia auriculiformis	Ibadan, Nigeria	3	6.92	Salako and Tian (2001)
	Ibadan, Nigeria	7	12.11	Salako and Tian (2001)
	Kerala, India (woodlot)	8.8	12.7–12.9	Kunhamu et al. (1994); Jamaludheen and Kumar (1999)
	Palakkad, Kerala, India (silvipasture; pruned)	5	6.27	George and Kumar (1998)
Acacia leptocarpa	Ibadan, Nigeria	3	7.60	Salako and Tian (2001)
	Ibadan, Nigeria	5	10.7	Salako and Tian (2001)
Acacia nilotica	Karnal, India (alkaline soil)	4	2.5	Gill et al. (1987)
	Karnal, India (alkaline soil)	5	3.8	Gill et al. (1987)
	Karnal, India (alkaline soil)	6	4.9	Gill et al. (1987)
	Karnal, India (alkaline soil)	7	5.7	Gill et al. (1987)
Ailanthus triphysa	Palakkad, Kerala, India (woodlot)	8.8	4.57	Jamaludheen and Kumar (1999)
	Palakkad (pruned silvipasture)	5	1.92	George and Kumar (1998)
Albizia stipulata–Citrus reticulata	Sikkim, India	—	3.7	Sharma et al. (1997)
Alnus nepalensis	Darjeeling, India	7	3.15	Sharma and Ambasht (1987)
	Darjeeling, India	17	5.20	Sharma and Ambasht (1987)
	Darjeeling, India	30	5.66	Sharma and Ambasht (1987)
	Darjeeling, India	46	5.79	Sharma and Ambasht (1987)
	Darjeeling, India	56	5.45	Sharma and Ambasht (1987)
Amomum subulatum + *Alnus nepalensis*	Sikkim, India	—	7.3	Sharma et al. (1997)
Amomum subulatum + Forest	Sikkim, India		4.6	Sharma et al. (1997)
Anthocephalus chinensis	Puerto Rico	26	8.1	Cuevas and Lugo (1998)
Artocarpus heterophyllus	Palakkad, Kerala, India	8.8	6.23	Jamaludheen and Kumar (1999)
Artocarpus hirsutus	Palakkad, Kerala, India	8.8	3.92	Jamaludheen and Kumar (1999)
Casuarina equisetifolia	Palakkad, Kerala, India	8.8	6.44	Jamaludheen and Kumar (1999)
	Palakkad (silvipasture; pruned)	5	2.31	George and Kumar (1998)
	Puerto Rico	1.5–3.5	8.61	Parrotta (1999)
Casuarina + *Eucalyptus*	Puerto Rico (50:50 mixture)	1.5–3.5	7.74	Parrotta (1999)
Casuarina + *Leucaena*	Puerto Rico (50:50 mixture)	1.5–3.5	9.98	Parrotta (1999)
Citrus reticulata	Sikkim, India	—	3.8	Sharma et al. (1997)
Coffea arabica + *Erythrina poeppigiana*	Turrialba, Costa Rica (inclusive of pollarded shade tree litter)	13	3.70	Glover and Beer (1986)

(*continued*)

TABLE 10.1 (Continued)
Total Litterfall and Standing Crop of Litter in Tree Plantations and Agroforestry Systems in the Tropics

Species or Agroforestry System	Location	Stand Age (years)	Litterfall (Mg ha^{-1} yr^{-1})	Source
Coffea arabica + *Erythrina poeppigiana* + *Cordia alliodora*	Turrialba, Costa Rica (inclusive of pollarded shade tree litter)	13	6.65	Glover and Beer (1986)
Cupressus lusitanica	Central highlands, Ethiopia	28	5.01	Lisanework and Michelsen (1994)
Dalbergia sissoo	India	—	4.75	Rajvanshi and Gupta (1985)
Dendrocalamus hamiltonii	Meghalaya, India (jhum fallow)	10	3.50	Toky and Ramakrishnan (1982)
	Meghalaya, India (jhum fallow)	15	3.90	Toky and Ramakrishnan (1982)
	Meghalaya, India (jhum fallow)	20	5.20	Toky and Ramakrishnan (1982)
Dendrocalamus strictus	Pauri Garhwal, UP, India (257–360 clumps ha^{-1}, 49%–62% ground coverage)	—	0.35–0.58	Joshi et al. (1991)
	East Mirzapur, UP, India (dry tropical bamboo savanna)	5[a]	7.18	Tripathi and Singh (1994)
	East Mirzapur, UP, India (dry tropical bamboo savanna)	1[a]	4.08	Tripathi and Singh (1994)
Eucalyptus globulus	Central highlands, Ethiopia (lignotubers)	40	5.83	Lisanework and Michelsen (1994)
Eucalyptus cf. *patentinervis*	Puerto Rico	26	11.12	Cuevas and Lugo (1998)
Eucalyptus robusta	Puerto Rico	1.5–3.5	5.42	Parrotta (1999)
Eucalyptus saligna	Puerto Rico	25	13.17	Cuevas and Lugo (1998)
	Hawaii, USA	4	7–9	Binkley et al. (1992)
Eucalyptus saligna + *Albizia falcataria* mixed stand	Hawaii, USA	4	12–13	Binkley et al. (1992)
Eucalyptus tereticornis	Karnal, India (alkaline soil)	4	1.02	Gill et al. (1987)
	Karnal, India (alkaline soil)			
	Karnal, India (alkaline soil)	5	1.07	Gill et al. (1987)
	Karnal, India (alkaline soil)	6	1.10	Gill et al. (1987)
	Pantnagar, India (associated	7	1.13	Gill et al. (1987)
	with aromatic grass)	4	4.6	Singh et al. (1989)
Eucalyptus + *Leucaena*	Puerto Rico (50:50 mixture)	1.5–3.5	8.87	Parrotta (1999)
Hardwickia binata	Jhansi, India (silvipasture)	23	8.15	Roy et al. (1998)
Hernandia sonora	Puerto Rico	26	8.96	Cuevas and Lugo (1998)
Hevea brasiliensis	Bendel State, Nigeria	23	10.23–13.67	Onyibe and Gill (1992)
Gigantochloa spp.	West Java, Indonesia (bamboo talun–kebun system)	Early fallow	2.0	Christanty et al. (1996)
	West Java, Indonesia (bamboo–talun–kebun system)	Mature stand	3.5	Christanty et al. (1996)

TABLE 10.1 (Continued)
Total Litterfall and Standing Crop of Litter in Tree Plantations and Agroforestry Systems in the Tropics

Species or Agroforestry System	Location	Stand Age (years)	Litterfall ($Mg\ ha^{-1}\ yr^{-1}$)	Source
Hibiscus elatus	Puerto Rico	26	13.7	Cuevas and Lugo (1998)
Juniperus procera	Central highlands, Ethiopia	40	10.87	Lisanework and Michelsen (1994)
Khaya nyasica	Puerto Rico	26	10.8	Lisanework and Michelsen (1994)
Leucaena leucocephala	Ibadan, Nigeria	3	8.78	Salako and Tian (2001)
	Ibadan, Nigeria	7	10.05	Salako and Tian (2001)
	Palakkad, Kerala, India (silvipasture; pruned)	5	2.30	George and Kumar (1998)
	Palakkad, Kerala, India (woodlot)	8.8	5.09	Jamaludheen and Kumar (1999)
	Puerto Rico	1.5–3.5	9.69	Parrotta (1999)
Paraserianthes falcataria (syn. *Albizia falcataria)*	Kerala, India	8.8	9.17	Jamaludheen and Kumar (1999)
	Hawaii, USA	4	18.0	Binkley et al. (1992)
Phyllanthus emblica	Kerala, India	8.8	5.18	Jamaludheen and Kumar (1999)
Phyllostachys pubescens	South China	—	3.1–5.0	Maoyi et al. (1990)
Pinus caribaea var. *hondurensis*	Puerto Rico	4	2.1–7.4	Lugo (1992)
	Puerto Rico	18.5	12.9–14.5	Lugo (1992)
	Puerto Rico	26	14.33	Cuevas and Lugo (1998)
Pinus elliottii var. *densa*	Puerto Rico	26	11.35	Cuevas and Lugo (1998)
Pinus merkusii	Merapi, Java, Indonesia	30	9.0	Gunadi (1994)
	Merbau, Java, Indonesia	25	4.0	Gunadi (1994)
Populus deltoides	Tarai, India	1	2.0	Lodhiyal and Lodhiyal (1997)
	Tarai, India	2	3.5	Lodhiyal and Lodhiyal (1997)
	Tarai, India	3	4.5	Lodhiyal and Lodhiyal (1997)
	Tarai, India	4	6.7	Lodhiyal and Lodhiyal (1997)
	Dehra Dun, India	13	3.08	Raizada and Srivastava (1986)
	Pantnagar, India (associated with aromatic grass)	4	4.5	Singh et al. (1989)
Pterocarpus marsupium	Kerala, India	8.8	3.42	Jamaludheen and Kumar (1999)
Senna siamea	Ibadan, Nigeria	3	7.78	Salako and Tian (2001)
	Ibadan, Nigeria	7	10.37	Salako and Tian (2001)
Sphaerobambos philippinensis	Davao del Norte, Philippines	4	6.72–12.58	Virtucio et al. (1994)
Swietenia macrophylla	Puerto Rico	17	10–12.1	Lugo (1992)
	Puerto Rico	40	5.40	Cintrón and Lugo (1990)
	Puerto Rico	49	10.7–14.1	Lugo (1992)
	Puerto Rico	26	9.80	Cuevas and Lugo (1998)

(*continued*)

TABLE 10.1 (Continued)

Total Litterfall and Standing Crop of Litter in Tree Plantations and Agroforestry Systems in the Tropics

Species or Agroforestry System	Location	Stand Age (years)	Litterfall (Mg ha^{-1} yr^{-1})	Source
Terminalia ivorensis	Puerto Rico	23	9.26	Cuevas and Lugo (1998)
Theobroma cacao + *Cordia alliodora*	Turrialba, Costa Rica (shade trees pollarded)	4.5	4.19	Alpizar et al. (1986)
T. cacao + *Erythrina poeppigiana*	Turrialba, Costa Rica (shade trees pollarded)	4.5	1.78	Alpizar et al. (1986)
T. cacao + *Hevea brasiliensis*	Thrissur, Kerala, India (excluding overstory litter)	7	5.32	Sreekala (1997)
T. cacao (no overstory)	Thrissur, Kerala, India (excluding overstory litter)	7	8.23	Sreekala (1997)
Triplochiton scleroxylon	Nigeria	Young stand	7.44	Orimoyegun (1985)
Natural fallow	Ibadan, Nigeria	—	7.7	Salako and Tian (2001)

[a] Time after last harvest; pruned means the trees were pruned to facilitate grass growth in the interspaces; information not available.

between categories. This probably masks any influence of the evergreen versus deciduous nature of trees on litterfall rates.

N-fixing species are widely extolled for their soil improving properties, which are partly related to their ability to produce nitrogen-rich litter (MacDicken, 1994). N-fixing species such as *Casuarina equisetifolia* and *Acacia auriculiformis* reportedly accumulate large quantities of organic matter on the forest floor (Mailly and Margolis, 1992; Kunhamu et al., 1994). Data from Jamaludheen and Kumar (1999) further exemplify this. They showed that exotic N-fixing species such as *A. auriculiformis*, *Paraserianthes falcataria*, and *C. equisetifolia* accounted for the three highest litterfall rates (6.44–12.69 Mg ha^{-1} yr^{-1}, Table 10.1) among nine multipurpose tree species studied. *Pterocarpus marsupium*, another indigenous legume, however, showed the lowest litterfall

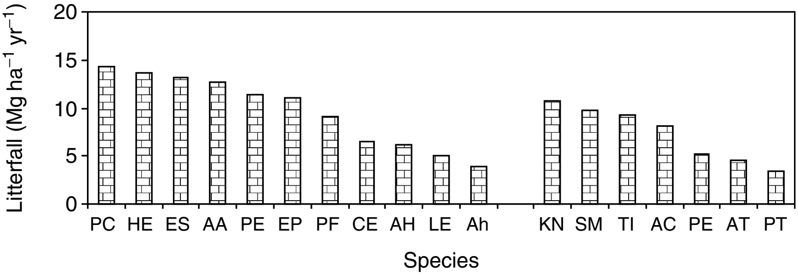

FIGURE 10.2 Annual litterfall of 18 evergreen and deciduous tropical trees. PC—*Pinus caribaea* var. *hondurensis*, HE—*Hibiscus elatus*, ES—*Eucalytpus saligna*, AA—*Acacia auriculiformis*, PE—*Pinus elliottii* var. *densa*, EP—*Eucalyptus* cf. *patentinervis*, PF—*Paraserianthes falcataria*, CE—*Casuarina equisetifolia*, AH—*Artocarpus heterophyllus*, LE—*Leucaena leucocephala*, Ah—*Artocarpus hirsutus*, KN—*Khaya nyasica*, SM—*Swietenia macrophylla*, TI—*Terminalia ivorensis*, AC—*Anthocephalus chinensis*, PE—*Phyllanthus emblica*, AT—*Ailanthus triphysa*, PT—*Pterocarpus marsupium*. (From Cuevas, E. and A.E. Lugo, *For. Ecol. Manage.*, 112, 263, 1998; Jamaludheen, V. and B.M. Kumar, *For. Ecol. Manage.*, 115, 1, 1999.)

(3.42 Mg ha^{-1} yr^{-1}; Table 10.1), denoting a paradox in the litter production potential of woody tropical legumes.

Some authors (e.g., O'Connell and Sankaran, 1997) also asserted that exotic plantation species, regardless of N fixing or not, generally have a higher standing crop of litter. This can perhaps be rationalized by their higher biomass production potentials and lower decay rates (explained elsewhere). Data presented in Figure 10.1 clearly show that the high litter-producing trees concomitantly showed higher biomass production potential suggesting that more than the geographic origin (i.e., indigenous vs. exotic), growth habit (deciduous vs. evergreen), and N-fixation ability, the potential for high growth rates determines litterfall rates. Although it cannot be reasoned that biomass production potential of species exerts a cause–effect relationship on litterfall, it is perhaps the best indicator of litterfall rates.

Implicit in this is also the possibility of differential litter production capacities for different clonal lines or provenances because of the variations in production potential and growth habits. Although data on litterfall potentials owing to clonal variations in forest trees are not readily available, in one study dealing with three clones of rubber (*Hevea brasiliensis*), Onyibe and Gill (1992) found that variations among tree clones in litterfall production were not statistically significant. More experimentation is, perhaps, necessary to make firm conclusions in this respect.

10.3.3 Species Mixtures

Since litterfall rates generally parallel the trend in biomass productivity, higher litter yield is probable in mixed species stands, as they are intrinsically more productive than monospecific stands (sensu. Binkley et al., 1992). However, most studies on litterfall in tropical plantations have been conducted in monospecific stands. A notable exception is that of Parrotta (1999), who in a comparative study of single- and mixed-species plantations of *C. equisetifolia*, *Eucalyptus robusta*, and *Leucaena leucocephala*, found that mixed-species stands had higher litterfall rates than monospecific stands, despite variations in species attributes (Table 10.1).

10.3.4 Site Characteristics

Fixed-site characteristics such as latitude, altitude, and aspect may strongly affect the litterfall dynamics. Perhaps there are three factors influencing productivity and biomass allocation strategies, and consequently, litter production along a latitudinal or altitudinal gradient—the energy budget, the hydrological regimes, and changes in plant growth form. The tropical zone is often characterized by a constant radiation surplus and general thermic uniformity; temperatures are often closer to the optimum for plants, and hence, it is reasonable to expect higher litterfall production rates there. In addition, the higher temperatures may accelerate leaf fall rates, especially when it is not limiting plant growth. Consistent with this, Gwada et al. (2000) showed that temperature increases between 20°C and 28°C stepped up leaf production and abscission rates in *Kandelia candel*, a mangrove species. In the moist forests of Western Ghats, Bhat and Murali (2001) also found that leaf abscission is more when the temperature increases and when the day length is short, signifying a higher amount of fine litterfall under warmer temperature and shorter photoperiodic regimes.

Rainfall and actual evapotranspiration determine the hydrological regime of a site. Sites with plentiful supplies of water and nutrients will allow trees to grow quickly and attain a large leaf area index, in turn producing more leaf litter. Paradoxically, reduced water availability triggers leaf fall. Thus, soil-water retention and soil fertility are important determinants of litterfall quantity and composition within the same climatic range (Facelli and Pickett, 1991). Other workers too (Swift et al., 1979; Bernhard-Reversat, 1993) have noted that the type of soil would generally determine the rate of litterfall and its subsequent decay dynamics. A limited amount of data also indicates that adverse soil parameters such as soil acidity, salinity, sodicity, and water logging may depress primary production and litterfall rates. For example, Eusse and Aide (1999) reported that litter production of *Pterocarpus officinalis* decreased along a gradient of soil salinity and was twice greater at the low-salinity site than at the high-salinity site.

10.3.5 TEMPORAL VARIATIONS

Litterfall for deciduous species especially is an episodic process, with conspicuous peaks corresponding either to the beginning or near the end of the dry period. A plausible explanation is that water or temperature stresses activate the *de novo* synthesis of abscissic acid in the foliage (Kumar and Deepu, 1992); thus annual or seasonal drought (Cintrón and Lugo, 1990) and hot winds may produce large pulses of leaf fall. Coincidentally, litterfall for most species follows a unimodal distribution pattern with a distinct peak either during the dry season (Raizada and Srivastava, 1986; Pascal, 1988; Joshi et al., 1991) or during the winter season (Gill et al., 1987; Cintrón and Lugo, 1990). In some cases it, however, coincided with the peak rainfall events, for example, the Puerto Rican plantations studied by Lugo (1992) and the *P. officinalis* stands examined by Eusse and Aide (1999). Although unimodal litterfall pattern is most common for tropical species (e.g., George and Kumar, 1998; Jamaludheen and Kumar, 1999), Gill et al. (1987) reported that litterfall in *Acacia nilotica* plantations on the highly alkaline soils of north India followed a bimodal trend, with the principal peak during the winter and a minor one in early summer. Species also may respond to seasonal changes in soil salinity (Twilley et al., 1986) and day length (Cuevas and Lugo, 1998; Bhat and Murali, 2001). Overall, within-year and year-to-year variations in tropical trees mirror pronounced climatic or edaphic cues.

10.3.6 PERTURBATIONS

Disturbances such as fire, wind, and hurricanes and damages due to droughts or diseases also induce large pulses of litterfall and may probably explain much of the observed seasonal and interannual variations (Bruederle and Streans, 1985; Adu-Bredu et al., 1997). High-velocity winds not only provoke premature abscission of already senescent leaves, but may also cause fall of other litter components (Caldentey et al., 2001). Windstorms are important in tree fall, but deposition of this component is highly variable in time and space (Sollins, 1982). Premature abscission of leaves by summer storm or through pathogenic infection (e.g., abnormal leaf fall in *H. brasiliensis* and other species) will not only change the seasonality of litterfall, but also ensures higher nutrient returns, as nutrient reabsorption from the prematurely shed foliage had not occurred.

10.3.7 TREE MANAGEMENT PRACTICES

Thinning, pruning, and fertilization are important especially in managed stands of high-value crops. As regards to thinning, Caldentey et al. (2001) reported that annual litter flux decreased by 50% two years after a shelterwood cut wherein 55% of the initial basal area was removed. Stand thinning thus lowers litterfall rates but soon the stand would be back at the plateau of litterfall, if crown closure were quickly regained. Pruning the laterals at the beginning of the crop-planting season is typical of agroforestry and the pruned trees usually yield less litter (excluding pruned materials). In a study involving four tropical species grown in silvopastoral system in the humid tropical regions of Kerala with periodical pruning, George and Kumar (1998) indicated that annual addition of litter ranged from 1.92 to 6.25 Mg ha^{-1}, which was substantially lower than the litterfall recorded in woodlots (unpruned) at the same location (Jamaludheen and Kumar, 1999). Moreover, pruning alters the leaf fall periodicity, especially if significant quantities of foliar biomass are removed in such operations; it, nonetheless, provides a large pulse of nutrient-rich green manure or fodder. Fertilization may enhance litterfall in tropical hardwood species, as it enhances the leaf biomass production. Experimental evidences are, however, variable. For instance, Tanner and Kapos (1992) reported that application of $N + P$ significantly increased litterfall in Venzuelan montane forests, 4 years after the initial application. Conversely, for conifers like *Pinus sylvestris*, Finer (1996) encountered a fertilization-induced reduction in needle litterfall due to increased needle longevity.

10.4 PROXIMATE COMPOSITION OF LITTER

Although most attention is on the leaf fraction because of its predominance in the litterfall process, Cuevas and Lugo (1998) suggested that other litterfall fractions (e.g., twigs, sloughed off bark, reproductive parts such as flowers and fruits) are important in terms of both magnitude of flux and the quality of inputs. Litterfall studies, however, combine fruit fall with other miscellaneous components including flowers, bark, or seeds; but fruit fall may be substantial in the tropics (Lugo and Frangi, 1993). Woody litter constitutes another significant entity in litterfall collections. It usually accounts for about 15% of the total annual litterfall (Cuevas and Lugo, 1998), and may increase as the stand age increases. Similarly, insect frass may be important during major pest outbreak periods. Yet, no specific attempts have been made to characterize the dynamics of woody litterfall or to quantify the nutrient return through insect frass, in a tropical plantation or agroforest context (but see Harmon et al., 1986, for a review on the dynamics of large woody debris in the temperate region).

On a final note, litterfall reported from the managed stands in the tropical region are tremendously variable. Environmental factors, species attributes, tree management, stocking levels, and age-structure cause variations in the quality and quantity of litter. In contrast to the well-studied temperate forest sites, the tropical environments also involve more species, soil types, greater annual rainfall, and longer growing seasons; and depict far more diversity in litterfall characteristics. Yet, the extent of scientific studies on litter dynamics in the tropics has been disproportionately lower than what the ecological benefits, and silvicultural importance of the process, would warrant. Part of the variability in litterfall reported (Table 10.1) might also be due to inconsistent methodology used by the experimentalists.

10.5 METHODOLOGICAL ASPECTS OF LITTERFALL STUDIES

As regards to the methodology for litterfall quantification, broadly, there are two problems, neither of which has been completely solved. These are (1) how to design a trap to accurately collect litterfall and (2) how to locate a network of litter traps to sample an area within acceptable limits of error. The first may be called litter trap design error, and the second, the sampling error.

10.5.1 LITTER TRAP DESIGN

Tropical ecologists, just as their counterparts elsewhere, use diverse sizes and shapes of litter traps for estimating litter production rates. Box type (square or rectangular) is the traditional design, but a circular construction is best as it minimizes the edge effects (Anderson and Ingram, 1989). Nonetheless, traps used by various investigators show considerable variability, as outlined below:

- 1 m^2 wooden traps, established 15–20 cm aboveground level by pegs at the corners and constructed with 0.5–1.5 mm mesh size net to minimize water retention and decomposition of litter (Proctor, 1983; Gill et al., 1987; Sharma and Ambasht, 1987; Onyibe and Gill, 1992; Tanner and Kapos, 1992)
- 50 × 50 cm frames located on the forest floor (Lugo et al., 1990)
- Conical traps 80 cm in diameter, 1 m high (Luizao and Schubart, 1987)
- Baskets (0.25 m^2) of fiberglass or fine mesh plastic screens (1 mm mesh) supported at ~1.5 m aboveground (Eusse and Aide, 1999; Parrotta, 1999)
- 50 × 50 × 10 cm deep wire screen baskets of 1.25 cm mesh galvanized wire screen and lined with fiber glass window screening of 1 mm mesh size (Cintrón and Lugo, 1990)
- 0.06 m^2 funnels of polyethylene net (Gundersen, 1998)
- Grids delineated on the forest floor using bamboo culms (Arunachalam et al., 1998a)

Indeed, such diversity in trap design complicates the comparison of data from different studies. A main problem to be considered in trap design is wind turbulence around the trap, which may move the litter in and out. The probability of litter moving in and out of the traps would be greater in the shallow box and grid designs. This problem of wind turbulence was, however, overcome when Hughes et al. (1987) designed a 15 L capacity trap with a 0.25 m^2 circular collecting surface. The sharply delineated edges further minimize the edge effect and facilitate 100% retention of collected litter even in strong winds. Yet another problem encountered in litterfall studies is the *in situ* decomposition of litter samples. Indeed, the box method results in greater *in situ* decomposition of litter, compared to suspended free drainage mesh base. This may not only lead to a slight underestimation of litterfall, but can also result in underrating nutrient accessions because of leaching. Other potential problems include tipping of the traditional box-type traps by the rapidly growing plants and by animal movements. Action by soil fauna and the effects of soil splash are also more when "low" traps are used. Therefore, the traps preferably should be 0.5 m aboveground, just to level with the understory vegetation.

10.5.2 SAMPLING ERRORS

Improper deployment of fewer litter traps per unit area is a problem in many studies. This may cause either underestimation or overestimation of litterfall, as the traps fail to capture variations in litterfall within the plots. Litter traps are usually placed on the ground along transects representing environmental gradients or other parameters. Most studies, however, do not give concrete information on deployment of litter traps within the plots. At least in some studies, the traps were seen installed nonrandomly invalidating the resultant comparisons.

As regards the number of litter traps, periodicity of sampling, and duration of litter collection, again the published reports seldom show any consistent pattern. Although Newbould (1967) recommended the use of at least 20 traps to achieve 5% standard error about the mean, many authors have used fewer traps (e.g., Sharma et al., 1997; McGrath et al., 2000). As a result, in most studies analyzed, more traps might have certainly improved the experimental design. Despite this, in plantation or agroforestry systems with regularly spaced trees of fewer species, within-stand variation in litterfall may be lower than that of the more heterogeneous natural forests.

Inadequate temporal scales also preclude assessment of the interactions of biological, chemical, and physical processes within the ecosystem. Periodicity of litter collection (sampling protocol) varies from weekly (Luizao and Schubart, 1987), fortnightly (Onyibe and Gill, 1992; Arunachalam et al., 1998a), monthly (Gunadi, 1994; Jamaludheen and Kumar, 1999) to six monthly intervals. Shorter intervals are preferred in the tropical regions to minimize *in situ* decomposition. Duration of litter collection also ranges from 4 to 6 months (Parrotta, 1999) to a few years, and in some cases up to 6 years (Pedersen and Bille-Hansen, 1999), with a mode value of 1 year. Although longer durations are desirable, especially when the study aims at characterizing inter-year variations, 1 year studies have the potential to give reasonable accounts on litter production, provided the climatological parameters of the study period are representative and that the stands are under steady-state conditions.

10.5.3 ANALYSIS OF LITTERFALL DATA

Statistical methods such as ANOVA are often used for comparing litterfall, without considering peculiarities such as correlation between successive measurements and heterogeneity of variances, which in turn, may lead to erroneous conclusions. Most litterfall data (collected from multiple traps repeated over a period) might, probably require a repeated measures design because the same traps are sampled from month to month. Furthermore, in straight comparisons involving univariate and multivariate solutions, Moser et al. (1990) noted that the multivariate approach always provided interpretations that are consistent with the univariate approach, and suggested that the former should be preferred.

10.6 LITTER DECOMPOSITION

Once on the forest floor, the litter is subjected to a variety of processes, which result in its disappearance. Many workers attempted quantifying forest litter decomposition (see reviews by Bray and Gorham, 1964; Singh and Gupta, 1977; Facelli and Pickett, 1991; Lavelle et al., 1993; Berg, 2000). Most studies, however, compare litter decomposition of a single species or different species within a site, or single species of different age groups (Melillo et al., 1982). A vast majority of these reports also signify homogenous stands in the temperate region, and studies on managed tropical ecosystems in general and mixed species agroforestry systems in particular are scarce.

Yet, the reported studies indicate that the tropics exhibit a rapid turnover of organic matter (range in *k* values for a wide range plantation trees and green manures: 0.03–2.3 month^{-1}; Table 10.2). Clearly, the decay rate coefficients stated for most tropical species are greater than those of the temperate coniferous litter (*k* values of 0.15–0.28 yr^{-1}; Monleon and Cromack, 1996, and 0.36–0.63 yr^{-1}; Cromack et al., 1991), and their half-lives are correspondingly low (Table 10.2). Implicitly, the tropical ecosystems decompose what their temperate counterparts consume over 1 year in less than a month. This rapid turnover of organic matter is believed to be responsible for the maintenance of a significant soil organic pool (humus) in the tropical ecosystems (Ola-Adams and Egunjobi, 1992) and their generally high productivity levels, despite most of them being sited on nutrient-poor soils.

A survey of the available literature also indicates that variability abounds in the litter decay rates of tropical species (Table 10.2). Differences in chemical quality of litter, variations in the biophysical environment, and soil microfaunal and macrofaunal activities (Olson, 1963; Swift et al., 1979; Nagy and MaCauley, 1982; Moore, 1986; Upadhyay et al., 1989) are widely regarded as causative effects in this respect. Yet another source of variability, however, is the differential experimental methods employed in such studies. A critique on the major determinants of litter decay and the methods used in litter decay studies with a view to evolve standard techniques for characterizing litter decomposition rates is attempted here.

10.6.1 SUBSTRATE QUALITY

Many previous workers reported that chemical and physical characteristics of litter are key regulators of decomposition (e.g., Swift et al., 1979; Heal et al., 1997). Accordingly, several functional relationships between breakdown rates of litter and its chemical nature (Swift et al., 1979; Bloomfield et al., 1993; Constantinides and Fownes, 1994; Giller and Cadisch, 1997; Russell and Vitousek, 1997; Maithani et al., 1998; Arunachalam et al., 1998b; Kwabiah et al., 1999, 2001) have been evolved. Although chemical attributes such as initial lignin, N, and P concentrations, lignin-to-N ratio, C-to-N, and C-to-P ratios are deemed as driving functions of the decomposition process, lignin and N concentrations attracted the most scientific attention. Lignin is particularly recalcitrant to enzyme degradation, and its intimate association with cellulose fibers results in masking of a large fraction of carbohydrate, which otherwise would be accessible to the leaf-associated microbes (Gessner and Chauvet, 1994). Hence, lignin content of litter is regarded as an important inverse index of decay rates (Palm and Sanchez, 1991; Couteaux et al., 1995; Mesquita et al., 1998; Berg, 2000; Kumar and Goh, 2000).

Despite numerous workers suggesting that initial lignin content is a reasonable predictor of decomposition rates for most temperate and some tropical species, there is yet no consensus regarding which chemical parameter is the best predictor of decomposability of tropical litter (Berg, 1986, 2000; Vitousek et al., 1994). In particular, Jamaludheen and Kumar (1999) found that despite lower lignin content, the decay coefficient (*k*) of *Phyllanthus emblica* was lower than that of other species with higher lignin concentrations, implying a predominant role for other chemical constituents in the decay process. According to Kumar and Deepu (1992), detrital N content of six tropical tree species is a better predictor of decay rate than lignin. Consistent with this, litter from N-fixing species decomposed faster than litter of non-N-fixing species (Binkley, 1992; Bernhard-Reversat, 1993; Sharma et al., 1997; Jamaludheen and Kumar, 1999). However, it cannot

TABLE 10.2
Monthly Litter Decay Rate Coefficients (*k*) and Half-Lives of Tropical and Subtropical Tree, Shrub, and Herbaceous Species in Plantations and Agroforestry Systems as Studied by the Litterbag Technique

Species	Location and Land Use Practice	Materials Used	*k*	Half-Life (Months)	Source
Acacia auriculiformis	Palakkad, Kerala, India	Freshly fallen leaves	0.16	4.2	Jamaludheen and Kumar (1999)
	Palakkad, Kerala, India (silvipasture)	Freshly fallen leaves	0.42	1.7	George and Kumar (1998)
	Thrissur, Kerala, India	Freshly fallen leaves	0.28	2.48	Kunhamu et al. (1994)
Acacia mangium	Thrissur, Kerala, India (homegarden)	Leaf	0.68	1.0	Hegde (1995)
	Thrissur, Kerala, India (open area)	Leaf	0.75	0.93	Hegde (1995)
Acacia spp.	India, Malaysia, Congo	—	0.087	8.0	O'Connell and Sankaran (1997)
Acioa barteri	Ozala, Anambra, Nigeria (bush fallow)	Leaf	0.116	6.0	Okeke and Omaliko (1992)
Ailanthus triphysa	Palakkad, Kerala, India	Freshly fallen leaves	0.31	2.2	Jamaludheen and Kumar (1999)
	Palakkad, Kerala, India (silvipasture)	Freshly fallen leaves	0.14	4.8	George and Kumar (1998)
Albizia spp.	India	—	0.139	5.0	O'Connell and Sankaran (1997)
Albizia stipulata	Sikkim, India (agrisilviculture)	Leaf	0.109	6.3	Sharma et al. (1997)
	Sikkim, India (agrisilviculture)	Twig	0.064	10.9	Sharma et al. (1997)
Alnus nepalensis	Sikkim, India (agrisilviculture)	Leaf	0.087	7.9	Sharma et al. (1997)
	Sikkim, India (agrisilviculture)	Twig	0.025	27.9	Sharma et al. (1997)
	Darjeeling, India (7 year old woodlot)	Forest floor litter	0.053	13.2	Sharma and Ambasht (1987)
	Darjeeling, India (17 year old woodlot)	Forest floor litter	0.07	9.9	Sharma and Ambasht (1987)
	Darjeeling, India (30 year old woodlot)	Forest floor litter	0.12	5.5	Sharma and Ambasht (1987)
	Darjeeling, India (46 year old woodlot)	Forest floor litter	0.068	10.2	Sharma and Ambasht (1987)
	Darjeeling, India (56 year old woodlot)	Forest floor litter	0.04	18.0	Sharma and Ambasht (1987)
Amomum subulatum	Sikkim, India (agrisilviculture)	Residue	0.073	9.5	Sharma et al. (1997)
Andropogon gayanus	Meta, Columbia	Litter—stored frozen (1 month) and dried (60°C)	0.045–0.076	9.12–15.4	Thomas and Asakawa (1993)
Arachis pintoi	Meta, Columbia	Litter—stored frozen (1 month) and dried (60°C)	0.049–0.128	5.4–14.1	Thomas and Asakawa (1993)

TABLE 10.2 (Continued)

Monthly Litter Decay Rate Coefficients (*k*) and Half-Lives of Tropical and Subtropical Tree, Shrub, and Herbaceous Species in Plantations and Agroforestry Systems as Studied by the Litterbag Technique

Species	Location and Land Use Practice	Materials Used	*k*	Half-Life (Months)	Source
Artocarpus heterophyllus	Palakkad, Kerala, India	Freshly fallen litter	0.22	3.1	Jamaludheen and Kumar (1999)
Artocarpus hirsutus	Palakkad, Kerala, India	Freshly fallen litter	0.21	3.4	Jamaludheen and Kumar (1999)
Bombax ceiba	India	—	0.139	5.0	O'Connell and Sankaran (1997)
Brachiaria decumbens	Meta, Columbia	Litter—stored frozen (1 month) and dried at 60°C	0.03–0.094	7.4–23.1	Thomas and Asakawa (1993)
Brachiaria dictyoneura	Meta, Columbia	Litter—stored frozen (1 month) and dried at 60°C	0.024–0.061	1.3–28.9	Thomas and Asakawa (1993)
Brachiaria humidicola	Meta, Columbia	Litter—stored frozen (1 month) and dried at 60°C	0.03–0.085	8.2–21.0	Thomas and Asakawa (1993)
Bridelia retusa	Kerala, India, *ex situ* field	Leaf	0.54	1.3	Kunhamu (1994)
Calliandra calothyrsus	Maseno, Kenya (*ex situ* field)	Fresh, fully expanded leaves (A)	0.365	1.9	Kwabiah et al. (2001)
	Maseno, Kenya (*ex situ* field)	A + urea enriched	0.365	1.9	Kwabiah et al. (1999)
	Maseno, Kenya (*ex situ* field)	A + TSP enriched	0.49	1.4	Kwabiah et al. (1999)
	Highlands of Sri Lanka (tea plantations)	Pruned foliage	0.221	3.1	De Costa and Atapattu (2001)
	Highlands of Sri Lanka (tea plantations)	Pruned stem	0.168	4.1	De Costa and Atapattu (2001)
Cassia siamea	Chipata, Eastern Zambia (in a maize field)	Fresh leaves	0.21	3.3	Mwiinga et al. (1994)
Casuarina equisetifolia	Palakkad, Kerala, India	Freshly fallen leaves	0.17	4.0	Jamaludheen and Kumar (1999)
	Palakkad, Kerala, India (silvipasture)	Freshly fallen leaves	0.67	1.3	George and Kumar (1998)
Centrosema acutifolium	Meta, Columbia	Litter—stored frozen (1 month) and dried (60°C)	0.03–0.052	13.3–23.1	Thomas and Asakawa (1993)
Centrosema pubescens	Nsukka, Nigeria (*in situ* bush fallow)	Leaf	0.125	5.5	Okeke and Omaliko (1992)

(*continued*)

TABLE 10.2 (Continued)

Monthly Litter Decay Rate Coefficients (*k*) and Half-Lives of Tropical and Subtropical Tree, Shrub, and Herbaceous Species in Plantations and Agroforestry Systems as Studied by the Litterbag Technique

Species	Location and Land Use Practice	Materials Used	*k*	Half-Life (Months)	Source
Croton megalocarpus	Maseno, Kenya (*ex situ* field)	Fresh, fully expanded leaves	1.13	0.6	Kwabiah et al. (2001)
	Maseno, Kenya (*ex situ* field)	Fresh, fully expanded leaves + urea enriched	1.46	0.47	Kwabiah et al. (1999)
	Maseno, Kenya (*ex situ* field)	Fresh, fully expanded leaves + TSP enriched	1.0	0.69	Kwabiah et al. (1999)
Cupressus lusitanica	Central highlands, Ethiopia	Senescent leaves	1.9	0.36	Lisanework and Michelsen (1994)
Dalbergia sissoo	India	—	0.11	6.3	O'Connell and Sankaran (1997)
Desmodium ovalifolium	Meta, Columbia	Litter—stored frozen (1 month) and dried (60°C)	0.03–0.052	13.3–23.1	Thomas and Asakawa (1993)
Dillenia pentagyna	Kerala, India	Litter—stored frozen (1 month) and dried (60°C)	0.33	2.1	Kumar and Deepu (1992)
Eucalyptus camaldulensis	Dehra Dun, India	Freshly fallen dry leaves	0.129	5.4	Bahuguna et al. (1990)
Eucalyptus globulus	Central highlands, Ethiopia	Senescent leaves	1.5	0.46	Lisanework and Michelsen (1994)
Eucalyptus tereticornis	Kerala, India (*ex situ* field)	Freshly fallen litter	0.061	11.2	Sankaran (1993)
	Kerala, India (laboratory)	Freshly fallen litter	0.045	15.4	Sankaran (1993)
Eucalyptus spp.	Different locations	—	0.075	9.2	O'Connell and Sankaran (1997)
Euphatorium innulifolium	Highlands of Sri Lanka (tea plantations)	Pruned foliage	0.460	1.5	De Costa and Atapattu (2001)
	Highlands of Sri Lanka (tea plantations)	Pruned stem	0.274	2.5	De Costa and Atapattu (2001)
Flemingia congesta	Highlands of Sri Lanka (tea plantations)	Pruned foliage	0.145	4.8	De Costa and Atapattu (2001)
	Highlands of Sri Lanka (tea plantations)	Pruned stem	0.098	7.1	De Costa and Atapattu (2001)
	Chipata, Eastern Zambia (maize field)	Fresh leaves	0.126	5.5	Mwiinga et al. (1994)
	Abidjan, Ivory Coast (in the open area)	Fresh leaf mulch with petioles	0.395	1.75	Budelman (1988)
Gliricidia sepium	Highlands of Sri Lanka (tea plantations)	Pruned foliage	0.70	0.99	De Costa and Atapattu (2001)

TABLE 10.2 (Continued)

Monthly Litter Decay Rate Coefficients (*k*) and Half-Lives of Tropical and Subtropical Tree, Shrub, and Herbaceous Species in Plantations and Agroforestry Systems as Studied by the Litterbag Technique

Species	Location and Land Use Practice	Materials Used	*k*	Half-Life (Months)	Source
Gliricidia sepium	Highlands of Sri Lanka (tea plantations)	Pruned stem	0.25	2.7	De Costa and Atapattu (2001)
	Papua, New Guinea	Mixture of new and old leaves; oven dried at 70°C	0.10	6.9	Hartemink and O'Sullivan (2001)
	Chipata, Eastern Zambia (maize field)	Fresh leaves	0.52	1.3	Mwiinga et al. (1994)
	Abidjan, Ivory Coast (in the open)	Fresh leaf mulch with petioles	0.94	0.73	Budelman (1988)
Glycine max	Central Taiwan (68 days old green manure crop)	Incorporated during wet season on raised beds	0.33	2.09	Thönnissen et al. (2000)
	Central Taiwan (68 days old green manure crop)	Incorporated—but low beds	0.44	1.58	Thönnissen et al. (2000)
	60 days old green manure crop	Incorporated—dry season raised beds	0.29	2.41	Thönnissen et al. (2000)
	60 days old green manure crop	Incorporated—dry season low beds	0.31	2.27	Thönnissen et al. (2000)
	Luzon, Philippines (74 days old green manure crop)	Incorporated—dry season low beds	0.12	5.75	Thönnissen et al. (2000)
	Central Taiwan (68 days old green manure crop)	Mulched, wet season, raised beds	0.21	3.25	Thönnissen et al. (2000)
	Central Taiwan (68 days old green manure crop)	Mulched, wet season, low beds	0.18	3.96	Thönnissen et al. (2000)
	Central Taiwan (60 days old green manure crop)	Mulched, dry season, raised beds	0.11	6.06	Thönnissen et al. (2000)
	Central Taiwan (60 days old green manure crop)	Mulched, dry season, low beds	0.09	8.02	Thönnissen et al. (2000)
	Luzon, Philippines (74 days old green manure crop)	Mulched, dry season, low beds	0.08	8.76	Thönnissen et al. (2000)
Grewia tiliifolia	Kerala, India (moist forest)	Freshly fallen leaves	0.34	2.0	Kumar and Deepu (1992)
Imperata cylindrica	Papua, New Guinea	Mixture of new and old leaves; oven dried at 70°C	0.06	11.5	Hartemink and O'Sullivan (2001)
Indiogofera tinctoria	Central Taiwan (68 days old green manure crop)	Incorporated during wet season on raised beds	0.32	2.14	Thönnissen et al. (2000)

(*continued*)

TABLE 10.2 (Continued)

Monthly Litter Decay Rate Coefficients (*k*) and Half-Lives of Tropical and Subtropical Tree, Shrub, and Herbaceous Species in Plantations and Agroforestry Systems as Studied by the Litterbag Technique

Species	Location and Land Use Practice	Materials Used	*k*	Half-Life (Months)	Source
	Central Taiwan (68 days old green manure crop)	Incorporated during wet season on low beds	0.32	2.17	Thönnissen et al. (2000)
Indigofera tinctoria	60 days old green manure crop	Incorporated—dry season, raised beds	0.38	1.82	Thönnissen et al. (2000)
	60 days old green manure crop	Incorporated—dry season, low beds	0.43	1.63	Thönnissen et al. (2000)
	Luzon, Philippines (74 days old green manure crop)	Incorporated—dry season, low beds	0.32	2.20	Thönnissen et al. (2000)
	Central Taiwan (68 days old green manure crop)	Mulched, wet season, raised beds	0.12	5.81	Thönnissen et al. (2000)
	Central Taiwan (68 days old green manure crop)	Mulched, wet season, low beds	0.14	5.13	Thönnissen et al. (2000)
	Central Taiwan– (60 days old green manure crop)	Mulched—dry season, raised beds	0.20	3.52	Thönnissen et al. (2000)
	Central Taiwan– (60 days old green manure crop)	Mulched—dry season, low beds	0.18	3.87	Thönnissen et al. (2000)
	Luzon, Philippines (74 days old green manure crop)	Mulched—dry season, low beds	0.07	9.65	Thönnissen et al. (2000)
Juniperus procera	Central highlands, Ethiopia	Senescent leaves	2.3	0.30	Lisanework and Michelsen (1994)
Lantana camara	Maseno, Kenya (*ex situ* field)	Fresh, fully expanded leaves	2.04	0.34	Kwabiah et al. (2001)
Leucaena leucocephala	Palakkad, Kerala, India	Freshly fallen leaves	0.29	2.4	Jamaludheen and Kumar (1999)
	Palakkad, Kerala, India (silvipasture)	Freshly fallen leaves	0.51	1.3	George and Kumar (1998)
	Chipata, Eastern Zambia (in a maize field)	Fresh leaves	0.39	1.76	Mwiinga et al. (1994)
	Abidjan, Ivory Coast (in the open)	Fresh leaf mulches with petioles	0.67	1.03	Budelman (1988)
Macaranga peltata	Thrissur, Kerala, India (*ex situ* field)	Leaves	0.41	1.68	Kunhamu (1994)
Mikania micrantha	Thrissur, Kerala, India (*ex situ* field)	Fresh biomass	0.478	1.45	Abraham (1999)
	Thrissur, Kerala, India (*ex situ* field)	Oven dried at 80°C for 1 day	0.418	1.66	Abraham (1999)
Paraserianthes falcataria	Thrissur, Kerala, India (*ex situ* field)	Freshly fallen litter	0.139	5.0	Sankaran (1993)
	Thrissur, Kerala, India (laboratory)	Freshly fallen litter	0.08	8.6	Sankaran (1993)

TABLE 10.2 (Continued)
Monthly Litter Decay Rate Coefficients (*k*) and Half-Lives of Tropical and Subtropical Tree, Shrub, and Herbaceous Species in Plantations and Agroforestry Systems as Studied by the Litterbag Technique

Species	Location and Land Use Practice	Materials Used	*k*	Half-Life (Months)	Source
	Palakkad, Kerala, India	Freshly fallen leaves	0.20	3.4	Jamaludheen and Kumar (1999)
Percopsis angolensis	Chipata, Eastern Zambia (maize field)	Fresh leaves	0.134	5.16	Mwiinga et al. (1994)
Phyllanthus emblica	Palakkad, Kerala, India	Freshly fallen leaves	0.18	3.9	Jamaludheen and Kumar (1999)
Phyllostachys pubescens	South China	Fallen leaves dried at 85°C for 48 h	0.034–0059	11.7–20.3	Maoyi et al. (1990)
Pinus kesiya	Arunachal Pradesh, India	Freshly fallen foliage	0.107	6.5	Arunachalam et al. (1998b)
Piper aduncum	Papua New Guinea	Mixture of new and old leaves oven dried at 70°C	0.119	5.8	Hartemink and O'Sullivan (2001)
Populus deltoides	India	—	0.087	8.0	O'Connell and Sankaran (1997)
Pongamia pinnata	Kerala, India (*ex situ* field)	Leaves	0.28	2.46	Kunhamu (1994)
Pterocarpus marsupium	Palakkad, Kerala, India	Leaves	0.25	2.8	Jamaludheen and Kumar (1999)
	Thrissur, Kerala, India (moist forest)	Leaves	0.44	1.6	Kumar and Deepu (1992)
Pueraria phaseoloides	Meta, Columbia	Litter—stored frozen (1 month) and dried (60°C)	0.033–0.06	11.5–21.0	Thomas and Asakawa (1993)
Quercus dealbata	Arunachal Pradesh, India	Freshly fallen foliage	0.073–0.103	6.7–9.5	Arunachalam et al. (1998b)
Quercus griffithii	Arunachal Pradesh, India	Freshly fallen foliage	0.115	6.0	Arunachalam et al. (1998b)
Rhododendron arboreum	Arunachal Pradesh, India	Freshly fallen foliage	0.064	10.8	Arunachalam et al. (1998b)
Schima khasiana	Arunachal Pradesh, India	Freshly fallen foliage	0.088	7.9	Arunachalam et al. (1998b)
Schleichera oleosa	Thrissur, Kerala, India (*ex situ* field)	Leaves	0.47	1.48	Kunhamu (1994)
Senna spectabilis	Highlands of Sri Lanka (tea plantations)	Pruned foliage	0.715	0.97	De Costa and Atapattu (2001)
	Highlands of Sri Lanka (tea plantations)	Pruned stem	0.179	3.9	De Costa and Atapattu (2001)
	Masena, Kenya (*ex situ*; field)	Fresh, fully expanded leaves	1.49	0.465	Kwabiah et al. (2001)
Sesbania sesban	Masena, Kenya (*ex situ*; field)	Fresh, fully expanded leaves	1.28	0.54	Kwabiah et al. (2001)
	Masena, Kenya (*ex situ*; field)	Fresh, fully expanded leaves + urea enriched	1.98	0.35	Kwabiah et al. (1999)

(*continued*)

TABLE 10.2 (Continued)

Monthly Litter Decay Rate Coefficients (*k*) and Half-Lives of Tropical and Subtropical Tree, Shrub, and Herbaceous Species in Plantations and Agroforestry Systems as Studied by the Litterbag Technique

Species	Location and Land Use Practice	Materials Used	*k*	Half-Life (Months)	Source
	Masena, Kenya (*ex situ*; field)	Fresh, fully expanded leaves + TSP enriched	1.25	0.56	Kwabiah et al. (1999)
Sesbania sesban	Chipata, Eastern Zambia (in a maize field)	Fresh leaves	0.39	1.76	Mwiinga et al. (1994)
Shorea robusta	Dehra Dun, India	Freshly fallen dry leaves	0.073	9.5	Bahuguna et al. (1990)
	India	—	0.126	5.5	O'Connell and Sankaran (1997)
Stylosanthes capitata	Meta, Columbia	Litter—stored frozen (1 month) and dried (60°C)	0.082–0.146	4.7–8.45	Thomas and Asakawa (1993)
Stylosanthes guianensis	Meta, Columbia	Litter—stored frozen (1 month) and dried (60°C)	0.027–0.066	10.5–25.7	Thomas and Asakawa (1993)
Tectona grandis	India (Different locations)	—	0.135	5.1	O'Connell and Sankaran (1997)
	Thrissur, Kerala, India (*ex situ*; field)	Freshly fallen litter	0.167	4.2	Sankaran (1993)
	Thrissur, Kerala, India (*ex situ*; field)	Freshly fallen litter (laboratory)	0.117	5.9	Sankaran (1993)
	Thrissur, Kerala, India (moist forest)	Freshly fallen leaves, in situ field	0.32	2.2	Kumar and Deepu (1992)
Terminalia paniculata	Thrissur, Kerala, India (moist forest)	Freshly fallen leaves, in situ field	0.29	2.4	Kumar and Deepu (1992)
	Thrissur, Kerala, India (*ex situ*; field)	Leaf	0.56	1.24	Kunhamu (1994)
Theobroma cacao	Thrissur, Kerala	Fresh litter	0.21	3.28	Sreekala (1997)
Tithonia diversifolia	Highlands of Sri Lanka (tea plantations)	Pruned foliage	0.615	1.13	De Costa and Atapattu (2001)
	Highlands of Sri Lanka (tea plantations)	Pruned stem	0.238	2.9	De Costa and Atapattu (2001)
	Maseno, Kenya (*ex situ*; field)	Fresh leaves	1.92	0.36	Kwabiah et al. (2001)
Xylia xylocarpa	Kerala, India (moist forest)	Freshly fallen foliage	0.35	2.0	Kumar and Deepu (1992)
Tropical ever green forests	Manaus, Brazil (forest)	Dead boles	0.167	4.1	Chambers et al. (2000)
Dry tropical forests	Yucatan peninsula, Mexico	Dead boles	0.197	3.5	Harmon et al. (1995)

Note: Monthly decay rate constants (*k*) were computed using the single exponential model (Olson, 1963), wherever it was not mentioned in the source reference *k* values mentioned in other time intervals were converted to a monthly basis for uniformity. All studies other than those described as "*ex situ*," "laboratory," and so on were conducted under in situ field situations (in pure or mixed stands, as the case may be); information not available.

be deduced that N level of litter has a rate-regulating effect. As litter decomposes, N concentration in the litter increases, often in proportion to the mass loss (Gessner and Chauvet, 1994). Thomas and Asakawa (1993), therefore, contended that lignin-to-N ratio is a better indicator of decomposition in nine green manure crops than N or lignin concentrations.

Notwithstanding the lignin and N control of the decay process, polyphenols bind proteins and the soluble organic N released from leaves. It also forms resistant complexes (Northup et al., 1995) and inhibits enzyme action (Swain, 1979). Phenolic compounds, particularly tannins (Stout, 1989), are thus important regulators of litter decay. Constantinides and Fownes (1994) confirmed this for fresh leguminous leaves and leaf–stem mixtures. They reported that soluble polyphenols or polyphenol-to-N ratios, correlated better with N release, despite (lignin + polyphenol): N ratio being the best index.

Nevertheless, before confirming the position regarding which structural chemistry parameter is the best predictor of decomposition, more species with a wider range of quality attributes should be analyzed. Accordingly, data have been compiled on initial lignin concentrations, N contents, lignin-N-ratios, and decay rate coefficients of 74 tropical species or mulch materials, which represent a disparate range of N and lignin contents. The interrelationships between the chemical parameters and decay rate coefficients (Figure 10.3) show that lignin control of the decomposition process is profound at levels above 15%, despite some “noise” in the data set. Similarly, N content of litter is likely to favor the decay only if it is above 3%, implying that chemical attributes regulate litter decay only at high initial levels.

One plausible explanation for the lack of any robust relationships (modest R^2 values) between structural chemistry attributes and decay rate coefficients, however, is the intersite variations in biochemical quality of litter. Variations in litter N, lignin, and polyphenol concentrations abound because of differences in germplasm, eco-climatic conditions (Johansson, 1994; Hartemink and O’Sullivan, 2001), seasonal variations (George and Kumar, 1998; Jamaludheen and Kumar, 1999), stand age, evergreen versus deciduous nature of the trees (Thomas and Asakawa, 1993), and the relative proportion of tender leaves and less-lignified stems in the mulch (Thönnissen et al., 2000).

Yet another critical concern is how the changing atmospheric CO_2 levels may alter the litter chemistry and how this would influence the decomposition process (Norby et al., 2001; Kumar et al., 2005). Owing to a small but predictable decline in litter N concentrations and increased lignin content, a decline in litter decay rates is probable under elevated CO_2 levels. Experimental evidences in this respect, mainly from the temperate regions, however, have been mixed, probably depending on factors such as litter chemistry (Van Ginkel et al., 1996; Norby et al., 2001) and length of the study period (Ball and Drake, 1997).

10.6.2 SITE QUALITY AND EXOGENOUS NUTRIENT ADDITIONS

Edaphic factors such as pH, nutrient availability, aeration, salinity levels, structure, and texture profoundly influence the organic matter decomposition process (Kumar and Goh, 2000). Tropical soils, which are particularly deficient in N and P, can potentially limit the microbial activity and thus the decomposition process (Berg, 1986). In view of this, Lethbridge and Davidson (1993) recommended exogenous nutrient supply to stimulate decomposition and mineralization. Effect of exogenous additions of N and P, however, may be species dependent (Kwabiah et al., 1999). For instance, decay rates of sesbania and croton were more enhanced by N addition than by P, whereas those of *Calliandra* were greater when P was added (Table 10.2), presumably because of variations in the intrinsic chemical nature of litter. Yet again, litter decomposition rates are more in neutral than in acid soils and hence liming acid soils is often recommended to accelerate litter decay (Condron et al., 1993).

10.6.3 TEMPERATURE AND SOIL MOISTURE

The Q_{10} (temperature quotient) values for N mineralization rate of native SOM in the temperature range of 5°C–35°C have been reported to be ~2 (Scholes et al., 1994), indicating a favorable effect

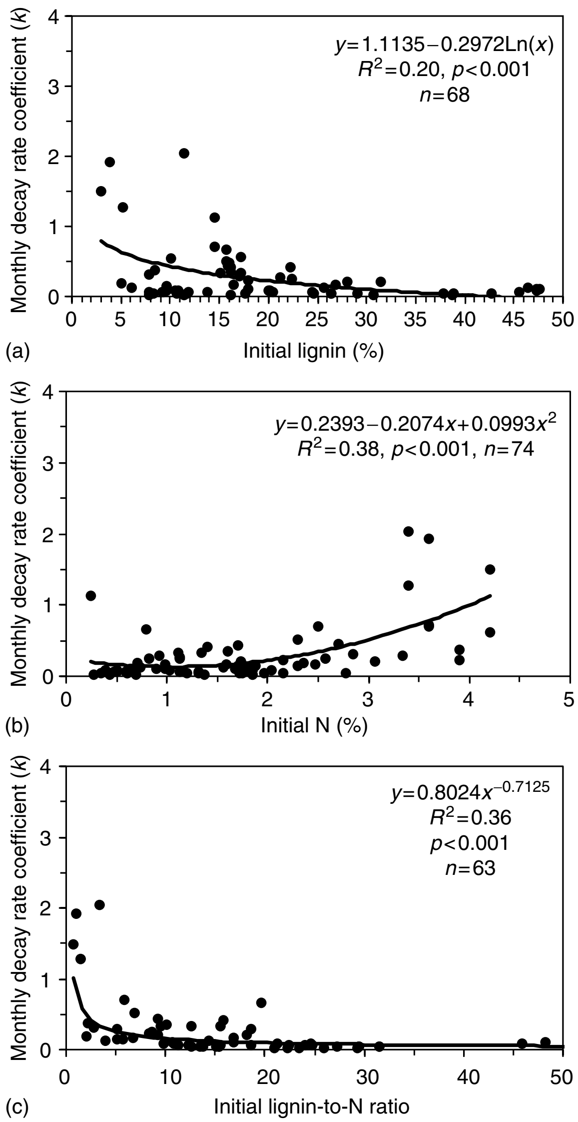

FIGURE 10.3 Relationships between monthly decay rate coefficients and initial lignin concentration (a), N concentration (b), and lignin-to-N ratio (c) of tropical litter. (Available data from various sources with best fitting equations are presented; From Kumar, B.M. and J.K. Deepu, *For. Ecol. Manage.*, 50, 181, 1992; Thomas, R.J. and N.M. Asakawa, *Soil Biol. Biochem.*, 25, 1354, 1993; Kunhamu, T.K., *Nutrient Content and Decomposition of Leaf Biomass of Selected Woody Species*, M.Sc. (For.) Thesis, Kerala Agricultural University, Thrissur, India, 1994; George, S.J. and B.M. Kumar, *Int. Tree Crops J.*, 9, 267, 1998; Arunachalam, A., K. Maithani, H.N. Pandey and R.S. Tripathi, *Forest Ecol. Manag.*, 109, 151, 1998b; Jamaludheen, V. and B.M. Kumar, *Forest Ecol. Manage.*, 115, 1, 1999; De Costa, W.A.J.M. and A.M.L.K. Atapattu, *Agroforestry Syst.*, 51, 201, 2001; Hartemink, A.E. and J.N. O'Sullivan, *Plant Soil*, 230, 115, 2001; Kwabiah, A.B., N.C. Stoskopf, R.P. Voroney and C.A. Palm, *Biotropica*, 33, 229, 2001.)

of temperature on the decay process. However, the temperature effect is moderated by litter or soil moisture content (Mugendi and Nair, 1997; De Costa and Atapattu, 2001). Consistent with this, many authors reported higher litter mass loss rates during the rainy season than during dry seasons (Facelli and Pickett, 1991; Thomas and Asakawa, 1993; Thönnissen et al., 2000).

In addition, there is also an ongoing debate about the impact of rising atmospheric concentrations of CO_2 on litter decomposition rates (Norby et al., 2001; Kumar et al., 2005 and the references therein). Presumably, the average global surface temperature would rise by 0.6°C–2.5°C in the next 50 years and 1.4°C–5.8°C in the next 100 years, with significant regional variations (USEPA, 2001). As regards the impact of this on litter decay, most workers have predicted changes in litter decomposition based on the "litter quality" hypothesis, and the associated effects of rising temperature on decomposition have been overlooked (Kumar et al., 2005). Overall, the effect of global warming on litter decay appears to be mixed—on the one hand, decay rates would increase as a direct consequence of increased atmospheric temperatures on the physiological reaction of litter decomposing biota (De-Neve et al., 1996). However, there is a cascading effect of rising evaporation rates and the consequential changes in precipitation, soil moisture availability, and litter quality. Nevertheless, direct evidences on these are scarce, and hence it is difficult to make firm conclusions.

10.6.4 Soil Microfaunal and Macrofaunal Activity

Greater earthworm activity (Anderson and Swift, 1983), higher bacterial cell counts and fungal hyphal lengths (Swift et al., 1979), and higher termite diversity (Lavelle et al., 1993) are frequently reported in the tropics. A vast majority of these soil organisms are also heterotrophs and, therefore, a close relationship between their activity and litter dynamics is expected (Janzen et al., 1992). The activity of soil biota, however, is variable among sites, and it may give rise to spectacular variations in litter decay rates. In particular, decomposition is dominated by drilosphere systems formed by associations between earthworms and soil bacteria, where soil-moisture regimes are favorable and termitosphere systems with increasing dryness (Lavelle et al., 1993, and the references therein). Termites feed on leaf and wood litter (ingest up to 30%–70% of the aboveground production; Josens, 1983), and when present can lead to rapid disappearance of the litter mass.

10.6.5 Lower Decay Rates of Tropical Plantations Than Native Forests

There is some evidence to show that accumulation of litter on the forest floor in tropical plantations is greater than that of native forests (Brassell and Sinclair, 1983; Bernhard-Reversat, 1987). Implicit in this is a potentially lower decay rate for plantations. Consistent with this, Louzada et al. (1997)—in a comparison involving *Eucalyptus* spp. and adjacent semideciduous Brazilian forests with 20 species—reported that the heterogeneous "forest litter in forest" decayed faster ($c = -0.00360$ g day^{-1}; $p < 0.01$) than "forest litter in eucalypt" ($c = -0.00302$ g day^{-1}). A plausible explanation for the higher decay rates in natural forests than plantations is the high degree of litter heterogeneity, in addition to the changes in biophysical environment of plantations vis-á-vis natural forests (see the discussion on perturbations in Section 10.6.6). Regarding litter heterogeneity, natural forests often contain a wide spectrum of species with disparate chemical or physical attributes for litter. The admixture of such diverse litter kinds promotes decomposability of even inherently "poor" litter materials. In addition, management concerns such as sustainability of forest plantations also favor species mixtures (Binkley et al., 1992; Kumar et al., 1998b; Nichols et al., 2001). However, studies on decay dynamics of mixed species litter are rare. Most attempts to characterize decomposition using litterbag technique have used monospecific litter, except when composite litter has been sampled from the forest floor.

Furthermore, O'Connell and Sankaran (1997)—based on a comparative analysis of 56 forest and 50 plantation sites in the tropics—found that litter from exotic plantations of eucalypt, pine, and casuarina decayed more slowly than did litter from native tropical forests. They attributed this to a lower nutrient status of the litter and soil, in addition to unfavorable environmental conditions. Despite this, a comparison of the data in Table 10.2 covering a wider range of tropical species shows that monthly k values for many exotics in the peninsular India such as *A. auriculiformis*

($k = 0.16$–0.42), *A. mangium* (0.68–0.75), *Casuarina equisetifolia* (0.17–0.67), *L. leucocephala* (0.29–0.51), and *Paraserianthes falcataria* (0.20) under *in situ* stand conditions were not substantially different from those of either man-made woodlots of native species (e.g., *Artocarpus heterophyllus*: 0.22, *A. hirsutus*: 0.21, *Bridelia retusa*: 0.54, etc.) or species in the natural forests (e.g., *Pterocarpus marsupium*: 0.44, *Tectona grandis*: 0.32, *Terminalia paniculata*: 0.56, *Xylia xylocarpa*: 0.35, etc.). Interestingly, Lugo (1992), when comparing tropical tree plantations with secondary forests in Puerto Rico, found that plantations decomposed more litter on an annual basis. Similar observations were reported by Bahuguna et al. (1990) and Lisanework and Michelsen (1994) also.

To conclude, both negative and positive effects have been reported in respect of plantation litter decay rates vis-á-vis those of natural forests. Reduced activities of soil biota consequent to removal of native vegetation (Critchley et al., 1979) and a concomitant increase in soil temperature (Hegde, 1995) may explain such opposing trends. In any case, the effects are likely to be temporary in a plantation until revegetation or canopy closure. Thereafter, as the decomposer populations build up and the physical environment are ameliorated, it may approach an equilibrium commensurating with species and site characteristics. The negative effects, however, may be protracted along an edge-to-interior gradient, especially if the disturbances persist.

10.6.6 DO PERTURBATIONS REDUCE LITTER DECAY RATES?

Litter usually accumulates on the forest floor following disturbances such as timber extraction, fire, mechanical clearing, and site preparation (Cortina and Vallejo, 1994; Grigal and Vance, 2000), signifying lower decay rates after perturbations. Previous studies in the temperate region have clearly showed that prescribed fires on sites that burned 0.3 and 12 years earlier (Monleon and Cromack, 1996) reduced litter decay, owing to the combined effects of fire on forest floor reduction, losses of understory plants, and possible changes in the patterns of fine root and mycorrhizal colonization. High-intensity disturbances, in particular, are negatively correlated with decomposer activities and may increase soil temperatures and evaporation rates. Consistent with this, Kumar and Deepu (1992) found the less-disturbed native forest sites with high stand basal area having faster decomposition rates than the more disturbed ones.

10.6.7 NATURE OF DECOMPOSING MATTER AND ITS PROCESSING

Most studies deal with senescing or freshly fallen leaf litter (Table 10.2). Handpicked leaves (Hartemink and O'Sullivan, 2001), leaves; and stems of prunings (De Costa and Atapattu, 2001) and herbaceous plant leaves cut at ground level (Hartemink and O'Sullivan, 2001) are also frequently used. It is, however, hard to ascertain what has been inferred as litter in some studies, but it is obvious that the dataset (Table 10.2) included leaf material that did not senesce normally or was picked off the plant before senescence was full. Fresh litter is very different from older, partly decomposed litter. Pruned leaves, by virtue of their nutrient-rich nature, are expected to decay faster than senescing leaves, from which substantial part of the foliar bioelements have been already reabsorbed. Furthermore, only little attention has been paid to the coarse litter fractions, specifically the stems and branch woods, which comprise a significant proportion of the litter of several species, although most of it is removed as firewood.

An additional source of discrepancy in the litter decay data reported in the literature may be the pretreatment of litter samples. Leaves for the litterbags are sometimes washed with distilled water and oven dried (Palm and Sanchez, 1991; Mwiinga et al., 1994; Mafongoya et al., 1998; Hartemink and O'Sullivan, 2001). Air drying or withering under the sun (De Costa and Atapattu, 2001) before packaging into the litterbags is also common. Oven drying, although increases the homogeneity of leaf moisture contents, is likely to retard the mass loss rates (Maoyi et al., 1990; Thomas and Asakawa, 1993; Taylor, 1998; Abraham, 1999; Table 10.2). Therefore, air drying under shade and concomitant moisture determination (at the time of transfer) is recommended for characterizing the litter decomposition dynamics. In addition, the litter mass retrieved at periodic intervals are typically washed with a fine jet of water to remove adhering soil particles or other extraneous matter, before drying and weighing (Anderson and Ingram, 1989; Hartemink and O'Sullivan, 2001 and others). Washing, however, may lead to overestimating the litter decay rates and may accelerate leaching of water-soluble carbohydrates and K.

Alternate methods include careful brushing to remove the soil particles adherent to the litter samples (Sharma et al., 1997; De Costa and Atapattu, 2001). Incomplete removal of extraneous matter and consequent underestimation of the decay process are, however, inherent problems here. Moreover, when the substrate is clayey and near saturation (e.g., marshes), it is hard to separate the fine clay particles that enter the litterbags, even by washing. Comminution and leaching of the hydrosolubles also exacerbate the problem of overestimation.

10.6.8 METHODOLOGICAL ASPECTS IN LITTER DECOMPOSITION STUDIES

Decomposition is generally evaluated by the litterbag technique, wherein known quantities of litter placed near the soil surface are allowed to decompose, and the residual mass is estimated (see reviews by Lousier and Parkinson, 1975; Woods and Raison, 1982; Kumar and Goh, 2000). Since the litterbag technique allows registering the litter mass loss in the field and the subsequent chemical and biological examination of the material, it is frequently used to obtain comparative information on different species at the same site for indexing decay rates and for evaluating treatment effects. Yet, as mentioned earlier, there is considerable variability in the procedures employed, especially in respect of the materials used for making the bags, mesh size, capacity, duration of the study, and sampling intensity.

Regarding the materials used for making litterbags, frequently, nylon bags, fiberglass screen, or polyvinyl bags 10 × 10 cm to 30 × 30 cm size (Palma et al., 1998) with 1–2 mm mesh size (Bocock et al., 1960) containing 5–20 g (dry weight) samples are used (Anderson and Ingram, 1989). Although no rigid principle governing the size of the bags or mesh size can be deduced from the available data, smaller mesh sizes in general restrict the soil–litter contact and are, therefore, likely to retard decay. Furthermore, Kwabiah et al. (2001) found that mass losses in coarse-mesh (5 mm) litterbags of 30 × 30 cm size sewn with nylon threads and fine-mesh (2 mm) bags with the opposite sides left unsealed were comparable with those of unconfined residues. This in turn suggests that mesh size need not be a constraint, if accessibility by soil invertebrates is unobstructed.

Yet another consideration is the depth of placement of litterbags. Litterbags are usually buried below the soil surface to allow maximum influence of the mesofauna and macrofauna. However, the depth to which such burials are made is often variable, for example, ~1 cm (Kwabiah et al., 2001), 2 cm (Jamaludheen and Kumar, 1999), 5 cm (De Costa and Atapattu, 2001), 10 cm (Jama and Nair, 1996), slightly covered with litter (Hartemink and O'Sullivan, 2001), and so on. Louzada et al. (1997), however, tied the litterbags to nylon ropes and left it on the forest floor. Decay rates of such "mulched" materials were, however, more variable between seasons and locations than those of the "incorporated" materials, as the latter experiences a generally more favorable environment (e.g., close soil contact, adequate moisture, etc.; Thönnissen et al., 2000). Indeed, slow decomposition rates during summer are probable for aboveground positioned litter, owing to lower litter moisture contents (Russell and Vitousek, 1997). None of the studies reported, however, examined the impact of differing burial depths on decay rates. Yet, if the objective is to evaluate the decay rates of confined litter, it should be preferably placed below the soil close to the surface.

Variations in sampling intensity and sample size (number of litterbags retrieved) also abound in the literature. For example, different workers retrieved litterbags ranging from 1 (Thomas and Asakawa, 1993; Louzada et al., 1997; Hartemink and O'Sullivan, 2001) to 1 (Kwabiah et al., 2001) at a time, and up to 10 bags in certain cases (De Costa and Atapattu, 2001). The Statistical

Law of "Inertia of large numbers" indicates that larger the size of the sample, more accurate the results are likely to be. Following this rule, 10 litterbags preferably should be sampled per replicate. Duration of litter decay studies reported in the literature is also variable with a mode value of 1 year. In a recent study, however, Thönnissen et al. (2000) used periods ranging from 77 to 113 days, which may be justified, as the study aimed to characterize nutrient release patterns of mulches or green manures during the grand growth phase of the crop.

With regard to experimental conditions, many studies have been conducted in the laboratory (Palm and Sanchez, 1991; Sankaran, 1993; Hegde, 1995; Handayanto et al., 1997; Lupwyi and Haque, 1998), or under *ex situ* field conditions with no crop after the fallow (Budelman, 1988; Sankaran, 1993; Kunhamu, 1994; Mwiinga et al., 1994; Kwabiah et al., 2001). Although such studies provide comparative information on different litters and for the time course of mass loss, the decay rates reported are either underestimates or overestimates. Presence or absence of an overstory or a crop particularly alters the microenvironment and therefore, may have a discernible (positive or negative) impact on the decay rates (Mwiinga et al., 1994; Hegde, 1995). *In situ* studies are, therefore, preferred if the objective is to characterize the intrinsic decay dynamics of litter.

10.6.8.1 Modified Litterbag Technique of Bubb et al. (1998)

Experimental artifacts due to exclusion of predatory and saprophagous microarthropods and fungal vegetative structures during the decomposition of confined materials have been expressed by many (e.g., Hagvar and Kjondal, 1981; Seastedt, 1984). Bubb et al. (1998), therefore, suggested a modified litterbag technique wherein the litter is allowed to be in direct contact with the mineral soil. In this process, access by soil fauna and flora, considered paramount in the decomposition process, is ensured. This technique employs a metal cylinder (thickness 2.5 mm, diameter 40 cm, and length 5 cm), which is covered aloft by a terylene mesh (1.5 mm) to exclude further additions of litter but to allow gas and moisture exchange. Before installation, all litter and litter fragments should be removed from the area to be encompassed by the cylinder by means of a fine-hair brush. The cylinder is driven $\approx$1 cm into the ground and secured by pegs. A known quantity (~200 g dry weight) of the sample is placed evenly over the area encompassed by the cylinder. Finally, the terylene cover is sealed over the cylinder and at predetermined intervals, the contents are retrieved for estimating the residual mass.

10.6.8.2 Tethered Leaf Technique

Yet another dilemma relating to placement of the litterbags is the trampling of the site and the associated microenvironmental changes. Furthermore, the experimental objective often may be to assess the decay rates of nonconfined litter. In this context, the "tethered leaf method" has been proposed as an alternative to the conventional litterbag technique (Vitousek et al., 1994; Russell and Vitousek, 1997). To simulate natural decomposing conditions, the leaves and stems are tied to cords running parallel to the forest floor and strung 1–2 m aboveground in this method. The advantage of this method included little or no alterations in the decomposition microenvironment and there is complete access for decomposers to the leaves. The main disadvantage, however, is that when the leaves reach the fragmentation stage, decomposition cannot be distinguished from comminution, so this method is suitable only for the initial decomposition stages of leaves and for large leaves or marcescent (do not abscise) leaflets. Such studies should also be terminated when mass loss by fragmentation is evident.

Again, a rough estimate of the turnover rates of nonconfined litter can be obtained by working out the ratio of litter production and its biomass on the ground. This represents a balance between litterfall accession and the rate of litter decomposition and determines the amount of accumulated forest floor litter under steady-state conditions. Turnover rates, $k = L/(X_L + L)$ where L is the annual litterfall and X_L is the mean annual standing crop (Reiners and Reiners, 1970).

10.6.9 ANALYSIS OF LITTER DECAY DATA

The general approach to the analysis of the decomposition data is the fitting of mathematical models to estimate constants that describe the loss of mass over time. Frequently, the single exponential (Olson, 1963) and double exponential models are used (Bunnell and Tait, 1974). The use of these equations, however, requires the assumption of steady-state conditions. Most studies show that residual litter mass declines exponentially with time, and it is generally assumed that the model for constant potential weight loss (Olson, 1963) represented by the equation $\frac{x}{x^{o}} = e^{-kt}$, where x is the weight remaining at time t, x^{o} is the original mass, e is the base of the natural logarithm, k is the decay rate coefficient, and t is time, is appropriate for characterizing litter decomposition. Half-lives ($t_{0.5}$) of decomposing litter samples are also estimated from the k values as $t_{0.5} = \frac{\ln(0.5)}{-k} = \frac{0.693}{-k}$ (Bockheim et al., 1991).

The fascination for single exponential model arises from the fact that a single constant (k) characterizes the mass loss. This also seems to follow the fundamental scientific principle of parsimony, which requires that of all the models that explain the data well, one should choose the simplest. However, the single exponential model is often regarded as an oversimplification and does not account for the intrinsic complexity of litter (McClaugherty and Berg, 1987) or the role of complex heterotrophic interactions on substrate quality (Pastor et al., 1987; Seheu, 1987). When applied to a heterogeneous substrate, they tend to underestimate early rates of decay when more labile compounds are present and overestimate the speed of later stages (Hunt, 1977).

The double exponential model is, therefore, thought to explain the decomposition process better than the single exponential model (Jama and Nair, 1996). Indeed, the time course of decomposition is a "cascade" process during which a given resource is progressively transformed into a set of secondary, tertiary, or higher-order resources (Bunnell and Scoullar, 1975). It probably involves a succession of active phases separated by periods of inhibition (Lavelle et al., 1993). Most authors, therefore, describe it as a two-step process (Swift and Anderson, 1989; Kumar and Deepu, 1992; Jama and Nair, 1996; Palma et al., 1998; Chambers et al., 2000). The first step is a rapid process in which 30%–50% leaf biomass decomposes in about 3–4 months. This includes the breakdown of the labile fractions such as hydrosolubles, nonlignified cellulose, and hemicelluloses, sugars, starches, and proteins that can be rapidly utilized by decomposers to give the "rapid release phase." A second decomposition step, which is much slower than the first one, probably because of the accumulation of more recalcitrant constituents such as cellulose, fats, waxes, and tannins, which are chemically bound to native lignin in the residual mass, follows this, justifying the fitting of the double decay model.

Wieder and Lange (1982), however, reported that a double exponential decay function does not take into account of any possible transfer of labile to recalcitrant fraction, but presents exponential decay of each faction and its sum as total mass. It is, therefore, imperative that fitting decomposition models should take into consideration the variations in litter quality. Materials with high nutritional status, especially N, and less of complex organic constituents, such as lignin and tannins, are expected to follow the single exponential or linear model, whereas decomposition of other materials may be better represented by the double exponential model.

10.7 NUTRIENT RELEASE FROM DECOMPOSING LITTER

Although quick-growing trees may actively withdraw soil nutrient reserves, especially during the early phase of growth (Kumar et al., 1998a), they may act as self-nourishing systems after canopy closure, via litter dynamics. According to Gregorich and Janzen (1998), in natural ecosystems, organic matter decomposition synchronizes with plant growth, and C and other nutrients are utilized in the system with maximum efficiency. In agroecosystems, however, no single organic material can be expected to release N in perfect synchrony to match the plant demand (Berg, 2000; Kumar and Goh, 2000; Seneviratne, 2000; TSBF, 2000). Nevertheless, high-quality materials (high N, low lignin, and low polyphenol) release a large proportion of N rapidly, in advance of the main period of

N uptake by the actively growing plants. Materials of ‘‘poorer quality’’ (high lignin or high polyphenol), however, release only a smaller total proportion of their N initially, and the remainder is released at a continuous slow rate.

10.8 RESEARCH NEEDS AND CONCLUSIONS

Establishment of forest plantations and agroforests will become increasingly significant in view of the rising rates of tropical deforestation and increasing demands for wood, fuel, and fodder, in addition to the potentially favorable impact of tree growth on carbon sequestration. Most man-made forests or agroforests in the tropics, however, are managed with little or no exogenous fertilizers or manures, and their productivity is often positively related to the organic matter turnover processes within the system. Litter dynamics play an important role in the biogeochemical cycling and supplies most essential elements required for stand growth, especially after canopy closure. This chapter reveals that litter production–decay relationships in managed tropical land use systems are complex, because of the dramatic effects of site, climate, vegetation type, disturbance, and other factors that affect productivity. These effects strongly determine the ability of land managers to manipulate litter dynamics. Site-specific information is, therefore, needed on potential litter yields or decomposition rates and response to disturbances and silvicultural factors such as spacing, pruning, and harvesting; but little is known about the nutrient cycling characteristics of many tropical trees and their interrelationships with site characteristics.

Although broad-spectrum trends have been observed in respect of mass loss rates and resource quality parameters, no distinctive relationship has been developed. Tropical litterfall or decay rates reported in the literature are also inconsistent. Variability in the methodologies and approaches adopted by different workers to quantify such relationships may explain this in part. There is thus a need for standardizing the methods for characterizing mass loss rates and their principal determinants. Rising atmospheric CO_2 levels and the concomitant increase in global temperature are also likely to impact litter chemistry and decay rates; yet, little concrete information exists on these. Simulation modeling is recommended, in addition to field experimentation, to predict the complex impacts of changing CO_2 levels and the associated rise in temperatures on litter dynamics, including the effects on litter chemistry, biota, evaporation, and precipitation rates.

Perhaps the greatest need for research in the near future is on the role of litter dynamics in averting the potentially adverse effects of nutrient limitation on plantations and agroforests or associated crops. Therefore, to manage short- and long-term nutrient availability, we need to understand the mineralization and immobilization patterns of litter-held nutrients. Hence, the value of understanding the dynamics of nutrient supply in plantations or agroforests and formulating nutrient budgets cannot be overemphasized. The return of plant nutrients to the soil and the subsequent recycling via plant uptake can, however, be manipulated via choice of species, for example, less or more readily decomposable litter-yielding species. Incorporating N-rich leaf materials from N-fixing plants may lead to quick release of nutrients to the associated crops. Furthermore, owing to litter heterogeneity, mixed-species stands and agroforests usually have higher litterfall rates than monospecific stands. Hence, more information is needed on how species selection can make differences in the management strategies proposed to ensure efficient use of available natural resources.

In addition to leaf litter, nutrient accretion also occurs through wood, twigs, branch, and fruit fall, besides fine root dynamics, abnormal leaf fall, and insect frass. Leaf fall during pathogenic infestation and insect outbreaks are rich in nutrient concentrations since nutrient reabsorption did not occur. However, only very few quantitative estimates are available on most of these forms of nutrient accretion. Although much attention was focused on litterfall and decomposition, litter removal from the forest floor, typical of tropics, has been seldom considered. The impact of litter removal from natural forests or plantations or agroforests and its interrelationships with the nutrient budget of forests, therefore, needs elucidation.

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11 Developments in the Research of the Potential of Agroforestry for Sustaining Soil Fertility in Zimbabwe

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CONTENTS

11.1 INTRODUCTION

Poor soil fertility is one of the major problems facing smallholder farmers in Zimbabwe (Smaling et al., 1997). Most of the farmers are located on granitic sandy soils that are inherently low in soil organic matter and nutrients essential for crop production, particularly N (Grant, 1976). Incorporation of plant residues in agricultural soils is a useful means to sustain soil organic matter content, and thereby enhance the biological activity, improve physical properties, and increase nutrient availability (Palm et al., 2001). Legume remains and animal manures form a potentially important source of nutrients for crop production in smallholder agriculture in sub-Saharan Africa.

The potential of these resources to contribute nutrients, especially N, for other crops is highly dependent on their N content and its release characteristics with respect to demand for uptake by the crops. Decomposition and N release from organic materials in the soil is influenced by several biotic and abiotic factors including the quality of residues (Swift et al., 1979; Cadisch and Giller, 1997). High-quality materials (high N content, low lignin, and polyphenols) are known to decompose quickly and could substitute mineral fertilizers in supplying N for annual crops (Mafongoya et al., 1998b). However, the quantity of high-quality legume materials on smallholder farms in southern Africa is pitifully small. Senesced legume materials are largely utilized. Because of retranslocation of nutrients, senesced leaves or litter have lower N and higher C:N ratio, are more lignified, and may show different mineralization rates compared with the fresh material. Fresh legume prunings (leaves and twigs) are most widely studied for decomposition patterns (e.g., Mafongoya et al., 1998a), and apparently little attention has been paid to N mineralization by the more available litter component.

Application of prunings from different agroforestry tree species has been advocated for as a potential alternative to alleviate the problem of nitrogen deficiency. One of the limitations associated with the use of pruning to supply of N to maize is the poor synchrony between N mineralization from the prunings and the demand by maize at different stages of crop development. When N release and maize demand do not match, there may be periods of temporal shortage or excess, even though the total amount of N mineralized at the end of the season would be sufficient to meet the total demand by the maize crop (Myers et al., 1997). This situation results in very low N-use efficiencies, particularly on the poorly buffered sandy soils, as much of the excess N mineralized during the periods of excess supply is lost through leaching.

Synchrony has been an issue of central focus in organic matter managed in sub-Saharan Africa over the last years, as it has been realized that there are limited opportunities for the smallholder farmers to produce sufficient organic resources to fertilize crops; hence, it is imperative that the little organic resources available are used efficiently (Myers et al., 1994; Palm et al., 2001). To address the problem of poor synchronization in agroforestry systems in Zimbabwe, several strategies have been tested under field, greenhouse, and laboratory conditions to explore the potential to improving the match between release of N from prunings with maize demand.

Section 11.3 discusses the capacity of various legume types, including agroforestry tree species, to cycle N and benefit rotational maize under the different edaphic conditions. Further, we provide a synthesis of the problems associated with legume materials as sole source of N for cereal production on smallholder farms. A detailed N budget for a case study of an improved fallow system with *Sesbania sesban* is given. The Section 11.4 presents advances made in developing strategies for improving N recovery by maize in agroforestry systems through (1) timing application of prunings, (2) methods of pruning application, and (3) manipulation of N-release patterns through mixing prunings of different quality. In Section 11.5, we report on the potential of improved fallows to improve soil physical factors. Fallows have been reported to increase crop yields, and this has been almost exclusively attributed to increased nitrogen availability. However, recent research work has shown that improved yields also stem from improved soil physical factors.

11.2 STUDY SITE

The results reported in this section are mainly drawn from experiments done at the International Centre for Research in Agroforestry (ICRAF) research station located at Domboshawa, Zimbabwe, or in greenhouse and laboratory experiments, on the soil taken from this site. Domboshawa is located 30 km northeast of Harare at 17°35′ S latitude and 31°14′ longitude. The soil at the site is a sandy clay loam, with 22% clay content and 73% sand in the top 20 cm. The CEC is 2 $\text{cmol}_c\ \text{kg}^{-1}$, and pH (0.01 M $CaCl_2$) is 4.9. When experiments were done at other sites, description of the sites is given in the relevant sections. The site experiences subhumid moisture regime with annual rainfall ranging between 750 and 1000 mm. Annual temperatures average 23°C.

11.3 N CYCLING IN AGROFORESTRY AND LEGUME-BASED CROPPING SYSTEMS

11.3.1 Quantifying N Flows under Legume Systems

Two field experiments were conducted at two sites with different soil texture: The first one was located at the field station of the ICRAF located at Domboshawa, Zimbabwe. Two maize crops were planted after a 2 year period under improved fallows with (1) *Sesbania sesban*, (2) *Acacia angustissima*, (3) *Cajanus cajan*, and (4) continuous maize cropping. After cutting the legumes, all woody plant materials (>5 mm diameter) were removed from the plots for firewood, whereas twigs (<5 mm diameter), leaves, and litter were left in the plots for incorporation. Soil samples were taken at 2 week intervals to 1.2 m depth to determine nitrate N dynamics during the cropping phase and to estimate quantities of nitrate leached from the rooting depth of annual crops. The closed soil chamber technique as described by Matthias et al. (1980) was used to estimate gaseous N losses as nitrous oxide. The second experiment was at a site with 4% clay content, and it involved a range of legumes that included soyabean, *Mucuna*, *Crotalaria*, and the agroforestry species, as described for the Domboshawa site. In both experiments, biological N_2-fixation rate of the legumes was determined using the natural abundance method (Peoples et al., 1989).

11.3.2 N Mineralization in Leaching Tube Incubations

Decomposition and N mineralization of different legume litters and two fresh prunings were determined in leaching tube incubations (Stanford and Smith, 1972). Leaching tube incubations take into account the initial rapid loss of organic and mineral constituents during decomposition and allow periodic leaching from the same tube over time (Sakala et al., 2000). The treatments were fresh prunings of *Sesbania* and *Acacia*, and senesced litter of *Sesbania*, *Acacia*, soyabean, *Mucuna*, and an unamended soil as the control. The soil was a granitic-derived sandy soil from a smallholder farm in Zimbabwe with 4% clay content. All the residues were added at a rate equivalent to 100 mg N of residues kg^{-1} soil after they were analyzed for initial N, lignin, polyphenols, and protein-binding capacity of their polyphenols (Table 11.1).

The tubes were leached on days 0, 4, 8, 16, 32, 48, 64, 90, and 120 with 100 mL of leaching solution (1 mM $CaCl_2$, 1 mM $MgSO_4$, 0.1 mM KH_2PO_4, and 0.9 mM KCl) in 50 mL aliquots

TABLE 11.1
Chemical Composition of Legume Fresh Prunings and Senesced Litter Used in the Experiments

Plant Material	%[a] N	%[a] P	%[a] Lignin	%[a] Polyphenols	PBC[b]	C:N ratio	Lignin:N
Sesbania leaves	3.2	0.28	4.5	1.9	28	14	1.4
Sesbania litter	1.6	0.21	5.4	0.8	20	21	3.4
Acacia leaves	4.8	0.57	7.1	5.7	160	10	1.5
Acacia litter	1.5	0.17	24.4	1.3	29	24	16.2
Soyabean stover	1.7	0.20	12.9	0.6	18	25	7.6
Mucuna litter	1.8	0.31	11.5	3.4	52	24	6.4

[a] %, mg per mg dry weight × 100.

[b] PBC, protein-binding capacity, μg BSA mg^{-1} plant sample (BSA, bovine serum albumin).

(Cassman and Munns, 1980). The leachates were analyzed for NH_4–N and NO_3–N. Net N mineralization was calculated by subtracting N released from the unamended soil from that released by the residue-amended treatments. Nitrogen mineralization from the plant residues as a function of time ($N\mathrm{min}_t$) was calculated from the difference in cumulative amounts of mineral N between soil treated with plant materials and the control at each sampling time divided by the total residue N added.

$$N\ \mathrm{min}_t = \frac{\mathrm{Min}\ N\ (\mathrm{treat})_t - \mathrm{Min}\ N\ (\mathrm{control})}{\mathrm{Total\ residue\ N\ added}}$$

11.3.3 BIOLOGICAL N_2-FIXATION RATES AND LEGUME PRODUCTIVITY

Estimates of biological N_2 fixation on two field experiments with different texture showed that legumes fixed a large proportion of their accumulated N on both sites, but the actual amounts of the fixed N were different (Table 11.2). *Mucuna* derived at least 96% of its total N from biological N_2 fixation, whereas soybean fixed 76% of its N as determined by the ^{15}N natural abundance on a sandy soil. At the same site, *Acacia*, *Sesbania*, and *Cajanus* fixed 79%, 84%, and 65% of their N, respectively. Although N_2-fixation rates were high for these woody legume species, total N fixed was small as these legumes grew poorly and produced little biomass. Total aboveground N input to the soil by soyabean through leaf litter and stover was 28 kg ha^{-1}. After accounting for the soil N exported through seed harvest, soyabean net N input was only 8 kg ha^{-1}. This was improved when root N was estimated. As expected, *Mucuna* had the greatest net N input into the system as no N was exported from the field through seed harvest. On the clay loam soil, *Sesbania* and *Acacia* had lower N_2-fixation rates, but overall N fixed and N cycling was significantly larger (Table 11.2).

TABLE 11.2

Biomass Production, Biological N_2 Fixation, and N Input through Litter and Stover from Selected Legumes Grown on a Sandy Soil and Clay Loam Soil in Zimbabwe

Legume	Leaves/Litter (Mg ha^{-1})	Recyclable N[a] (kg ha$-^{1}$)	% N from N_2 Fixation	Net N Input[b] (kg ha^{-1})
		Sandy soil		
Soyabean	1.7	28	76	8 (+16)
Crotalaria paulina	0.2	4	46	2
Mucuna	3.9	87	96	84 (+26)
Cajanus cajan	0.4	7	65	4
Acacia angustissima	nd[c]	nd	79	nd
Sesbania sesban	nd	nd	84	nd
		Sandy clay loam		
Cowpea	2.5	48	58	17 (+13)
Cajanus cajan	5.3	115	84	82 (+21)
Sesbania sesban	5.7	152	55	38 (+29)
Acacia angustissima	9.9	218	56	69 (+60)

Source: Adapted from Chikowo, R., P. Mapfumo, P. Nyamugafata and K.E. Giller, *Agric. Ecosyst. Environ.*, 102, 119, 2004a.

[a] Aboveground plant accumulated N (soil N + N_2-fixed N) returned to the soil in form of litter and leaves.

[b] Amount of N_2 fixed and returned to soil through aboveground nonwoody components—soil-derived N exported through woody parts and grain. Values in parentheses are estimates of additional amounts of N_2 fixed in roots.

[c] nd, not determined. These legumes grew poorly on the sandy soil site.

Sesbania, *Cajanus*, and *Acacia* are legume species that have produced high subsequent maize yields (Mafongoya and Dzowela, 1999). *Acacia* is hypothesized to be especially efficient in recycling nutrients as its regrowth capabilities after fallow clearance means that it can be pruned regularly during maize cropping, and the prunings are spread in the field where they act as green manure or mulch. Herbaceous green manure legumes like *Mucuna* grown specifically for soil fertility restoration have not been widely adopted by farmers in southern Africa (Snapp et al., 1998), but there has been some success in Benin (Becker and Johnson, 1998). The lack of a direct usable food product is the principal disincentive in farmers readily adopting green manuring. In addition, unless farmers add some P fertilizer to the legume or to the cereal crop following a green manure crop, there is no outright guarantee of massive crop responses to the legume N in the soils, in which crop growth is often constrained by multiple nutrient deficiencies and unreliable rainfall (Grant, 1981). As green manures compete for land resources with other food crops, it will be important to ascertain from farmers how these fit in the farmers' cropping cycles. Soil fertility management, largely, is a function of socioeconomic processes associated with a household and its management.

11.3.4 N-Mineralization Patterns

The chemical composition of the materials used in the N-mineralization study varied widely (Table 11.1). *Acacia* had the highest concentration of N and polyphenols, and consequently the narrowest C:N ratio. Additionally, *Acacia* polyphenols had the largest activity as indicated by large protein-binding capacity. Except for *Acacia* litter, all the other materials had lignin concentrations <15%. Total N was <2% for all senesced litters, indicating large variations between the fresh and the litter materials from the same species. The % N mineralized from high-quality *Sesbania* prunings was 55% compared with 27% for the *Sesbania* litter after 120 days of incubation under leaching conditions (Figure 11.1). During the same period, fresh prunings of *Acacia* released only 12%, whereas its litter released 9% of the added N. Despite the large differences in total N concentration of the *Acacia* prunings and litter, the total mineralized N at the end of the incubation period was similar. The small proportion of N mineralized from the *Acacia* prunings was associated with the high activity of the polyphenols as indicated by the large protein-binding capacity (Table 11.1). Most of the senesced litter materials showed N immobilization between days 4 and 60 of incubation, and then net mineralization. *Sesbania* litter immobilized N for a 2 week period only and then mineralized N slowly over the next 90 days.

N mineralization was greatly depressed for *Acacia* prunings when compared with *Sesbania* prunings. *Acacia* prunings had high concentration of polyphenols, which were very active in binding protein (Table 11.1). The presence of inhibitory compounds is an important component of litter quality. Polyphenols may also interfere with enzyme function or decomposer metabolism, resulting in reduction in decomposition of other litter constituents (Paustian et al., 1997). High activity of polyphenols, as indicated by their high protein-binding capacity, has been shown to be highly correlated with reduced N-mineralization rates (Handayanto et al., 1997). Earlier, Palm and Sanchez (1991) had also attributed the differences in N release by various tropical legumes to polyphenols in the legumes, and showed that polyphenol:N ratio was a good predictor of N mineralization.

N mineralization from the fresh prunings was higher than that from litter of the same species. Much of the high energy-soluble carbon compounds that support microbial activity are translocated from leaves during senescence, before abscission and leaf fall, and litter becomes more lignified (Constantinides and Fownes, 1994). The N concentrations were also significantly lower (Table 11.1). All these factors contribute to reduced N release. In a review of decomposition and N-release patterns of tree prunings and litter, Mafongoya et al. (1998a) showed that litter materials had lower nutrient concentrations compared with green foliage of the same legume species, and this was correlated with decomposition rates.

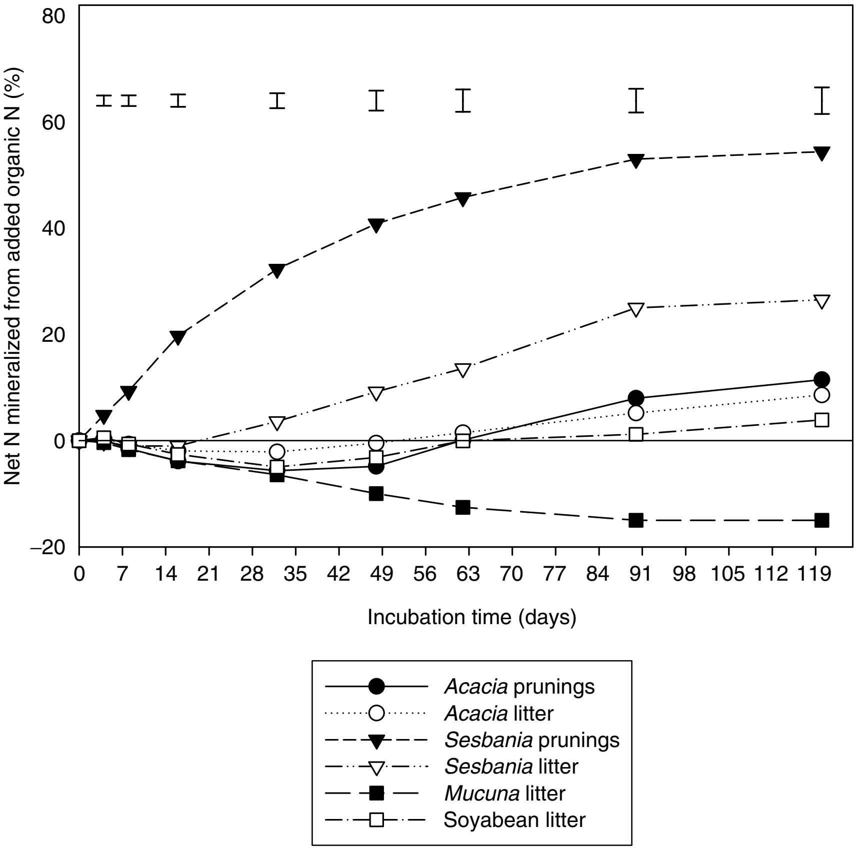


FIGURE 11.1 N release by various legume materials in a leaching tube incubation experiment. Bars represent least significant differences.

11.3.5 NITRATE LEACHING AND N_2O EMISSIONS AFTER IMPROVED FALLOWS

Nitrate N increased substantially in the topsoil at the beginning of the rains for *Sesbania* and *Acacia* plots in the two top 20 cm layers (Figure 11.2). Topsoil nitrate increased from 3.2 to 34 kg N ha^{-1} in the *Sesbania* and from 6.7 to 29 kg N ha^{-1} for *Acacia* plots during the period from preseason sampling to week 1. There were no treatment differences in nitrate concentration in the week 1 samples of the soil from the layers below 40 cm (Figure 11.2). The increase in nitrate concentration in topsoil observed for the *Sesbania* and *Acacia* treatments was not sustained as concentrations decreased rapidly only 3 weeks after planting maize. The decrease in topsoil nitrate concentration was accompanied by a relative increase at depths below 40 cm. There was a clear evidence of nitrate movement down the profile as the bulge became more pronounced with depth and time. Beyond the third week after planting, the bulk of the nitrate had moved to the 100–120 cm layer or beyond. Poor N recovery following incorporation of high-quality legume prunings, such as *Sesbania*, is partly a result of rapid N release and subsequent leaching before crop root systems sufficiently develop (Figure 11.2). We could estimate that up to 30 kg N ha^{-1} was leached from the top 40 cm in the early weeks following planting of maize. Materials with perfect synchrony of N release and plant N uptake demand are apparently not available. Mineral N availability in topsoil following legume tree-improved fallows increases significantly at the start of the rains following a long dry season. Such temporary flushes in mineral N are not directly related to the quality of legume materials as described earlier, but other N sources such as of microbial biomass turnover and labile organic matter. Conversely, senesced materials release N too slowly, and this is inadequate to sustain a high N-demanding crop.

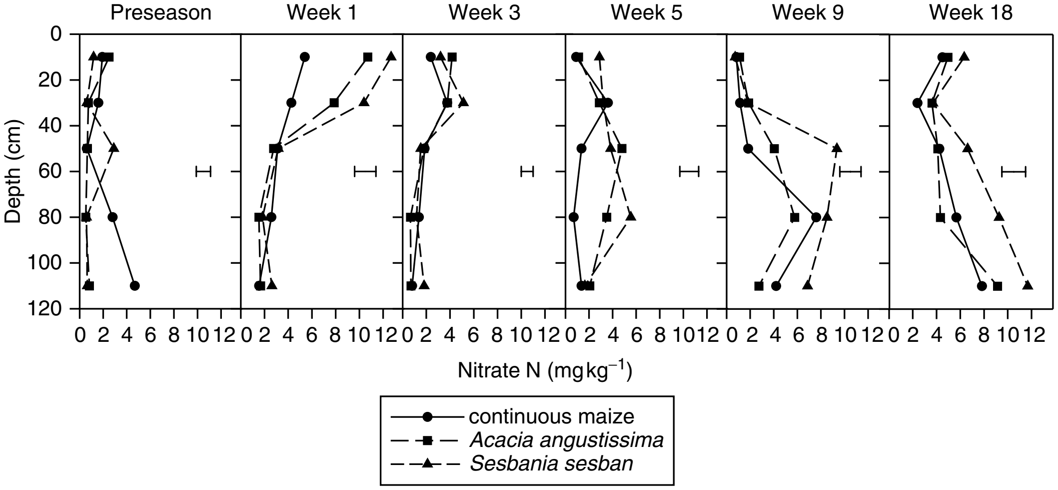


FIGURE 11.2 Nitrate N dynamics following plots that had maize, and 2 year fallows of *Acacia* and *Sesbania* at Domboshawa, Zimbabwe. Each error bar represents LSD ($p < 0.05$) for all depths for the respective sampling dates.

N_2O emissions following 2 years improved fallows on a sandy loam soil, indicating that legumes increased N_2O emissions compared with unfertilized maize monoculture, but losses were <0.5 kg ha^{-1} for a period of 56 days (Table 11.3 Chikowo et al., 2004b). Under the less-reducing conditions of the open textured soils, a large proportion of the gaseous N could have been lost in the form of nitric oxide. In the humid tropics of western Kenya, N_2O emissions on a heavier textured soil were also substantially higher following improved fallows (Table 11.3). Though legumes result in increased N_2O emissions, the quantities lost are small and would not contribute significantly to the poor N recovery that has been widely reported under organic fertilization. Although the total emissions in both cases may be of little economic consequence to total N left available to the crop,

TABLE 11.3

N_2O Emissions in Field Experiments That Involved Improved Fallows on a Sandy Loam Soil in Subhumid Zimbabwe and on a Silty Clay Loam Soil in the Humid Tropics of the Western Kenya Highlands

Location, Soil Type and Measurement Duration	Treatment	Amount of N Input (kg ha^{-1})	Total N_2O–N Emission (g ha^{-1})
Zimbabwe (sandy loam) (56 days)	Unfertilized maize	0	60
	Acacia angustissima	215	180
	Sesbania sesban	152	240
	Sesbania[a] NT	152	290
Kenya[b] (silty clay loam) (84 days)	Unfertilized maize	0	230
	Sesbania sesban	115	1940
	Sesbania/*Macroptilium*	215	4130
	Macroptilium	360	1630
	Crotalaria grahamiana	288	1860

[a] *Sesbania* NT, *Sesbania sesban* plots that were not tilled.

[b] Data for Kenya adapted from Millar et al., unpublished results.

these quantities of N_2O may be of significance to climate change, if the improved fallow technology is adopted at a wide scale.

11.3.6 NUTRIENT RELEASE DILEMMA OF ORGANIC MATERIALS

Although nutrient quality explains and predicts N release under controlled conditions (Cadisch and Giller, 1997; Handayanto et al., 1997), field environmental factors drive a substantial part of the realities that the farmers face. There are complex factors involved in the N transformations in soils once organic materials are incorporated, and quality parameters as well as the environment dictate N release. Synchronization of N mineralization and crop demand will be difficult to achieve for many organic resources as either N release precedes plant demand or it is too slow. For example, data for nitrate dynamics in the field under maize in sequence with improved fallows showed nitrate in profiles moving well in advance of the crop demand (Figure 11.2). Specifically, the rapid mineral N accumulation in *Acacia* plots after 1 week sharply contradicts its N-release characteristics as controlled by high lignin content for its litter and the high protein-binding capacity of its polyphenols for the prunings. A few weeks into the cropping season, mineral N originating from most organic materials used falls short of the amounts required to sustain high N demands by crops.

At the end of the 2 years of improved fallows, there are mixture of plant materials at various stages of decomposition and the beginning of rain triggers mineralization. Managing N release in this scenario will remain difficult. When fresh prunings are used, as in biomass transfer systems, the release of N is in most cases similar to that of litter that has been in the field for a long time. Certain tree prunings decompose too quickly, and this decomposition is not synchronized with the demands of the cultivated crops. Materials with large active polyphenol amounts like *Acacia* prunings would decompose too slowly, again in asynchrony with plant uptake (Figure 11.1). Though slow-decomposing legume materials would not supply adequate N for immediate crop requirements, promoting their use could be an opportunity to build up and maintain reasonable soil organic matter contents on sandy soils.

11.3.7 NITROGEN BUDGET FOR AN IMPROVED FALLOW SYSTEM

Figure 11.3a summarizes N cycling in an improved fallow system of *Sesbania* as related to the specific seasons we carried out our experiment. The combined N uptake by the two maize crops following *Sesbania* fallows was 45 kg ha^{-1}. When the 22 kg N ha^{-1} used by the unfertilized maize treatment in the two seasons is subtracted from Figure 11.3a, then N from *Sesbania* prunings and litter recovered by the two subsequent maize crops can be estimated to be 23 kg, representing a N recovery rate of 15% N of that applied. It should, however, be pointed out that N-use efficiency by maize was greatly depressed by cutworm infestation during the first season, and by drought during the second season. An earlier experiment on the same field resulted in high maize yields following improved fallows (Table 11.4). This kind of information is useful though, as it shows the level of risk that farmers have to cope with as legume technologies get integrated in the farming system.

For the continuous maize cropping, crop production is supported through native soil organic matter decomposition. If the decomposition coefficient is assumed to be 2%, the soil with 0.06% organic N in the top 0.2 m layer releases about 30 kg N ha^{-1} $year^{-1}$. Even with very high N-use efficiency, this amount would only support modest crop yields (Figure 11.3b). With no substantial annual organic material additions (except from weeds), the capacity of soil to supply N gradually decreases, and so does the production intensity that can be supported. We have established that gaseous N emissions are small under unfertilized conditions, and we propose that a substantial proportion of the mineralized N from soil organic matter will be leached as unfertilized maize in a P-deficient soil has a poor root system to absorb the N. Though soil losses due to erosion are higher under maize monoculture than following improved fallows, the eroded soil has poor nutrient content, and total N losses are thus small.

Although N_2-fixation rate may not be that variable, it is clear that other components of the cycle are largely a function of the quality of the cropping seasons. It may be expected that in the absence

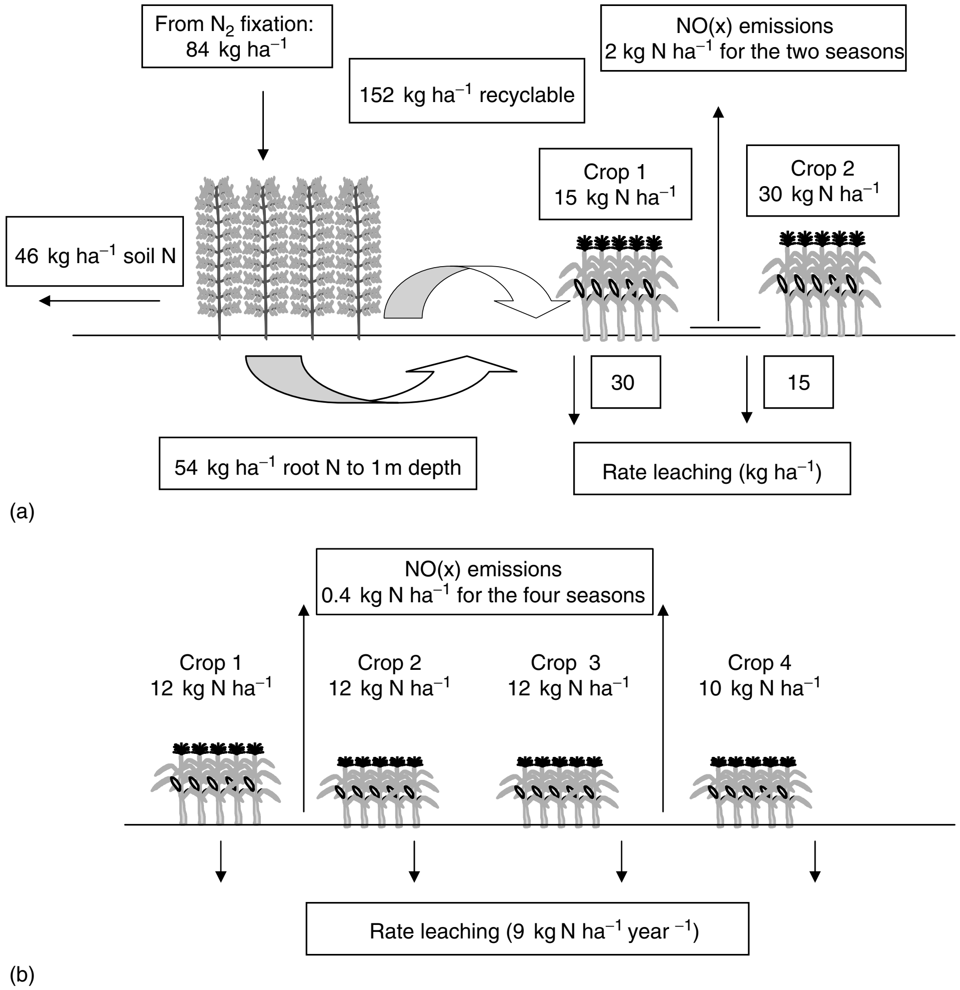


FIGURE 11.3 Nitrogen flows and use efficiency in (a) *Sesbania sesban*–maize–maize cropping sequence and (b) maize monoculture with no fertilizer application. It was assumed that the N mineralization from SOM was equal in both cases at 30 kg N ha^{-1} year^{-1} (2% of native SOM).

of pests and when rainfall is more evenly distributed, N-leaching losses will be smaller and N-use efficiency will be larger. Losses of N through erosion will also depend on the rainfall intensity, and its partitioning into infiltration and runoff. N losses due to erosion are small during immediate cropping season after fallow termination, as infiltration rates are large. However, such losses markedly increase in subsequent cropping seasons as the physical effects of improved fallows on water infiltration dynamics disintegrate (Nyamadzawo et al., 2003).

11.4 STRATEGIES FOR IMPROVING SYNCHRONY BETWEEN N RELEASE FROM PRUNINGS AND DEMAND BY MAIZE

11.4.1 Time of Application

As evoked in the previous section, N release from organic material is mainly influenced by chemical composition, soil and environmental properties (moisture and temperature), and management of prunings, including time and rate of application (Swift et al., 1979). Of these factors, timing

TABLE 11.4

Maize Grain Yields (Mg ha^{-1}) for Two Immediate Seasons of Cropping following 2 Year Improved Fallows at the Same Site, Dombosahawa, Zimbabwe

	Phase 1		Phase 2	
Legume Species	**Season 1 (1995)**	**Season 2 (1996)**	**Season 1 (2001)**	**Season 2 (2002)**
Sesbania sesban	4.9	3.7	0.67	1.30
Acacia angustissima	2.9	1.3	0.91	0.58
Cajanus cajan	3.4	3.0	1.20	1.11
Unfertilized maize	1.2	1.3	0.85	0.62
LSD (0.05)	0.4	0.5	0.18	0.14
Rainfall (mm)	672	715	1218	461

Sources: Phase 1—Adapted from Chikowo, R., P. Mapfumo, P. Nyamugafata and K.E. Giller, *Plant Soil*, 259, 315, 2004a.
Phase 2—Adapted from Mafongoya, P.L. and B.H. Dzowela, *Agroforestry Syst.,* 47, 139, 1999.

of application is easiest to control since it is directly related to farmer management practices. A greenhouse pot experiment was established to explore the interaction between the quality of prunings and the time of application (Mafongoya et al., 1996a). Prunings of three multipurpose tree species representative of poor-, medium-, and high-quality parameters were selected for use in this experiment. *Flemingia macrophylla* prunings were selected in the poor-quality category. *Flemingia* prunings decompose slowly and exhibit prolonged N-immobilization patterns because of highly lignified leaves and a high polyphenol concentration. In the medium-quality pruning category, *Calliandra calothyrsus* prunings decompose slowly and have low net N mineralization due to polyphenols with a high protein-binding capacity (Mafongoya et al., 1998b). A species with high-quality parameters selected for use in the study was *Cajanus*. This species has a high N content and polyphenols with a low protein-binding capacity and as a result release N at high rates.

Oven-dried prunings were applied to the pots containing 7.8 kg of soil at rates equivalent to 5 t ha^{-1} at planting, 2 weeks after planting, and 4 weeks after planting. Maize plants were harvested for biomass determination at 6 week and 3 week intervals thereafter up to 18 weeks. To remove the limiting effects of P and K, P was applied as single super phosphate at 40 kg ha^{-1} and K was applied at 60 kg ha^{-1} as muriate of potash to all treatments.

Results from this experiment showed that at the different times of application, uptake of N by maize was highest from *Cajanus* prunings followed by *Calliandra* and *Flemingia* (Figure 11.4). Uptake of N from *Cajanus* and *Calliandra* prunings was highest when the prunings were applied at planting and least when they were applied later at 4 weeks after planting. Uptake of N from the poor-quality *Flemingia* pruning was low at all application times, and there was no significant difference in maize N uptake across all pruning application times.

The results indicated that the influence of application time of prunings on N uptake by maize was strongly dependant on the quality of prunings. Application of the high- and medium-quality pruning at planting led to the highest nitrogen use efficiency, which was most likely due to them releasing most of their N before the period of maximum requirement of the N by the maize crop. Maize growth and N uptake are slow up to 6 weeks after emergence; the demand for N by the crop then increases thereafter until the time of tussling. Application of these pruning at 2 and 4 weeks after planting could have resulted in low accumulation of N during the period of high demand, leading to the overall low N uptake and use efficiency. These results were consistent with those obtained in other field studies that also showed that N was used most efficiently when prunings were applied at planting (Mulongoy et al., 1993). The low N uptake from the *Flemingia* prunings at all

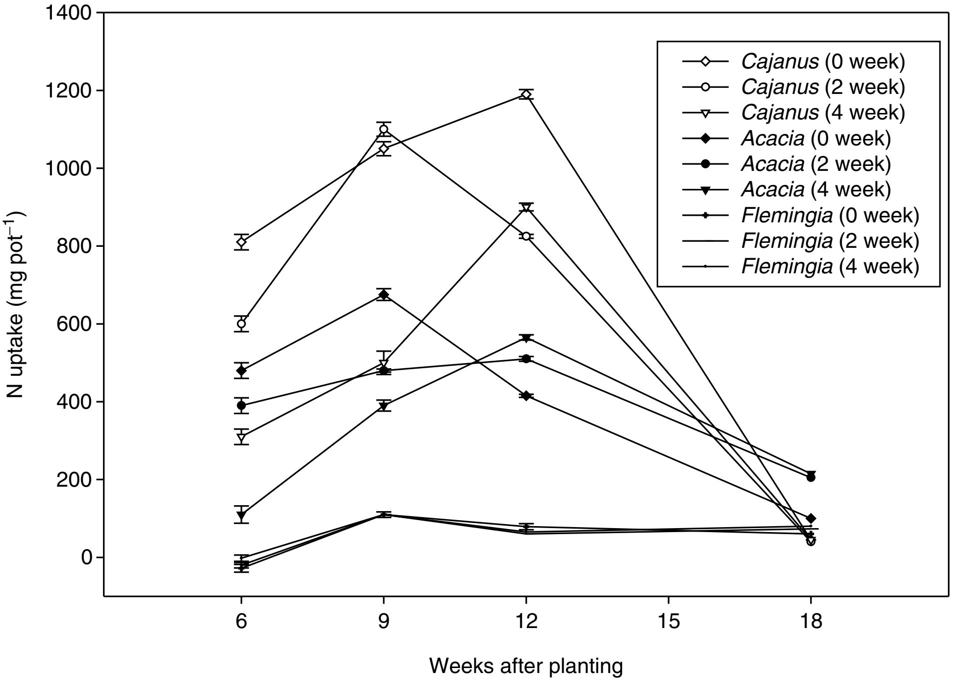


FIGURE 11.4 Nitrogen uptake by maize as affected by pruning type and time of planting at Domboshawa, Zimbabwe. (From Mafongoya, P.L., P.K.R. Nair and B.H. Dzowela, *Agroforestry Syst.*, 35, 57, 1996a.)

application times were attributed to the poor quality of these prunings, which could have resulted in initial immobilization of N and overall low net N mineralization.

The results from the greenhouse pot experiment gave a strong case for evaluating the effect of timing of pruning application for improving uptake of N by maize under more practical field conditions (Mafongoya et al., 1996b). Under field conditions, decomposition is subject to fluctuating moisture as opposed to optimal moisture maintained in greenhouse pot experiment. In addition, under field conditions, mineralized N is prone to leaching, unlike the greenhouse where leaching is not a factor. Application of *Calliandra* prunings at planting led to the higher N uptake and grain yield compared with late application, which was consistent with the results from the greenhouse pot experiment. However, influence of time of application of *Leucaena leucocephala* prunings (high quality, similar with *Cajanus*) on N uptake and grain yield was not significant. It can therefore be deduced that application of slow-decomposing *Calliandra* prunings at planting ensured that the period of maximum N release coincides with the period of maximum demand. As for the fast-decomposing *Leucaena* prunings, rapid release of N could have taken place before the period of peak demand leading to high losses, irrespective of early or late application.

11.4.2 METHOD OF PRUNING APPLICATION

Further field studies were conducted to evaluate the interaction between the quality of prunings and two methods of application, that is, surface and incorporation (Mafongoya and Nair, 1996). It was hypothesized that incorporation of prunings may improve N uptake by maize from low-quality pruning as it enhances biomass decomposition. In addition, incorporation of prunings can potentially reduce N losses by volatilization. A field experiment in which prunings from of five leguminous tree species considered appropriate for Zimbabwe conditions were tested was conducted at Domboshawa. Two of the species were in the high-quality category (*Cajanus* and

Leucaena) and two in the medium category (*Acacia* and *Calliandra*), whereas the fifth was the poor-quality Miombo woodland prunings dominated by *Brachystegia spiciformis*. The prunings were applied at a rate of 5 t ha^{-1} at the time of planting. The prunings were either surface applied or incorporated into the top 15 cm using hand hoes.

Maize grain yield was strongly influenced by the method of application, and the method of application had contrasting effects depending on the quality of the prunings (Figure 11.5). Incorporation of the *Acacia*, *Cajanus*, and *Leucaena* gave higher yield than surface application. On the contrary, incorporating the *Calliandra* and Miombo woodland prunings had no significant effect on maize grain yields. The improved maize grain yields when *Acacia*, *Cajanus*, and *Leucaena* were incorporated were related to higher decomposition rates of these prunings due to enhanced contact of the prunings with the soil. This made larger amount of N available for maize uptake and thus the higher grain yields. Increased maize yield when the prunings were incorporated may also be attributed to reduced N losses through volatilization. It should, however, be stressed that the influence of incorporation of pruning may differ depending on the intensity of rainfall, particularly during the initial weeks of the season.

11.4.3 MANIPULATION OF N-RELEASE PATTERN THROUGH MIXING PRUNINGS OF DIFFERENT QUALITY

Selecting and mixing prunings of different quality is another possible option for improve synchrony when prunings are used as a source of N for maize (Chapman et al., 1988; Mafongoya et al., 1998a). Application of high-quality prunings can result in release of N too soon leading to high losses, whereas poor-quality prunings release N too slow to satisfy requirement by maize. The underlying principle behind mixing the prunings of different quality is that since N release is linked to protein-binding capacity of the prunings, mixtures of pruning with low and high protein-binding capacities can have the effect of delaying N release from high-quality prunings with low protein-binding capacity. Accelerated N release can then occur later as the polyphenols are degraded. It is also hoped

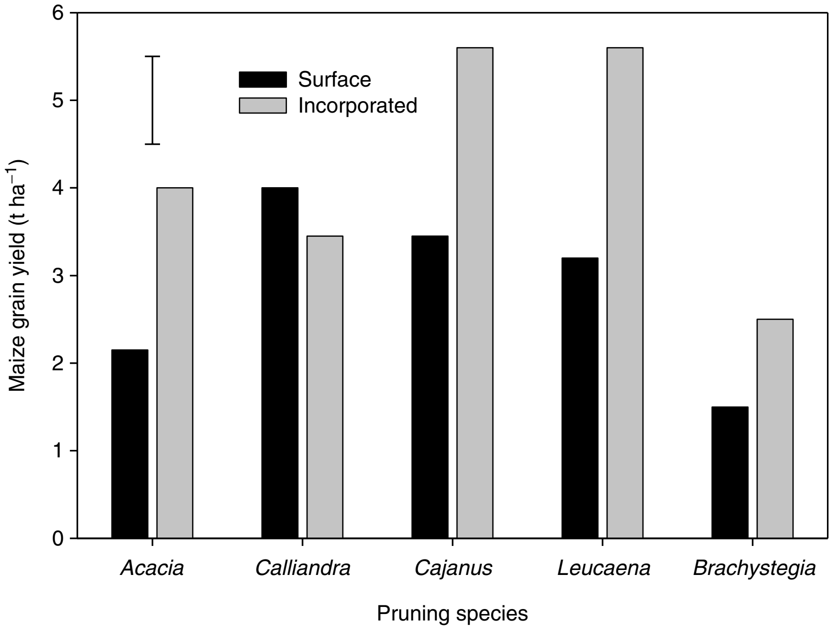


FIGURE 11.5 Maize grain yields as affected by method of application of prunings at Domboshava, Zimbabwe. (From Mafongoya, P.L., P.K.R. Nair and B.H. Dzowela, *Agroforestry Syst.*, 35, 47, 1996b.)

that in cases that the prunings of different quality release N independently, early N requirement by maize is met from the immediate N release from the high-quality pruning, whereas the requirement during later stages of growth is satisfied from the slow-decomposing low-quality prunings.

Preliminary assessment of the potential to manipulate N release by mixing prunings of different quality was done in a laboratory incubation experiment (Zingore et al., 2003). The experiment was conducted to determine the N-release patterns of *Tithonia diversifolia* leaves, which are fast decomposing and are characterized by rapid rates of N mineralization, and its mixtures (1:1) with medium-quality pruning of *Acacia* and *Calliandra* and the low-quality prunings of *Flemingia*. The % N released in 84 days were highest for the high-quality prunings of *Tithonia* that release 30% of the initial N, and least for the low-quality prunings of *Flemingia* that exhibited net N immobilization (Figure 11.6a). The medium-quality prunings of *Acacia* and *Calliandra* had similar proportions of N release of about 10%. Different mixtures showed contrasting effects of interaction with respect to N mineralization. The mixture between *Tithonia* and *Acacia* and that between *Tithonia* and *Flemingia* showed weak interaction when the measured N released was not significantly different from the predicted (Figure 11.6b and d). The mixture of *Tithonia* and *Calliandra* showed

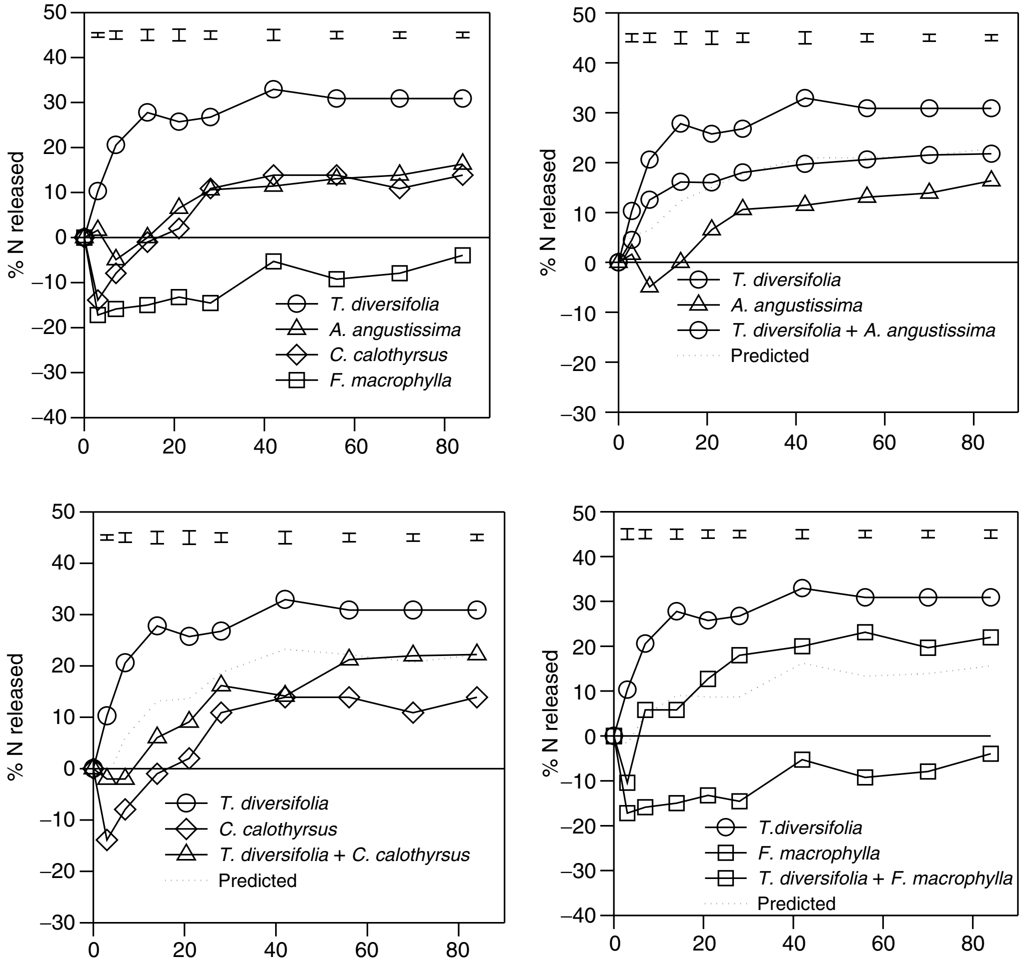


FIGURE 11.6 Cumulative nitrogen mineralization from prunings of different quality and their mixtures in a sandy soil from Domboshawa, Zimbabwe. (From Zingore, S., P. Mafongoya, P. Nyamugafata and K.E. Giller, *Agroforestry Syst.*, 57, 199, 2003.)

TABLE 11.5
Maize Grain Yields following Application of Prunings, Some of Their Mixtures and Mineral N Fertilizer at Domboshawa, Zimbabwe

	1997–1998 Season		1998–1999 Season	
Treatment	**Measured**	**Predicted**	**Measured**	**Predicted**
Tithonia diversifolia	1.7 d	—	1.9 c	—
Acacia angustissima	3.4 a	—	2.8 b	—
Calliandra calothyrsus	2.4 c	—	2.2 c	—
Flemingia macrophylla	2.5 c	—	2.1 cd	—
Tithonia diversifolia + *Acacia angustissima*	2.8 b	2.6	2.8 b	2.4
Tithonia diversifolia + *Calliandra calothyrsus*	2.5 c	2.2	1.4 d	2.1
Tithonia diversifolia + *Flemingia macrophylla*	1.2 e	2.1	1.4 d	2.0
Mineral N fertilizer	2.8 b	—	3.5 a	—
Control	1.2 e	—	1.5 d	—
SED ($p < 0.05$)	0.23		0.20	

Source: From Zingore, S., P. Mafongoya, P. Nyamugafata and K.E. Giller, *Agroforestry Syst.*, 57, 199, 2003.

Note: Means within a column followed by the same letter are not significantly different at $p < 0.05$.

a strong negative deviation from the predicted. This was a result of the initial delay in N release, which was followed by faster N release, and by 84 days the N released from the mixture closely matched the predicted (Figure 11.6c).

Testing of the effect of the sole pruning and their mixtures on maize yields was then undertaken under field conditions. The prunings were added at 5 t ha^{-1} and incorporated into the top 15 cm by hand hoeing in the first season and their residual effects were monitored in the second season. Medium- and low-quality prunings produced significantly ($p < 0.05$) higher maize grain yields in the first season ranging between 2.4 and 3.4 t ha^{-1} compared with *Tithonia*, which produced 1.7 t ha^{-1} (Table 11.5). This suggested better synchrony in N release and uptake by maize with medium- and low-quality prunings than high-quality prunings. The only mixture that indicated improved synchrony in the first season was the mixture of *Tithonia* and *Calliandra*. This was possibly due to the delayed N release from *Tithonia*, as indicated in the laboratory incubation experiment. Overall, the study showed that mixing prunings of different quality produced different patterns of N mineralization, some of which were complex and unpredictable.

11.5 IMPACT OF AGROFORESTRY ON SOIL PHYSICAL PROPERTIES AND SOIL WATER DYNAMICS

11.5.1 General Approach

Soil physical factors were determined following establishment of improved fallows. The main treatments were fallows of *Acacia* (a coppicing fallow) and *Sesbania*, natural fallow (NF) and continuous maize. Biomass produced at fallow termination was 10 t and 5.7 t ha^{-1} for *Acacia* and *Sesbania*, respectively. *Acacia* produced an additional 1.5 t ha^{-1} as coppices during the cropping phase. The plots were divided into two subtreatments of conventional tillage (CT) and no-till (NT) at fallow termination.

Soil samples for aggregate stability were collected in October 2000 (at fallow termination), in October 2001, and in October 2002. Samples were collected at the three depths using metal core rings and the soil was then air dried before it was sieved into different fractions that were used in determining aggregate stability (Nyamadzawo et al., 2003). A double-ring infiltrometer was used for measuring saturated hydraulic conductivity (Anderson and Ingram, 1993). Measurements were done after harvesting the first maize crop under both CT and NT subplots. Infiltration rates were plotted and fitted to Kostiakov equation (Hillel, 1982). The results were expressed as infiltration rate (cm h^{-1}) against cumulative time (h) and final infiltration rates.

Total effective porosity and pore sizes per unit area were calculated using a method by Watson and Luxmoore (1986). Unsaturated hydraulic conductivity measurements were carried out on an initially dry soil and measurements were done after harvesting the first maize crop. Tension infiltrometer measurements were carried out at tensions of 5 and 10 cm, which excluded pores >0.06 and 0.03 cm in diameter, respectively, from the transport process. The hydraulic conductivity, macroscopic capillary length (λ_c), and mean pore size were calculated according to a method in the CSIRO (1988) manual. Water retention measurements were done for samples collected in October 2000 and October 2002. Samples were collected from the following depths: 0–5, 10–15, 20–25, and 35–40 cm using metal ring cores. Soil cores saturated with water were equilibrated at the following suctions; 0, 5, and 10 using tension tables, and 33, 200, and 1500 kPa using pressure plate.

Rainfall simulations were run for 3 h or until steady-state runoff was attained (until a constant rate of runoff was collected) at fallow termination in October 2000. Runoff intensity (mm h^{-1}) was periodically measured by sampling water flowing from each plot. In 2001 and 2002, simulated rainfall was applied to dry soil for 30 min durations (modal duration for storms received in the country) (Meteorological Services Department, 1976). Dry runs were conducted on a dry soil and wet runs at the same spot the following day to reduce the time required to achieve steady-state infiltration rate.

11.5.2 FALLOWING AND SOIL PHYSICAL PROPERTIES

Improved fallows resulted in increased soil aggregation and pore spaces, and porosity, and these are in turn linked to available water capacity, permeability, and soil drainage, an observation that was also made by Alegre and Rao (1996) and Lal (1989). Superior aggregate stability was related to soil organic matter content (Table 11.6). Organic matter influences physical properties of soils as it supplies the gums, which bind the particles together. From the analysis that was done using soil from Domboshawa, aggregate stability was higher in fallows than in continuous cropped land regardless of the method used as shown in Figure 11.7, where stability tests were carried out over a 2 year cropping period after fallowing. The same trend was observed using other different methods, which included the water drop test, aggregate stability after different wetting treatments, and the

TABLE 11.6
Relationship (R^2) between Different Methods of Assessing Aggregate Stability and OC, Infiltration Rate Decay Coefficient, and Steady-State Infiltration Rates

	OC[a] versus Aggregate Stability	Aggregate Stability versus Steady-State Infiltration Rate	Infiltration Rate Decay Coefficient versus Aggregate Stability
I_{ma}[b]	0.79	0.86	0.78
Water drop method	0.40	0.64	0.89
Fast wetting	0.79	0.75	0.70
Mechanical agitation	0.77	0.60	0.64
Slow wetting	0.70	0.80	0.88
EDC[c]	0.42	0.38	0.72

[a] OC, organic carbon.

[b] I_{ma}, macroaggregation index from the water stability test.

[c] EDC, easily dispensable clay.

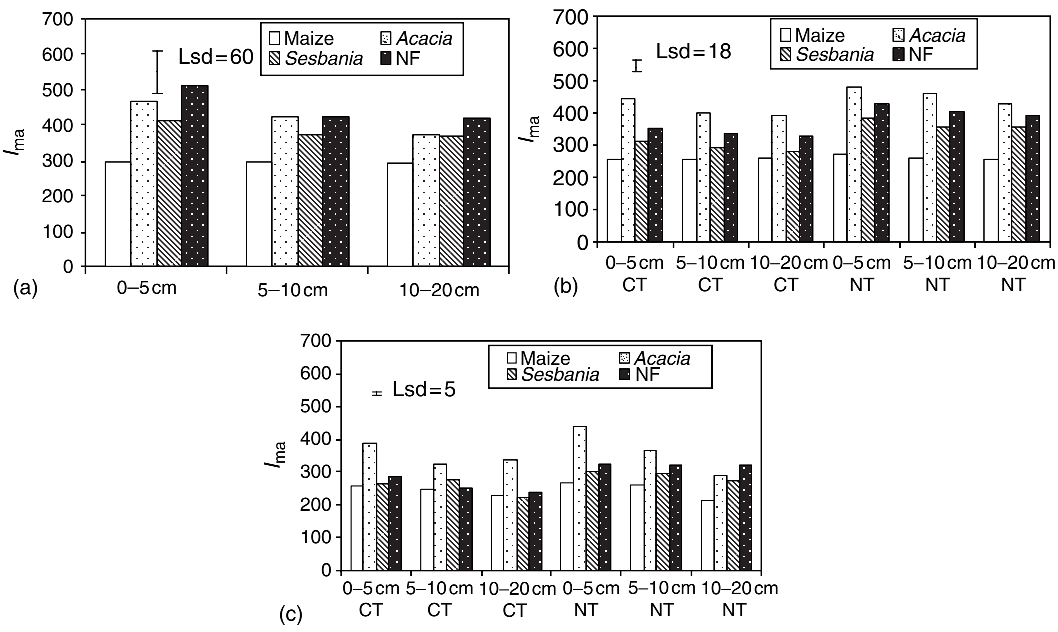
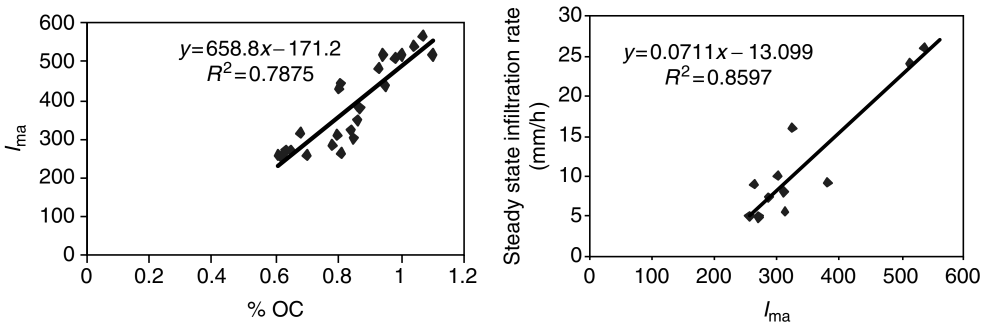


FIGURE 11.7 Effects of fallows and continuous maize on macroaggregation index. (a) October 2000, (b) October 2001, and (c) October 2002.

easily dispersible clay method. Aggregate stability using different methods was significantly higher in fallows than in continuous maize.

Aggregate stability was largely related to soil organic carbon (Figure 11.8a), and higher aggregate stability was in turn related to high infiltration rates using simulated rainfall (Figure 11.8b), using the tension infiltrometer and double ring infiltrometers (Figure 11.9). Better aggregation in fallows also resulted in increased porosity when compared with continuous maize cropping. The numbers of pores per unit area were significantly higher in fallow treatments than in continuous maize (Table 11.7). Pores were higher in fallows than in unit area in continuous maize at both 5 and 10 cm tensions.

Using data from tension measurements under unsaturated conditions, there were significant differences in mean pore sizes between treatments, with fallow treatments having higher mean pore



(a) Percent OC vs I_{ma} pooled data 2000–2002 (b) I_{ma} vs steady-state infiltration rate (2000–2002)

FIGURE 11.8 Regression functions of (a) I_{ma} versus OC and (b) I_{ma} versus infiltration rate decay coefficients.

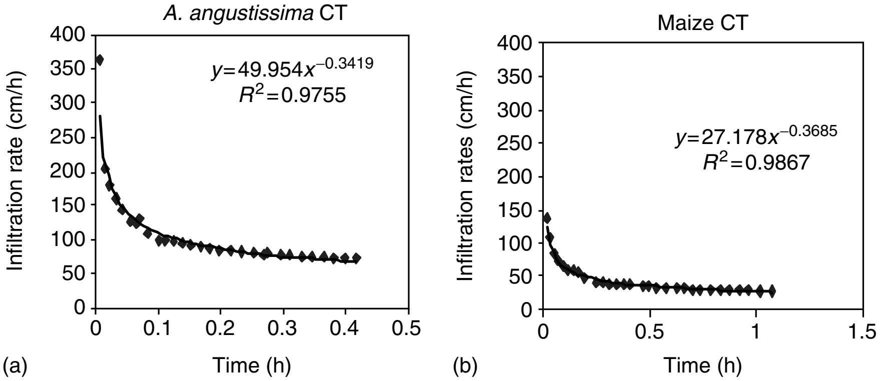


FIGURE 11.9 Infiltration rates using double ring infiltrometer in (a) *Acacia* and (b) continuous maize in October 2001. Symbols show the plotted data and the line is the fitted curve using the Kostiakov model. From the equation, y is the infiltration rate (cm h^{-1}) and x is the time (h).

sizes (0.07–0.09 mm) relative to continuous maize, which had 0.03 mm (Table 11.8). There were no significant tillage effects on mean pore sizes (Table 11.8). There were also significant differences in unsaturated hydraulic conductivity between treatments at 5 cm tension in the order *Acacia* > NF > *Sesbania* = continuous maize (Table 11.8). However, there were significant differences in unsaturated hydraulic conductivity between the 5 and 10 cm tensions.

11.5.3 WATER DYNAMICS UNDER IMPROVED FALLOW SYSTEMS

Soil and water conservation was greater under improved fallows relative to continuous maize cropping. Time to ponding and runoff was longer under fallows than under maize (Figure 11.10). The same trend was observed across different sites in both Zimbabwe and Zambia. Improved fallows resulted in significantly higher infiltration rates. Steady-state infiltration rates were not achieved in *Acacia* and NF but reached 24 and 5 mm h^{-1} in *Sesbania*, and continuous maize at fallow termination (October 2000). At the beginning of the second season (October 2001), steady-state infiltration rates could not be obtained again for both *Acacia* and NF under CT and

TABLE 11.7
Macroporosity Parameters Estimated from Ponded, Tension, and Poiseuille's Equation

	Total Effective Porosity (m^3/m^3)		Pores (m^{-2})	
Treatment	**5 cm**	**10 cm**	**5 cm**	**10 cm**
Continuous maize CT	6.2 b	107 a	256 b	3938 b
Continuous maize NT	4.8 a	96 a	175 d	2689 a
Acacia angustissima CT	10.2 c	157 b	379 a	6074 d
Acacia angustissima NT	12 d	179 b	443 e	6494 d
Sesbania sesban CT	7.2 b	115 a	285 c	4521 c
Sesbania sesban NT	12 d	180 b	439 e	8911 f
NF CT	12 d	181 b	443 e	7065 e
NF NT	9.9 c	160 b	387 d	6177 d
LSD ($p < 0.05$)	1.1	34	16.5	535

Note: Means in the same column followed by the same letter are not significantly different at $p = 0.05$.

TABLE 11.8
Unsaturated Hydraulic Conductivity (K_o) and Pore Sizes

	K_o (cm h^{-1})		Mean Pore Size Diameter (mm)	
	5 cm Tension	10 cm Tension	5 cm Tension	10 cm Tension
Maize CT	0.50 a	0.25 a	0.03 a	0.02 a
Maize NT	0.45 a	0.30 a	0.03 a	0.03 a
Acacia angustissima CT	0.80 b	0.23 a	0.07 b	0.08 b
Acacia angustissima NT	0.90 bc	0.26 a	0.09 b	0.07 b
Sesbania sesban CT	0.65 ab	0.26 a	0.08 b	0.09 b
Sesbania sesban NT	0.74 b	0.20 a	0.09 b	0.07 b
NF CT	0.8 b	0.45 a	0.08 b	0.07 b
NF CT	0.8 b	0.45 a	0.08 b	0.07 b
LSD ($p < 0.05$)	1.9		0.03	

Note: Means in the same column followed by the same letter are no significantly different at $p = 0.05$.

NT as infiltration rates were >35 mm h^{-1}. Steady-state infiltration rates averaged 21, 14, and 5 mm h^{-1} 2 years after fallow termination (October 2002) for NF, *Sesbania*, and continuous maize but remained >35 mm h^{-1} for *Acacia*.

Runoff losses from the same simulation measurements were significantly lower ($p < 0.05$) in fallows than those in continuous maize. These losses were 44% in continuous maize compared with 22% in *Sesbania* and none in *Acacia* and NF plots after 30 min at fallow termination (October 2000). After 1 year, the average runoff loss across conventionally and NT plots increased to 57% for continuous maize, 30% for *S. sesban*, and no losses for *Acacia* and NF. In October 2002, runoff losses were 63%, 61%, and 45% for continuous maize, *S. sesban*, and NF, respectively. In *Acacia*, there were no runoff losses throughout the study.

Fallows also resulted in improved water retention. There were significant differences in water retention among treatments in October 2000 at low suctions. Soil moisture retention varied with treatments, with continuous maize having the lowest water retention and *Acacia* with the highest water retention for both October 2000 and 2002. However, there were no differences in soil water retention at suction >33 kPa, across all treatments for both October 2000 and 2002. For both October 2000 and 2002, water retention decreased as depth increased; however, there were no treatment differences in water retention at depths >25 cm, at suction >33 kPa, and no significant tillage effects for October 2002 (Table 11.9).

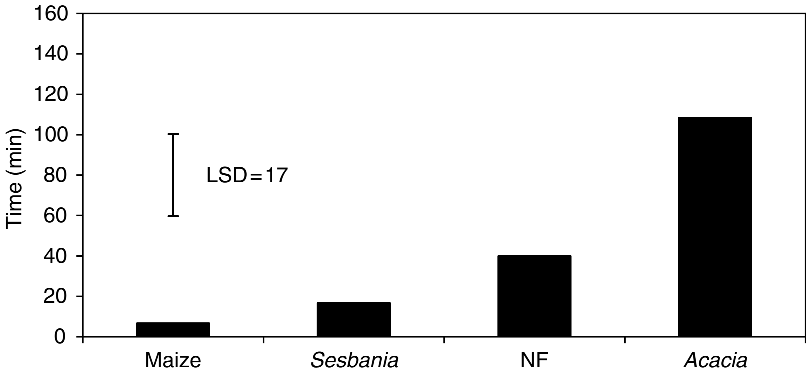


FIGURE 11.10 Time to ponding at Domboshawa at fallow termination, October 2000.

TABLE 11.9
Volumetric Water Content Held between 5 and 33 kPa

	0–5 cm	10–15 cm	20–25 cm	35–40 cm
Maize	12.3 c	10.1 b	10.2 b	5.8 a
Acacia angustissima	17.2 d	15.5 d	11.3 b	7.2 a
Sesbania sesban	13.5 c	11.7 bc	9.4 b	5.8 a
Natural fallow	16.2 d	12.8 c	10.7 b	5.7 a
LSD ($p < 0.05$)	1.9			

Note: Values are averages for October 2000 and October 2002. Means in the same column followed by the same letter are no significantly different at $p = 0.05$.

11.6 CONCLUSIONS

Generalizing N inputs and N-loss pathways is rather difficult because both processes are controlled to a large extent by environmental and biophysical factors. Large proportions of N came from N_2 fixation on the sandy soil, but these were large proportions of small amounts of N. Overall N cycling was therefore poor. On heavier soils, agroforestry technologies will improve soil fertility, and given favorable environmental conditions, lead to increased subsequent crop yields. The gaseous N losses are likely to be low in porous soils where anaerobic conditions are less severe, and the combined N oxides will probably not exceed 2 kg ha^{-1}. The greatest threat to the N accumulated by the legumes is through leaching driven by the high-intensity rainfall and the asynchronous nature in the release of N and its demand by crops. We have further confirmed that the total N content is a poor index to use for N mineralization prediction in some legume prunings like *Acacia* that have highly active polyphenols. The slow release of N due to initial immobilization by a number of legume materials we used means that crop fertilization with legume materials alone will not support high productivity. There is need for fertilizer N in addition to organic N, and the strategy has to be the use of all available resources, as improved fallow systems alone may not be attractive to farmers.

Judicious management of pruning is required to improve N-use efficiency in biomass transfer and alley-cropping systems, where pruning is used as a source of N for annual crops such as maize. Incorporation of prunings at planting led to accumulation of large amounts of N before the period of peak demand by maize. However, this may be risky in seasons with early high rainfall intensities as the accumulated N is prone to leaching before uptake. Mixing prunings of different qualities showed potential for improving synchrony, but the trends produced by different mixtures were variable and unpredictable. The effect of this technique on maize under field conditions may also be highly dependant on the distribution of rainfall.

Fallowing improved soil physical properties, that is, aggregate stability and soil porosity, during the fallowing phase benefited the crop during the years of cropping. The results also showed that fallows improved infiltration rates, hydraulic conductivity, and water retention relative to continuous maize cropping, and this is important for it increases water availability during cropping, and the farmer will benefit from increased crop yields.

We recommend a holistic evaluation of the impact of agroforestry encompassing multi-nutritional benefits, soil physical properties, and socioeconomic benefits to fully appreciate the role of agroforestry in sustaining soil fertility on smallholder farms. For accelerated adoption of promising agroforestry technologies by smallholder farmers, we also suggest identification of niches suitable for the technologies on the smallholder farms that are characterized by spatial variability of soil fertility.

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12 Soil Sustainability in Agroforestry Systems: Experiences on Impacts of Trees on Soil Fertility from a Humid Tropical Site

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CONTENTS

12.1 INTRODUCTION

The problem of maintaining soil fertility in the long term has become an increasingly important topic in the management of agroforestry systems as tree–crop combinations are often established on low-fertility soils. Inclusion of woody components in a production system can provide benefits from the tree products and functions (timber, fuelwood, leaf mulches, the fencing function in a living fence, etc.) and from their potential ecological advantages, especially their nutrient cycling abilities. The choice of a tree species will often depend on whether both productive and ecological advantages can be achieved in the same system, and in some cases one prevailing function, either productive or environmental, may be desired.

Agroforestry systems are especially important in regions where commercial fertilizers are expensive or unavailable, because of their ability to recover, recycle, or efficiently utilize nutrients. This ability is often linked to mechanisms associated with woody or perennial species that recycle nutrients mainly through litterfall and decomposition. Although agroforestry systems can be profitable if established immediately after forest clearing, they often require a number of years to become profitable when established on degraded lands. For this reason, capital-limited farmers on poor soils may require subsidies to enable the establishment of agroforestry systems (Montagnini et al., 2006).

Some management strategies to conserve nutrients of a site and to improve the sustainability of agroforestry systems consist of planting tree species that do not have an elevated demand for nutrients (Wang et al., 1991; Montagnini and Sancho, 1994). There can be a large difference between the efficiency of nutrient use by tree species. For example, in studies realized in Puerto Rico, species of *Casuarina* were two times more efficient than species of *Leucaena* for nitrogen (N), three to four times more efficient than species of *Albizia* and *Leucaena* for potassium (K), and approximately two times more efficient than all the other species studied for magnesium (Mg) (Wang et al., 1991). In other studies of the suitability of tree species for agroforestry in Brazil, Argentina, and Costa Rica, large differences in nutrient use efficiency between species were found (Montagnini, 2001). This information was used to draw recommendations for the use of the species in agroforestry systems in the three regions under study.

The use of different tree species can increase or decrease the nutrients of a site, which is also influenced by the type of management. The extraction of nutrients with the harvest of tree products is especially critical to the productivity of agroforestry systems. Knowledge of nutrient content in each of the tree parts can be a guide for management considerations at the time of harvesting the trees, in particular the parts of the tree that are left behind or taken away from the site.

In this chapter, aboveground biomass, concentrations of nutrients in aboveground biomass, rate of litterfall, rate of litter decomposition, and soil nutrients were examined in tree plantations of native species from humid regions of the Neotropics. The information is used to determine their suitability for combinations with crops, as well as their impacts on soil fertility, and to offer management recommendations for the conservation of nutrients over the long term.

12.2 METHODOLOGY

12.2.1 EXPERIMENTAL SITE

The research took place at La Selva Biological Station, Costa Rica (10 22′ N, 83 59′W, 35–127 m.a.s.l.). The mean annual temperature is 24°C and the mean annual precipitation is 4000 mm. The tree plantations were established in 1991–1992 in an area of abandoned pastureland. The area was cleared in the mid1950s and grazed until 1981, a land-use pattern common in the region. The area is on flat, uniform terrain (<1 m average difference between lowest and highest points). At the time of clearing for the plantations, the area was covered with shrubs and early successional trees interspersed with patches of grass and ferns.

The soils are Fluventic Dystropepts, derived from volcanic alluvial soil; they are deep, are free of rocks, have low to medium organic matter content (2.5%–4.5%), have moderately heavy texture, are acidic ($\text{pH} < 5.0$), and are not very fertile (Montagnini and Porras, 1998). In comparing soil chemical characteristics before planting, results showed that there were no significant differences among blocks within each plantation (Montagnini et al., 1993). According to standards set by the Costa Rican Ministry of Agriculture, fertility levels of the site were too low for conventional agriculture (Montagnini and Porras, 1998).

The plantations consist of 12 native tree species: Plantation 1: *Jacaranda copaia* (Aubl) D. Don (Bignoniaceae), *Vochysia guatemalensis* Donn. Sm. (Vochysiaceae), *Calophyllum brasiliense* Cambess (Clusiaceae), and *Stryphnodendron microstachyum* Poepp. and Endl. (Fabaceae-Mimosoideae); Plantation 2: *Terminalia amazonia* (J. Gmel.) Exell (Combretaceae), *Dipteryx panamensis* (Pittier) Record and Mell (Fabaceae-Papilionoideae), *Virola koschnyi* Warb. (Myristicaceae), and *Paraserianthes guachapele* (Kunth) Harms. (Fabaceae-Mimosoideae); Plantation 3: *Hyeronima alchorneoides* Allemao (Euphorbiaceae), *Balizia elegans* (Ducke) Barneby and Grimes (Fabaceae-Mimosoideae), *Genipa americana* L. (Rubiaceae), and *Vochysia ferruginea* Mart. (Vochysiaceae). Plots of 32×32 m^2 are in random blocks with four repetitions and six treatments:

pure plots of each species, a mixed plot with four species, and a plot of natural regeneration where no trees were planted (Montagnini et al., 1995; Montagnini and Porras, 1998; Montagnini, 2001). Three of the species had problems with disease, pests, or adaptability to the site, which were noticed at a relatively early age (2–3 years after planting). In *S. microstachyum* plots, anthracnosis caused the death of all the trees in the pure plots and in a majority of the mixed plots. Other plantations in the region did not appear to have had this problem. In *P. guachapele* plots, pocket gophers affected the roots, causing complete mortality in pure plots and almost complete mortality in mixed plantations. This species is not planted by farmers in the region, so comparison with other plantations was not possible. *G. americana* appeared to have low adaptability to the site or to growing conditions, with poor growth and high mortality but with no apparent cause of disease or pest. These three species suffered levels of mortality such that they have not been included in the most recent measurements of these plantations (Alice et al., 2004; Petit and Montagnini, 2004, 2006; Redondo and Montagnini, 2006). Trees of *C. brasiliense* had heavy mortality in pure plots at a later age (15 years after planting). The agent causing mortality was not identified; most individuals in mixed plots survived (personal observation, June 2006).

12.2.2 ABOVEGROUND BIOMASS AND NUTRIENT CONTENT OF THE TREES

In a study designed to estimate the quantity of nutrients in the aboveground biomass, half of the trees in the plots were removed in a thinning treatment at 6 years (Shepherd and Montagnini, 2001). In each plot, three trees were selected to measure the biomass. Trunks, branches, and leaves were separated, weighed on site, and subsamples were taken to an oven at 70°C. The ratio dry weight:wet weight was used to correct data from the field. The average biomass per tree was multiplied by the number of trees per hectare, correcting for tree mortality, to obtain biomass per hectare. The chemical analysis of different parts of plant tissue was performed at the laboratories of the Center for Agricultural Research (Centro de Investigaciones Agronómicas, CIA) at the University of Costa Rica (Universidad de Costa Rica) in accordance with the standard procedures for the analysis of plant tissues. The concentrations of N, P, Ca, Mg, and K were multiplied by the biomass of the corresponding parts to obtain kilogram per hectare of each nutrient and tree part, by species, by pure plot, and by mixed plot (Stanley and Montagnini, 1999; Montagnini, 2000a).

12.2.3 RATE OF LITTERFALL AND DECOMPOSITION AND NUTRIENT RELEASE TO CROPS

Other studies in the same experimental site have examined the rate of litterfall and litter decomposition, the accumulation of the litterfall on the plantation floor, the nutrient content of litterfall and forest-floor litter, and the release of nutrients from the litterfall to the soil (Byard et al., 1996; Kershnar and Montagnini, 1998; Horn and Montagnini, 1999; Stanley and Montagnini, 1999). These studies were done on the plantations 3 to 5 years after their establishment. Litterfall was measured using litter traps and collecting material every 2 weeks for 12 months in each plot of every plantation. Decomposition was measured using bags filled with litter that were placed on site and were collected every 2 weeks for 12 months. The decomposition constants were calculated for the litterfall of each species. The quantity of forest-floor litter was measured every 3 months using 30 × 30 cm PVC squares that were placed on top of the plantation floors, collecting all the material and taking it to an oven to calculate dry weight. Greenhouse trials were performed using leaves of each species as mulch, which was added to small pots seeded with maize in order to observe the response of the maize to the treatment (height growth) and also the quantity of nutrients in the biomass of the maize. The litterfall and forest-floor material were analyzed for N, P, Ca, Mg, and K at the Center for Agricultural Research (Centro de Investigacions Agronómicas, CIA) of the University of Costa Rica (Universidad de Costa Rica), in accordance with the standard procedures for nutrient analysis of plant tissues used in the country (Stanley and Montagnini, 1999; Montagnini, 2000a, 2000b). Concentrations of N, P, Ca, Mg, and K were multiplied by the quantity of litterfall or forest-floor litter to obtain kilogram per hectare for each nutrient.

12.2.4 Soil Sampling and Analysis

For the studies of soil fertility, soils were sampled in each forest plot: pure, mixed, and natural regeneration. Composite samples were taken with a Dutch-type soil auger every year for the first 5 years of the plantations and again in 2003 when the plantations were 11 to 12 years old. In the first 5 years of soil sampling, soils were sampled down to 60 cm depth. In 2003, samples were taken from the top soil only (0–5 and 5–15 cm) because results of previous sampling showed that most differences in soil parameters among treatments were found in the top soil (Stanley and Montagnini, 1999; Montagnini, 2000a).

The samples were processed for pH, acidity, exchangeable Ca, Mg, K, extractable P, organic matter, total N, and extractable minor elements: Cu, Zn, Mn, and Fe. The soils were analyzed in the laboratories of the CIA of the University of Costa Rica (Universidad de Costa Rica), using methodologies that are in accordance with standard procedures for soil analysis in the country (Stanley and Montagnini, 1999; Montagnini, 2000a, 2000b).

12.3 RESULTS AND DISCUSSION

12.3.1 Nutrient Content in Aboveground Biomass of the Trees

In Plantation 1, the pure plots of *J. copaia* had higher quantities of N, P, and Mg in the tree biomass than the other treatments, whereas the pure plots of *Vochysia guatemalensis* accumulated greater quantities of K and Ca (Montagnini, 2000b). For *J. copaia*, the harvest of the trunks would eliminate around 54% of the N content of the tree, but around 80% of the P, K, Ca, and Mg. For *V. guatemalensis*, the harvest of the trunks would remove less than 30% of N, but 50%–60% of the total content of Ca, K, Mg, and P. The branches and foliage accounted for between 25% and 35% of the total aboveground biomass, but in general contained around 50% of the nutrients of the aboveground biomass.

In Plantation 2, mixed plots had greater nutrient content in biomass for all studied elements, and both mixed and pure plots of *T. amazonia* had greater quantities of P and Mg in the trunk (Montagnini, 2000b).

In Plantation 3, the branches and foliage—considered together—accounted for between 25% and 35% of the total biomass, but around 50% of the total tree nutrients. In this plantation, pure plots of *H. alchorneoides* and *B. elegans*, and mixed plots of four species had greater accumulation of total nutrients in the tree biomass per hectare (Stanley and Montagnini, 1999).

The losses of nutrients during the harvest can be much greater than the inputs of nutrients to the soil via the mineralization of soil minerals or rainfall, especially when rotations are very short (Fölster and Khanna, 1997). In addition, the nutrient content of plant tissues is fairly variable. The results of this study show the nutrient concentration in tree tissues occur in the following order: foliage > branches > trunks. Although branches and foliage combined only represented 25%–35% of the total tree biomass, they represented approximately 50% of the total tree nutrients (Stanley and Montagnini, 1999; Montagnini, 2000b).

In order to reduce the nutrient loss associated with harvests, conservation of the tree components should be done in the following order of priority: (1) foliage, (2) branches, and (3) trunks. If branches and foliage are left on site at the moment of the harvest, instead of the entire tree, the nutrient loss of the harvest is reduced by almost half. The branches and foliage left behind also serve as mulch and help to improve soil conditions.

The quantity of nutrients in branches and foliage varies according to the nutrient, the species, and the site. The management of harvest residues, keeping in mind the different nutritional contents of plant tissues, is an important facet of the nutritional management of plantations (Wang et al., 1991; Montagnini and Sancho, 1994; Fölster and Khanna, 1997; Nykvist, 1997; Stanley and Montagnini, 1999; Montagnini, 2000a, 2000b).

12.3.2 LITTERFALL AND DECOMPOSITION AND NUTRIENT RELEASE TO CROPS

In Plantation 1, the litter of *V. guatemalensis*, *J. copaia*, and the mixed plantation decomposed the fastest. Less than 16% of the initial weight remained after 12 months in litter bags (Byard et al., 1996). The litter of *C. brasiliense* had the slowest decomposition, with 23% of the initial weight remaining in the litter bags after 12 months in the field. The litterfall and accumulation of litter on the forest floor was elevated in plantations of *J. copaia*, even though litter accumulation in the soil varied over the course of the year. The mixed plantations showed average levels of litterfall and accumulation of litter. The litter of *S. microstachyum*, used as mulch to fertilize corn plants, contributed the most to growth and recapturing N. These results have implications in reference to the use of these species in agroforestry systems.

In Plantation 2, the litter of *T. amazonia* decomposed the fastest; no litter was left in the bags after only 6 months in the field (Kershnar and Montagnini, 1998). After 12 months, the leaf litter of *D. panamensis*, *P. guachapele*, and mixed plantation decomposed completely, whereas 15% of the original *Virola koschnyi* litter still remained. Litterfall was greatest in plantations of *T. amazonia* (872.9 gm^{-2}), followed by *D. panamensis*, *V. koschnyi*, and the mixed plantations. *P. guachapele* had the lowest quantity of litterfall (236.0 gm^{-2}). The accumulation of litter on the floor was greatest in plots of *V. koschnyi* and *D. panamensis*. Litterfall and litter accumulation on the floor fluctuated less in mixed plantations than in pure plantations. The litter of *P. guachapele* and *D. panamensis*, used as mulch to fertilize corn plants, were the most beneficial for growth, followed by the litter of mixed plantations.

In Plantation 3, the litter of *B. elegans* decomposed the fastest, the leaf litter of *Vochysia ferruginea* decomposed the slowest, and the leaf litter of mixed plantations had an average rate of decomposition (Horn and Montagnini, 1999). The litterfall was greatest in *V. ferruginea* plantations (867.2 gm^{-2}), *G. americana* had the least (386.7 gm^{-2}), and mixed plantations had an average quantity (660 gm^{-2}). The leaf litter used as mulch to fertilize corn plants was beneficial for growth in all cases, with the exception of *G. americana*.

The large quantity of leaf litter produced by *V. guatemalensis*, *T. amazonia*, *H. alchorneoides*, and *V. ferruginea* makes these species useful for protecting soil against erosion. Mixed plantations offer the combined benefits of these species: protection against soil erosion in the case of abundant leaf litter and slow decomposition, and rapid release of nutrients to the soil in the case of species with high nutrient content and quick decomposition. In addition, mixed plantations have other advantages such as promoting biodiversity and product diversification (Guariguata et al., 1995; Carnevale and Montagnini, 2002; Cusack and Montagnini, 2004).

The importance of litter accumulation on the floor as a storage compartment of nutrients varied over time. When biomass of the floor litter reached its maximum, its total content of N, Ca, and Mg were approximately equal or greater than that of the trunk for all species, with the exception of *B. elegans*. For *B. elegans*, the floor litter consistently represented a very low proportion of the nutrient content of the biomass (Stanley and Montagnini, 1999).

12.3.3 IMPACT OF TREE PLANTATIONS ON SOIL FERTILITY

Five years after planting, decreases in the content of P, K, and Ca in the soil became apparent in pure plots of fast-growing tree species, such as *J. copaia* and *Vochysia guatemalensis*, with greater accumulation of nutrients in the aboveground biomass (Montagnini, 2000a). However, in other cases there were beneficial effects upon the soil: for example, increases in Ca in the soil under *T. amazonia* and *Virola koschnyi*, both species with a high content of Ca in their foliage and elevated rates of annual litterfall (Kershnar and Montagnini, 1998). In a similar fashion, soils under *Vochysia ferruginea* had greater concentrations of Ca, Mg, and higher organic matter in comparison to the other species. This result is consistent with other studies that include this species (Montagnini and Sancho, 1990; Montagnini and Sancho, 1994; Stanley and Montagnini, 1999).

The mixed plantation plots had average values for the nutrients examined, and even improved conditions for some soil nutrients, such as P (Montagnini, 2000a). In some cases, there were lower values for nutrients in mixed plantations than in pure plantations, as was the case for Ca and Mg (Stanley and Montagnini, 1999). This suggests that in mixed plots, soils have a more balanced nutrient status as a result of the complementary effect on nutrient cycling of the different species participating in the mixture.

Measurements over a long period are necessary to determine the effects of tree species on soils. When the plantations were 11–12 years old, results indicated that although many of these trends continued, some new ones were observed (Tables 12.1 through 12.3). For example, in Plantation 1, the soil under *Vochysia guatemalensis* had higher pH, less acidity, and greater Mg than other treatments, and a high concentration of Ca even though this difference was not statistically significant (Table 12.1). On the contrary, the soils under *C. brasiliense* had lower values for the same parameters. These results appear to be related to the high capacity of *V. guatemalensis* to recycle cations, given its high quantity and rapid decomposition of leaf litter, whereas the opposite happens with *C. brasiliense*. The value of soil pH in *V. guatemalensis* (5.03) was higher than in previous measurements; the values of soil Ca, Mg, K, organic matter, and N were similar and the values of P were less.

In Plantation 2, the results from measurements at 11 years showed that soils under *D. panamensis* had greater K and under *Virola koschnyi* a lesser value was found for this nutrient, whereas there were no statistically significant differences between treatments in the other parameters studied (Table 12.2). With respect to past measurements, the result for *D. panamensis* was similar; the values of soil pH were similar, whereas those of cations and P were lower. The values for soil organic matter and N were greater than those found in previous measurements.

In Plantation 3, the results at 11 years of age indicated that soils under mixed plantations had the greatest quantity of organic matter, followed by *H. alchorneoides* and *Vochysia ferruginea* (Table 12.3). The quantities of organic matter and N, in general, were greater than those found when measured previously.

From these results, it seems that at 11–12 years of age when plantations approach maturity, the top soil has accumulated organic matter and nitrogen from litter recycling under the plantations' canopies. Values for other soil parameters were higher or lower than in earlier measurements depending on the species. It would be interesting to perform additional soil sampling when plantations approach the end of their rotation cycle, estimated to be 15–25 years depending on the species (Petit and Montagnini, 2004; Petit and Montagnini, 2006).

12.3.4 SUITABILITY OF THE SPECIES TESTED FOR THEIR USE IN AGROFORESTRY SYSTEMS

From the species of Plantation 1, it appears that both *V. guatemalensis* and *J. copaia* would be good species for agrosilvopastoral systems, due to their good growth in pure and mixed conditions. Their canopy characteristics allowed for enough illumination to favor the growth of abundant understory, also permitting the growth of pasture grasses (Montagnini et al., 2003). Among the grasses found under the canopy of these and the other species of these experiments were the native *Cynodon nlemfuensis* (pasto estrella), *Paspalum fasciculatum* (gamalote), and the exotic (naturalized) *Panicum maximum* (Guinea grass), *Pennisetum purpureum*, *Brachiaria* spp., *Melinis minutiflora* (calinguero or San Juan), and *Ischaemum indiana* (retana). Except for gamalote, these species are grazed by beef cattle, although improved grasses would be needed to increase cattle productivity of these silvopastoral systems (Montagnini et al., 2003).

Of the four species of Plantation 1, *V. guatemalensis* had the highest rates of litterfall, and its litter decomposed relatively quickly (Byard et al., 1996) therefore nutrient release from this species could favor growth of associated crops or pastures. *V. guatemalensis* is probably the tree species that is most frequently planted by farmers in the Caribbean lowlands of Costa Rica and knowledge exists regarding several aspects of this species' domestication, including seed collection and germination, vegetative propagation, and preliminary stages of tree genetic improvement (Montagnini et al., 2003).

TABLE 12.1

Plantation 1: pH, Acidity, Concentrations of Calcium, Magnesium, Potassium, Phosphorus, Organic Matter, and Nitrogen and Minor Elements: Copper, Zinc, Manganese, and Iron in Soils under Four Forest Species in Pure Plantations, Mixed Plantations of the Four Species 12 Years after Planting, and Natural Regeneration Plots (Not Planted) at La Selva Biological Station

	pH			Acidity (cmol L^{-1})			Ca (cmol L^{-1})			Mg (cmol L^{-1})			K (cmol L^{-1})			P (mg L^{-1})			Organic Matter (%)			N (%)			Cu (mg kg^{-1})			Zn (mg kg^{-1})			Mn (mg kg^{-1})			Fe (mg kg^{-1})		
Treatment and Depth	**Mean**	**SE**	**Sig**	**Mean**	**SE**	**Sig**	**Mean**	**SE**	**Sig**	**Mean**	**SE**	**Sig**	**Mean**	**SE**	**Sig**	**Mean**	**SE**	**Sig**	**Mean**	**SE**	**Sig**	**Mean**	**SE**	**Sig**	**Mean**	**SE**	**Sig**	**Mean**	**SE**	**Sig**	**Mean**	**SE**	**Sig**	**Mean**	**SE**	**Sig**
Jacaranda copaia																																				
0–5 cm	4.43	0.02	b	2.34	0.10	b	1.08	0.44	a	0.53	0.03	ab	0.16	0.02	a	7.05	1.10	a	10.56	0.74	a	0.54	0.03	a	16.43	1.43	a	2.14	0.18	ab	11.93	0.63	a	639.03	81.11	a
5–15 cm	4.45	0.03	b	2.39	0.04	b	0.72	0.41	a	0.29	0.02	ab	0.10	0.01	ab	6.27	0.96	a	7.49	0.70	a	0.41	0.03	a	17.95	2.42	a	1.84	0.15	a	9.86	0.93	a	607.31	96.51	a
Vochysia guatemalensis																																				
0–5 cm	5.03	0.00	a	1.17	0.01	c	1.64	0.16	a	0.84	0.02	a	0.14	0.01	a	5.04	0.07	a	11.15	0.43	a	0.50	0.01	a	15.37	3.38	a	2.05	0.16	ab	9.96	0.42	a	284.30	29.02	b
5–15 cm	4.78	0.07	a	1.72	0.05	c	0.78	0.09	a	0.43	0.01	a	0.09	0.01	b	4.59	0.12	a	7.08	0.52	a	0.37	0.02	a	18.88	2.79	a	1.94	0.02	a	9.77	0.45	a	365.22	11.01	a
Calophyllum brasiliense																																				
0–5 cm	4.15	0.06	bcd	3.43	0.11	a	0.32	0.07	a	0.26	0.03	b	0.11	0.00	a	10.31	1.70	a	9.45	0.28	a	0.46	0.02	a	15.96	18.95	a	1.93	0.31	b	14.51	4.55	a	561.81	106.84	ab
5–15 cm	4.23	0.05	c	3.02	0.17	a	0.18	0.03	a	0.15	0.01	b	0.07	0.00	b	8.88	1.56	a	6.43	0.29	a	0.35	0.02	a	1.36	4.46	a	1.90	0.37	a	11.63	4.23	a	499.27	94.07	a
Mixed 1																																				
0–5 cm	4.70	0.14	bcd	2.01	0.34	ab	1.24	0.39	a	0.76	0.17	a	0.14	0.01	a	6.29	1.60	a	10.79	0.32	a	0.53	0.01	a	16.72	0.97	a	2.72	0.26	ab	16.30	2.15	a	311.88	55.39	ab
5–15 cm	4.60	0.04	a	2.33	0.20	b	0.60	0.14	a	0.42	0.09	a	0.09	0.01	ab	6.70	2.13	a	7.11	0.34	a	0.39	0.02	a	21.18	2.00	a	2.91	0.50	a	17.65	4.67	a	378.26	15.34	a
Regeneration 1																																				
0–5 cm	4.48	0.05	bc	2.63	0.33	ab	1.00	0.23	a	0.86	0.13	a	0.22	0.05	a	12.53	3.45	a	9.82	0.65	a	0.51	0.02	a	17.69	1.13	a	3.31	0.55	a	19.14	5.67	a	530.97	141.42	ab
5–15 cm	4.53	0.07	bc	2.84	0.37	ab	0.46	0.09	a	0.44	0.05	ab	0.14	0.03	a	8.98	1.96	a	6.78	0.52	a	0.38	0.03	a	20.60	0.79	a	2.36	0.20	a	10.80	2.78	a	540.52	197.26	a

Note: Averages, standard error (SE), and statistical significance (Sig). The differences between treatments for each parameter and depth are statistically significant ($P < .05$) when the means are followed by different letters.

TABLE 12.2

Plantation 2: pH, Acidity, Concentrations of Calcium, Magnesium, Potassium, Phosphorus, Organic Matter, and Nitrogen and Minor Elements: Copper, Zinc, Manganese, and Iron in Soils under Four Forest Species in Pure Plantations, Mixed Plantations of the Four Species, 11 Years after Planting, and Natural Regeneration Plots (Not Planted) at La Selva Biological Station

	pH			Acidity (cmol L^{-1})			Ca (cmol L^{-1})			Mg (cmol L^{-1})			K (cmol L^{-1})			P (mg L^{-1})			Organic Matter (%)			N (%)			Cu (mg kg^{-1})			Zn (mg kg^{-1})			Mn (mg kg^{-1})			Fe (mg kg^{-1})		
Treatment and Depth	**Mean**	**SE**	**Sig**	**Mean**	**SE**	**Sig**	**Mean**	**SE**	**Sig**	**Mean**	**SE**	**Sig**	**Mean**	**SE**	**Sig**	**Mean**	**SE**	**Sig**	**Mean**	**SE**	**Sig**	**Mean**	**SE**	**Sig**	**Mean**	**SE**	**Sig**	**Mean**	**SE**	**Sig**	**Mean**	**SE**	**Sig**	**Mean**	**SE**	**Sig**
Dipteryx panamensis																																				
0–5 cm	4.40	0.07	a	1.91	0.22	a	0.94	0.25	a	0.52	0.11	a	0.19	0.01	a	5.60	0.10	a	9.38	0.70	a	0.48	0.03	a	17.58	1.08	a	2.32	0.20	a	36.76	7.42	a	508.07	130.80	a
5–15 cm	4.50	0.07	a	1.88	0.34	a	0.55	0.15	a	0.33	0.07	a	0.13	0.02	a	4.27	0.05	a	6.27	0.32	a	0.36	0.01	a	19.90	1.29	a	2.22	0.23	a	26.91	6.09	a	458.43	136.10	a
Virola koschnyi																																				
0–5 cm	4.38	0.04	a	2.51	0.16	a	0.58	0.06	a	0.40	0.01	a	0.11	0.01	b	5.14	0.53	a	11.01	0.15	a	0.50	0.01	a	17.30	1.19	a	2.33	0.44	a	40.16	6.17	a	493.44	7.51	a
5–15 cm	4.38	0.04	a	2.33	0.14	a	0.32	0.06	a	0.21	0.01	a	0.07	0.01	a	4.92	0.19	a	6.82	0.39	a	0.36	0.01	a	22.16	1.40	a	2.13	0.08	a	30.78	7.99	a	501.26	21.65	a
Terminalia amazonia																																				
0–5 cm	4.35	0.05	a	2.55	0.30	a	0.58	0.03	a	0.54	0.05	a	0.13	0.01	ab	7.22	0.79	a	8.29	0.50	a	0.45	0.01	a	17.22	0.60	a	2.30	0.12	a	38.51	10.56	a	566.68	119.00	a
5–15 cm	4.45	0.05	a	2.44	0.32	a	0.37	0.06	a	0.33	0.01	a	0.09	0.01	a	5.66	1.13	a	6.19	0.51	a	0.36	0.02	a	20.01	0.91	a	2.69	0.44	a	28.06	8.63	a	517.29	88.71	a
Mixed 2																																				
0–5 cm	4.43	0.07	a	2.19	0.35	a	0.51	0.08	a	0.41	0.03	a	0.12	0.02	ab	4.52	1.14	a	8.40	1.20	a	0.45	0.05	a	19.05	1.68	a	2.58	0.39	a	43.08	11.84	a	448.72	99.69	a
5–15 cm	4.48	0.05	a	2.17	0.30	a	0.38	0.08	a	0.33	0.07	a	0.11	0.02	a	5.32	0.79	a	6.77	0.69	a	0.38	0.03	a	19.88	1.51	a	2.69	0.57	a	30.32	8.35	a	385.96	34.94	a
Regeneration 2																																				
0–5 cm	4.38	0.07	a	2.08	0.08	a	0.87	0.06	a	0.72	0.16	a	0.17	0.03	ab	5.95	0.94	a	9.91	0.37	a	0.52	0.01	a	18.43	1.71	a	3.29	0.91	a	45.04	9.60	a	424.73	23.80	a
5–15 cm	4.45	0.06	a	2.01	0.22	a	0.46	0.04	a	0.36	0.08	a	0.11	0.02	a	4.49	0.55	a	6.60	0.36	a	0.38	0.02	a	20.23	0.95	a	2.48	0.29	a	30.11	6.75	a	384.80	57.52	a

Note: Averages, standard error (SE), and statistical significance (Sig). The differences between treatments for each parameter and depth are statistically significant ($P < .05$) when the means are followed by different letters

TABLE 12.3

Plantation 3: pH, Acidity, Concentrations of Calcium, Magnesium, Potassium, Phosphorus, Organic Matter, and Nitrogen and Minor Elements: Copper, Zinc, Manganese, and Iron in Soils under Four Forest Species in Pure Plantations, Mixed Plantations of the Four Species 11 Years after Planting, and Natural Regeneration Plots (Not Planted) at La Selva Biological Station

	pH			Acidity (cmol L^{-1})			Ca (cmol L^{-1})			Mg (cmol L^{-1})			K (cmol L^{-1})			P (mg L^{-1})			Organic Matter (%)			N (%)			Cu (mg kg^{-1})			Zn (mg kg^{-1})			Mn (mg kg^{-1})			Fe (mg kg^{-1})		
Treatment and Depth	**Mean**	**SE**	**Sig**	**Mean**	**SE**	**Sig**	**Mean**	**SE**	**Sig**	**Mean**	**SE**	**Sig**	**Mean**	**SE**	**Sig**	**Mean**	**SE**	**Sig**	**Mean**	**SE**	**Sig**	**Mean**	**SE**	**Sig**	**Mean**	**SE**	**Sig**	**Mean**	**SE**	**Sig**	**Mean**	**SE**	**Sig**	**Mean**	**SE**	**Sig**
Genipa americana																																				
0–5 cm	4.53	0.10	a	2.11	0.54	a	0.82	0.19	a	0.81	0.20	ab	0.24	0.05	a	7.43	0.55	a	9.00	0.24	ab	0.49	0.02	a	20.79	2.57	a	2.67	0.59	ab	37.27	14.68	a	596.44	116.05	a
5–15 cm	4.43	0.06	a	2.30	0.33	a	0.45	0.11	a	0.36	0.08	ab	0.11	0.01	ab	6.34	1.09	a	5.84	0.48	a	0.34	0.04	a	24.88	4.28	a	2.46	0.43	a	23.95	9.71	a	487.18	124.33	a
Vochysia ferruginea																																				
0–5 cm	4.60	0.25	a	2.06	0.44	a	0.73	0.25	a	0.35	0.12	b	0.15	0.03	b	6.02	0.00	a	12.18	0.16	ab	0.53	0.00	a	16.41	0.14	a	2.38	0.29	ab	35.97	19.79	a	404.71	120.65	a
5–15 cm	4.45	0.14	a	2.30	0.38	a	0.28	0.09	a	0.16	0.01	b	0.07	0.00	b	5.84	0.36	a	6.02	0.55	a	0.32	0.04	a	23.33	0.49	a	2.72	0.22	a	32.54	13.40	a	519.30	134.64	a
Balizia elegans																																				
0–5 cm	4.35	0.10	a	2.49	0.22	a	0.56	0.04	a	0.43	0.06	b	0.17	0.02	ab	6.76	0.95	a	9.35	0.59	b	0.50	0.05	a	16.81	1.69	a	1.74	0.31	b	42.95	19.41	a	616.81	133.63	a
5–15 cm	4.38	0.06	a	2.38	0.35	a	0.29	0.07	a	0.22	0.03	b	0.08	0.01	b	5.26	0.72	a	5.61	0.32	a	0.33	0.03	a	21.77	3.86	a	2.01	0.58	a	37.34	17.15	a	520.85	146.41	a
Hyeronima alchorneoides																																				
0–5 cm	4.35	0.06	a	2.42	0.28	a	0.98	0.21	a	0.70	0.09	ab	0.16	0.02	ab	7.03	0.25	a	12.70	1.55	ab	0.56	0.11	a	16.49	1.75	a	2.46	0.23	ab	42.46	14.36	a	662.69	49.57	a
5–15 cm	4.30	0.06	a	2.48	0.21	a	0.25	0.06	a	0.21	0.03	ab	0.08	0.01	b	7.70	0.91	a	6.19	0.55	a	0.34	0.05	a	23.46	3.40	a	2.43	0.38	a	24.16	6.34	a	660.43	80.92	a
Mixed 3																																				
0–5 cm	4.33	0.13	a	2.62	0.42	a	0.98	0.18	a	0.67	0.05	ab	0.16	0.04	ab	6.46	0.47	a	14.67	1.74	a	0.64	0.10	a	15.98	2.01	a	2.70	0.51	ab	47.00	19.64	a	584.31	148.11	a
5–15 cm	4.35	0.10	a	2.93	0.37	a	0.23	0.06	a	0.23	0.04	ab	0.09	0.01	b	4.97	0.65	a	6.25	1.03	a	0.33	0.08	a	28.11	5.16	a	4.54	1.49	a	32.64	11.06	a	533.21	190.65	a
Regeneration 3																																				
0–5 cm	4.58	0.06	a	1.90	0.26	a	1.12	0.10	a	1.02	0.15	a	0.31	0.07	ab	7.59	0.47	a	9.78	0.47	b	0.50	0.01	a	20.61	3.07	a	4.11	0.64	a	72.70	29.32	a	526.40	97.86	a
5–15 cm	4.55	0.03	a	1.97	0.34	A	0.58	0.14	a	0.46	0.11	a	0.16	0.04	ab	5.42	0.68	a	5.60	0.64	a	0.30	0.10	a	25.47	4.52	a	3.18	0.56	a	42.44	15.34	a	490.96	161.19	a

Note: Averages, standard error (SE), and statistical significance (Sig). The differences between treatments for each parameter and depth are statistically significant ($P < .05$) when the means are followed by different letters

This species is planted in agroforestry systems by farmers, for example, in combination with annual crops such as cassava (Haggar et al., 1999).

In contrast, and in spite of its good growth, farmers are not planting *J. copaia* due to its poor timber quality. However, *J. copaia* is highly appreciated and planted in other countries of Latin America (e.g., Colombia), where it also grows as a native tree.

Since *Calophyllum brasiliense* has very good timber quality but slower growth, a good alternative would be to combine it in silvopastoral systems, so that the earlier earnings from the cattle products could help offset the relatively high maintenance costs and longer rotation times. The authors have observed cattle grazing under *C. brasiliense* plantations in a private property located in the region. Pruning practices are needed for this species to maintain good tree form and also to let enough light to reach the understory and allow the growth of pastures.

Among the species tested in Plantation 2, *T. amazonia* and *Virola koschnyi* appear as good species for combination in agroforestry systems, due to their good growth and good timber quality. Understory vegetation was abundant under *T. amazonia* (Montagnini, 2001), suggesting a good potential for its combination with crops. *V. koschnyi* also encourages abundant understory vegetation (Montagnini, 2001). Beneficial effects on some soil nutrients have been reported under *V. koschnyi* and *T. amazonia* (Montagnini, 2000a, 2000b), again suggesting that these species would aid in soil restoration of degraded lands. *T. amazonia* had the highest rates of litterfall, and its litter decomposed the fastest among the four species tested in this plantation, suggesting fast nutrient release from litter to soil under this species (Kershnar and Montagnini, 1998). *T. amazonia* is currently used in agroforestry systems in the country in combination with agricultural crops (Haggar et al., 1999).

D. panamensis, with its good timber quality and its slower growth, could also be combined with cattle to help offset the higher costs of plantation maintenance and longer rotation times. People are attracted to planting this species because its timber price has recently increased in local markets, its extraction from natural forests has been banned and its fruits are the main food source for the green macaw, an endangered species in the country. *D. panamensis* is also being used in agroforestry systems by farmers in Costa Rica (Haggar et al., 1999).

H. alchorneoides and *Vochysia ferruginea* appeared the most promising species for agroforestry combinations from the species tested in Plantation 3. In fact, *H. alchorneoides* is one of the species that have been used the most for combination with cattle in the region. This species encourages abundant understory (Carnevale and Montagnini, 2000, 2002), and results of nutrient-cycling studies have shown that growth of test crops was favored from nutrient release from its litter (Horn and Montagnini, 1999). Under plantation conditions in the experimental site, *V. ferruginea* had abundant leaf litter production that covered the ground and protected against soil erosion (Stanley and Montagnini, 1999; Horn and Montagnini, 1999). This dense litter cover may not favor the growth of pastures under its canopy; however, this effect could be compensated with wider spacing such as is generally used in silvopastoral systems.

Vochysia guatemalensis, *C. brasiliense*, *T. amazonia*, *Virola koschnyi*, *D. panamensis*, *H. alchorneoides*, and *Vochysia ferruginea* are currently planted by farmers in the region, and they are all being used in silvopastoral combinations with beef cattle when the trees reach about 5 years of age and their canopy becomes more open and allows the growth of natural grasses (Montagnini et al., 2003). Three of the species tested in the present experiments are not recommended for agroforestry systems due to poor growth or pest problems (*S. microstachyum*, *Paraserianthes guachapele*, *G. americana*). Further observations are needed to confirm if *C. brasiliense* maintains good health and growth in plantations other than in the experimental setting at La Selva.

Balizia elegans (a N-fixing species) and *J. copaia* have good growth and combining abilities but due to their low timber value in local markets farmers in the country do not currently prefer them. However, planting *J. copaia* may become more popular as timber scarcity increases in the country and people decide to turn to fast-growing plantation species of good performance.

B. elegans may also turn into a preferred species for agroforestry due to its N-fixing ability. Its sparse canopy allows for combinations with crops or pastures. Its small leaves decompose rapidly and release nutrients to the soil (Horn and Montagnini, 1999). Almost no litter can be found on the soil under the canopy of this species due to its fast decomposition (personal observation, June 2006). Although not the best species for soil protection against erosion, this characteristic points to this species as a good candidate for combination with agricultural crops.

The faster growing species of these experiments, such as *Vochysia guatemalensis*, *Virola koschnyi*, and *H. alchorneoides*, can help recover degraded pastures by rapid establishment of tree cover (2–3 years). For the slower-growing species, such as *C. brasiliense* or *D. panamensis*, combination with cattle helps offset the higher costs of establishment and maintenance of the tree plantation, and their higher timber value will result in higher revenues in the long term.

In all three plantations, the mixtures performed very well in terms of tree growth. A mixed-species treatment may combine beneficial effects of the different species' characteristics with the additional advantage of product diversification, an important factor among the small farmers of the region (Montagnini et al. 2005). In addition, for the species that had pest or disease problems, at least one-third of the individuals survived in mixed plantation, whereas mortality was total in pure plots. Therefore, mixed plantations can help survival by sheltering or protecting the trees against pest and disease problems.

12.4 CONCLUSIONS AND RECOMMENDATIONS

The calculation of the nutrient balance of tree plantations can help in the selection of tree species for tree–crop combinations by favoring mechanisms of nutrient recycling and conservation of site nutrients. The results of the present study indicate that to reduce the nutritional costs of tree harvesting, the tree foliage and branches should be left in situ. By doing so, it is possible to reduce by half the nutritional loss from harvesting.

The effect of the tree species of an agroforestry system upon the nutrients of a site depends upon the balance between the uptake of nutrients by the biomass and the recycling of these nutrients to the soil. The mixed plots had average values for the studied nutrients, and even improved the conditions of some soil nutrients such as P and organic matter.

It is interesting to observe that the soil under *Vochysia guatemalensis* had higher pH, less acidity, and greater Mg than other treatments, a result that was only observed for samples taken when plantations were 12 years old. Initially these effects were not observed, and it is important to know the impact of this species on a site given that it is a species preferred by producers in northern Costa Rica for reforestation. Also, soils under plantations of *V. ferruginea* had larger quantities of organic matter, a result that held constant over the long term and which is consistent with other studies that include this species. The long term results showed that soils under *D. panamensis* had greater K, which was also consistent with prior results.

The large quantity of leaf litter produced by *V. guatemalensis*, *T. amazonia*, *H. alchorneoides*, and *V. ferruginea* makes these species useful for protecting soils against erosion. Mixed plantations offer the combined benefits of these species: protection of soils against erosion in the case of abundant leaf litter, and slow decomposition and rapid release of nutrients to the soil in the case of species with high nutrient content and fast decomposition. In addition, mixed plantations have other additional advantages, such as promoting biodiversity and product diversification and helping in protecting the trees against pest and disease attacks.

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13 Root Competition for Water between Trees and Grass in a Silvopastoral Plot of 10 Year Old *Prunus avium*

Philippe Balandier, François-Xavier de Montard, and Thomas Curt

CONTENTS

13.1 INTRODUCTION

In temperate Europe, fast-growing broad-leaved trees such as wild cherry (*Prunus avium* L.) supply highly valued wood with a veneer end use. The wild cherry tree has a high light requirement (Ruchaud, 1995), which makes it a species potentially well adapted for agroforestry purposes where trees are planted with very wide spacing to allow intercropping or grazing (Balandier and Dupraz, 1999). Cattle or sheep maintain grass and shrubs at low height and add an income from animal products for the owner. With the help of tree pruning (Balandier, 1997), such a silvopastoral system has proved efficient in producing straight knot-free quality boles (Balandier et al., 2002).

Agroforestry practice requires that the biological and physical relationships between the different components of the system (for instance tree and crop or pasture) generate a favorable balance between negative and positive interactions (Anderson and Sinclair, 1993). In other words, the trees must utilize resources that the crop does not (Cannell et al., 1996) and *vice versa*. This is also called the niche theory: two or more species must use resources differently if they are to coexist on a site (Kelty, 1992). However, though often postulated, such a relationship has seldom been demonstrated, particularly as regards interactions at the root level (Kelty, 1992), namely competition for water, nutrients uptake, allelopathy, etc.

The wild cherry tree is a species known to be sensitive to intra- and interspecies competition (i.e., other trees, shrubs, or grass; Collet et al., 1993), which adversely affect its growth and the quality of its wood (Le Goff et al., 1995). Therefore, when it is associated with a crop or pasture in agroforestry, the question arises of whether the balance of interactions will be positive.

The basic mechanisms that lead to growth impairment of wild cherry in competition with grass or shrubs are not fully known (Lucot, 1997). Most studies have been indirect: the elimination of weeds or shrubs around trees has a positive effect on growth in height and especially in diameter or biomass for young trees (Monchaux, 1979; Frochot and Lévy, 1980; Britt et al., 1991; Collet and Frochot, 1992; Campbell et al., 1994; Le Goff et al., 1995; Balandier et al., 1997; Cain, 1997; Davis et al., 1999) and a positive effect on root growth (Larson and Schubert, 1969). Interactions between trees and weeds or shrubs, although demonstrated practically, need to be more fully described in terms of specific processes to form a basis for improving tree management (Nambiar and Sands, 1993). Some functional physiological studies have been conducted on very young trees, but often in containers or not in natural conditions (Collet et al., 1996; Jäderlund et al., 1997; Johnson et al., 1998; Mohammed et al., 1998). For instance, the leaf water potential of trees in association is often more negative than that of trees in bare soil (e.g., *Juglans regia* L. with *Trifolium subterraneum* L., Pisanelli et al., 1997; *Pinus strobus* L. with *Populus tremuloides* Michx., Boucher et al., 1998; *Quercus robur* L. and *Fagus sylvatica* L. with natural herbaceous vegetation, Löf, 2000). Tree transpiration, leaf CO_2 assimilation, and leaf conductance can also be altered by herbaceous competition (*Pinus radiata* D. Don with *Dactylis glomerata* L., Miller et al., 1998; *J. regia* with *Lolium perenne* L., Picon-Cochard et al., 2001).

Girardin (1994) concluded from a study on 4 year old wild cherry trees that as this species has a very shallow root system, it suffers badly from competition by grass. However, the study was indirect and the true depth of the tree root system was not measured directly. Even so, all the studies conducted suggest that trees do suffer from such competition, to different extents depending on the competing species (Nambiar and Sands, 1993; Miller et al., 1998; Dupraz et al., 1999; Coll et al., 2003) and that this competition can reduce their growth and sometimes prevent their establishment.

Allelopathy, the release of toxic chemicals in the environment by a plant or a tree is other possible negative interference, which can reduce either tree growth or grass production. In agroforestry systems, some trees were characterized as probably having an allelopathic inhibitory effect (e.g., *Juglans* sp., *Eucalyptus* sp., Gallet and Pellissier, 2002). Many grasses were also reported to have such similar effects (Qasem and Foy, 2001). However, nothing is mentioned on a potential allelopathic effect of the wild cherry tree or the main herbaceous species composing the pasture (see Section 13.2) in the study reported here (Qasem and Foy, 2001), except perhaps for *Holcus lanatus*.

Much work has been done on competition between trees and grass in agroforestry systems with pine (e.g., Nambiar and Sands, 1993; Yunusa et al., 1995, for *P. radiata*) and *Eucalyptus* (Eastham and Rose, 1990 for *Eucalyptus grandis* Maiden) and for warm climates (Scholes and Archer, 1997; Balandier, 2002). However, the literature is much more scant for temperate climates and broad-leaved species such as wild cherry.

Here we report on interactions at the root level between trees and grass in a temperate silvopastoral system with 10 year old broad-leaved wild cherry trees in natural conditions. Competition for light and for nitrogen in such a system has already been reported (De Montard et al., 1999; Méloni, 1999). Nutrients other than nitrogen are present in the soil in supraoptimal values and competition for them was unlikely. Therefore, this chapter focuses on interactions for water. We studied not only the aerial growth of the tree but also its water status, its root growth through direct measurement, time course of volume soil water content, and the interactions between these different parameters to understand more fully and so better manage the water competition relationships between tree and grass.

13.2 MATERIALS AND METHODS

13.2.1 EXPERIMENTAL PLOT

The experiments took place in a silvopastoral plot of 2.9 ha in Auvergne, Central France (approximate latitude 46°N and longitude 3°E), in hilly country, at an elevation of 810 m a.s.l. The plot slopes moderately (from about 8%–15%). Two year old wild cherry trees (*Prunus avium* L.) were planted directly with minimum tillage of the soil in March 1989 at 200 stems ha^{-1} (6×8 m) on a permanent pasture grazed by sheep. For practical reasons during the experiment—from 1997 to 1999—the sheeps were kept out of the experimental plot (about 1000 m^2) and the pasture was regularly cut by hand to simulate sheep browsing. The main species of the pasture were orchard grass (*D. glomerata* L.), hairy oat grass (*Avena pubescens* Huds.), yellow oat-grass (*Trisetum flavescens* [L.] P. Beauv.), velvet grass (*H. lanatus* L.), Erect Brome (*Bromus erectus* Huds.), red fescue (*Festuca rubra* L.), white clover (*Trifolium repens* L.), Bush vetch (*Vicia sepium* L.), common yarrow (*Achillea millefolium* L.), and Germander speedwell (*Veronica chamaedrys* L.). Trees were weeded with glyphosate (3.6 g L^{-1}) during the first 4 years after planting (i.e., from 1989 to 1992) within a radius of 0.6 m around their trunk to ensure firm rooting. None of the trees in this study were pruned.

13.2.2 CLIMATE AND SOIL

Average annual rainfall was 835 mm, fairly evenly distributed throughout the year but sometimes with pronounced drought periods (e.g., about 15 March–07 May, 4–11 June, 18–25 June, 15–22 July, and 30 July–20 August in 1997, 10 May–09 June and 09–30 July in 1998, and 30 May–09 July in 1999). The mean annual temperature was about 9°C. The soil was a slightly acid granitic brown soil (brunisolic order—Orthic B, Canadian soil classification 1998; $\mathrm{pH_{water}} = 5.8$, the organic matter ranged from about 65 g kg^{-1} in the upper soil layer to 6 g kg^{-1} in depth which corresponds to a moderately fertile soil) topped by a thin basaltic colluvium, and soil depth reached up to 180 cm. On average, the first layer (about 0–15 cm) of the soil displayed a sandy-silt texture with a micro-lumpy structure. The proportion of coarse elements (i.e., >2 mm) was about 10%. The compactness was low. The second layer (15–40 cm) had the same texture (sandy-silt) but was more compact with a high density and coarser elements (40%); the structure was heavier. The next two layers had a silty-sand texture with a heavy structure and a high proportion of coarse elements (60%–70%). Taking into account the proportion of the coarse elements, the calculated total available water content of the soil (Baize and Jabiol, 1995) to a depth of 120 cm deep was about 85 mm. Among the different trees, there were some small differences in soil layer depth and compactness. Wherever possible, we tried to take into account these small variations when analyzing growth data. For each layer of soil, the soil water content corresponding to the wilting point (pF of 4.2 or 16 atm., i.e., by convention, the soil potential over which plant roots cannot extract water) was assessed after establishing curves of "soil potential–soil water content" (Lucot, 1997); on average, for a 20 cm thick layer, the soil water content at the wilting point is about 12 mm. Apparent density was also calculated from soil samples at different depths ($d =$ total soil sample dry weight/soil sample volume, g cm^{-3}).

13.2.3 EXPERIMENTAL DESIGN

Observations and measurements were made on eight trees selected among the most vigorous ones (i.e., trees that had heights and trunk diameters in the upper quartile). In this way, we avoided puny trees, for which poor growth may be due to disease and not due to competition with grass. At the beginning of the experiment (spring 1997), the average height of the trees was 6.5 m and the average trunk diameter at 1.3 m was 8 cm. Three trees were weeded (grass suppression, T – G treatment) in March 1997 with glyphosate (3.6 g L^{-1}) in a 4 m radius around the tree trunk to form a control with no grass competition. Their growth was compared with that of five trees maintained in grass (T + G treatment). Two control plots (or subplots, 100 m^2 each) were installed about 30 m away from the trees; a plot with only grass and no tree (G treatment) and a plot with bare soil (BS treatment). For the T – G trees, regular treatments with glyphosate (3.6 g L^{-1}, one treatment every year at the beginning of the growing season) and manual harrowing (several times in the year) were carried out for 3 years to keep the soil grassfree. All the trees were regularly treated against aphids and *Blumeriella jaapii* (with, respectively, deltamethrine 0.00075 g L^{-1} and doguadine 0.72 g L^{-1}).

13.2.4 MEASUREMENTS

13.2.4.1 Tree Dimension

Tree trunk girth at breast height (1.3 m) and total height of each tree were measured manually every week from 1997 (when trees were 10 years old) to 1999. In addition, for trunk diameter increment, an automatic electric sensor (LVDT type, Solarton DF 2.5) was fitted to the trunk of each tree at about 1.3 m height to record daily variations in trunk diameter: contraction in the day was due to water loss through transpiration flow, and increase during the night was due to water uptake and growth (Améglio and Cruiziat, 1992). The sensor was accurate to less than 2.3 μm.

13.2.4.2 Tree and Grass Water Status

Predawn (ψ_p) and midday (ψ_m) leaf water potentials of tree and grass were measured each week with a pressure chamber (Scholander et al., 1965). The grass cover was made up of several species. As we were unable to make water potential measurements on all the species present, we chose the most representative species based on abundance for these measurements, that is, *Avena pubescens* in 1997 and *D. glomerata* in 1998 and 1999.

13.2.4.3 Grass and Tree Root Growth

Grass and tree root densities and elongations were calculated using rhizotrons. Three rhizotrons were installed in April 1997 in three directions at 1.1, 2.2, and 3.3 m from the trunk of a T – G treatment tree and from the trunk of a T + G treatment tree. One rhizotron was set up in the G treatment. In 1998, two additional rhizotrons were installed 2.2 m from a T – G tree and a T + G tree. Each rhizotron was 1.25 m deep and 1.0 m wide. Such a dimension was necessary to assess 10 year old tree root systems. The number of rhizotrons was voluntarily limited, given their dimension, to avoid disrupting too much tree growth. In spite of some disadvantages such as modified microclimatic conditions (Taylor et al., 1990; Vogt et al., 1998), rhizotrons allow sequential measurements to be made of the same roots without any destruction (Lopez et al., 1996). Minirhizotrons were not used because they are much more expensive and require numerous long tubes to estimate such large root systems accurately (Franco and Abrisqueta, 1997).

13.2.4.4 Soil Water Content

Volume soil water content was measured every week in 20 cm thick layers to a depth of 80 cm with a TDR probe (Time Domain Reflectometry IMKO device). The TDR probe used was a tube type adapted for measurements in permanent thin-walled plastic tubes. Thin-walled tubes were driven vertically into the soil with the help of an auger. Measurements were made every week by lowering the probe into the tubes with a stop measurement every 20 cm to a maximum depth of 80 cm. Three tubes were placed 1.1, 2.2, and 3.3 m (i.e., at the same distance as rhizotrons from tree trunks) from each T − G tree and each T + G tree. Control tubes were driven below the G and BS treatments. The use of the TDR technique is a proven method for measuring soil water content accurately with limited disturbance of the soil and root distribution (Werkhoven, 1993; Todoroff and Langellier, 1994; Mastrorilli et al., 1998).

13.2.4.5 Data Analysis

It was not possible to perform all the measurements in this experiment on more than eight trees, which was already a large task; a thorough statistical analysis was therefore impossible. However, as all the measurements were done at tree scale, it was nevertheless possible to link individual tree growth to each tree's local conditions: soil characteristics, evolution of soil water content, depth and density of tree and grass roots, etc. Hence, the response of each individual tree was analyzed taking into account the "treatment" variable (with or without grass) as a first explanatory variable and the microsite conditions for each individual tree as a secondary, or covariate factor. Variations in soil water content, which are less sensitive to the initial conditions than absolute values, were set as a cofactor to explain tree growth. In the same way, the relative growth rate (RGR) was calculated for the different tree growth variables (height, diameter, root elongation, etc.), to take into account the initial size of the tree in its growth response (Causton and Venus, 1981; Collet et al., 1996). RGR (day^{-1}), for instance for girth, for a given period of time t_1 to t_2 (in number of days) was calculated by:

$$\mathrm{RGR} = \frac{[(C_2 - C_1)/(t_2 - t_1)]}{C_1}, \tag{13.1}$$

where
C_1 is the girth at t_1
C_2 is the girth at t_2

Relationship between tree growth and causal variables (i.e., soil water content) was based on regression analysis using the general linear model (Statgraphics plus 5.1 software).

Each value of water potential was the mean of three leaf measurements sampled in different parts of the crown of each tree. The value for grass was the mean of 8–10 leaves sampled on different grass clumps. Each TDR value (i.e., for a 20 cm layer from a particular tube) was the mean of three measurements made in three different directions.

13.3 RESULTS

13.3.1 ABOVEGROUND TREE GROWTH

During the whole study period, T − G trees displayed a much better height and especially girth growth than T + G trees (Figure 13.1) and differences tended to increase with time. After 3 years, the T − G tree girth increment was about twice that of T + G. Over the season, girth RGRs (Figure 13.2) showed some global variations according to tree phenology (i.e., in general, RGR increased at the beginning of the season and decreased at the end), and also that the girth RGRs of T−G trees were often greater than T + G girth RGRs, especially during the drought periods (e.g., 4–11 June, 18–25 June, and 15–22 July in 1997; similar data were found in 1998 and 1999).

13.3.2 SOIL WATER CONTENT

Volume soil water content fluctuated according to rainfall events and treatments (Figure 13.3). Only data of 1997 are presented, the same soil water patterns being recorded in 1998 and 1999. Only the variations of the 0–20 cm and 40–60 cm soil layers are presented, the 20–40 cm soil layer showing results intermediate between the 0–20 and 40–60 cm soil layers, and the 60–80 cm soil layer

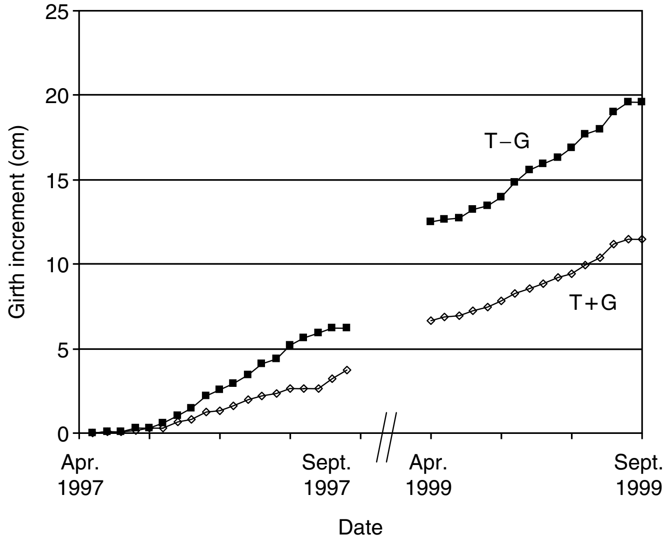

FIGURE 13.1 Mean tree girth increment at breast height for the T − G and T + G treatments for the period 1997–1999 (data of 1998 not shown).

showing no variation. For the 0–20 cm layer, the soil water content of the T − G treatment was about the same as the BS treatment (Figure 13.3a). In contrast, the soil water contents of the G and markedly for the T + G treatment were much lower than under the BS and T − G treatments, and fluctuated widely according to rainfall events. In the 40–60 cm deep layer, soil water content was much more stable than in the 0–20 cm deep layer (Figure 13.3b) and showed only small variations following some isolated rainfall events for the T − G treatments. In this layer, the soil water content was globally low in comparison with the 0–20 cm layer (between 20 and 30 mm for the T + G treatment). As observed in the 0–20 cm deep layer, we recorded the same hierarchy among the treatments regarding soil water content in the 40–60 cm deep layer, BS > T − G > G > T + G.

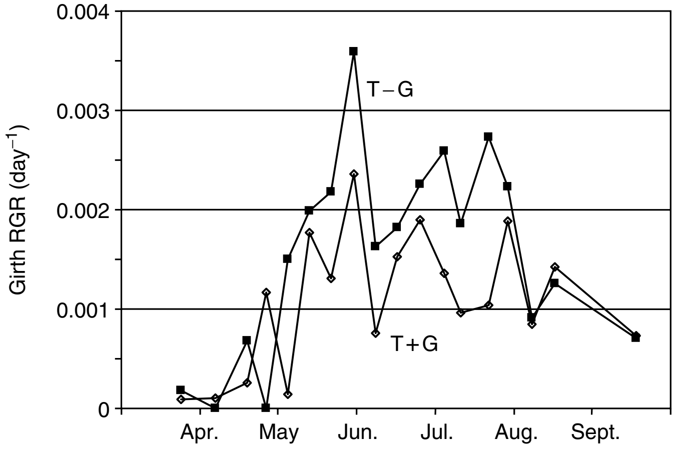

FIGURE 13.2 Mean tree girth RGR time course for the T − G and T + G treatments over the season: example for 1997. Each point corresponds to the RGR between two consecutive dates.

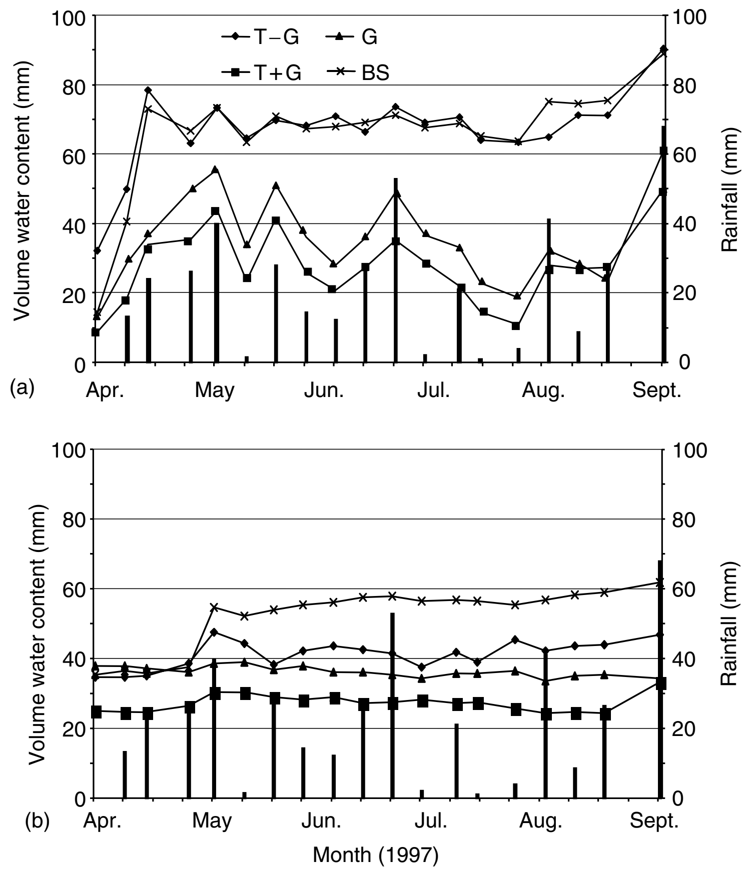

FIGURE 13.3 Volume soil water content dynamics (example for 1997) for the different experimental conditions. Values for the T − G and T + G trees are those at 2.2 m from the trunk. (a) 0–20 cm deep layer and (b) 40–60 cm deep layer. (Results of the other layers not shown.)

Focusing on the periods corresponding to marked differences in the T − G and T + G girth RGRs (Figure 13.2), it is clear that these periods corresponded to ranging degrees of soil water deficit according to date and treatment (Table 13.1).Total soil water contents for the 0–80 cm deep layer were always greater for the T − G treatment (about 200 mm) than for the T + G treatment (under 140 mm), resulting in a high availability of water for trees of this T − G treatment and a corresponding high tree RGR (Table 13.1, RGRs were always greater than 0.00163 day^{-1}). When the amount of available water decreased severely (less than 80 mm, i.e., close to the wilting point), girth growth also decreased and even stopped in some particularly pronounced droughts (data not shown). Pooling all the data, a close relationship between girth RGR (10^{-3} day^{-1}) and water availability (WA) (mm) was established:

$$\begin{aligned} \text{RGR} &= 0.0177\ \text{WA} - 0.8083, \\ R^2 &= 0.68, \\ n &= 12. \end{aligned} \tag{13.2}$$

Figure 13.4 shows in detail the variations of soil water content a few days before and after 22 July 1997, a period of severe drought, according to soil layer depth, rainfall event, and treatment.

TABLE 13.1

Measured Total Soil Water Content (mm) Using the TDR Probe for the 0–80 Deep Layer at 2.2 m from Tree Trunk, Calculated Soil Water Content (mm) Corresponding to the Wilting Point of the Same Layer (See Section 13.2) and Resulting Water Content Available for Plant (Total Water Content–Wilting Point Water Content) for Three Different Dates and Associated Girth RGRs (Year 1997)

Treatment	T − G			T + G		
Date	11 June	25 June	22 July	11 June	25 June	22 July
Total soil water content (mm) as measured with TDR probe for the 0–80 cm deep layer (1)	212	205	198	131	117	105
Soil water content (mm) corresponding to the wilting point for the 0–80 cm deep layer as deduced from "soil potential–water content" curves (2)	53	53	53	42	42	42
Resulting soil water content (mm) available for plant (1–2)	159	152	145	89	75	63
Girth RGR (10^{-3} day^{-1})	2.18	1.63	1.86	1.3	0.75	0.95

Variations in the soil water content were much more marked for the T − G than for the T + G treatment (Figure 13.4). For the T − G treatment (Figure 13.4a), the soil water content of the 0–20 cm deep layer varied according to the rainfall events: increase with rainfall, decrease with dry period. For the deepest layers, there was a time lag between precipitation events and increase in the soil water content. The water percolation toward the deepest layers sometimes took several days. For the T + G treatment (Figure 13.4b), no such variations were observed, nor was any water transfer toward the deepest layers observed. Only the 0–20 cm deep layer showed some small variations. It seems that all the water coming from rainfall events was taken up in this 0–20 cm layer as there was no variation in the deepest layers.

13.3.3 WATER STATUS OF TREES

Table 13.2 gives the tree and grass leaf water potentials for two consecutive dates of measurement in 1998: 23 July and 30 July, respectively, before and after a period of water deficit. The total amount of rainfall water between 11 June and 9 July was 40 mm; there was then no rainfall for 2 weeks till 23 July and a rainfall event of 15 mm between 23 and 30 July.

The mean value of ψ_m for trees was very negative and some individual values were as low as −2.5 MPa for some trees. Despite this severe stress during daytime, the much less negative values of ψ_p indicated that the trees rehydrated themselves partially during the night (Table 13.2). However, there was a significant difference between trees of the T − G and T + G treatments in predawn leaf water potential, whereas values for the midday water potential were insignificantly different (Table 13.2). Clearly T − G trees rehydrated themselves overnight more than T + G trees.

The recorded tree diameter microvariations between 6 and 28 July 1998 (Figure 13.5) confirmed the leaf water potential measurements: tree contraction during the day reached 0.5 mm (e.g., 20 July—day 201) indicating marked water stress. However, while T + G tree growth was greatly reduced during this period (Figure 13.5), T − G trees continued to display an impressive growth due

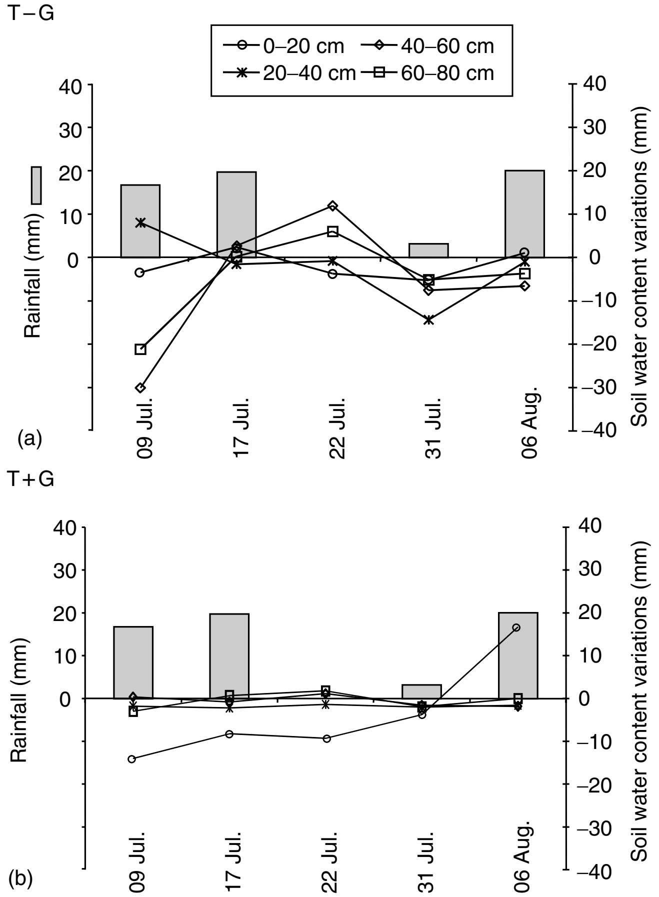

FIGURE 13.4 Relative variations (according to the initial value at the beginning of the season) of volume soil water content (mm) as measured by TDR probe at 2.2 m from tree trunk for the period around the 22 July 1997, which was a dry one, according to rainfall events and treatment. (a) T − G and (b) T + G. Each curve corresponds to a soil layer.

TABLE 13.2

Mean Predawn (ψ_p) and Midday (ψ_m) Leaf Water Potential for Trees and Grass (*Dactylis glomerata*) for Two Dates in 1998 (see text for more details)

	23 July		30 July	
	ψ_p in Mpa (±SD)	ψ_m	ψ_p	ψ_m
T−G	−0.32 (0.05)*[a]	−1.64 (0.31)	−0.28 (0.02)*[a]	−2.08 (0.26)
T+G	−0.74 (0.19)	−1.77 (0.21)	−0.57 (0.11)	−1.92 (0.21)
Grass close to the tree[b]	−2.98 (0.72)	−3.77 (0.11)	−0.84 (0.42)	−2.75 (0.49)
Grass far from the tree	−2.06 (1.41)	−3.01 (0.30)	−0.34 (0.08)	−2.16 (0.12)

[a] * Indicates a significant difference between T−G and T + G with a risk level of 5%.

[b] Grass close to the tree is grass in a radius of 1 m around the tree trunk. Grass far from the tree is grass about 3 m from the tree.

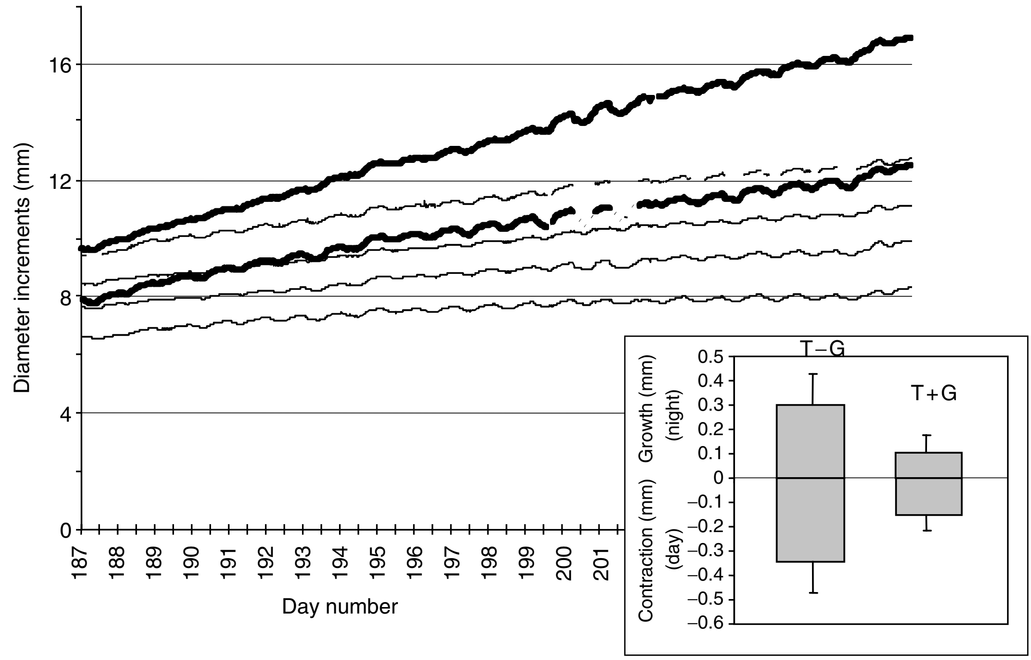

FIGURE 13.5 Tree diameter microvariations between 6 and 28 July 1998 for the T−G trees (thick line) and the T + G trees (thin line), each curve represents one tree, and in insert, comparison between the T−G and T + G tree diameter mean (and standard deviation) net growth during the night and contraction during the day for three days (199, 200, and 201).

to good rehydration during the night (i.e., the water balance between night and day was strongly positive for T − G trees but near zero for T + G trees).

Grass ψ_p and ψ_m for 23 July were strongly negative (Table 13.2). Grass located far from the tree (3 m from the trunk) was always less stressed than grass close to the tree (1 m from the trunk) although the relationship was not statistically significant because of a wide dispersion of the water potential values for grass. After the 15 mm rainfall event between the two dates (23 and 30 July), grass ψ_p for 30 July reverted to a less negative value although ψ_m values were always very negative though increasing (Table 13.2). The trees did not benefit from this rainfall as much as the grass: their ψ_p values were barely less negative and their ψ_m values were more negative than the values of 23 July.

13.3.4 ROOT GROWTH

Rhizotron data showed that grass roots grew mainly in the first 60 cm of soil, with a peak in the 20–40 cm layer, but there were some roots growing even at a depth of 100 cm (Figure 13.6). Tree roots grew mainly 20–80 cm deep, with a peak in the 40–60 cm layer but there were also some roots growing at a depth of 100 cm. There was practically no tree root elongation in the top layer in contrast to grass. Tree root elongation was high at 1.1 m from the trunk and decreased rapidly at 2.2 and 3.3 m from the trunk (Figure 13.6). Irrespective of the depth, the total length of the roots emitted by the grass was much higher than that of the trees (Figure 13.7). The total root length of the grass alone was higher than that of the grass under trees, and the T − G trees emitted longer roots than the T + G trees. Therefore, it seems that in the T + G treatment, the soil space was a limiting factor and both tree and grass root growth was limited. Although the grass root system was longer than the tree root system, the roots of the trees grew faster than those of the grass (Figure 13.8), and it was the

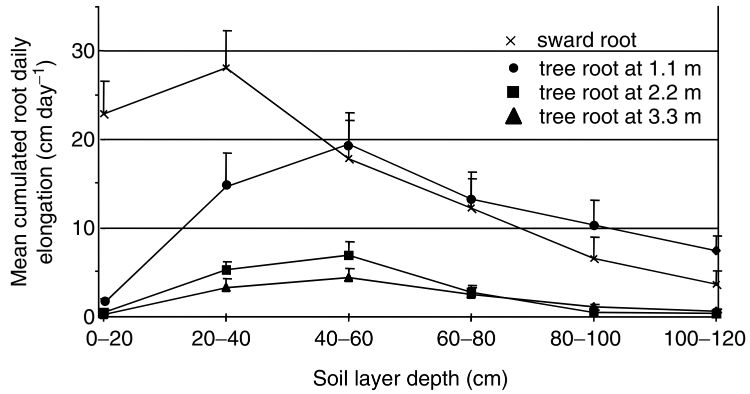

FIGURE 13.6 Mean cumulated root daily elongation (cm day^{-1}) on the rhizotron window according to layer depth of grass and tree of the T + G treatment for the period from 1 May to 6 July 1998. Tree root elongation is given for three distances from tree trunk. Vertical bar represents half the standard deviation.

T − G tree roots that had the highest growth rate. Lastly, the grass emitted a very high number of roots: 10 times more than the trees (Figure 13.9). Therefore, from Figures 13.6 through 13.9, one can conclude that the grass had a very high number of roots but with a relative low growth rate, concentrated in the upper horizons, and the tree emitted only a few roots but with a high growth rate, mainly colonizing the deepest horizons.

Both tree and grass root growth was driven by the soil WA (Figures 13.10 and 13.11). When the soil water content was greater than about 50 mm for a 20 cm depth layer, tree root RGR increased significantly (Figure 13.10). In this figure, we can also see some low RGR at high water levels; they corresponded to early and late low root growth in the season, that is, May and July (Figure 13.8). Grass roots showed the same response to WA, and the maximum RGR was observed for deeper and deeper horizons as the upper horizons became drier and drier as the season progressed (Figure 13.11). In some particularly dry periods, we recorded some grass root deaths in the upper horizons, whereas a high root growth rate was recorded for the deepest horizons (Figure 13.11).

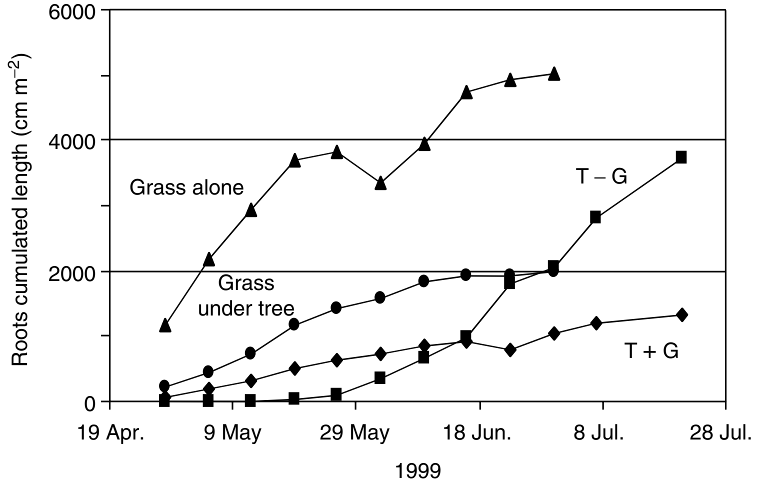

FIGURE 13.7 Root cumulated length (cm m^{-2}) along the 1999 season as recorded on the rhizotron windows for the different treatments.

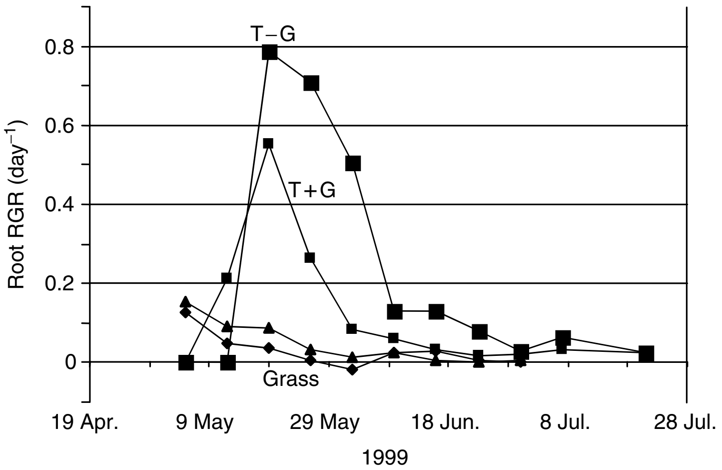

FIGURE 13.8 Root relative growth rate (RGR, day^{-1}) along the 1999 season as recorded on the rhizotron windows for the different treatments. Each point corresponds to the RGR between two consecutive dates.

13.4 DISCUSSION

In this study on the interactions at the root level between 10 year old wild cherry and pasture, data showed that there was clearly a separation of the soil horizons used by the two partners (niche separation strategy, Casper and Jackson, 1997); the grass mainly grew in the upper layers and the tree in the deepest layers, although there was not a strict separation between the two root systems. Moreover, the tree and the grass displayed two different strategies in colonizing the soil space: the tree emitted a rather small number of roots but they grew very fast; the grass emitted a very large number of roots but they grew rather slowly. The length root density per soil volume unit was much greater for the grass than for the trees in the T + G association. The grass root density in the first layer was so abundant that it formed an almost impenetrable cover (data not shown). In contrast, the tree roots were not very dense, but they colonized the soil to a depth of more than 2 m (Lucot, 1997). Such a distribution of the root systems

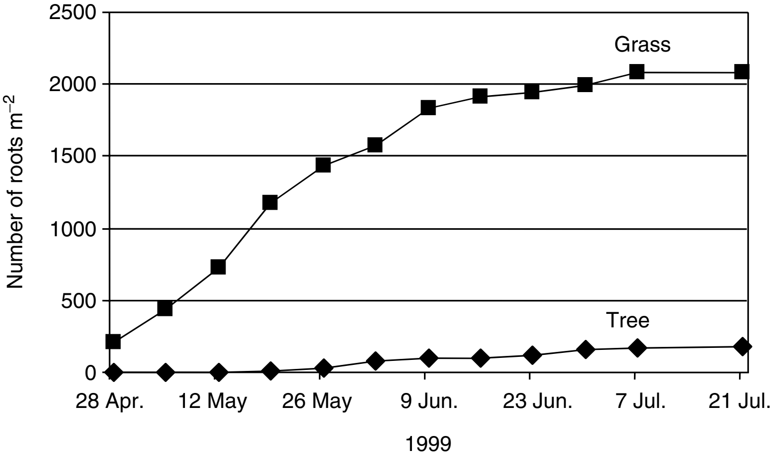

FIGURE 13.9 Cumulated number of roots for the trees and the grass along the 1999 season as recorded on the rhizotron windows.

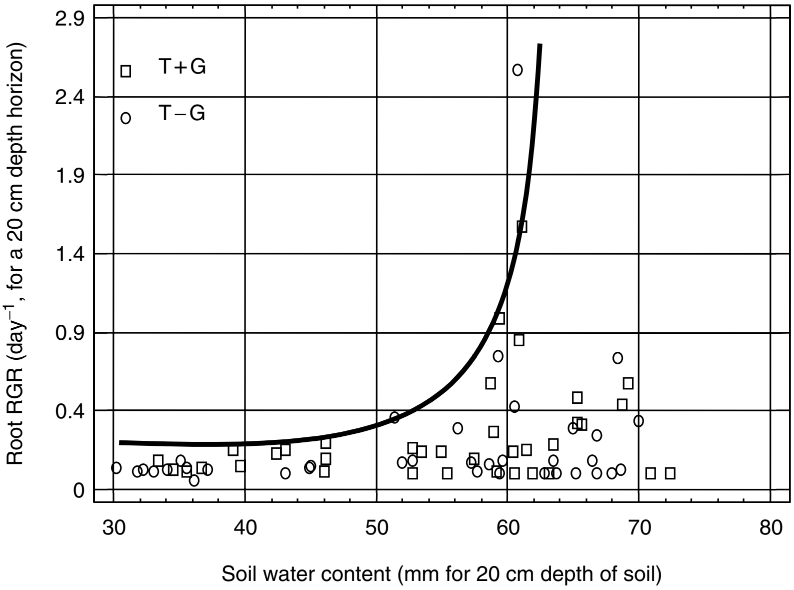

FIGURE 13.10 Tree root relative growth rate (RGR, day^{-1}) calculated from the rhizotron windows for the T + G and T − G treatment in 1999 according to soil water availability (mm). Each point corresponds to a given date and a given 20 cm depth soil layer and its corresponding soil water content.

in the soil between grasses and woody species has already been reported (by Dawson et al., 2001 for very young wild cherries; by Burch et al., 1997 for a pine-hardwood forest; by Eastham et al., 1990 for *E. grandis*; and by Casper and Jackson, 1997 in a general way).

However, even with this different root distribution in the soil between the two plant types, the presence of grass reduced tree diameter (and to a lesser extent, growth height) by at least 30% as

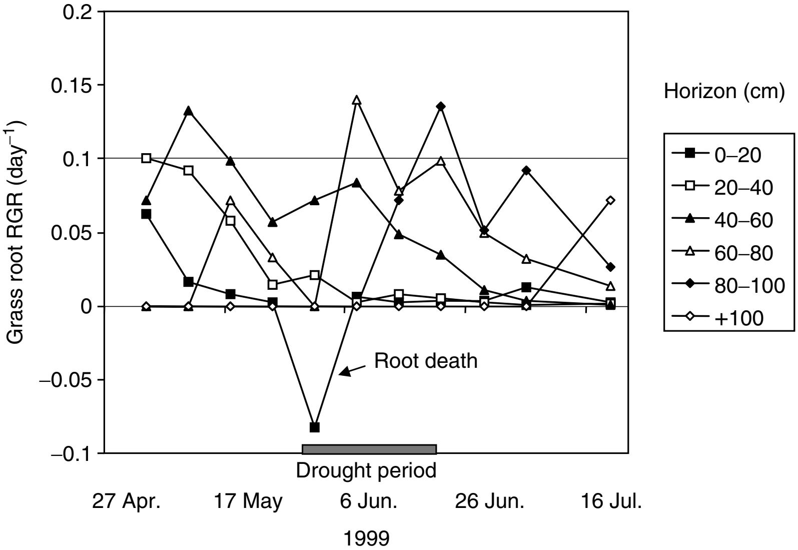

FIGURE 13.11 Grass root relative growth rate (RGR, day^{-1}) calculated from the rhizotron windows for the G treatment in 1999 according to the soil layer depth (0–20 cm, 20–40 cm, etc.).

already noted in other studies or for other species (Monchaux, 1979; Frochot and Lévy, 1980; Collet and Frochot, 1992; Le Goff et al., 1995; Cain, 1997; De Montard et al., 1999). In some circumstances of severe water deficit and especially for young trees with a poorly established root system, grass competition can even kill the tree (Balandier et al., 1997). The aboveground part of the tree was not the only part that was affected by the presence of grass; tree root growth (and also grass root growth) decreased in the wild cherry tree with grass treatment (T + G), as observed for other species (e.g., *J. regia*, Picon-Cochard et al., 2001).

This study provides new data concerning the mechanisms that lead to tree water stress when trees are planted with grass, even for 10 year old trees with deep roots, which ought to insure them against water competition (Nambiar and Sands, 1993). The recorded data showed that the T + G trees were more strongly affected by water deficit than trees alone (T – G). In the case of severe drought (available soil water content close to the wilting point, Table 13.1), the girth growth of the T + G trees sometimes stopped (RGR close to 0) and for higher values of soil water content, there was a good relationship between tree girth RGR and soil water content (Equation 13.2). Even when the drought was not very pronounced, different leaf water potentials between T – G and T + G trees were recorded, indicating that trees with grass were more stressed than trees alone. Therefore, rainfall and grass seem to be very important in accounting for the water variations during the growth season. Trunk diameter microvariations between night and day confirmed these observations and showed that T – G trees grew better than T + G trees even when drought increased (as demonstrated by the increase in the trunk contraction range during the day). This indicated that T – G trees took up more water than T + G trees.

Volume soil water content in the 0–20 cm soil layer was the same in the T – G and bare soil (BS) treatments, indicating that the tree took little water from this layer. Observations made with the rhizotron confirmed this: whether associated or not with grass, tree roots were very weakly developed in the upper soil layer. Tree roots were mainly distributed 20–80 cm deep, suggesting that the trees took up water, essentially in those layers. Comparing ψ_p with the mean soil water potential of each layer (ψ_s) (data not reported here) showed that the dynamics of ψ_p tended to follow those of ψ_s for the 20–60 cm deep layers (Lucot, 1997). This suggests that the trees essentially extracted water in the 20–60 cm deep soil layer, and confirms the analysis of the soil water content time course and root localization in the soil. However, wild cherry can have roots as deep as 2 m in the soil (Bienfait, 1995; Lucot, 1997). Consequently, it may be that in very pronounced water stress, wild cherry can also stock up with water during the night from the very deep layers to survive (Badot et al., 1994). Data recorded here support this possibility, at least for T – G trees: although the leaf water potential was very negative during the day (to -2.50 MPa) and trunk microvariations showed a strong contraction of the trunk indicating a pronounced water deficit, ψ_p was close to zero and the trunk increased notably during the night, indicating that the tree had found a water supply, probably in the very deep layer. None of the methods we employed enabled us to record water dynamics or root elongation in these very deep layers: we were not able to drive tubes for the TDR measurements deeper than 0.8–1 m because of the presence of large stones in the deepest soil layers. For practical reasons, rhizotrons were also limited to a depth of 1.2 m. This imposes a limit on this study. The advantages and disadvantages of rhizotrons have already been discussed elsewhere (Taylor et al., 1990). Another explanation for a pronounced water stress during the day and a water recovery at night could be the very heterogeneous nature of the soil (numerous large stones); in this case, some water "pockets" may have supplied the tree during the night but were not sufficient to bear the transpiration flux during the day (Améglio and Archer, 1996).

Unlike the trees, the grass mainly colonized the 0–20 cm first soil layer and to a lesser extent the deeper layers and displayed a greater total root elongation as also recorded in other studies (Nambiar and Sands, 1993). This gave it an advantage in the uptake of rain water. As we recorded, soil water content below the T – G tree increased following rainfall events, whereas it showed only few variations for the T + G treatments (Figure 13.4), grass roots close to the surface obviously removed a large amount of rainfall water as shown by many other results in this study (e.g., the recording of the leaf water potentials after a small rainfall reported in Table 13.1 clearly shows that only the grass was able to benefit from a small rainfall event: grass ψ_p and ψ_m were significantly less negative after the rainfall event, whereas tree ψ_p and ψ_m hardly changed or even became more negative). Consequently, the deepest soil horizons might have gradually dried up. Moreover, following Davies (1987) for other tree species, even when tree and grass roots colonized the same soil layer, grass roots would absorb water faster than tree roots, owing to their better physiological ability to take up and transport water (Casper and Jackson, 1997). Lastly, as tree root growth is largely driven by the soil WA as shown in this study, the grass, in taking up the rain water preferentially, could maintain a fairly high root growth rate (Figure 13.11), allowing it to prospect more soil volume as soil dryness increased (Figure 13.11), whereas tree roots in the deepest horizons were maintained at a low growth rate, owing to the water uptake by grass roots, and so had a low potential for soil prospecting.

The part played by the water in the interaction mechanism between tree and grass is shown here. However, tree and grass growth are obviously influenced by other factors, associated or not with water, such as nitrogen availability (De Montard et al., 1999). How nitrogen and water act together remains to be studied: a low level of water can limit nitrogen uptake by plant roots, and a high level of nitrogen can increase a tree's resistance to drought.

13.5 CONCLUSION

As stated in Section 13.1, one of the principles of agroforestry is that the different components of the system—here trees and grass—use different resources, or get resources from different locations or at different times, so that the total available resources of the field are utilized. In this study on the association between 10 year old wild cherry and pasture, this assumption seems to be justified, at least at the root level, as there was clearly a separation of the soil horizons used by the two partners; the grass mainly grew in the upper layers, and the tree in the deepest layers, although there was no strict separation between the two root systems. Moreover, the tree and the grass displayed two different strategies in colonizing the soil space: the tree emitted a rather small number of roots but they grow very fast; the grass emitted a very large number of roots but they grew rather slowly. Even so, when wild cherry trees are in the presence of grass, they can suffer severe competition for water, even though they have roots in the deepest horizon that grass roots cannot colonize. This may be the consequence of an almost complete withdrawal of soil water coming from rainfall by the grass roots in the upper soil layer, gradually drying up the deeper soil layers. Hence, the filling of all the soil layers with water in spring is fundamental for tree growth and, in the case of severe water deficit, tree survival. Of course in the worst water climate conditions, weeding the trees, and particularly young trees, even in a small radius around the trunk, can favor better tree growth or survival and so help optimize the agroforestry system.

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Part III

Models in Agroforestry

14 Relationship of Three-Dimensional Structure to Shelterbelt Function: A Theoretical Hypothesis

Xinhua Zhou, James R. Brandle, Eugene S. Takle, and Carl W. Mize

CONTENTS

14.1 INTRODUCTION

Shelterbelts or windbreaks are widely used to provide numerous benefits to agricultural producers, home-owners, and society (Brandle et al., 2000). In the sheltered zone, they reduce windspeed and alter the microclimate. From an agricultural perspective, they reduce wind erosion and increase crop yield and quality, resulting in improved economic return. Shelterbelts can be used to manage snow, distributing it across a crop field or storing it in a narrow drift. They provide protection for homes, reducing energy demands for heating or cooling and provide islands of habitat for numerous types of wildlife in an area otherwise dominated by agricultural crops (Johnson and Beck, 1988).

The degree of windspeed reduction and the resulting microclimate change depends on the structure of a shelterbelt (Heisler and DeWalle, 1988). Characterized by its external shape and internal element arrangement, the structure of a shelterbelt determines the path of boundary-layer flow over the shelterbelt and the momentum sink and shear stress source of boundary-layer flow through the shelterbelt. Different planting combinations of tree species, spacings, and patterns form numerous types of shelterbelts with diverse external shapes and complicated internal element arrangements. The windspeed reduction and microclimate patterns around different types of shelterbelts are very different (Woodruff et al., 1963; Sturrock, 1969, 1972). Thus, the function of a shelterbelt determines the appropriate design for a given shelterbelt. Although the overall arrangement of shelterbelts on the landscape is a major factor in determining shelterbelt effectiveness (Zhou and Sun, 1994), it is the structure of a shelterbelt that determines how the barrier will influence wind flow and microclimate. In order to achieve a given level of protection, the relationship between structure and the resulting windspeed changes must be better understood (Wilson, 1987; Jiang et al., 1994). Once this relationship is established, the structure can be manipulated to achieve a desired goal (Cao et al., 1981; Brandle et al., 2003).

In earlier research, wind fields in the lee of a shelterbelt were predicted by relating its optical porosity to the leeward windspeed (Borrelli et al., 1989; Loeffler et al., 1992). Optical porosity is defined as the percentage of open spaces as seen in a view perpendicular to the shelterbelt and is a two-dimensional (2D) representation of the shelterbelt structure. When a shelterbelt has no significant width dimension, such as a slat fence, optical porosity explicitly means the open portion of the fence through which airflow passes, providing an excellent measure of its structure and a reasonable approximation of the path of the wind flow through the barrier (Baines and Peterson, 1951). Optical porosity is highly correlated to the fence resistant coefficient (k_r, the ratio of across-fence pressure difference over aerodynamic pressure of the approaching flow). Hörner (1965) empirically described the relationship as $k_r = 0.5[3/(2\phi) - 1]^2$ where ϕ is optical porosity. This relationship was successfully used by Wilson (1985, 1987) to describe the drag force in the equations of motion and to predict boundary-layer flows near a slat fence. The resistant coefficient is based on fence porosity, which for a thin barrier is optical porosity.

In the case of a tree shelterbelt, which has a significant width dimension, optical porosity underestimates the path of wind flow through the barrier. Furthermore, two shelterbelts with similar optical porosities may have very different external characteristics in terms of height, width, and cross-sectional shape and very different internal amounts of vegetative surface area and volume. Given one row of green ash (*Fraxinus pennsylvanica* Marsh.) and two rows of eastern red cedar (*Juniperus virginiana* L.), at least two types of three-row shelterbelts can be established by planting one row of green ash in the middle or in either outside row. All may have similar optical porosities, but very different external shapes. As a result, these shelterbelts may produce very different wind flow patterns, depending on wind direction.

Most recently, vegetative surface area density, defined as vegetative surface area per unit canopy volume (S_{AD}), was used to parameterize the drag force in the equations of motion for simulation of boundary-layer flows near a shelterbelt (Wang and Takle, 1995), yielding:

$$\frac{\partial \bar{u}_i}{\partial t} + \bar{u}_j \frac{\partial \bar{u}_i}{\partial x_j} + \bar{u}_j \frac{\partial \overline{u_i' u_j'}}{\partial x_j} = -\frac{1}{\rho_0} \frac{\partial \bar{p}_1}{\partial x_i} - C_D S_{AD} \left| \bar{\vec{U}} \right| \bar{u}_i \tag{14.1A}$$

and

$$\frac{\partial \bar{u}_j}{\partial x_j} = 0, \tag{14.1B}$$

where

subscript i or j ($i, j = 1$, 2, or 3) denotes the dimensions of width, length, and height
overbar ($\bar{\ }$), the temporal average
prime ($'$), the departure of a variable from its mean
C_D, the drag coefficient per unit vegetative surface area
p_1, the instantaneous air pressure
t, time
u_i, the air velocity in the ith dimension

$|\vec{U}| = \sqrt{u_1^2 + u_2^2 + u_3^2}$, the total air velocity
x_i, the coordinate in the ith dimension
ρ_0, the reference air density

In these equations, the vegetative surface area density is used as a structural descriptor of the shelterbelt. As a spatial function in three dimensions, vegetative surface area density describes the amount and arrangement of vegetative surface area within a shelterbelt canopy. In addition, it can indicate the height, width, and cross-sectional shape of that shelterbelt by its domain boundary outside of which it is continuously zero. Wang and Takle (1996, 1997a, 1997b) were able to simulate aerodynamic response of hypothetical shelterbelts with various vegetative surface areas, several distributions of vegetative surface area, and different external characteristics. They found significant differences in wind fields with varying levels of vegetative surface area density (Wang and Takle, 1997b) but very little difference in aerodynamic response with various arrangements of that surface area, different shelterbelt widths (Wang and Takle, 1996), or cross-sectional shapes (Wang and Takle, 1997a).

In contrast, field experience tells us that snow distribution around tree shelterbelts with different planting patterns but similar overall vegetative surface areas is different. Similarly, observations in field and wind tunnel studies showed that shelterbelts with similar internal structural components, but different cross-sectional shapes, had significant differences in leeward minimum relative windspeed, extent of the leeward protected distance, average relative windspeed reduction over a given leeward distance from a barrier, and recovery rate of windspeed from its minimum toward the equilibrium (Woodruff and Zingg, 1952, 1953; Gandemer, 1979; Cao et al., 1981). Inconsistencies of the simulated results with field experience and the observations from field or wind tunnel studies lead to questions about the ability of vegetative surface area density alone to describe the overall aerodynamic structure of a tree shelterbelt.

In related studies, cubic density, defined as solid volume per unit medium volume, and cubic porosity, defined as void volume per unit medium volume, were considered as aerodynamic structural descriptors of a porous medium. Gross (1987) used cubic porosity to parameterize the drag force in the equations of motion to simulate the flow fields around isolated trees with different crown shapes. Kozeny (1927) formulated the pressure decrease of fluid flow across a porous medium using cubic density and surface area density assuming that a porous medium was represented by an assemblage of crooked channels of various cross sections and definite lengths (Carman and Malherbe, 1950; Scheidegger, 1974; Coulson et al., 1978). These studies suggest that a measure of vegetative volume should be included in the full description of the overall aerodynamic structure of a tree shelterbelt.

In review of the numerous field and wind tunnel studies observing the flow fields associated with shelterbelt structure, Zhou et al. (2005) proposed that the overall aerodynamic structure of a tree shelterbelt should be defined in three dimensions by the external characteristics of height, width, and cross-sectional shape and by the internal amounts and arrangements of vegetative surface area and volume, as well as the shape of the vegetative elements. The spatial functions of vegetative surface area density [$S_{AD}(x_1, x_2, x_3)$] and cubic porosity [$\phi_C(x_1, x_2, x_3)$] were suggested as two structural descriptors for the quantitative characterization of this overall three-dimensional (3D) aerodynamic structure. To improve accuracy in predicting windspeed, pressure, and turbulent stress around a shelterbelt as influenced by its overall structure, both structural descriptors need to be incorporated into the equations of motion and the continuity equation.

14.2 EQUATIONS OF MOTION IN A POROUS SHELTERBELT CANOPY

Similar to an air parcel in the atmosphere (Holton, 1992), an air parcel in a shelterbelt canopy is acted on by four forces: (1) a pressure gradient, (2) surface stress, (3) gravity, and (4) the Coriolis force, which is negligible in boundary-layer flows near a tree shelterbelt (Schlichting, 1979).

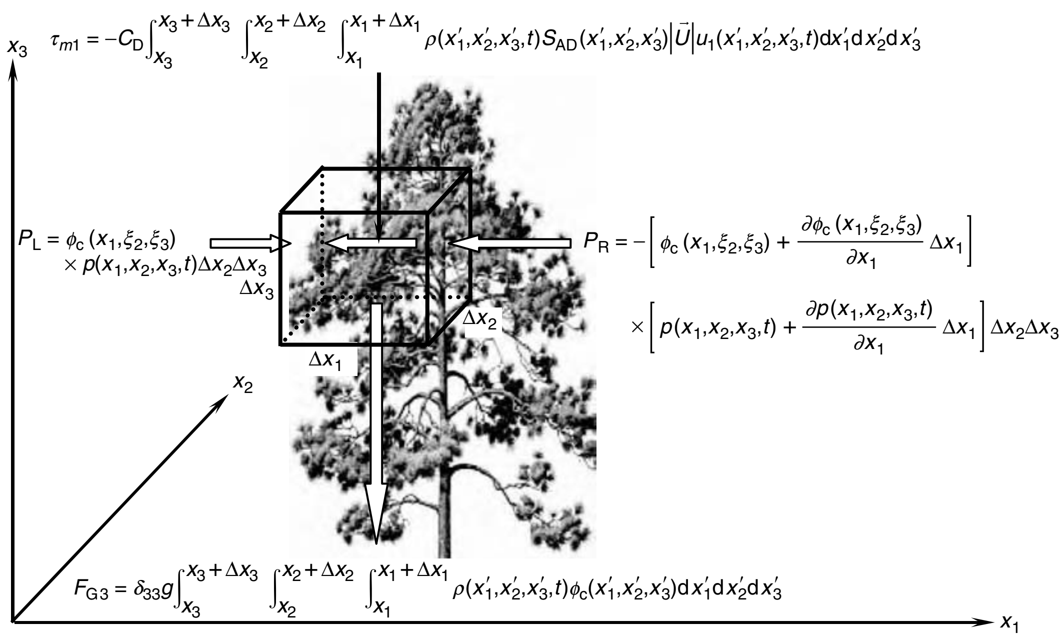

FIGURE 14.1 Illustration of the forces on air inside the cube in a porous canopy in the x_1 direction where C_D is the drag coefficient per unit vegetative surface area; F_{G3}, the gravitational force; g, acceleration due to gravity; $p(x_1, x_2, x_3, t)$, pressure; P_L and P_R, the pressure forces on air inside the cube from its left and right sides, respectively; $S_{AD}(x_1, x_2, x_3)$, vegetative surface area density; $u_1(x_1, x_2, x_3, t)$, air velocity in the x_1 direction; $|\vec{U}|$, the total air velocity; δ_{33}, the Kronecker delta coefficient; ϕ_C (x_1, x_2, x_3), cubic porosity; $\rho(x_1, x_2, x_3, t)$ air density; τ_{m1}, the drag force due to the stresses produced at the solid/pore interfaces in the x_1 direction; ξ_2 and ξ_3, appropriate values in a given domain; and $\Rightarrow$, force direction.

Figure 14.1 illustrates the remaining three forces, which act on air inside a permeable cube filled with vegetative elements: (1) the pressure gradient across the two sides perpendicular to the x_1 direction, (2) the stresses produced at the solid-pore interfaces in the x_1 direction, and (3) gravity in the x_3 direction.

The void volume inside the cube (V_0) is the integration of cubic porosity over the cube. Using the intermediate value theorem (Burden and Faires, 1993), this volume can be expressed as:

$$V_0 = \int_{x_3}^{x_3+\Delta x_3}\int_{x_2}^{x_2+\Delta x_2}\int_{x_1}^{x_1+\Delta x_1} \phi_C(x_1', x_2', x_3')\mathrm{d}x_1'\mathrm{d}x_2'\mathrm{d}x_3' = \phi_C(\xi_1,\xi_2,\xi_3)\Delta x_1\Delta x_2\Delta x_3, \qquad (14.2)$$

where

$x_i \leq \xi_i \leq x_i + \Delta x_i$ ($i = 1$, 2, and 3)
ξ_i is the appropriate value in a given domain

Inside the cube, the mean open area of the cross sections perpendicular to the x_1 direction (A_{m1}) is the ratio of its internal void volume to its dimension in that direction:

$$A_{m1} = \frac{V_0}{\Delta x_1} = \phi_C(\xi_1,\xi_2,\xi_3)\Delta x_2\Delta x_3. \qquad (14.3)$$

The open area on the left side of the cube (A_L) is the limit of this equation, giving (Coulson et al., 1978):

$$A_{\mathrm{L}} = \lim_{\Delta x_1 \to 0} A_{m1} = \phi_{\mathrm{C}}(x_1,\xi_2,\xi_3)\Delta x_2 \Delta x_3. \tag{14.4}$$

The pressure force on air inside the cube from its left side in the x_1 direction (P_{L}) can then be expressed as (see Figure 14.1):

$$P_{\mathrm{L}} = p(x_1,x_2,x_3,t)A_{\mathrm{L}} = p(x_1,x_2,x_3,t)\phi_{\mathrm{C}}(x_1\xi_2,\xi_3)\Delta x_2 \Delta x_3, \tag{14.5}$$

where $p(x_1, x_2, x_3, t)$ is the spatially and temporally dependent pressure.

Similarly, the pressure force on air inside the cube from its right side in the x_1 direction (P_{R}) is the product of the open area and pressure on that side (see Figure 14.1):

$$P_{\mathrm{R}} = -\left[p(x_1,x_2,x_3,t) + \frac{\partial p(x_1,x_2,x_3,t)}{\partial x_1}\Delta x_1\right]\left[\phi_{\mathrm{C}}(x_1,\xi_2,\xi_3) + \frac{\partial \phi_{\mathrm{C}}(x_1,\xi_2,\xi_3)}{\partial x_1}\Delta x_1\right]\Delta x_2 \Delta x_3. \tag{14.6}$$

The stresses on an air cube in the atmosphere can be represented by nine stress components τ_{ij} ($i, j = 1$, 2, and 3), where subscript i indicates the axis normal to the face on which the stress acts and subscript j, the direction of the stress (Schlichting, 1979). In the case of air mixing with vegetative elements inside the cube in Figure 14.1, the nine stress components cannot describe the stresses produced at the solid–pore interfaces. Wilson (1985) lumped the form drag and skin friction at the solid–pore interfaces together as a momentum source in the equations of motion for his numerical simulation of boundary-layer flows near a 2D fence. Following this procedure, one can lump all stresses produced at the solid–pore interfaces inside the cube together and seek an alternative to represent the drag force due to these stresses on airflow. The lumped stresses representing the drag force on airflow inside the cube in the x_1 direction can be expressed by the commonly used formula (Thom, 1971; Wang and Takle, 1995):

$$\tau_{m1} = -C_{\mathrm{D}} \int_{x_3}^{x_3+\Delta x_3} \int_{x_2}^{x_2+\Delta x_2} \int_{x_1}^{x_1+\Delta x_1} \rho(x_1',x_2',x_3',t) S_{\mathrm{AD}}(x_1',x_2',x_3') \left|\vec{U}\right| u_1(x_1',x_2',x_3',t)\,\mathrm{d}x_1'\,\mathrm{d}x_2'\,\mathrm{d}x_3', \tag{14.7}$$

where

τ_{m1} is the lumped stress (the first subscript "m" denotes that orientations of the surfaces exerted by the stresses are mixed, and the second subscript "1" indicates that the stress is in the x_1 direction)

$\rho(x_1, x_2, x_3,t)$, the spatially and temporally dependent air density;

$u_1(x_1, x_2, x_3,t)$, the spatially and temporally dependent air velocity in the x_1 direction

The air mass in the cube (m_{a}) is given by:

$$m_{\mathrm{a}} = \int_{x_3}^{x_3+\Delta x_3} \int_{x_2}^{x_2+\Delta x_2} \int_{x_1}^{x_1+\Delta x_1} \rho(x_1',x_2',x_3',t)\phi_{\mathrm{C}}(x_1',x_2',x_3')\,\mathrm{d}x_1'\,\mathrm{d}x_2'\,\mathrm{d}x_3', \tag{14.8}$$

Applying Newton's Second Law, the force on air inside the cube in the x_1 direction (F_1) can be expressed as:

$$F_1 = m_{\mathrm{a}} \frac{\overline{D} u_1(x_1,x_2,x_3,t)}{\overline{D} t} = P_{\mathrm{L}} + P_{\mathrm{R}} + \tau_{m1}. \tag{14.9}$$

Substituting Equations 14.5 through 14.8 into this equation and dropping the second order terms of Δx_1 lead to:

$$
\begin{aligned}
&\frac{\overline{D}u_1(x_1,x_2,x_3,t)}{\overline{D}t}\int_{x_3}^{x_3+\Delta x_3}\int_{x_2}^{x_2+\Delta x_2}\int_{x_1}^{x_1+\Delta x_1}\rho(x'_1,x'_2,x'_3,t)\phi_C(x'_1,x'_2,x'_3)\mathrm{d}x'_1\mathrm{d}x'_2\mathrm{d}x' \\
&= -\left[p(x_1,x_2,x_3,t)+\frac{\partial\phi_C(x_1,\xi_2,\xi_3)}{\partial x_1}\ \phi_C(x_1,\xi_2,\xi_3)\ \frac{\partial p(x_1,x_2,x_3,t)}{\partial x_1}\right]\Delta x_1\Delta x_2\Delta x_3 \qquad (14.10)\\
&\quad - C_D\int_{x_3}^{x_3+\Delta x_3}\int_{x_2}^{x_2+\Delta x_2}\int_{x_1}^{x_1+\Delta x_1}\rho(x'_1,x'_2,x'_3,t)S_{AD}(x'_1,x'_2,x'_3)\left|\vec{U}\right|u_1(x'_1,x'_2,x'_3,t)\mathrm{d}x'_1\mathrm{d}x'_2\mathrm{d}x'_3.
\end{aligned}
$$

The first term on the right-hand side of this equation is the pressure gradient force on air inside the cube from both sides perpendicular in the x_1 direction (see Figure 14.1). However, only the pressure force on the open area of either side can act on air inside the cube. Therefore, this term is the force on air inside the cube due to the interaction of cubic porosity with pressure. The last term on the right-hand side of Equation 14.10 is the drag force due to the stresses, produced at the solid–pore interfaces, on air inside the cube. As x_1, x_2, and x_3 become sufficiently small, Equation 14.10 becomes the equation of motion in a porous canopy for the x_1 direction:

$$\frac{\overline{D}u_1}{\overline{D}t} = -\frac{1}{\rho}\left[\frac{p}{\phi_C}\frac{\partial\phi_C}{\partial x_1}+\frac{\partial p}{\partial x_1}\right] - C_D\frac{S_{AD}}{\phi_C}\left|\vec{U}\right|u_1. \qquad (14.11)$$

Similarly, the equation for the x_2 direction can be derived. Taking into account the gravitational force, the equation for the x_3 direction can also be derived. In tensor notation, the three equations of motion in a porous canopy can be written as:

$$\frac{\overline{D}u_i}{\overline{D}t} = -\frac{1}{\rho}\left[\frac{p}{\phi_C}\frac{\partial\phi_C}{\partial x_i}+\frac{\partial p}{\partial x_i}\right] - C_D\frac{S_{AD}}{\phi_C}\left|\vec{U}\right|u_i - \delta_{3i}g, \qquad (14.12)$$

where

g, is the acceleration due to gravity

δ_{3i}, the Kronecker delta coefficient (Frederic and Chang, 1965)

14.3 CONTINUITY EQUATION FOR AIRFLOW THROUGH A POROUS SHELTERBELT CANOPY

The divergence and convergence of airflow through a cube filled with porous media consisting of trunk, branches, leaves, and seeds are illustrated in Figure 14.2. If air enters the cube with the same velocity in all three directions, more air flows into the cube from the side where cubic porosity is high than from the side where this porosity is low. If air leaves the cube with the same velocity in all three directions, more air flows out of the cube from the side where cubic porosity is high than from the side where this porosity is low. This divergence and convergence of airflow in a shelterbelt canopy as influenced by its volume heterogeneity can be mathematically described.

In Figure 14.2, the mass (m_L) of air entering the cube through the left slice over a time interval (Δt) is:

$$m_L = \rho(x_1,x_2,x_3,t)\phi_C(x_1,x_2,x_3)u_1(x_1,x_2,x_3,t)\Delta t\Delta x_2\Delta x_3. \qquad (14.13)$$

At the same time, the mass (m_R) of air flowing out of the cube through the right slice is:

$$m_R = \rho(x_1+\Delta x_1,x_2,x_3,t)\phi_C(x_1+\Delta x_1,x_2,x_3,t)u_1(x_1+\Delta x_1,x_2,x_3,t)\Delta t\Delta x_2\Delta x_3. \qquad (14.14)$$

Thus, the difference in the mass of air into and out of the cube over a time interval (Δt) in the x_1 direction (ΔM_1) is:

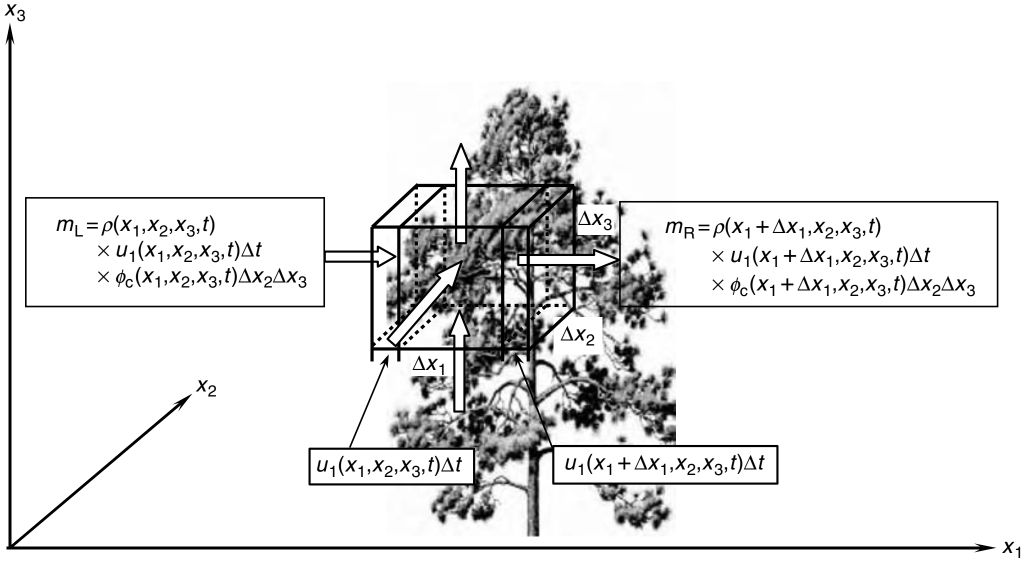

FIGURE 14.2 Air mass flow in the Lagrangian control cube where m_L is the mass of air into the cube through the left slice over a time interval (Δt); m_R, the mass of air out of the cube through the right slice over Δt; $u_1(x_1, x_2, x_3, t)$, air velocity in the x_1 direction; $\rho(x_1, x_2, x_3, t)$, air density; $\phi_C(x_1, x_2, x_3)$, cubic porosity; and $\Rightarrow$, the direction of air motion.

$$\begin{aligned}\Delta M_1 = m_L - m_R = \big[&\rho(x_1,x_2,x_3,t)\phi_C(x_1,x_2,x_3)u_1(x_1,x_2,x_3,t) \\ &- \rho(x_1+\Delta x_1,x_2,x_3,t)\phi_C(x_1+\Delta x_1,x_2,x_3,t)u_1(x_1+\Delta x_1,x_2,x_3,t)\big]\Delta t\Delta x_2\Delta x_3.\end{aligned} \tag{14.15}$$

Similarly, the difference in the mass of air into and out of the cube over the same time interval (Δt) either in the x_2 direction (ΔM_2) or in the x_3 direction (ΔM_3) can be formulated. According to the principle of conservation of mass, the local mass change rate per unit pore volume is equal to the net convergence of mass per unit volume. Thus:

$$\begin{aligned}\frac{\partial\rho(x_1,x_2,x_3,t)}{\partial t} &= \lim_{\Delta x_1,\Delta x_2,\Delta x_3,\Delta t\to 0}\frac{\Delta M_1+\Delta M_2+\Delta M_3}{\Delta t\int_{x_3}^{x_3+\Delta x_3}\int_{x_2}^{x_2+\Delta x_2}\int_{x_1}^{x_1+\Delta x_1}\phi_C\left(x_1',x_2',x_3'\right)\mathrm{d}x_1'\mathrm{d}x_2'\mathrm{d}x_3'} \\ &= \frac{1}{\phi_C}\left[\frac{\partial(\rho\phi_C u_1)}{\partial x_1}+\frac{\partial(\rho\phi_C u_2)}{\partial x_2}+\frac{\partial(\rho\phi_C u_3)}{\partial x_3}\right].\end{aligned} \tag{14.16}$$

The boundary-layer flows moving at a speed less than 0.1 Mach number can be considered as incompressible (Schlichting, 1979). Accordingly, the air density is a constant with respect to time:

$$\frac{\partial\rho(x_1,x_2,x_3,t)}{\partial t} = 0. \tag{14.17}$$

In tensor notation, Equation 14.16 can be rewritten as:

$$\phi_C\frac{\partial u_j}{\partial x_j} + u_j\frac{\partial\phi_C}{\partial x_j} = 0. \tag{14.18}$$

This is the continuity equation for airflow through a porous shelterbelt canopy.

14.4 DISCUSSION

Using the Reynolds average and the Boussinesq approximation (Dutton and Fichtl, 1969), Equations 14.12 and 14.18 for nonhydrostatic flow can be expressed in mean flow and turbulence terms:

$$\frac{\partial \bar{u}_i}{\partial t}+\bar{u}_j\frac{\partial \bar{u}_i}{\partial x_j}+\frac{\partial \overline{u_i'u_j'}}{\partial x_j}=-\frac{1}{\rho_0}\left[\frac{\bar{p}_i}{\phi_\mathrm{C}}\frac{\partial \phi_\mathrm{C}}{\partial x_i}+\frac{\partial \bar{p}_i}{\partial x_i}\right]-C_\mathrm{D}\frac{S_\mathrm{AD}}{\phi_\mathrm{C}}\left|\vec{\bar{U}}\right|\bar{u}_i \tag{14.19A}$$

and

$$\phi_\mathrm{C}\frac{\partial \bar{u}_j}{\partial x_j}+\bar{u}_j\frac{\partial \phi_\mathrm{C}}{\partial x_j}=0. \tag{14.19B}$$

These equations clearly address the relationship of the overall 3D shelterbelt structure, as described by the spatial functions of vegetative surface area density (S_AD) and cubic porosity (ϕ_C), to its aerodynamic influences, as described by mean air velocity ($\bar{u}_i$), mean air pressure ($\bar{p}_i$), and turbulent stress $\left(\overline{u_i'u_j'}\right)$. Using some assumptions for structural description and air motion, these equations are derived based on Newton's Second Law and the principle of conservation of mass. The relationship of the 3D structure to shelterbelt aerodynamics, as addressed by the equations, is a theoretical hypothesis.

14.4.1 VALIDATION OF ASSUMPTIONS FOR STRUCTURAL DESCRIPTION AND AIR MOTION

Considering the total volume of a porous shelterbelt canopy as a collection of numerous grid cells, Zhou et al. (2005) used vegetative surface area density and cubic porosity in each grid cell to describe the 3D aerodynamic structure of a shelterbelt with the assumption that vegetative elements are randomly and motionlessly distributed within a grid cell. The random assumption suggests that either vegetative surface area density or cubic porosity in one grid cell can be represented by one value. This assumption has been used previously to develop the structural descriptions of a whole canopy (de Wit, 1965; Campbell and Norman, 1989). The motionless assumption means that vegetative surface area and volume as well as element compositions in a grid cell do not change with respect to time, allowing either of two structural descriptors to be expressed as a spatial function independent of time. This assumption is consistent with the assumption of any canopy flow model that ignores momentum transfer to plant motion (Wilson and Shaw, 1977; Raupach and Shaw, 1982).

The stresses at the solid–pore interfaces on airflow are formulated (Equation 14.7) with the assumption that the stress under a given air velocity and air density is proportional to vegetative surface area. This assumption is universal in the study of canopy aerodynamics (Thom, 1971; Mayhead, 1973; Holland et al., 1991a, 1991b; Lee, 2000). Moreover, the continuity equation in a porous canopy (Equation 14.18) is developed with the assumption that the airflow in a shelterbelt canopy is incompressible (Equation 14.17). This assumption is common for boundary-layer flows (Schlichting, 1979).

14.4.2 AVAILABILITY OF METHODS FOR SOLUTION TO THE DEVELOPED EQUATIONS

A numerical method for solution to the equations, such as Equation 14.1A and Equation 14.1B, was developed for simulation of boundary-layer flows near a shelterbelt (Hagen et al., 1981; Wilson, 1985; Wang and Takle, 1997b; Patton et al., 1998). The developed equations do not bring any new unknown variables to the problem of predicting the flow fields near a tree shelterbelt with known structure. Equations 14.19A and 14.19B are identical to Equations 14.1A and 14.1B if one ignores the vegetative volume in a shelterbelt canopy (i.e., $\phi_C = 1$), or if the distribution of vegetative volume is uniform (i.e., $\partial\phi_C/\partial x_i = 0$). Recently, a method to estimate the 3D aerodynamic structure of a tree shelterbelt, as described by the spatial functions of vegetative surface area density and cubic porosity, was documented (Zhou et al., 2002). Therefore, the methods for solution to the developed equations are readily available.

14.4.3 ABILITY OF THE SPATIAL FUNCTIONS OF VEGETATIVE SURFACE AREA DENSITY AND CUBIC POROSITY FOR THE STRUCTURAL DESCRIPTION OF A SHELTERBELT

The description of tree shelterbelt aerodynamic structure is thoroughly discussed by Zhou et al. (2005). They used the spatial functions of vegetative surface area density [$S_{AD}(x_1, x_2, x_3)$] and cubic porosity [$\phi_C(x_1, x_2, x_3)$] to describe the amounts and arrangements of vegetative surface area and volume within a shelterbelt canopy. Both are capable of reflecting the geometric shape and size of individual elements because plant elements with different shapes and sizes have different surface to volume ratios. Furthermore, because $S_{AD}(x_1, x_2, x_3) = 0$ and $\phi_C(x_1, x_2, x_3) = 1$ beyond the width, height, and the external envelope of a tree shelterbelt, the functions constrain the drag force terms within the canopy envelope. Likewise, the influence of the volume gradients, as described by the term $\partial\phi_C/\partial x_i$, on the divergence and convergence of airflow is constrained within the shelterbelt canopy and disappears outside its envelope, allowing the developed equations to take the aerodynamic influence of the external characteristics of a shelterbelt into account. Therefore, the spatial functions of vegetative surface area density and cubic porosity have the ability to describe the overall 3D aerodynamic structure of a tree shelterbelt, including its internal structure and external characteristics.

14.4.4 APPLICATION OF THE DEVELOPED EQUATIONS

Equations 14.19A and 14.19B provide the theoretical means to better understand air motion, as influenced by the overall 3D aerodynamic structure of a shelterbelt, and support our basic understanding of the aerodynamics of the internal structural components of a shelterbelt. For a given vegetative volume within a shelterbelt canopy, greater vegetative surface area creates a stronger canopy drag force on the airflow (Grant and Nickling, 1998; Gillies et al., 2000). On the other hand, for a given surface area (S_{AD}) and volume arrangement ($\partial\phi_C/\partial x_i$) within a shelterbelt canopy, more vegetative volume creates greater canopy drag force on the airflow. In other words, for a given vegetative volume within a canopy, vegetative elements with smaller average size produce stronger canopy drag force. The volume heterogeneity, as measured by the gradients of cubic porosity ($\partial\phi_C/\partial x_i$) not only determines the canopy drag force (see the first term on the right-hand side of Equation 14.19A), but also orients the divergence and convergence of airflow within a shelterbelt canopy (see Equation 14.19B).

These equations can be used to test the aerodynamics of individual structural components of a shelterbelt. Given the amount and arrangement of vegetative volume in a shelterbelt canopy and external characteristics of the shelterbelt, the aerodynamics of the vegetative surface area can be tested at varying amounts and arrangements. Vice versa, the aerodynamics of vegetative volume can also be tested over a range of amounts and arrangements. How the volume in a porous medium influences air velocity, air pressure, and turbulent stress is theoretically addressed by Equations 14.19A and 14.19B. Zhou et al. (2002) recently quantified the cubic density of a green ash shelterbelt as between 0.00004 and 0.02330 $m^3\ m^{-3}$ depending on position within the shelterbelt. This small magnitude of cubic density raises the question of whether or not the vegetative volume in a shelterbelt canopy is great enough to have a significant influence on wind flow. Using Equations 14.19A and 14.19B, this influence can be numerically tested, which is currently underway as part of our ongoing research to determine how shelterbelts influence wind flow. Regardless of the outcome, a better understanding of the relationship between shelterbelt structure and wind flow will result.

14.4.5 POTENTIAL FOR IMPROVING THE PREDICTION OF BOUNDARY-LAYER FLOWS NEAR A TREE SHELTERBELT

Wilson and Mooney (1997) have raised the issue that current turbulence models underestimated the sharp speedup in airflow over the barrier, consequently showing a slower recovery rate of leeward windspeed from its minimum toward equilibrium flow. Current models do not include any term to address the aerodynamic influence of a structural component of volume. As a result, numerical simulations of the wind fields around a shelterbelt using these models allow more air to pass through the barrier. Due to the law of air continuity, this increased flow must be matched by a reduction in air converging at the top, leading edge of the barrier (Cleugh, 1998) and by a weaker Coanda effect in the lee (Plate, 1971). As a result of these flow changes, simulations using the current models result in an overestimation of windspeed in the lee and an underestimation of the rate of recovery to equilibrium flow. Equations 14.19A and 14.19B, by including the aerodynamic influence of the structural component of volume, have the potential to address this issue for improving the prediction of boundary-layer flows near a tree shelterbelt.

14.5 CONCLUSION

A set of equations describing the aerodynamic influence of overall shelterbelt structure described by the spatial functions of vegetative surface area density and cubic porosity was derived using the classic laws and principles of physics. The validation of the assumptions used in the derivation and the availability of methods to solve the equations, combined with the ability of the spatial functions to enhance the structural description, make the equations have significant applicability to research on shelterbelt aerodynamics and potential to improve the prediction of boundary-layer flows near a shelterbelt. The research will advance our theoretical understanding of the relationship between 3D structure and shelterbelt function, and will help guide the development of optimal designs for individual landowner's objectives.

14.6 APPENDIX

Prime ($'$), Departure of a variable from its mean.
Overbar ($\bar{\ }$), Averaged value.
A_L, Open area on the left side of the cube in Figure 14.1 for airflow to pass.
A_{m1}, Inside the cube in Figure 14.1, mean open area of cross sections perpendicular to the x_1 direction.
C_D, Drag coefficient per unit vegetative surface area.
F_1, Force in the x_1 direction.
F_{G3}, Gravitational force (see Figure 14.1).
g, Acceleration due to gravity at the earth surface.
i and j, As subscripts of x and u, i and j denote the width, length, and height dimensions for 1, 2, and 3, respectively.
k_r, Resistant coefficient.
m_a, Mass of air inside the cube in Figure 14.1.
m_L and m_R, In Figure 14.2, the mass of air into the cube through its left slice and out of the cube through its right slice over a time interval (t), respectively.
p_1, p, and $p(x_1, x_2, x_3, t)$, Instantaneous pressure.
P_L and P_R, Pressure forces on air inside the cube in Figure 14.1 from its left and right sides, respectively.
S_{AD} and $S_{AD}(x_1, x_2, x_3)$, Vegetative surface area density.
t, Time.
u_i or u_j, Air velocity in the ith or jth direction.

$|\vec{U}|$, Total air velocity.
V_0, Void volume inside the cube in Figure 14.1.
x_i or x_j, Coordinate in the ith or jth direction.
δ_{3i} or δ_{33}, Kronecker delta coefficient.
Δ, Small increment.
ΔM_1, ΔM_2, and ΔM_3, Differences in the mass of air into and out of the cube in the x_1, x_2, and x_3 directions, respectively.
ϕ, Optical porosity.
ϕ_C and $\phi_C(x_1, x_2, x_3)$, Cubic porosity.
ρ and $\rho(x_1, x_2, x_3, t)$, Air density.
ρ_0, Reference air density.
τ_{ij}, Stress in the jth direction on the face perpendicular to the ith direction.
τ_{m1}, Stress produced at the solid–pore interfaces in the x_1 direction.
ξ_1, ξ_2, and ξ_3, Appropriate values in a given domain.

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15 Modeling a Field Shelterbelt System with the Shelterbelt Agroforestry Modeling System

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CONTENTS

15.1 INTRODUCTION

Field shelterbelts (ones used primarily to protect crop fields) have been an important component of agroecosystems in certain parts of the world for centuries. The primary microclimatic influence of shelterbelts is windspeed reduction (Caborn, 1957; Grace, 1977; McNaughton, 1988), which can improve crop production (Stoeckler, 1962; Kort, 1988), reduce wind erosion (Tibke, 1988), reduce the movement of fugitive pesticides and fertilizer (Tibke, 1988), reduce odor emissions from animal enclosures (Tyndall and Colletti, 2000), and increase economic returns (Brandle et al., 1992a). Other documented shelterbelt effects include increased wildlife habitat (Johnson and Beck, 1988; Johnson et al., 1994), sequestration of carbon (Brandle et al., 1992b), increased abundance of natural enemies of insect pests (Dix et al., 1995), and improved aesthetics (Sutton, 1992).

Although shelterbelts produce multiple effects and some of those effects have been intensively studied, estimation of the benefits and costs of a specific shelterbelt on an individual farm is a

TABLE 15.1
Shelterbelt Effects, the Impact of Shelterbelts on the Effects, and How to Simulate or Quantify the Benefits and Costs of the Effects

Effect	Impact of Shelter	How to Simulate/Quantify
Crop yield	Yield increase or decrease, depending upon growing season conditions	Use model described in this chapter
Carbon sequestration	Trees and soil sequester C; fuel usage is reduced	Directly measure or use yield equations; straightforward calculations
Wind erosion	Reduces erosion where it is a problem	WBECON methodology (Brandle and Kort, 1991); WEPS model (Wagner, 1996) when it includes multiple subregions
Fugitive pesticides	Reduces spread of chemicals but can also kill trees	Cannot be done presently
Wildlife	Increased numbers of animals which can be a benefit or a cost	Only willingness to pay estimates can be made by landowners
Natural enemies of insect pests	Little impact unless there is a dense network of belts	Cannot be done presently
Odor emissions	Some reduction possible	Cannot be done presently
Aesthetics	Potentially valuable contribution	Only willingness to pay estimates can be made by landowners

complex task that is best done by a computer model. In 1991, our research group began developing a system to estimate site-specific effects, benefits, and costs for sheltered fields that produce maize (*Zea mays* L.) and soybeans (*Glycine max* L.) that can be used to help develop tools that individuals can use for decision making. The system is called SAMS, which stands for Shelterbelt Agroforestry Modeling System (Mize, 2000), and it is being developed as a useful tool for landowners who are interested in establishing shelterbelts on their land. As far as we know, SAMS is the only model that has been developed specifically to predict crop yield and other effects in a shelterbelt system (a shelterbelt and adjacent cropland). Although SAMS was initially developed to predict crop yield in a sheltered field, we are incorporating benefits and costs associated with carbon sequestration and increased wildlife usage into SAMS. Eventually, SAMS will include the benefits and costs of other effects discussed in this chapter (see Table 15.1). SAMS is being developed as a web site and currently has the URL of http://oriole.ae.iastate.edu/sams.

Other shelterbelt modeling research includes Easterling et al. (1997) who used EPIC (Erosion-Productivity Impact Calculator) to evaluate the potential of shelterbelts to ameliorate climate change-induced crop stress. Also, WBECON (Brandle and Kort, 1991; Kort and Brandle, 1991) is a model that evaluates economic aspects of shelterbelts; however, it uses published crop yield curves, representing an average response over a variety of fields, to estimate yield response for individual farms, whereas SAMS estimates crop yield curves for individual fields.

This chapter describes methods of estimating the benefits and costs associated with some shelterbelt effects and explains how these effects are or will be incorporated into SAMS. The chapter does not present an in-depth discussion of the effects. Two recent comprehensive reviews of shelterbelt technology provide good background material on shelterbelt effects (Nuberg, 1998; Brandle et al., 2000).

For many farmers the most important shelterbelt effect, often the only effect they expect, is increased crop yield. As a result, much of the research on shelterbelts has focused on yield and associated microclimatic variation across sheltered fields. Our discussion on modeling a shelterbelt system will begin with modeling crop yield.

15.2 MODELING CROP YIELD ACROSS A SHELTERED FIELD

Although shelterbelt systems have been studied for decades, as far as we know, there has been no attempt until recently to develop a model capable of simulating crop yield across individual sheltered fields. Our research group began developing such a model in 1991 (Mize and Qi, 1994). As a modeling framework, we divided a shelterbelt system into three components: the crop being grown in the sheltered field; the microclimate, which influences crop yield; and the shelterbelt, which influences the microclimate across the field (Figure 15.1). Using these three components, a model to simulate crop yield across a sheltered field could be developed by combining: (1) a shelterbelt model that would simulate characteristics of a shelterbelt, (2) a microclimatic model that would use the simulated shelterbelt characteristics to simulate microclimate at specific locations across a sheltered field, and (3) a crop model that would use the simulated microclimate to simulate yield at those locations. Predicted yields would be combined to estimate a field level yield.

Next, we briefly describe crop, microclimatic, and shelterbelt modeling and how such models are being used to simulate crop production in SAMS.

15.2.1 Crop Models

Models capable of simulating crop yield for most major agronomic crops have existed for at least 25 years. Early models simulated yield using empirical approaches, in which yield was computed as a function of major inputs, such as planting date, cumulative rainfall during the growing season, and yield potential. In the mid 1970s, researchers began to develop more process-based crop growth

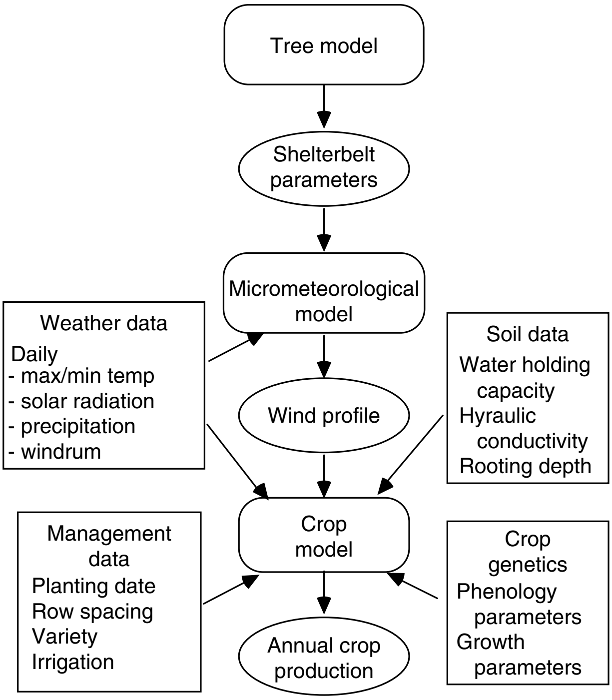

FIGURE 15.1 Conceptual model of components used to predict crop production in a sheltered field.

models. These models extended the mathematical representation of plant growth by simulating daily rate of plant growth and integrating this rate over the season to ultimately compute final yield. Processes, such as photosynthesis, respiration, carbon partitioning, and soil water and nutrient stresses, are computed by solving mass balance differential equations and are calibrated to field data. Examples of these models include CERES (Jones and Kiniry, 1986), CROPGRO (Hoogenboom et al., 1994), and SUCROSE (Kuelen et al., 1982). Our research group works primarily with the DSSAT (Decision Support Systems for Agrotechnology Transfer) models (Hoogenboom et al., 1994). Inputs to these models include daily weather (maximum or minimum temperature, rainfall, solar radiation, windrun—optional), soil properties (water holding capacity, rooting depth), crop genetics (development and reproductive rates, stress response, photosynthesis parameters), and management practices (planting date, row spacing, variety traits) (Figure 15.1). These models have been used to simulate influence of global climate change and crop management on maize and soybean production and are being used widely in Iowa to simulate optimum management practices in row crop production (Paz et al., 1998; Sexton et al., 1998; Garrison et al., 1999).

Simulation of maize and soybean yield under sheltered conditions requires crop growth models that are sensitive to varying microclimates across a sheltered field. The DSSAT models, including the CROPGRO soybean and CERES-Maize models (Hoogenboom et al., 1994), respond to daily maximum and minimum temperatures and windrun, all of which are influenced by a shelterbelt, through the use of the Penman Monteith equation for simulation of evapotranspiration. These models recently have been used to simulate the impact of shelterbelts on maize and soybean yield across a field. Qi et al. (2001) used the soybean model to simulate the potential yield response in a sheltered environment using long-term historical weather data. They used a theoretical wind response function to compute the windspeed reduction at different distances from the shelterbelt using actual unsheltered windspeed data for 14 years. Mize et al. (2005) used the CERES-Maize and SOYGRO models to simulate effects of microclimatic changes in windspeed and temperature due to shelterbelts on maize and soybean yield. They used empirical functions derived from measurements of windspeed and temperature to compute daily windrun and daily maximum and minimum temperatures at a location near a shelterbelt from windrun and daily maximum and minimum temperatures at an open site. These served as inputs to the maize and soybean models for a 2 year period for a field in Indiana, and both models showed sensitivity to microclimatic differences.

15.2.2 MICROCLIMATIC MODELS

Numerical modeling of turbulent flow and microclimate near shelterbelts is a recent development. In 1995, Wang and Takle introduced a new approach for simulating flow fields in the vicinity of shelters (Wang and Takle, 1995) and subsequently applied the model to shelters of variable porosity, irregular shapes, and oblique orientations to the wind (Wang et al., 2001). They also added soil layers and extended the model to include moisture and temperature. This allowed for simulation of differential patterns of heat and moisture flux across sheltered areas and provided a more complete set of microclimate factors for evaluating a wider range of sheltering influences. The model requires information about shelterbelt characteristics, including shelter height, width, cross-sectional shape, and specific surface area (area of leaves and branches per unit volume) of each cell in a grid representing a cross-section of the barrier.

The Wang and Takle (1995) model (WT model) captures many of the general features of windspeed and temperature in a shelterbelt system, but it has not been tested with field data, because appropriate field data, until recently, have not been available. While waiting to evaluate the WT model, we decided to pursue an empirical approach to predicting microclimate across a sheltered field. Intensive measurements of microclimate (windspeed and temperature) across various sheltered fields have been made, and the data are being used to develop regression equations to predict microclimate at specific distances from a shelterbelt using open-field (part of the field not under the influence of a shelterbelt) measurements as predictors (Mize et al., 2005). The statistical models are complementary to the first-principles of the dynamical WT model and allow prediction of microclimate across a sheltered field that can be used for testing the WT model, when information about individual shelterbelts becomes available. If the WT model adequately predicts microclimate across a sheltered field, it will be used in SAMS. Otherwise, we will continue making intensive measurements and developing regression equations to predict microclimate for a variety of shelterbelts.

15.2.3 SHELTERBELT MODELS

Because of the importance of tree and forest growth in forest management, models to predict their growth have been developed for more than 200 years (Fernow, 1907). Few forest growth models would be useful for predicting characteristics of shelterbelts, however, because they are based on data from expansive forests. Such models do not apply to narrow plantings, typical of shelterbelts in which most, if not all, trees are on the edge of the "forest." Individual tree growth models have been developed for many species and are capable of predicting some shelterbelt characteristics, tree height in particular.

Presently, we are categorizing shelterbelts into what we call "types of shelterbelts." All shelterbelts that have the same number of rows, the same species composition and spatial arrangement, and similar spacing are considered the same "type." Models to predict shelterbelt characteristics are being developed for various types of shelterbelts.

One of the most important shelterbelt characteristics is the height of the tallest row in the shelterbelt (H). Many tree height growth models have been developed, but almost all require a measure of the quality of the environment, often site index (Avery and Burkhardt, 2002), which is seldom available for agricultural soils. We will develop height growth equations for groups of similar soil types by collecting height growth data on previously established shelterbelts on the various soil groups. Additionally, SAMS will allow users to indicate the expected height of their shelterbelt at age 50 to allow a better estimation of shelterbelt height for the user's field.

If regression equations are used to predict microclimate across a field, only H needs to be estimated for each type of shelterbelt. If, however, we use the WT model, estimates of shelter width, cross-sectional shape, and specific surface area will be needed for each type of shelterbelt. Shelter width can be estimated by developing equations to predict crown radius by using data that can be collected with height growth data. Cross-sectional shape will be estimated by using data collected for each type of shelterbelt at different ages. The most difficult characteristic to estimate will be the specific surface area, and we are trying two methods to estimate that. The first method uses a series of equations developed from data collected during intensive sampling of individual shelterbelts (Zhou et al., 2002). The second involves actually estimating specific surface area on a sample of cells within individual shelterbelts. If the WT model is shown to be effective, the method developed by De Reffye et al. (1995) will be evaluated as a technique for estimating surface area.

15.2.4 SIMULATING CROP YIELD IN SAMS

Because of the importance of maize and soybeans in the Midwestern United States, CERES MAIZE and SOYGRO are the crop models presently used in SAMS. Other crop models can be easily incorporated. A conceptual model for predicting crop yield (Figure 15.1) shows the crop models to be at the heart of SAMS. Both crop models require four input files: crop management, crop genetics, soils, and weather. Presently, crop management and crop genetics are assumed to be uniform across a field, so only one set of crop management practices and crop genetics information is used to simulate production during an individual growing season. Presently, SAMS allows a user to select one of various generic soil groups to represent the soil in a specific field. The relationship between yield and distance from a shelterbelt has been studied thoroughly (Kort, 1988) and usually varies from a curved line, such as that shown in Figure 15.2, indicating a yield increase due to shelter, to a horizontal line, indicating no shelterbelt influence. To simulate response across a field in SAMS,

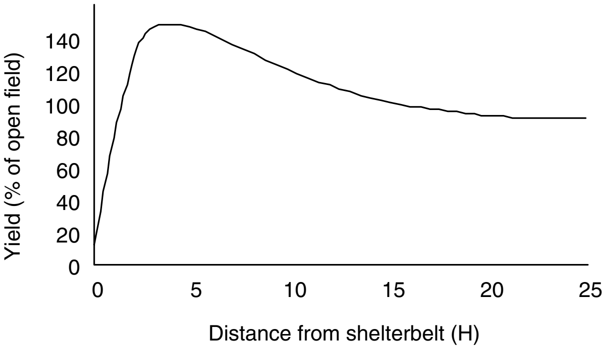

FIGURE 15.2 Example of yield variation across a shelterbelt.

yield is estimated at eight distances (1H–25H with 25H assumed to represent an unsheltered location) from the shelterbelt, which requires that a daily weather file be created for each distance to reflect differences in microclimatic differences at each distance.

Yield per hectare on a field-level basis is calculated by multiplying the estimated yield per hectare at each distance by the proportion of the field represented by that distance and summing the values. An estimate of the yield per hectare in the field without a shelterbelt is calculated using the yield estimate for 25H.

Estimated yield under shelter varies from year to year, being strongly influenced by weather during the growing season. To estimate an average yield response, SAMS is run with weather data from multiple years. In the United States, there is a national network of recording weather stations that has collected weather data for many years, which allows simulation of yield with weather data collected relatively close to any area within the north central portion of the country where maize and soybeans are generally grown. Many parts of the world lack such long-term data, which would prevent simulation of yield for multiple years.

Presently, SAMS assumes that each shelterbelt is straight, which is usually true, and infinitely long, which is obviously not true. At this time, we are not able to model microclimate for shelterbelts that are not straight or near the ends of shelterbelts. Microclimate near the ends of shelterbelts is not the same as across the field near the center of a shelterbelt.

Although we have limited data for testing the ability of the crop models to predict yield across sheltered fields, to date both CERES MAIZE and SOYGRO have clearly shown yield differences associated with microclimatic changes caused by shelter (Qi et al., 2001; Mize et al., 2005).

15.3 SIMULATING AND QUANTIFYING OTHER EFFECTS

15.3.1 Carbon Sequestration

Shelterbelts sequester carbon in the crowns, stems, and roots of the plants in the shelterbelt, and soil under shelterbelts is often higher in carbon than the soil surrounding it. Brandle et al. (1992b) provided some of the first estimates of the storage potential of shelterbelts. They estimated the average carbon storage of a 20 year old, single row conifer windbreak (9.2 mg km^{-1}) and a 20 year old, single row hardwood windbreak (5.4 mg km^{-1}). Kort and Turnock (1999) estimated that the carbon reserve in Canadian prairie shelterbelts varied from 24 to 104 mg km^{-1}, depending on species and age.

Individual shelterbelts are only useful for about 50 years, however, and at the end the trees are often burned, releasing the carbon to the atmosphere for no net sequestration, except for the root system, which will slowly decompose. However, a network of variously aged shelterbelts will sequester a substantial amount of carbon, above- and belowground, while the network exists.

Aboveground sequestered carbon in a shelterbelt can be estimated relatively easily by using well-established techniques used to estimate forest biomass (Avery and Burkhardt, 2002). Estimates for a specific shelterbelt could be developed by first measuring the heights and diameters of a sample of trees, which would be combined with equations to estimate tree biomass to estimate biomass per hectare. Adjustments would be needed to reflect the difference in biomass of forest grown trees compared to shelterbelt grown trees, which tend to have considerably more branches. As carbon sequestration estimates are developed for different types and ages of shelterbelts, equations similar to what foresters describe as yield equations (Avery and Burkhardt, 2002) could be developed that would estimate sequestered carbon for shelterbelts at different ages. This is being done for the prairie states in the United States (J. Brandle, 2002, Univ. of Nebraska, personal communication).

Belowground biomass in a shelterbelt is considerably more difficult to estimate than aboveground biomass because it is much more difficult to measure and its stability varies greatly, depending upon the size of the material. Presently, SAMS estimates aboveground tree biomass but not belowground biomass. Belowground biomass will be added when reliable estimates can be made relatively easily.

The major advantage of using shelterbelts relative to the carbon budget is the indirect benefits that flow from planting fewer acres of crops. Windbreaks are placed on land that is cropped annually, and by removing such land from row crop production, annual fuel usage is reduced, resulting in a reduction of carbon dioxide emissions. Over all, the potential of these indirect benefits is about double the amount of direct carbon storage in the wood of the shelterbelt (Brandle et al., 1992b), and these benefits remain even after a shelterbelt is removed.

Although the amount of carbon sequestered by a shelterbelt or not released because the land occupied by a shelterbelt was not farmed can be estimated with reasonable accuracy, the market for that carbon has not been well established in many parts of the world (Fischer et al., 1998). Until a carbon market becomes well established for shelterbelts, carbon sequestered in shelterbelts does not have a monetary value, although its potential value can be estimated by using current prices being paid in areas where there is a market. This is the approach used in SAMS.

15.3.2 WIND EROSION

Reduced wind erosion is a well-documented effect of shelterbelts (Brandle and Kort, 1991). In some areas, reduced erosion due to shelter is more important than increased yield. There are at least two methods to quantify the impact of wind erosion in a shelterbelt system.

WBECON (Brandle and Kort, 1991; Kort and Brandle, 1991) is being modified to value a shelterbelt's influence in reducing wind erosion. In the program, a landowner or a professional will be asked to estimate the annual loss in productivity due to erosion, usually somewhat less than 0.1% but sometimes higher. Regardless of the rate, it is applied to the unprotected part of the field each year. During the first 7 years after establishing a shelterbelt, the entire field is assumed to be unprotected. Starting in the eighth year, some of the field is assumed to be protected from wind erosion. It begins to receive a yield benefit and is no longer subject to a yield reduction due to erosion. The size of the protected area, a multiple of H, increases each year, and the size of the erosion-prone area decreases each year until the maximum protected area is reached. Then the unprotected area continues to lose productivity, whereas the protected area receives the yield benefit of the windbreak.

The Wind Erosion Prediction System (WEPS) is a model that simulates wind erosion (Hagen et al., 1995). It is a process-based, daily time-step computer model that uses weather files similar to those used by the crop models. When WEPS is capable of dividing a field into subregions, it will be a powerful tool for evaluating the influence of shelterbelts on wind erosion in individual fields. Even when that happens, the erosion predictions will need to be converted into estimates of the impact on crop yield, as is being planned for WBECON. Presently, SAMS does not account for the effect of shelterbelts on wind erosion, but it will be added in the future.

15.3.3 MOVEMENT OF FUGITIVE PESTICIDES AND FERTILIZER

The movement of pesticides and fertilizers is a significant problem in some agricultural areas (Tibke, 1988). It is, however, only a concern for relatively few days per year and a problem only when those days are windy, but shelterbelts clearly can reduce the drift problem. On the other hand, herbicide drift has killed shelterbelt trees in some areas.

The WT model that we are evaluating for use in predicting microclimate across sheltered fields can also be used to simulate the effect of a shelterbelt on chemical drift across a field. The water vapor equation in the microclimate version of the WT model provides an analog for a trace gas constituent equation. Challenges to this approach include specification of the source function (e.g., surface flux of pesticide or other vapor into the atmosphere) and rate of capture of the fugitive vapor in the shelterbelt. Until these challenges are met, the benefits and costs of shelterbelts in reducing movement of pesticides and fertilizers cannot be simulated and will not be accounted for in SAMS.

15.3.4 RESPONSE OF WILDLIFE

Trees and shrubs are seldom found in or near agricultural fields used to produce annual crops. Introducing either of them creates structure, which is particularly attractive to birds, and cover, which attracts other animals, such as small mammals and deer. Thus, shelterbelts generally attract birds and some other animals (Johnson and Beck, 1988). Shelterbelts can serve as corridors for wildlife, if they are not too isolated from woodlands, and if they contain conifers, they offer winter cover that reduces winter mortality.

Although we know that wildlife are attracted to shelterbelts, estimating the actual numbers and species of wildlife in a particular shelterbelt is difficult. The length, number of rows, species composition, and age of the shelterbelt are important factors in determining which wildlife species might be found in a shelterbelt. Wildlife usage also depends considerably upon the condition of areas around a shelterbelt (Johnson and Beck, 1988; Beecher et al., 2002).

As simulating the response of wildlife to a shelterbelt is difficult, so is estimating the value that a farmer would be willing to pay for increased wildlife in a shelterbelt system. Many farmers are interested in having more birds around, but deer and raccoons use shelterbelts and eat crops near shelterbelts. In the Midwestern United States, shelterbelts are often excellent places for pheasant hunting in the fall after crops have been harvested (Cable and Cook, 1990). Birds, however, can do substantial damage to crops such as sunflowers. Thus, wildlife attracted by shelterbelts can represent a benefit or a cost.

SAMS will incorporate the value of the response of wildlife to a shelterbelt by allowing a landowner to indicate an estimate of the annual value of the wildlife. On the basis of responses to willingness-to-pay (WTP) questions (Drake, 1992) that will be presented to landowners in the Midwestern United States, users will be presented with the maximum, minimum, and average values and allowed to use one of these values or input their own WTP value.

15.3.5 ABUNDANCE OF NATURAL ENEMIES OF INSECT PESTS

There are a variety of natural enemies, including insects, birds, rodents, and spiders, that control insect pests in agricultural settings (Dix et al., 1995). Shelterbelts offer natural enemies food and foraging sites, protection from the elements, travel corridors, reproductive habitat, and overwintering sites for some species.

In farms with a relatively dense network of shelterbelts, natural enemy populations might be high enough to have a significant impact on insect pests and result in savings due to reduced pesticide use (Dix et al., 1995). As the influence of natural enemies decreases with distance from the shelterbelt, shelterbelts would have to be relatively close together to have a significant impact on pesticide use. However, shelterbelt networks, when they exist, typically separate shelterbelts by at least 20H. This density of shelterbelts is probably too low to support adequate numbers of natural enemies to have a significant impact on insect pests.

Little research is being done on the influence of shelterbelts on natural enemies. When more is done and estimates of the value of natural enemies are developed, their effect and value will be incorporated into SAMS.

15.3.6 REDUCTION OF ODOR EMISSIONS FROM ANIMAL ENCLOSURES

Shelterbelts potentially are useful for reducing livestock odor, because of their ability to scavenge particulate matter. Most odorous compounds associated with animal enclosures are easily absorbed onto and carried by particulate matter (Hammond and Smith, 1981; Hammond et al., 1981). Particulate matter emanating from animal enclosures tends to form a plume that stays at or very near ground level (Smith, 1993). When odor plumes interact with shelterbelts, some particulate matter is adsorbed onto tree leaves, some settles out on the leeward side of shelterbelts and is easily incorporated into the soil, and some is mixed with air above the plume, that is, diluted. The result is a reduction in the amount and concentration of particulate matter in odor plumes. Conifers may be more effective particle traps than deciduous species (Smith, 1984) and can be more efficient at removing particulate matter, because leaves are on the trees all year.

Although shelterbelts can reduce odor from animal enclosures, estimating the amount of reduction and the value of that reduction cannot be done presently but is being evaluated (Joe Colletti, 2002, Iowa State University, personal communication). When benefits and costs associated with odor reduction can be estimated, they will be incorporated into SAMS.

15.3.7 AESTHETIC VALUE

Many agricultural regions, particularly in the Midwestern United States, have evolved into highly homogenous landscapes to facilitate mechanized production. This goes against a key factor of aesthetic values—landscape variety (Berry, 1977; Hodge, 1991). Landscape variety includes functional and visual diversity at a local level, such as riparian forests for water protection and wildlife habitat and shelterbelts for erosion control and landscape corridors. With special reference to shelterbelts, Ronneberg (1992) noted that studies have suggested "*Visual diversity... (is) preferred to open landscape*."

Most people would agree that shelterbelts increase the aesthetics of many agricultural settings; however, assigning a monetary value to the aesthetic value of shelterbelts is difficult. Considering the old saying, "*beauty is in the eye of the beholder*," some farmers would be willing to pay solely for the beauty it adds to the landscape, whereas other farmers would see no value in its aesthetic contribution to the landscape. Although the aesthetic value of a shelterbelt to an individual farmer is too difficult to predict, a range of values will be estimated using WTP as described in the section on valuing wildlife. SAMS will incorporate the value of the aesthetic impact of a shelterbelt in the same manner that the value of wildlife is handled.

15.3.8 OTHER EFFECTS

There are other shelterbelt effects, and some will be discussed briefly here. Although shelterbelts might increase beneficial insects, they could also harbor insects and diseases that damage the crops and weeds, which would shed seed onto the field and increase weed problems. Shelterbelts increase biodiversity, and sometimes reduce water erosion by functioning as a filter strip or a vegetative terrace. Coniferous shelterbelts could reduce heat loss from farm animals. These effects could be important in some situations but are probably not of much economic value on most farms.

Another, more interesting, effect is that shelterbelts can control snow deposition. The primary impact of that would be increasing water availability in certain parts of the field, which presently is not accounted for in SAMS. It is an important effect that we need to incorporate.

SAMS does allow users to enter estimates of the value of the effects just mentioned and others. Users are allowed to enter the value for “Other Values or Benefits” that can apply to anything to which the user wants to estimate a positive or negative value.

15.4 FINANCIAL AND ECONOMIC ANALYSES DONE BY SAMS

Financial and economic analyses associated with shelterbelt costs are relatively straightforward, although there are many scenarios that can be evaluated. The major problem is quantifying some benefits, such as aesthetics and wildlife.

At the SAMS web site, aside from information about basic crop management practices and characteristics of the shelterbelt to be simulated and the field in which it would be located, users are asked to enter a discount rate and costs for site preparation, planting, replanting, and maintenance. They also need to enter costs for capital, management, and up to four other costs. Additionally, they can enter their estimates of the values for aesthetics and wildlife and a lump sum value for all other effects for which they wish to estimate a value.

With these values entered, SAMS will estimate crop yield at multiple locations in the field for each of the years for which weather data are available. The yield estimates are used to estimate field-level yield with and without a shelterbelt. The field-level yield estimates are combined with the costs and benefits entered by the user to estimate the marginal net value of the shelterbelt and the break-even yield, the yield increase that the shelterbelt needs to cause to break even with the shelterbelt. Also, the average maize and soybean yield across the field when the shelterbelt is mature will be calculated and displayed.

SAMS is being developed for individuals to help decide upon establishing a shelterbelt and, as such, the value associated with a particular effect is meant to reflect the value to the individual, not society. Some of the effects of shelterbelts, such as improved aesthetics, control of odor, and increased wildlife, have value to the public as well as to an individual. Policy makers could use SAMS to estimate the cost of incentives needed to encourage farmers to establish shelterbelts that could produce public goods.

15.5 FUTURE OF MODELING FIELD SHELTERBELT SYSTEMS

As global warming gradually changes the climate around the world, current agricultural practices will become increasingly unsuccessful in some areas (McCarthy et al., 2001). Shelterbelts probably will be useful in compensating for some of the impact of climatic changes. To be efficiently used, however, much more needs to be known about the impact of shelterbelts and the impact of global warming on the growth of the shelterbelt.

As most agricultural fields contain multiple soil types, SAMS, which presently only accepts one soil type in a field, will need to be modified to accept multiple soil types. Within 2 years, we plan to modify SAMS so that a user can indicate up to four soil types within a field and the relative distribution within the field.

Our research group will continue developing SAMS with a primary focus on estimating crop yield and a secondary focus on estimating the benefits and costs of other effects, such as carbon sequestration. We are broadening the area over which we are collecting maize and soybean yield data and are developing plans to collect yield data for other annual crops, such as sugar beets (*Beta vulgaris* L.). Any crop that has a growth model that subscribes to the IBSNAT format and has windrun as an input variable could be incorporated into SAMS, although yield data from sheltered fields will be needed to evaluate the system’s ability to predict sheltered yield for each new crop.

Interestingly, once the crop model for a species is incorporated into SAMS, it could be used to simulate the effect of shelterbelts on crop yield wherever the crop is grown. But before SAMS could be used to simulate a crop’s response to shelter in a particular area, the characteristics of the shelterbelts in that area will need to be quantified, because species used in shelterbelts vary substantially around the world.

Although SAMS presently is being developed to simulate a single shelterbelt, it will eventually simulate the effects of a network of shelterbelts, which can be on a single farm or developed by cooperating neighbors or in countries like Russia and China where networks of shelterbelts exist. Research is being started to simulate crop yield near the ends of shelterbelts. Eventually, we anticipate developing a system that uses GIS to allow an individual with appropriate training to work with individual farmers to evaluate the impact of shelterbelts at the farm level.

In summary, we have made good progress in developing a system to simulate crop production across a sheltered field and are beginning to develop techniques to estimate a few other shelterbelt effects. We plan to incorporate other crops and shelterbelt effects into SAMS and believe that it will be a useful tool for landowners who are interested in establishing shelterbelts on their land. As global warming continues, SAMS will be useful to an increasing number of landowners looking for options to help them manage their farmland.

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Part IV

Ecological Economics

16 Social and Economic Implications of Agroforestry for Rural Economic Development in Temperate Regions

Chris J. Doyle and Tony Waterhouse

CONTENTS

16.1 INTRODUCTION

Throughout this chapter "agroforestry" is reserved to define those systems of land use, in which trees are planted with an understory of either crops (silvoarable systems) or pasture for grazing animals (silvopastoral systems). In contrast, "plantation forestry" is used to define conventional forestry systems, involving planting of closely spaced trees to produce unbroken stands of woodland. Both systems of land use are covered by the term "farm forestry." Although the planting of agricultural land with trees forms the background context for this chapter, the specific focus is on agroforestry.

As a system of land use, agroforestry has become an increasingly attractive land-use option in temperate regions as countryside policy objectives have been broadened in response to concerns for the environment, demographic movements, changes in rural economies, and shifts in land use. Recent moves to make greater use of agroforestry practices in temperate agriculture have been driven by a perception that it can help satisfy many different policy objectives, including improved farm financial viability, agricultural diversification, environmental impact mitigation, land and water rehabilitation, sustainable use of marginal land, and natural habitat regeneration (Williams et al., 1997). For policy makers, it is also seen as providing a mechanism for encouraging change in long-accepted agricultural practices over a time period, which is sufficiently long to be acceptable to farmers (Hislop and Sinclair, 2000). In particular, agroforestry is often regarded as being easier to "sell" to farmers than conventional forestry, which has had limited adoption (Doyle and Thomas, 2000). Thus, while there are some farmers who are enthusiastic about tree planting, in countries like the United Kingdom the majority feel that the benefits of farm forestry cater more for the forester than the farmer (Hislop and Claridge, 2000).

Whether agroforestry will play a significant role in the future of rural employment in temperate regions will depend on three factors:

1. The willingness of farmers to plant tree;
2. The willingness of the government to subsidize farm forestry; and
3. Public attitudes to forestry in general and agroforestry in particular.

The first of these will depend strongly on the perceived economic benefits of agroforestry. The second factor, namely the government attitude to agroforestry, is likely to depend on the perceived wider local and regional benefits arising from increased farm forestry in general and agroforestry in particular. Within Europe, governments have introduced a variety of tree planting and management schemes to bring about policy objectives ranging from agricultural diversification through to habitat creation and enhancement of biodiversity. Among the arguments used to support these objectives is that extra revenue can be generated directly from the adoption of farm forestry and indirectly via value-added activities occurring beyond the farm in both the upstream and downstream sectors. Specifically, forestry activity is perceived as providing replacement incomes for farmers, as they move out of some existing forms of production, and it is hoped that this will enhance rural prosperity, while delivering a range of environmental benefits. It is these nonmarket benefits connected with amenity, habitat, landscape, and animal welfare that will determine the third factor, namely public attitudes.

Against this context, the chapter examines the research findings in temperate regions in relation to four interrelated issues, namely:

1. The economics of agroforestry;
2. The attitudes of farmers to agroforestry;
3. Government policies toward agroforestry; and
4. Public perceptions of the nonmarket benefits of farm forestry.

16.2 ECONOMICS OF AGROFORESTRY

Although a number of studies in the last 15–20 years of the potential profitability of agroforestry have been conducted in Europe, the United States, and Australasia, the results of these studies have tended to be inconclusive. In general, the studies have shown that the potential depends on a large number of issues, including (1) tree species, (2) land type, (3) assumptions regarding the impact over time of canopy closure on the production of the understory crop, and (4) the choice of the discount rate.

High levels of public financial support for agriculture have made agroforestry a relatively unattractive option for farmers, particularly in Europe. One of the few studies to conclude unequivocally that agroforestry was economically attractive was that by Dupraz et al. (1995). This involved a study of farms in the Midi-Pyrénées Province of France, which traditionally had both agriculture and forestry. However, the majority of the European studies have been inconclusive. One of the earliest by Doyle et al. (1986) examined the comparative economics of widely spaced lowland silvopastoral systems using ash (*Fraxinus excelsior* L.), involving grazing the understory crop with sheep in southern Britain. In general, it was found that, where the level of inorganic nitrogen (N) fertilizer applied to the grass was low, namely between 0 and 100 kg N ha^{-1} $year^{-1}$, agroforestry compared favorably to sheep production on its own. However, at high N application rates to grassland of over 150 kg ha^{-1} $year^{-1}$, the converse was true. Whether the trees were grown until they achieved a trunk diameter suitable for firewood or timber was also important. If the trees were felled for firewood, then devoting the land solely to sheep production offered a better projected return than agroforestry, regardless of the nitrogen application.

More importantly, the results were sensitive to the choice of the discount rate. Because the returns from agroforestry were spread over many years, whereas those from sheep production occurred within a shorter time period, it was necessary to use discounted cash-flow techniques. This involved reducing all future costs and revenues to a single equivalent *present* value, taking account of the fact that money arising at different points in time has a different perceived value to the decision-maker (Ritson, 1978; Nix, 1984). The differing value of equivalent sums of money arising at different points in time was captured by the discount rate (Doyle et al., 1986); the higher the discount rate, the less valuable in present terms were returns at a more distant future date. At discount rates below 5% agroforestry was projected to be economically more attractive than sheep production over the length of a tree rotation. However, in the United Kingdom at least, the normal discount rate used to evaluate government projects was around 10%, and at this rate agroforestry was not competitive with the returns from agriculture.

Later studies have been equally inconclusive. Sibbald (1990) projected that agroforestry schemes would yield between 86% and 108% of the returns from hill sheep farming on improved land in the West of Scotland. In Wales, Thomas (1990) reported that the range in relative economic returns of cereal-grass-poplar (*Populus interamericana* L.), compared to farming alone, was 52%–65%, except where a sharp improvement in the price of timber relative to agricultural commodities was postulated. A rather more favorable comparison with pure agriculture was obtained by Willis et al. (1993), following the introduction of improved species of poplar (*P. interamericana* L.). Their study indicated that agroforestry systems might yield long-term economic returns ranging from 31% to 125% of those obtained from sheep farming alone, depending on site conditions and the eligibility of agroforestry for government grants. Thomas and Willis (2000) projected returns from agroforestry ranging from 94% to 104% of returns from lowland sheep farming in Northern Ireland, from 76% to 90% of the returns from hill sheep farming in Scotland and 85% to 98% of the returns from lowland cereal farming in southern England. Finally, in a recent large-scale agroforestry trial conducted in the Highlands of Scotland on a hill sheep farm, Waterhouse et al. (2002) found that income net of costs was increased by 33%, compared to sheep rearing alone, in the first full year of operating the system. Moreover, over the first 6 years covering the establishment period, the authors confidently predicted that the cumulative surplus (income minus costs) from the agroforestry system would be nearly twice that obtained under the preexisting conventional sheep system. However, these gains were only being achieved because of associated major changes in both the way the sheep flock was managed and its fecundity.

In Australia, New Zealand, and North America, assessments of the comparative economics of introducing agroforestry to farms have been more favorable. In New Zealand, Knowles and West (1988) showed that hill systems based on grazing sheep under widely spaced radiata pine (*Pinus radiata* L.) could improve the economic returns for many farm businesses. Both Knowles et al. (1991) and Dupraz et al. (1992) reported internal rates of return on agroforestry investments of 12%. Evaluations of agroforestry carried out by Etherington and Matthews (1983) and Bulman (1991) and Loane (1991) all showed that introducing agroforestry on farms in the South and East of Australia could improve long-term profits. In many instances, agroforestry was the only perceived way to prevent further soil and water degradation on the farms and raise agricultural productivity (Moore and Bird, 1997). Silvopastoral systems similar to those in New Zealand have also been developed in the southeastern United States based mainly on slash pine (*Pinus elliotii* L.) (Zinkhan and Mercer, 1997). In addition, silvoarable production systems have been developed based either on interculture of crops in fruit and nut orchards or, more recently, widely spaced rows of trees in arable fields, particularly black walnut (*Juglans nigra* L.) (Garrett et al., 1991). Although a number of studies have suggested that silvoarable (Kurtz et al., 1984; Kurtz et al., 1991; Garrett et al., 1994; Kurtz et al., 1996) and silvopastoral (Dangerfield and Harwell, 1990; Clason, 1995) systems could significantly improve farm returns for U.S. farmers, Zinkhan and Mercer (1997) have noted that, from an economic standpoint, agroforestry was still an unproven system in the United States.

Part of the reason for the apparent differences between the economic assessments from Europe and elsewhere lies in three factors. First, at least in New Zealand and Australia, levels of government price support for agricultural commodities have been lower than within the European Union, so that forestry as an activity has always been more competitive with agriculture. Second, although the European studies rested on projections from mathematical models and looked at general farming situations, the counterparts in Australia, New Zealand, and the United States were based more on experimental observations in highly specific situations. Thus, the former tended to focus on introducing agroforestry on high-grade farmland, whereas the latter were concerned more with marginal areas. This is reflected in the fact that the agroforestry systems in temperate countries outside Europe were often less complex than their European counterparts, which might, like Willis et al. (1993), involve both cereals and grass being grown in rotation under the trees. Nevertheless, even in Australia (Campbell White & Associates and Black, 1999) and the United States (Zinkhan and Mercer, 1997), it is clear that agroforestry is considered a risky land-use system, as it does not always produce larger returns than traditional agricultural enterprises.

Part of the reason that agroforestry is an economically unproven land-use system is that in nearly every case, assessments of the long-term economic benefits of converting traditional arable or grasslands to agroforests have had to rest on projections from mathematical models (Dupraz and Newman, 1997). This is because of the absence of experiments conducted over the entire tree rotation, even in regions like Australia (Moore and Bird, 1997), where longer experimental data sets are available. Because of the lack of reliable observational data, the results from the models are subject to considerable uncertainties. In particular, differences among studies in the apparent economic returns from agroforestry can be ascribed to differences in assumptions within the models regarding the effects of trees on intercrop productivity and the tree growth rates. Thus, the model by Doyle et al. (1986) predicted that grass growth under the tree canopy would decline to zero by year 40 of the rotation. In contrast, Dupraz and Newman (1997) used pasture productivity decay functions, which only reduced grass growth by 16% over the rotation. A similar decline was also postulated by Thomas and Willis (2000), who assumed that the livestock carrying capacity of silvopastoral systems would decline by up to 25% over a 30 year rotation, whereas cereal yields under silvoarable systems would fall to about 50% of equivalent open field crops over the same period. Likewise, there have been differences in the assumptions made about tree growth rates. In some cases, it has been assumed that better growth of individual trees in an agroforest may occur (Doyle et al., 1986) due to low and late tree-to-tree competition. However, it is equally plausible that the slower growth of individual trees may result from intercrop competition and from a less protected microclimate than in a full forest (Dupraz and Newman, 1997). Experiments by Dupraz (1994) and Sibbald and Agnew (1995) have suggested that both possibilities may occur. Thomas (1991) assumed that the height growth of poplars (*Populus* spp.) was independent of tree spacing, but this could be an oversimplification for widely spaced trees. Dupraz et al. (1995) made the assumption that at harvest, the bottom log of trees would have identical volume in an agroforest and a plantation forest. Hence, the wood productivity of the agroforest would be proportional to the final density of trees at harvest time. In contrast, Thomas (1991) calculated the bottom log volume with an assumed taper function applied to a basal diameter deduced from time and spacing of the trees, so that individual trees in a plantation forest and an agroforest have different volumes. As a consequence of these differing assumptions, the various bioeconomic models developed in the last 10 years have forecasted different effects of intercrop productivity and timber yields, with obvious consequences for forecasts of the economic benefits of agroforestry.

In addition, perceptions of the potential profitability have tended to be governed by assumptions about the relative movement of timber and agricultural prices in future decades. Implicit in nearly all the European assessments has been the assumption that agricultural commodity prices will fall due to trade liberalization, whereas timber prices will rise, due to increasing restrictions on the harvesting of tropical hardwoods. Thus, Thomas and Willis (2000) have assumed that timber prices could improve by up to 2% a year, whereas agricultural prices could fall by 2% a year. Over a 30 year rotation, this

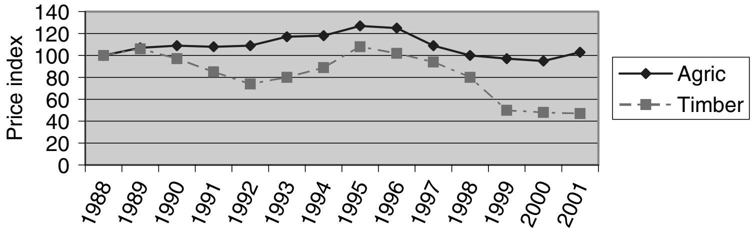

FIGURE 16.1 Comparative trends in the price indices for agricultural commodities (Agric) and timber in the United Kingdom over the period 1988–2001 (1988 = 100).

could mean that agricultural prices could decline in aggregate by 46% and timber prices could rise by 80%. However, relative price trends in the United Kingdom over the last decade for timber versus agricultural prices suggest that this optimism about timber prices is unfounded. Figure 16.1 shows the "all product" price index for agricultural commodities published by the Department for Environment, Food and Rural Affairs (2001) and the price index for standing coniferous woodland published by the Forestry Commission (2001) over the period 1988–2001. If anything is indicated by these price trends, it is that timber prices have been falling rather than rising in relative terms in the last decade.

In summary, generalizing about the economic potential of agroforestry is very difficult given the current state of knowledge in many temperate regions of the world. Certainly, in Europe it is difficult to provide confident forecasts of the comparative financial returns from agroforestry systems, especially given that such forecasts involve guessing future agricultural and timber prices in 25 or 30 years time. Instead, the evidence is that in temperate regions agroforestry under certain physical and environmental situations may be expected to produce better long-term returns than land-use systems solely based on grazing livestock. Thus, there is growing experimental evidence that agroforestry can increase the income potential of the farm business. However, predicting the conditions under which this will be the case is still problematic. For this reason, sentiment, rather than hard-nosed economics, may play a major part in determining whether farmers in temperate regions of the world adopt agroforestry.

16.3 FARMER ATTITUDES TOWARD FORESTRY AND AGROFORESTRY

Even if agroforestry can be unequivocally demonstrated to be more profitable than conventional agriculture, it is not a foregone conclusion that farmers will embrace it. Although farmers may state that they want to improve profits, it is a mistaken assumption that this means that they will adopt any new land use that is shown to be more profitable. As Moore and Bird (1997), Newman and Gordon (1997), and Zinkhan and Mercer (1997) have observed, a farmer will not suddenly switch to farm forestry to improve his income. Instead, his reaction will be to look at ways of modifying *existing* enterprises to secure an increase in profitability. Concentrating on the economics of agroforestry also ignores the fact that the reasons why farmers plant trees are many and varied (Matthews et al., 1993; McAdam et al., 1997; Thomas and Willis, 1997; Zinkhan and Mercer, 1997; Joannides, 1998). In North America (Matthews et al., 1993; Zinkhan et al., 1998) and Australia (Moore and Bird, 1997), the potential value of the wood produced is certainly a major factor in decisions to introduce agroforestry schemes on farms. However, environmental considerations are increasingly important (Zinkhan and Mercer, 1997; Sheldrick and Auclair, 2000; Binning et al., 2002). In Europe, studies by Appleton and Crabtree (1991) and McAdam et al. (1997) have shown that landscape, wildlife conservation, game, and shelter are all more important than increasing farm income, when deciding whether to plant trees.

These observations indicate that farmer perception and attitudes to agroforestry are likely to be central to its uptake. A number of studies, undertaken in recent years in the United Kingdom (Scambler, 1989; Sidwell, 1989; Appleton, 1990; Bishop, 1990; Gasson and Hill, 1990; Johnson, 1992; Dibden and Uzzell, 1992; Williams et al., 1994; Thomas and Willis, 1997), in the United States and Canada (Williams et al., 1997; Zinkhan and Mercer, 1997), and in Australia (Moore and Bird, 1997) to examine farmer attitudes to planting trees, have suggested that a high proportion of farmers regard agroforestry as an ‘‘inappropriate’’ use of productive land and as ‘‘irrelevant’’ as an alternative source of income. This widely held view derives from four perceptions. First, based on observation of natural forests, trees are regarded as slow growing, so they are not considered to be appropriate to the immediate financial pressures on the farm business. Second, afforestation is perceived as leading to an irreversible land-use change and is seen as limiting the future options for land use. Thus, in a survey of Northern Irish farmers, McAdam et al. (1997) found nearly a third of those surveyed expressed the view that agroforestry would give rise to long-term difficulties, associated with stump removal, interference of tree roots with drainage, and difficulties of working with machinery. Likewise, 10% of U.S. farmers questioned about the problems of silvopastoral systems stated that they considered it incompatible with other land uses (Zinkhan, 1996). Third, farmers frequently see agroforestry as a reversal of the conventional wisdom that producers should specialize (Newman and Gordon, 1997). Finally, forestry is usually seen as a ‘‘last-resort’’ activity, which is best suited to marginal areas of land. However, many of the European studies have indicated that agroforestry is most profitable on better quality land (Dupraz et al, 1995; Thomas and Willis, 2000), because the growth rate of the trees is predicted to be better. This is counterintuitive as far as farmers are concerned and many find the idea of afforesting good quality land a psychological barrier.

Institutional factors also act as a restraint on farmers’ enthusiasm for agroforestry. First, where the farmer is merely a tenant, renting the land from the landlord, the law may severely reduce the attraction of growing trees. Thus, in the United Kingdom, the trees on tenanted land legally belong to the landowner, so there are few incentives for tenants to manage woodlands to produce utilizable timber (McKnight, 1996). This is reinforced by the fact that controversial legal rulings have fuelled concern among tenant farmers that utilizing farm woodlands for anything other than ‘‘agricultural’’ use actually contravenes the terms of their agreement in respect of renting the land. As a result, the tenant farmer is dissuaded from even discussing the planting of trees with the landlord. Second, and just as significantly, commodity support programs in some countries actually constrain tree planting because land planted under trees is explicitly excluded from area payments and so reduces the land area qualifying for support (Williams et al., 1997). In other cases, the existing system of grants may militate against the adoption of agroforestry, even though it encourages conventional forestry on farms (Willis et al., 1993; Bullock et al., 1994). Thus, in the United Kingdom, generally, rates of grant for agroforestry schemes are merely determined on a pro rata basis, in terms of the number of trees per hectare relative to conventional forestry systems. However, this method of payment may fail to recognize the proportionately higher costs of agroforestry production (Thomas and Willis, 1997). In the United Kingdom, only Northern Ireland has a different approach to the public funding of agroforestry. Since 1995, farmers in this region investing in agroforestry have been entitled to 50% of the grant eligible for conventional forestry. As a consequence, farmers in Northern Ireland are reported to feel that the level of grants for agroforestry is adequate (McAdam et al., 1997).

Added to both the institutional and attitudinal restraints on farmers adopting agroforestry is a lack of knowledge. This expresses itself at three levels. First, there is a huge lack of awareness about agroforestry systems among the farming community in temperate parts of the world. In a survey of Northern Irish livestock farmers in 1997, it was revealed that 36% of those responding had never heard of agroforestry and the majority of the remainder only knew about it through newspaper articles (McAdam et al., 1997). In a slightly earlier survey of Canadian farmers in Ontario, Matthews et al. (1993) found that only 37% knew anything about silvopastoral systems and only 4% had ever heard of silvoarable systems. Second, even among advisers to farmers, there is perceived to be a lack of technical information about agroforestry. McAdam et al. (1997) identified that both farmers and advisers stated that they needed more information on issues related to the best types of land, the best varieties of trees, and the best methods of weed control before they would feel confident about recommending and adopting agroforestry. Added to this, it is clear that many farmers do not have the practical skills, relating to the harvesting, processing, and marketing of timber, to effectively manage agroforestry systems (Newman and Gordon, 1997). Only in countries, like Chile, Australia, and New Zealand, where a relatively more widespread practice of agroforestry based on combining one or two tree species with grazed pasture has developed, is there a view that there is sufficient technical information available not to obstruct farmer adoption (Thomas et al., 1990; Moore and Bird, 1997).

So, for many temperate farming regions, the adoption of agroforestry is constrained by a lack of scientific knowledge. The management of such systems is inherently more complex than conventional farming activities (Sinclair et al., 2000), as they involve managing the interactions between trees, crops, and animals. Moreover, there is no single blueprint for agroforestry systems. Instead there are many different niches for trees, which may be occupied by a variety of tree species. Newman and Gordon (1997), Williams et al. (1997), and Mercer and Miller (1998) have all expressed the view that further research is needed to answer some of the practical issues involved in establishing and managing agroforestry systems in temperate areas. In particular, Newman and Gordon (1997) have stated that further empirical proof is required before it is really possible to assert whether temperate agroforestry is either more agronomically or more environmentally efficient than monocultures. Williams et al. (1997) put it slightly differently by saying that a more systems-oriented approach to agroforestry was required and this involved overturning the conventional notion that research precedes extension. In their view, such systems would only begin to make progress when researchers actually worked alongside farmers.

To summarize farmer attitudes to agroforestry, the position over large parts of Europe and North America may be stated as one in which there may be a will to consider agroforestry systems, but there is a credibility problem over the relevance and practicality of such systems, compounded by a lack of knowledge and expertise among both farmers and their advisers about agroforestry management. For these reasons, it is unlikely at the current point in time that farmers in Europe or North America will adopt such systems on a widespread basis without strong government encouragement in the form of grants for establishing the systems on farms and funding to support further research and extension activities. As Zinkhan and Mercer (1997) noted in respect of the United States, reducing the financial uncertainties associated with agroforestry is probably the most critical factor for expanding it.

16.4 GOVERNMENT POLICIES TOWARD AGROFORESTRY

The motives for governments promoting and supporting agroforestry in temperate regions are diverse, including

1. a need to diversify farm incomes in order to reduce dependence on public financial support;
2. pressure to withdraw land from farming, because of either growing agricultural surpluses or a need to reduce soil erosion brought on by intensive agriculture; and
3. an expectation that agroforestry will have wider employment opportunities for the rural sector, helping to sustain marginal rural communities.

For many governments in Europe, the United States and Australasia, agroforestry is widely perceived as a “win-win” strategy for landowners, who need to seek improved farm incomes through enterprise diversification, while mitigating the negative environmental impacts of intensive agriculture (Williams et al., 1997). However, governments remain ambivalent about whether agroforestry can make a significant immediate impact on farm incomes or the well-being of the wider rural economy (Buck, 1995) and, as a result, few have introduced specific policies to support agroforestry, as distinct from farm forestry, in general. States, like the Victorian State government in Australia, which have formal agroforestry programs, tend to be the exception. In Victoria, there is an aim to establish 30,000 ha of plantation and agroforestry on farmland by 2020 (Moore and Bird, 1997). Much more common is the situation that is to be found in the United States and Canada, where there has been a gradual move in conservation and land management programs to develop policies, which do not discriminate against agroforestry (Williams et al., 1997). With a shift toward more sustainable farming systems, there has been a growing interest in exploring integrated approaches to land use (Joannides, 1998), which offer the prospect of economically viable farming systems and are also environmentally sustainable. For similar reasons, agroforestry has attracted government attention in Mediterranean Europe. There it is seen not only as a diversification opportunity, but also as a land-use system that will maintain fodder production in areas where landowners would otherwise plant forests and exclude pastoral activities altogether, and a means of increasing summer forage production in dry areas (Dupraz and Lagacherie, 1990). However, as a sustainable land-use system, agroforestry is perceived as only one among many, and probably not the one with the most potential. The truth is that governments will only begin to support agroforestry in a positive way, if it can be shown that it either delivers nonmarkets benefits in the form of environmental goods that the general public value or it has wider income and employment benefits, which compensate for its uncertain farm income effects. The first of these issues is considered later in this chapter, so attention is now focused on the second.

Any expansion of agroforestry will be expected to have social and economic benefits beyond the farm gate, in terms of both employing people to plant and harvest the timber on the farms and to process the harvested timber into wood products. Thus, Waterhouse et al. (2002) mapped the spatial distribution of the goods and services procured by a single agroforestry project in the Scottish Highlands and showed that there was a significant boost to the local, rural economy. However, generalizing these socioeconomic benefits is difficult, because of the complex nature of the land-use system and the paucity of actual studies. One of the few attempts to look at the wider socioeconomic effects of agroforestry in temperate regions was conducted by Doyle and Thomas (2000) and extended by Doyle (2002). The starting point for the analysis was that one way to gauge the impacts was to consider agroforestry as an agricultural system, based predominantly on grassland farming, with timber as a ‘‘minor’’ component. A measure of the wider social and economic benefits could then be assessed from the employment directly and indirectly supported by a grass-based livestock system. On this basis, Doyle and Thomas (2000) reported that in the United Kingdom for every person employed in agroforestry on farms, between 0.5 and 1.3 were employed in allied industries. However, whether the implied total (gross) employment effects, reported in this study, of 2–4 jobs per 100 ha of agroforestry can be regarded as a measure of the net benefits to society is debatable. If, in the absence of agroforestry, the land has no alternative productive use, then the *gross* employment effects would be a correct measure of the wider socioeconomic gains. However, if the land used for agroforestry primarily displaces traditional grass-based livestock farming, the *net* social gains from the introduction of agroforestry would largely be linked to the ‘‘added’’ forestry component of the system.

An approximate estimate of the socioeconomic gains that might be realized from the ‘‘forestry’’ component of agroforestry systems can be obtained by reviewing the work done on conventional forestry systems. Studies commissioned by the Forestry Commission in Great Britain have shown that for every job in forestry a further 0.8 jobs are created elsewhere in the economy. Based on the evidence that 1 man is directly employed in managing timber production for every 100–250 ha of woodland (Central Statistical Office, 1997), this suggests that every 100 ha of forestry creates 0.7–1.8 jobs in total. This is only 35%–45% of the estimated gross employment created by livestock farming. However, the relatively small observed employment impacts associated with conventional forestry may underestimate the potential gains from agroforestry. In particular, recent estimates of the employment impact of tree planting on farms in Scotland, have suggested that for every person

TABLE 16.1
Additional Income and Employment Generated beyond the Farm-Gate by Farm Forestry

Type of Woodland Planting	Additional Income per £1 of Farm Income Generated	Additional Jobs per Job Created on Farms
Commercial coniferous planting and maintenance	0.54	0.58
Farm forestry planting and maintenance	1.67	1.79
Timber harvesting	1.97	0.77

Source: http://www.forestry.gov.uk/website/oldsite.nsf/byunique/HCOU-4U4JMJ

employed full-time in timber production a further 1.8 are engaged elsewhere in the economy (see Table 16.1). This is consistent with an earlier U.K. study by Slee and Snowdon (1996) that suggested that farm-based forestry schemes supported 1–2 additional jobs per man employed in forestry. They also showed that, where employment creation was a specific objective of the rural development program, then figures of 3–4 jobs per person employed in forestry were achievable. Based on these observations, the potential employment impacts of farm woodland planting schemes may be nearer to 2–4 people per 100 ha.

However, regardless of whether agroforestry leads to direct job creation on farms, in so far as it increases the incomes of farm households, it will have an economic effect on the wider local economy. Recent unpublished studies, commissioned by the Forestry Commission in Scotland, have indicated that farm woodland planting and maintenance increased incomes outside farming by US$1.7 for every additional US$1 of farm income. The corresponding figure for timber harvesting was US$2 for every additional US$1 of farm income. For agroforestry, the benefits may be proportionately larger, in that the trees could add value to existing grassland systems by either increasing agricultural output or increasing returns per unit of output (Doyle and Thomas, 2000). Especially in hill areas, the presence of agroforestry may increase the shelter provided for animals with benefits in terms of output.

However, given the very limited commercial experience with agroforestry systems in many temperate regions of the world, considerable caution is needed in projecting the socioeconomic benefits, until actual evidence becomes available. In the first place, the scale of planting is likely to be modest for the foreseeable future and consequently some of the benefits connected with processing the timber will only arise if there is sufficient timber in an area to justify setting up processing facilities. Second, the potential socioeconomic gains from agroforestry may not easily be realized. Doyle and Thomas (2000) in a review of farmer surveys noted that few producers expected woodland planting to significantly boost their total farm income. If this is true, then agroforestry would be unlikely to stimulate increased farm spending and in turn increased demand for off-farm services.

16.5 PUBLIC PERCEPTIONS OF THE NONMARKET BENEFITS OF FARM WOODLANDS

The preceding discussion shows that the benefits to society from increased agroforestry may be linked as much to nonmarket benefits, associated with landscape, habitat creation, wildlife conservation, and recreation, as to any economic benefits, such as employment. Although there have been questions about whether there are unequivocal environmental gains from farm forestry (House of Commons Environment Committee, 1993), the evidence is growing that it can generate environmental and amenity benefits. In particular, Swain (1987) has stressed that much agroforestry takes place on poorer quality and unimproved grassland. Because another tier is added to the vegetation

TABLE 16.2
Summary of the U.K. Evidence on the Impact of Agroforestry Systems on the Diversity of Fauna and Flora

Impacts Relative to Agricultural Systems	Agroforestry System	
	Silvoarable	Silvopastoral
Consistent increase	Small mammals, flying arthropods, hoverflies, slugs, flora	Birds, spiders, flora
Consistent decrease	Aphids	
Both increases and decreases	Carabid beetles	
No effects found or no information available	Birds, spiders, staphylid beetles	Small mammals, carabid beetles, staphylid beetles, flying arthropods, aphids, hoverflies, slugs

Source: After McAdam, J., *Agroforestry in the UK*, Forestry Commission, Edinburgh, UK, 2000.

canopy and the ground vegetation is not shaded out completely, the wildlife implications are not as severe as with conventional forestry. Recent works by McAdam (2000) and Burgess (1999) in the United Kingdom and Borsboom et al. (2002) in Australia have begun to confirm that agroforestry may have positive effects on biodiversity.

In a comprehensive survey of trials in the United Kingdom, McAdam (2000) observed that in established agroforestry systems, where the tree canopy did not have a significant effect on ground vegetation or output, the presence of the trees attracted small mammals, some invertebrate groups, and slugs and enhanced the diversity of ground flora. Silvopastoral systems also encouraged birds, either through the spatial habitat diversity created, or through the increased levels of invertebrates, which act as a feed source. A summary of the ecological effects of agroforestry systems in the United Kingdom is provided in Table 16.2. In silvoarable systems, Burgess (1999) reported an increase in the populations of small mammals and flying arthropods with the introduction of agroforestry. Finally, in the study by Borsboom et al. (2002), which reported on the effects of agroforestry on fauna biodiversity in Queensland, Australia, replacing pure pasture by agroforestry systems, involving eucalypts, was observed to increase all vertebrate species. The effects tended to be most marked in small planting blocks of 10 ha or less, where there was some connectivity to some form of natural habitat.

At the same time, the planting of small blocks of trees in agroforestry systems can be expected to improve the visual appearance of the landscape by compartmentalizing the land area and reducing the appearance of openness. However, the planting needs to be sympathetic to the landscape if it is to prove aesthetically pleasing (Bell, 2000). Equally, some environmental benefits can be anticipated from reductions in nitrogen use and lower stocking intensities that are generally the consequences of switching from all livestock to agroforestry systems (Doyle et al., 1986; Lloyd, 1990).

The significance of the nonmarket benefits of agroforestry has been recognized in the United Kingdom (Slee and Snowdon, 1996), in the United States (Zinkhan et al., 1998) and in Australia (Binning et al., 2002). In Australia in particular, the government has recognized that unless farmers can capture the nonmarket benefits of agroforestry, there may be insufficient incentives for farmers to adopt agroforestry on a large enough scale to tackle the serious environmental problems in many river catchments. Binning et al. (2002) estimated that the traditional mix of policies might be successful in fostering community participation and partnerships to generate an increase in agroforestry from 5% to 10% or even 15% land cover over 10–20 years. However, in some areas dry-land salinity problems require more than 30%–40% of the landscape to be planted with trees. They concluded that larger market-based incentives reflecting the nonmarket environmental and ecological benefits of agroforestry were necessary, if farmers were to accelerate rates of tree planting. These conclusions have been reinforced by Campbell White & Associates and Black (1999), who found in a survey of nine case studies, that only five of the farms could expect a clear increase in income from agroforestry, although all the properties showed strong positive environmental gains.

The problem, however, is how to ensure land managers are financially rewarded for the environmental services that agroforestry provides. Binning et al. (2002) examined the opportunities for creating markets for services like carbon sequestration, salinity mitigation, biodiversity enhancement, and water-quality improvement. However, they noted that creating markets for environmental services would be difficult. Specifically, property rights to the intangible services would need to be assigned and a scarcity of environmental goods would need to be created. The latter aspect is important, as the market value of services and goods is a function of their scarcity. Binning et al. (2002) suggested that one way of creating such a scarcity was to impose limits on vegetation clearance. Farmers in Australia would only be able to clear new land, if they created a habitat, which compensated for the loss of biodiversity. They themselves might not create this habitat, but would pay another farmer, who might be planting trees, to create it. In other words, they would purchase a "biodiversity credit" from this farmer, in the same way that it is proposed that countries might buy "carbon credits" to offset their carbon dioxide emissions. However, though such methods will create a market value, which will attract private investors, the environmental benefits produced by agroforestry will take time to be realized. In the short run, the government will have to pay for these services. This in turn means that the general public as taxpayers must be willing to fund the payments to farmers.

This raises the question of just how much is the public willing to pay for the environmental services. Slee and Snowdon (1996) attempted to do just this for the United Kingdom. They specifically sought to quantify the benefits arising from recreation provision, increased wildlife diversity, and landscape enhancement from farm forestry. Using estimates by Willis and Benson (1989), Garrod and Willis (1992), Hanley and Spash (1993), and Spash and Hanley (1994), Slee and Snowdon (1996) estimated the recreational, conservational, and landscape benefits of afforestation to the public. In addition, the benefits of increased woodland planting as a sink for carbon, counteracting the continued rise in atmospheric carbon dioxide concentrations, were also considered using values provided by Pearce (1991). This exercise showed that the social value of forestry could be significantly improved when these nonmarket benefits were included. Thus, the discounted benefits, assessed at a discount rate of 8% over a 30 year rotation, in respect of recreation and wildlife conservation for a conventional woodland in Scotland, were put at between US\$215 and US\$930 ha^{-1}. The benefits of carbon sequestration were worth another US\$260–US\$320 ha^{-1}. In contrast, the discounted net benefits of timber production on its own were negative and worth between *minus* US\$1430 and *minus* US\$2430 ha^{-1}. Thus, the nonmarket benefits represented an important and significant social justification for any public funding for agroforestry. More recently, Campbell White & Associates and Black (1999) explicitly studied the economic value of environmental gains from agroforestry for 10 farms across Australia. On the basis of projected declines in the productivity of the farm, if trees were not planted, their calculations indicated that, on average, the environmental benefits were worth five times the value of the gains from timber and livestock activities. For four of the case-study farms, the environmental benefits were the prime justification for adopting agroforestry. However, as Campbell White & Associates and Black (1999) stressed, these nonmarket values are more a perception in the farmer's mind than a tangible asset at the moment.

16.6 LOOKING INTO THE FUTURE

This review of the social and economic implications of agroforestry for rural economic development in temperate regions underlines four key points:

1. It is difficult to demonstrate that agroforestry unequivocally increases profits of arable and livestock farmers. In Europe, the benefits of adopting silvopastoral and silvoarable systems

depend very much on the individual circumstances of the farm. Unless there is a significant rise in timber prices relative to those for conventional agricultural commodities, or a rise in the level of government grants afforded to agroforestry, agroforestry will struggle to be the preferred diversification opportunity for most farmers, who will primarily look at ways of modifying existing enterprises to improve profitability. Elsewhere, such as in the United States and Australia, the reasons why agroforestry may prove attractive are less connected with economic returns and more related to the urgency of undertaking measures to mitigate environmental degradation. Even in these countries, as both Zinkhan and Mercer (1997) and Binning et al. (2002) note, the spread of agroforestry may be inhibited by the lack of government grants.

2. Accordingly, farmer attitudes toward farm forestry and the environment are likely to play a large part in determining the uptake of agroforestry in the immediate future. In this respect, it is worth noting that in large parts of the temperate world there is widespread ignorance of agroforestry and, even where farmers are aware of it, they are inhibited from adopting it, because of its apparent complexity. As both Buck (1995) and Williams et al. (1997) observed, for a multipurpose land use such as agroforestry, it is essential that the institutional framework for providing advice to farmers integrates all the different agencies that are involved. Too often agroforestry development requires the farmer to interact with a whole series of agencies connected with farming, forestry, and the environment, which have overlapping remits. At the same time, if agroforestry is to be a viable diversification opportunity for a large number of farmers, then a series of technological problems in relation to tree species, crop management, and livestock performance will need to be solved through an interdisciplinary approach. At the same time, to successfully marry the expertise from numerous agencies with the diverse interests of landowners means that a reliance on the conventional "linear" technology transfer model needs to be replaced with a more pluralistic approach, in which scientists and farmers interact continuously and farm practices serve to inform the research.

3. In creating a favorable economic climate for agroforestry and in ensuring that the institutional framework for advising farmers on such systems is not fragmented, governments will play a major role. Clearly, for governments, the attraction of agroforestry depends on the extent to which it addresses three distinct policy problems, namely (1) finding viable diversification opportunities for farming, (2) helping to sustain employment in the wider rural community, and (3) providing a mechanism for mitigating environmental degradation. In terms of the wider socioeconomic policy agenda, the problem at the moment is that examination of the socioeconomic implications of agroforestry has been comparatively neglected (Mercer and Miller, 1998). However, from the limited research undertaken, it is already clear that, although agroforestry has the potential to stimulate off-farm employment, whether the jobs are created locally depends on there being a coordinated and integrated strategy for farm forestry in rural areas. Without this coordination, the likelihood is that few of the jobs generated in the forestry-related industries will be in peripheral rural areas.

4. With the economic viability of agroforestry for many farmers being marginal, the nonmarket benefits of agroforestry, connected with recreation, amenity, and conservation, are as important as income or employment effects. As shown by Doyle and Thomas (2000) and Campbell White & Associates and Black (1999), the value of the nonmarket effects may exceed the value of the forestry and agricultural output on some farms. However, for the imputed value of the nonmarket effects to be tangible for the farmer, it is necessary to create a market for environmental goods. In the long term, it may be possible for the governments to create such markets and attract private investors, but in the short term, the government itself will have to act as the buyer for environmental services, through the mechanism of grants. This means that the public as taxpayers needs to be willing to pay for

environmental services. At the moment, just how much the public is willing to pay for such services, when the payments become transparent, is a matter for debate in Europe, Australia, and the United States.

In summary, agroforestry as a land use offers considerable potential in the temperate regions of the world, as governments look to ways to diversify farming in directions that are considered both environmentally and economically sustainable. However, notwithstanding the potential to improve the profitability of farming in certain situations and to assist wider rural economic regeneration, until farmers are better informed about it and the public financial support for agroforestry improves, the uptake of agroforestry will be inhibited. The logic for introducing agroforestry may be compelling, but the economic case is still unproven.

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17 Forest Patches in Northeast Luzon (the Philippines): Their Status, Role, and Perspectives for Conservation in Integrated Land-Use Systems

Denyse J. Snelder and Gerard A. Persoon

CONTENTS

17.1 FOREST PATCHES IN TROPICAL LAND-USE SYSTEMS

17.1.1 USE AND CONSERVATION OF FOREST PATCHES UNDER PRESSURE

In the literature, the term *forest patches* generally refers to a diversity of habitats and management conditions, ranging from patches of trees in pastures, gallery forest along creeks, and lines of trees along field boundaries to managed forests and large fragments of primary forests. It includes both forest remnants and remnant forests. As opposed to forest remnants, ‘‘which may be patches of living dead or even single trees,’’ the remnant forests are characterized as forests in the true sense of the word, that is, ‘‘naturally regenerating forest communities’’ of any size (Alcorn, 1996). From a management perspective, the remnant forests are ‘‘largely unused fragments of old-growth forest, including forest intentionally uncleared for watershed protection, wildlife conservation, and other benefits, or patches that simply have not been cleared yet because of inaccessibility or recency human occupation" (Schelhas and Greenberg, 1996). Forest patches thus vary in structure and species composition because of differences in management and local biophysical conditions, but they all have one characteristic in common: trees and woody shrubs predominate in the aboveground biomass.

Throughout Southeast Asia, rural communities use forest patches in different ways. They provide a variety of food and household products, including fruit, leafy vegetables, meat, fuelwood, construction materials, and medicine. Harvesting times are not confined to one season but spread throughout the year, reducing risk and meeting emergency needs, particularly during times of low agricultural work or wage labor. Moreover, forest patches may serve various crucial ecological functions, such as firebreaks in the grasslands and soil and water conservation along watersheds and hillslopes (Snelder, 2001a). They may contribute to natural regeneration and serve as a refuge area for remaining wild life and a relatively wide range of species, as discussed elsewhere in the literature (Greenberg, 1996).

Yet, forest patches seem increasingly threatened by two forms of gradual change in resource exploitation: the intensification of agriculture and fuelwood extraction. The term agricultural intensification is used in different ways (Hunt, 2000), but generally defined as a higher crop output per unit area and per unit time. Although the latter description suggests less land is needed to feed one person, implying a reduction in land claims for agriculture because of which more forest may be conserved, the reverse may be true as population increases (see also Van Noordwijk et al., 1997). For example, during the second half of the previous century in the Philippines, the migration of landless and jobless farmers toward areas of extensive land use has resulted in the conversion of large tracts of forests into agricultural fields (Kummer, 1992; Eder, 1996; Cruz, 1997; Garrity et al., 1997). These changes may follow pathways of different directions. There may be an overall trend toward ecosystem simplification where land is increasingly used for cash-crop cultivation and leftover forests are suffering from its negative side effects (such as use of pesticides, chemical fertilizers, and herbicides and burning of fallow fields and crop residues). Yet, agricultural intensification may also be accompanied by diversification, depending on the initial situation, the biophysical environment, the subsistence needs and market conditions, the policy environment, and the sociocultural context (Schelhas, 1996). For example, in a study of two pioneer settlements in South Luzon, Philippines, Fujisaka and Wollenberg (1991) found that agricultural intensification led to diverse systems of root and mixed-perennial cropping (including citrus and other fruit trees, coffee, cacao, and banana), rather than systems of monocropping. Yet the change to diverse cropping systems took place under specific conditions, that is, after annual cereal and cash cropping proved not to be viable (due to poor soil, weeds, pests, etc.) and initial funds for field establishment were derived from small-scale logging and charcoal making. Similarly, Conelly (1992) reported that the farmers of the frontier community Napsaan in Palawn, Philippines, shifted from unproductive short fallow-swidden systems to diverse productive farming systems with small-scale irrigated agriculture and increased labor efficiency. The study represents, however, a short-term assessment at the point of transition. Conelly (1992) predicts that, in the long run, diversity may decline and yield may decrease leading to higher cost of labor and fertilizer input, and thus lower labor efficiency.

Fuelwood extraction forms another threat to the preservation of forest patches in agricultural landscapes. In the 1970s, fuelwood harvesting was proposed as the major cause of deforestation worldwide, a belief that led to the so-called Fuelwood Gap Theory in the 1970s (for a review see FAO, 1997). The theory postulated a gap between the rate of fuelwood consumption and the rate of sustainable wood supply, for many countries. It was assumed that most fuelwood was derived from natural forests, although available data on fuelwood sources were still scarce and incomplete at that time. More studies that are recent give, however, a rather different view on the origin of most fuelwood and its association with processes of deforestation. For example, Soussan et al. (1992) and FAO (1997) state that local communities rarely fell trees for fuel use and mainly depend on trees close to their houses to meet daily wood-energy demands. Likewise, in case of shortages, communities may establish fuelwood plantations as practiced by farmers in Cebu in the South of the Philippines (FAO, 1993b).

Practices of forest development, utilization, and management are subject to changes. For example, Wiersum (1999), discussing the various evolutionary stages of forest utilization and management in tropical regions, states that communities show increasing interest in forest development activities as forest resources become scarce, whereas private incentives to restore the availability of wood resources in the form of plantations arise. Domesticated trees are planted in forest homegardens to provide the majority of people's needs, whereas the remaining forest is kept to provide other products and services, such as watershed protection, restoration of soil fertility, fire control, and shelter for livestock.

Economic progress and ongoing development of infrastructure in urban and rural areas further affect forest utilization and increase the accessibility of other types of cost-effective energy sources. Yet in the Philippines, fuelwood and charcoal usage is likely to increase in the near future. These expectations are not only based on the growing population but also on the high prices of alternative fuels such as kerosene and natural gas. Cruz (1997) calculated that the total fuelwood consumption in 1985 (28.5 million cubic meters) was roughly equivalent to the volume of wood lost from deforestation and considerably higher than the volume of commercial timber harvested in the same year (3.6 million cubic meters).

17.2 CASE STUDY OF FOREST PATCHES IN NORTHEAST LUZON

The natural forest patches in the agricultural landscape of Northeast Luzon are leftover from the *Dipterocarp* rain forests once covering most parts of the region, or, in many cases, have spontaneously established themselves on former forested land, grassland, and cropland, after changes in land-use and management styles. The previous closed-canopy forests have been subject to different processes of fragmentation and degradation, like elsewhere in the Philippines (FAO, 1993a).

The closed-canopy forests of the remote Sierra Madre Mountains (Figure 17.1) have been degraded relatively recently, within a rather short time span, by commercial logging between 1950 and 1990 (Van den Top, 2003). The degradation of the rain forest in the more accessible hilly areas at the foot of the Sierra Madre Mountains has been a rather gradual process that started even before the eighteenth century. At that time, shifting cultivation was commonly practiced in forested areas throughout the region for the production of rice, millet, corn, root crops, and vegetables (De Jesus, 1980). Agricultural practices changed over time with a steady growth of the regional population and the spread of commercial crops, such as, tobacco in the eighteenth century. The growing cycles of crop cultivation, the reduction of fallow periods, and the repeated burning for land preparation caused a decrease in soil fertility and lowered crop yields. Consequently, where organic inputs remained absent or were inadequate, fields were eventually abandoned and left for cattle grazing (De Jesus, 1980; Scott, 1994). Nowadays, the hilly areas are part of the agricultural landscape of the Cagayan Valley (Figure 17.1), consisting of grasslands and cultivated fields with dispersed patches of forest.

These so-called derived grasslands are generally perceived as unproductive land traditionally used for marginal agriculture and extensive animal husbandry. Within these grasslands, forest patches mainly occur in the form of woody patches on hillslopes and watersheds, gallery forests along streams and rivers, and hillslope forests covering complete (or parts of) watershed and catchment areas (Snelder, 2001b). They were left untouched by logging companies because of their low value in terms of timber species. Although their spatial distribution and coverage may seem insignificant at first sight, the conservation and management of these woody elements may prove to be of high significance for rural livelihood and programs of reforestation and grassland rehabilitation.

Yet, there is evidence that, in more recent decades, forest patches are under mounting pressure with the growing population and shortages of cultivable land. The knowledge of forest patches, that is, their status and level of exploitation and management is, however, insufficient to be conclusive about the continued existence of these fragmented wood resources in the near future. Table 17.1 shows data on the land-use distribution within the Cagayan Valley for 1990–1991 and 2002–2003.

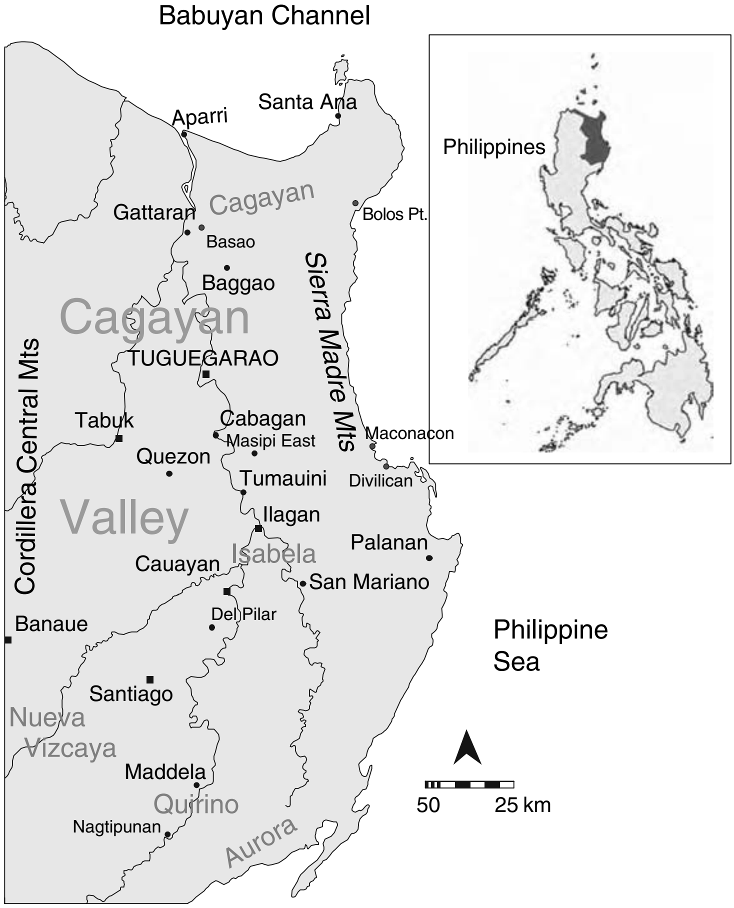

FIGURE 17.1 The location of the study area in Northeast Luzon, Philippines.

The data suggest a 2% increase in farm area throughout the valley and a 14% decrease in broad-leaved, mostly lowland *Dipterocarp*, forest, excluding tree plantations and mangrove forests. Isabela province, the largest of the five provinces comprising the valley region, contains the largest part of this forest, that is, 401,844 ha in 2003 or 39% of the 1,024,383 ha of the valley's forest cover in total (Forest Management Bureau, 2006). It is also the province with the largest number of farm holdings, that is, 129,715 in 2002 (40% of the total number of farm holdings in the valley region; NSO, 2006), covering 240,600 ha. It is, however, not clear from the values in Table 17.1 what happened with the lost forest. In Isabela, the total area under crop cultivation has also decreased since 1991, and likewise the total area composed of rangeland, grazing land, or pasture decreased considerably. Although the latter change corresponds to the government's effort to have reduced the area under "unproductive" grassland to almost 69% by 2020 (Boquiren, 1992), the outcome of the same policy directed at the conversion of this land into agroindustrial cropping is not clearly reflected in the regional or provincial crop data. These ambiguities in land use are on the one hand related to the regional and provincial data summaries masking true local land-use conversions. On the other hand, complexities are generated because of the various government institutions in charge of land-use and census operations (e.g., Department of Agriculture (DA), Department of Environment, National Economic and Development Authority (NEDA) and Natural Resources, National Statistics Office (NSO)). Although in theory the institutions cover distinct areas of operation, in practice there is considerable overlap created by, for example, areas that are double, reclassified, or not

TABLE 17.1

Changes in Land-Use Distribution in the Cagayan Valley (Region 2) and the Isabela Province between 1990–1991 and 2002–2003, for Both "Forestlands" and "Alienable and Disposable (A & D) Lands"

	Cagayan Valley[a]		Isabela Province[b]	
	A & D Land (ha)			
Year	**1991**	**2002**	**1991**	**2002**
Area of farms[c]	530,143	540,812	263,339	240,600
Seasonal crops[c]	443,515	437,006	229,608	206,621
Permanent crops[c,d]	46,266	55,898	19,400	19,718
Forest[c]	2,625	2,654	764	724
Meadows or pastures[c]	10,723	8,787	6,563	1,344
Other lands[c]	17,827	25,495	2,423	8,830
	Forestland (ha)			
Year	**1990[e]**	**2003[f]**	**1990**	**2003**
Forest	1,193,561	1,024,363	454,500	401,844
Old growth, secondary	969,300	981,811[g]	372,700	383,691[g]
Mossy	102,200	64,750[g]	81,800	42,006[g]
Pine	1,600	1,600[g]	0	0[g]
Watershed reserves	120,461	65,157[g]	0	9,330[g]
Mangrove	5,500	8,602	400	1,258
Plantation	102,876	120,026[g]	19,429	14,196[g]
Grazing land or pasture	70,629	27,792[g]	31,420	14,141[g]
Rangeland	351,192	264,730[g]	144,943	114,575[g]

[a] Total land area of 2,683,996 ha.

[b] Total land area of 1,066,456 ha.

[c] The 1991 and 2002 farm and land-use data are based on, respectively, the 1991 and 2002 censuses published by NSO (1994, 2006); see also http://www.census.gov.ph

[d] Major permanent crops include banana (*Musa* sp.), calamansi (*Citrus microcarpa*), coconut (*Cocos nucifera*), coffee (*Coffea arabica*), and mango (*Mangifera indica*).

[e] All 1990 data on forestland are based on 1990 data of the DENR (personal communication) and partly published in CVPED (1992).

[f] All forest-related 2003 data derived from Forest Management Bureau (2006), http://forestry.denr.gov.ph

[g] Data for 2000 based on DENR 2000 data (personal communication); total forest area for 2000 is 1,113,318 ha for the Cagayan Valley and 435,027 ha for Isabela province.

classified at all. Moreover, figures on land use are compiled in different ways, using different methods and data-gathering techniques. For example, all the land that is classified as forestland is public land and under the sole jurisdiction of the Department of Environment and Natural Resources (DENR), yet, not all forestland is indeed covered by forest. Part of this land is set aside for grazing or developed for crop cultivation as a special component of programs like the Integrated Social Forestry Program and the Forest Grazing Land Management or Pasture Lease Agreement (PLA). Reversely, not all forest-covered land is public land. A small part of it, for example, 8,696 ha (about 2%) of the 411,804 ha of forest-covered land in Isabela (2003 data, including plantations; Forest Management Bureau, 2006), is alienable and disposable (A & D) land. Similarly, the data on grass-covered land generate confusions: the figures for the total area under grass vary depending on their assessment by the DENR, that is, figures solely based on public lands (e.g., 351,192 ha of rangeland for 1990; DENR, personal communication), or the DA or NEDA, figures based on both public and A & D land (e.g., 480,209 ha of grassland or shrubland; CVPED, 1992), or by the NSO with figures based on area of farms on A & D land (e.g., 10,723 ha of land under permanent meadows or pastures; NSO, 1994).

In addition to the confusions associated with the area under forest cover, as determined by several institutions, information about the quality of the remaining forest is scarce. Although distinction is made between different types of forests, little is known about the canopy status and species composition of forests up to 2006 (a positive development is the distinction between open and closed forests for both A & D and forestland that is being made in recent publications; Forest Management Bureau, 2006). Hence, questions related to the current status of forest quality and the effects of agriculture, fuelwood collection, and logging operations are difficult to address.

More insight information on forest resources is crucial for finding answers to a series of questions related to the sustainable use and exploitation of these natural resources and the preservation of biological diversity in forest and agricultural areas. For example, will agricultural intensification lead to the degradation and removal of forest patches and the simplification of remaining land-use systems, or, is it more likely that the forest patches are integrated into the existing land-use systems given the ever-increasing demand for tree products? Are these natural wood resources sufficient to meet the energy needs of both rural and urban households? Regional urban areas extend rapidly and may add pressure on the rural fuelwood resources, putting forest patches at great risk. Although the gathering of tops and branches may have been sufficient to meet local needs, the felling of whole trees may be inevitable to meet urban demands. Alternatively, fuelwood gatherers may turn, mostly illegally, to the more distant Sierra Madre forest to meet the fuelwood demands or establish fuelwood plantations as practiced in Cebu.

In this chapter, we examine the status—in terms of quality and quantity—and the role of forest patches in agricultural systems and evaluate the comparative effect of forest use and agricultural intensification on the conservation of forest patches in the Isabela province of the Cagayan Valley region in Northeast Luzon. Forest patches are defined as areas dominated by woody plants, including at least two adjacent trees or shrubs of 1 m or taller with touching, or close to touching, canopies and surrounded by grasses and herbaceous plants or cultivated crops (Snelder, 2001a). Moreover, in this chapter, we distinguish the following types of patches:

1. Isolated, relatively recent clumps of natural vegetation on slopes and watersheds that cover relatively small areas (e.g., up to 400 m^2)
2. Gallery forest patches that are larger in size (up to 0.5 km^2 or even more) and stretch out along creeks and cover parts of hillslopes at the intersections between creeks that join further downstream
3. Hillslope forest patches that cover even larger areas including watersheds or whole catchment areas

Homegarden conglomerates in and around villages and towns, a prominent category of man-made forest patches, fall outside the scope of this chapter (and will be discussed elsewhere) that is directed at natural (as opposed to planted or domesticated) forest vegetation.

17.2.1 PHYSICAL AND SOCIOECONOMIC ENVIRONMENT OF STUDY AREA

The forest patches discussed in this chapter occur in the hilly grassland zone stretching out between 50 and 250 (up to 450) m.a.s.l. mainly in the north–south direction, in the northeastern part of the Cagayan Valley. The hilly zone is located between the lowlands with intensive cash-crop cultivation in the West and the mountainous uplands with closed-canopy rain forest in the East. The latter includes the Northern Sierra Madre Natural Park Reserve, one of the 10 protected areas in the Philippines under the National Integrated Protected Areas Systems (NIPAS) Act (DENR, 1997).

The climate is classified as seasonally humid rain forest (A_m of the Köppen system), with a growing season of 7–8 months and a short dry season starting in December and ending in April (mean annual rainfall: 1500–2500 mm). Typhoons cross the area regularly, mostly at the onset of the rainy season in May or toward the end in October and November. Out of the 20 tropical cyclones passing the Philippines on average each year, one-third traverse Northeast Luzon with maximum wind speeds of 65–290 km per h and maximum 24 h rainfall of 140–818 mm (PAGASA, 2001). The forest patches cover soils developed over different types of sedimentary rock exposed, at various sites, along slopes. The sedimentary rocks are part of the Lubuagan and Cabagan Formations of Miocene and Pliocene age, respectively (Bureau of Mines and Geo-Sciences, 1982; Snelder, 2001a).

Northeast Luzon is one of the least populated regions in the country, with an average of 105 persons per km^2 according to the last population census made in 2000 (NSO, 2001, 2006). However, the population is unequally distributed throughout the region, with 5–30 persons per km^2 in the isolated forested mountains and coastal areas and 150–600 persons per km^2 in the cultivated lowlands along the Cagayan River. The annual population growth of 2.25% is lower than the national average of 2.36%, both recorded for the 1995–2000 period (NSO, 2001, 2006).

A mixture of ethnic groups, both migrants and autochthonous residents, inhabits the area. In the distant past, most lowland communities, particularly Ibanag, Gaddang, Itawi, and Yogad (De Jesus, 1980, 1982) lived in towns and villages on the banks of the Cagayan river where they were protected from flood events. Nowadays, settlements of mainly Ilocano and Ifugao migrants are also present in the hilly grasslands and increasingly scattered throughout forested uplands. According to the 1980 statistics (Cruz et al., 1988; Van den Top, 2003), about 1.13 million people (59% of the total regional population) lived in upland areas throughout Northeast Luzon. Since 1980, the number of people in upland areas must have increased substantially, given the rapid natural growth and the in-migration of landless farmers. No recent data on migration flows are, however, available.

The regional economy is concentrated on agriculture: for the period 2000–2005, 60% of the working population found employment in the agricultural sector on average, followed by 7% in the industrial sector, and 32% in the service sector (NEDA, 2006). The most common farming systems are irrigated rice cultivation, hybrid corn (and upland rice) cultivation, and *kaingin* farming (slash-and-burn agriculture with 2–3 years of crop cultivation and fallow periods varying between 8 years in forested areas and 1–4 years in areas with grass fallows). The former system is mainly practiced in the lowlands and the latter two in the hilly lowlands and uplands. In addition to crop cultivation, extensive cattle grazing is practiced on ranches in hilly lowlands. The cattle are mainly kept for meat production. The same is true for poultry and pigs, but these animals are kept in cages on compounds and farms.

The forest patches at the foot of the Northern Sierra Madre Mountains are classified under different land-use categories. All lands with slopes of 18% (10°12′) or greater are officially classified as public forestland. This land is under the jurisdiction of the Philippine government. The forests on hillslopes of >50% or above 1000 m.a.s.l. and forest patches along stream and rivers (up to a maximum width of 20 m along both river sides) are protected by law. Although the intention is to reserve all public forestland for nature parks or forestry purposes, substantial areas are deforested and converted into grasslands with patches of forest, banana plantations, and cultivated fields. Forest patches on slopes of 18%–50% fall in the production area category and are mostly located on rangeland (unmanaged grassland or brushland) or on grazing land (managed grasslands, ranches). The latter is covered by either a PLA or a Forest Land Grazing Lease Agreement (FLGLA). These agreements are issued by the DENR for a period of 25 years (renewable for another 25 years). Forest patches on nonpublic forestland are privately owned or fall in the land-use category "alienable and disposable" (A&D), that is, agricultural land that includes land to be titled (see also Table 17.1).

17.2.2 METHODS

The research is based on field measurements of natural forest patches in Isabela province (Figure 17.1) and supplemented with data from a series of interviews and secondary data from various institutions. Two study areas were selected to examine changes in land use during the last decades and determine their effect on forest patch quality and quantity. One area covers thinly populated grassland (31,700 ha with an estimated average of 80 persons per km^2) southeast of Cabagan town (39,500 inhabitants, 2000 census) and the other more densely populated grassland with cultivated fields (11,100 ha and 180 persons per km^2) just south of Ilagan (119,500 inhabitants). The soils in both areas are similar, mainly *Eutric Cambisols* and *Regosols*, with patches of *Ferric Luvisols* according to FAO Soil Classification (Snelder, 2001a) and developed in sedimentary rock formations (marine sandstone and tuffaceous fine sandstone and shales of Miocene–Pliocene age).

Data on local land use in 1950 and 1980 in the study areas near Ilagan and Cabagan were determined with the aid of aerial photographs (scale 1:40,000 and 1:60,000, respectively). In addition, administrative records and data on agriculture, land use, and wood-energy use for the 1990s and 2000s were collected at the regional, provincial, and local offices of DENR, DA, NSO, NEDA, and various municipalities.

The floristic composition of the most representative natural forest patches are presented in this chapter, the results of a broader forest patch study being presented elsewhere (Snelder, 1997; Brekelmans, 2000; Snelder, 2001b). The forest patches are located in the hilly lowlands in two small catchment areas (6–10 ha each) just southeast of Cabagan. The catchment areas are <2 km apart and have soils developed in the same sedimentary rock formation. Both are covered with grassland but they differ in management and grazing intensity: one area has been subject to light grazing and occasional burning and most of its forest patches cover areas of 51–100 m^2, whereas the other area has been intensely grazed and yearly burned since 1986, and most of the forest patches encountered here cover areas of only 0–50 m^2.

Species richness was determined for all woody patches in both catchments, with each patch serving as a sampling unit. A gallery forest near a creek along the lower boundary of the catchment was also investigated for comparative study purposes. In this forest, sample plots were used. The size of the plots (10 × 10 m each) was adjusted to those most frequently encountered among woody patches. Environmental characteristics of sampling sites were also recorded (for detailed method description see Snelder, 2001b).

Semistructured and open interviews were conducted among farm households (over 100) and key informants of fuelwood gatherers and middlemen to gather in-depth information about changes in land use, cultivation practices, farmers' motivation and decision-making, forest-patch utilization, fuelwood sources, extraction, and marketing. These data, and the data on species frequency for fuelwood bundles, were collected and summarized together with students (Brekelmans, 2000; Op de Laak, 2000; Van Rees, 2000) within the framework of the Cagayan Valley Programme on Environment and Development, a research and education program of the College of Forestry and Environmental Management (Isabela State University, Philippines) and the Institute of Environmental Sciences (Leiden University, the Netherlands).

In this chapter, a distinction is made between agricultural intensification and agricultural expansion. The former refers to land-use intensification through technology adjustments (including frequency of cropping, crop species diversity, high-yielding crop varieties, and use of mechanical traction on a given plot of land) and crop output. The latter is related to intensification by increasing the total area under cultivation (i.e., input of land).

17.3 FOREST PATCHES AND AGRICULTURAL EXPANSION

Changes in local land use can be illustrated by comparing the data derived from the analysis of aerial photographs taken from the study areas near Ilagan and Cabagan in 1950 and 1980 (Figure 17.2a and b). For the area south of Ilagan, the 1980 data show a slight increase (12%) in the area under grass. Grasslands (with woody patches) mainly extended at the expense of gallery and hillslope forests (i.e., 41% of the area under grass in 1980 was forest in 1950). The increase in the area under grass (and woody patches) was counterbalanced by the conversion of grassland into cultivated fields (i.e., 26% of area under grass in 1950) with annual cropping systems mainly based on corn, rice, and

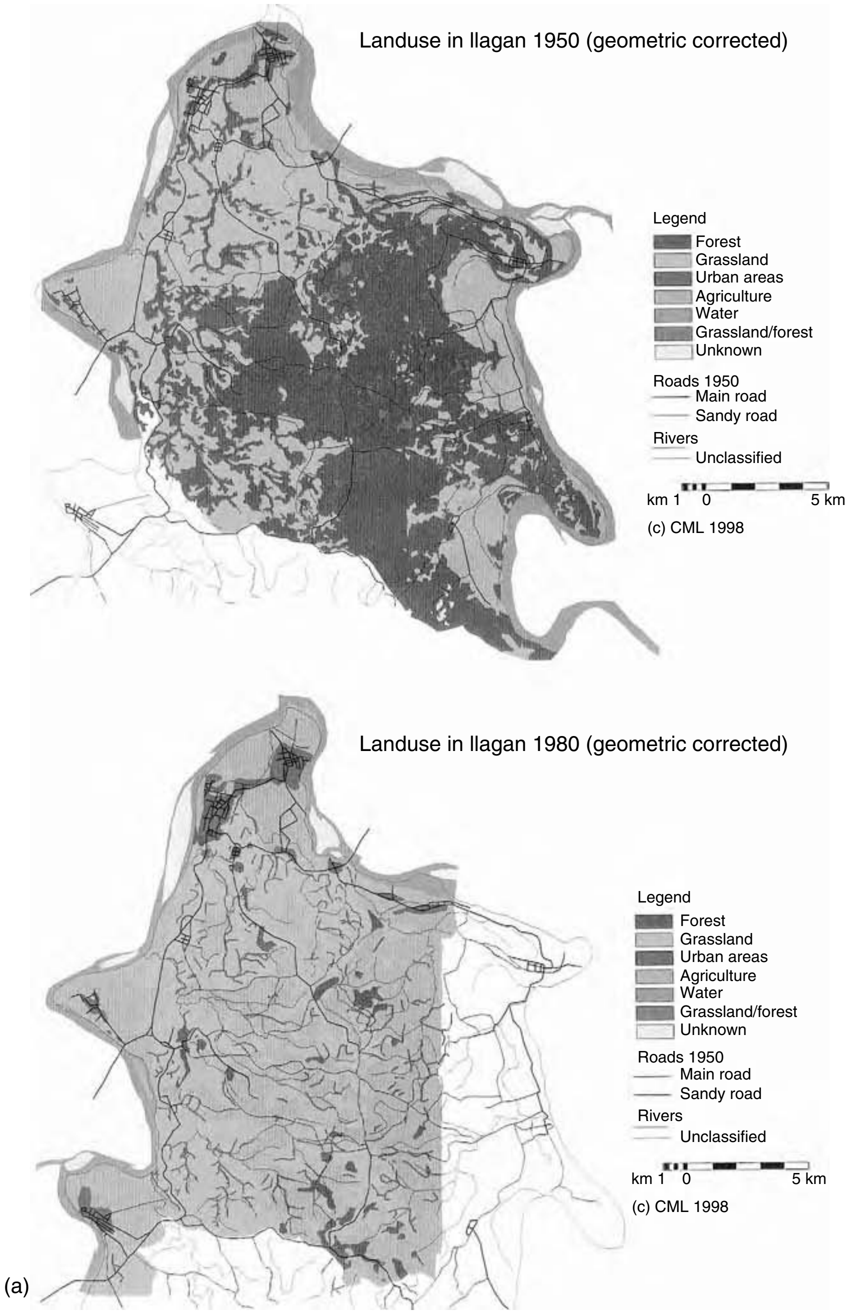

FIGURE 17.2 (a) Land-use change in Ilagan 1950–1980.

(continued)

legumes. The land-use changes in grassland near Cabagan were rather different, showing effects of reforestation. Between 1950 and 1980, the area under grass (and woody patches) diminished by 31%. Grasslands were converted to forests or a mixture of forest and grassland on the one hand and to cultivated fields on the other. The conversion of grasslands into forests did, however, not induce an overall increase in forest: forest cover declined by 30% because of the extension of cultivated

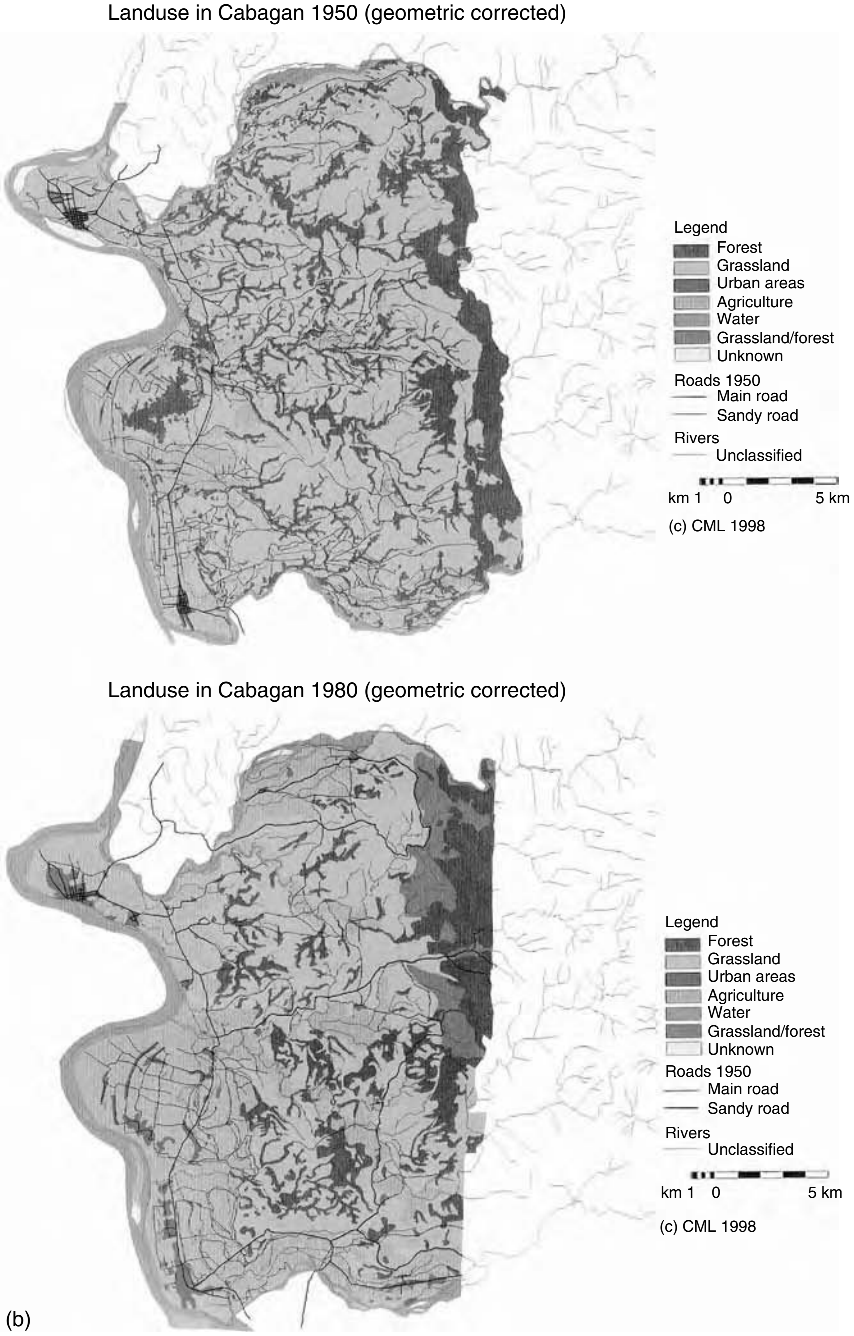

FIGURE 17.2 (continued) (b) Land-use change in Cabagan 1950–1980.

fields onto forested land. Forest decline in the more densely populated area south of Ilagan was, however, more alarming: 92% of the 1950 forest cover (composed of gallery and hillslope forests) was lost by 1980 because of both increased cultivation and conversion into grassland. The more recent data on land-use distribution for Isabela province as presented in Table 17.1 show similar trends. Moreover, in both areas, agriculture also extended in the 1980s covering an estimated 58% of the area in Ilagan and 43% in Cabagan by 1987 (NSO, 1992).

17.4 FOREST PATCHES AND AGRICULTURAL INTENSIFICATION

Although the increase in the area used for farms slowed down (Cagayan Valley) or stagnated (Isabela province) toward the year 2002, the number of farms continued to grow resulting in smaller areas per farm (Table 17.2). This observation corresponds with field reports of farm households reporting a lack of arable land. With increasing shortage of cultivable land, farmers are more and more forced to intensify their agricultural practices to produce sufficiently for household and marketing purposes. One sign of land-use intensification is the dwindling area of fallow land: although in 1980 still 27,597 ha of land was left idle in Isabela (Table 17.2), there was considerably less idle land in 2002.

The introduction of various new farm technologies over the past decades has also clearly affected farm properties and land-use intensification processes in Northeast Luzon. For example, in the late 1960s after the start of the Green Revolution, high-yielding rice varieties were introduced and spurred the construction of irrigation systems and the use of chemical fertilizers throughout the region in the following years (IBON, 2000). One of these systems is the National Irrigation Authority (NIA) Tumauini Irrigation System located in the Cabagan study site (Isabela province) discussed in the previous section. The scheme was constructed in 1974, and expanded in both the

TABLE 17.2

Changes in Selected Farm Characteristics between 1971 and 2002 for Isabela Province, Cagayan Valley, Philippines

	Isabela Province			
Farm Characteristics	**1971**	**1980**	**1991**	**2002**
Total number of farms	69,704	94,659	130,756	129,715
Number of farms (partly) owned	46,071	65,846	92,267	58,078
Number of farms leased or tenanted or other	23,633	28,813	45,024	6,424
Total area of farms (ha)	235,356	240,479	261,684	240,600
Average farm area (ha)	3	3	2	2
Average number of parcels per farm	n.r.	2	3	n.r.
Temporarily fallow (ha)	n.r.	27,597	846	2,322
Forest on farm holding (ha)	n.r.	4,136	764	724
Irrigated farm area (ha)	50,190	68,173	112,634	128,054
Area planted with rice (ha)	141,734	162,883	228,756	189,252
Number of farms planting rice	n.r.	59,188	83,464	72,807
Area planted with corn (ha)	66,876	65,957	161,133	177,846
Number of farms planting corn	n.r.	24,936	68,179	67,677
Number of farms using:				
High-yielding crop varieties	n.r.	40,569	96,566	n.r.
Chemical fertilizers[a]	n.r.	n.r.	104,686	1,344,607
Pesticides	n.r.	n.r.	108,626	n.r.
Irrigation pumps	n.r.	6,224	3,205	18,944
Number of hand and wheel tractors used	1,857	13,714	52,458	81,374
Number of plows used	74,715	96,745	116,220	95,701

Source: Based on NSO, *1990 Census of Agriculture, Cagayan Valley. Final report*, NSO, Manila, Philippines, 1994; NSO, *Census 2000*, http://www.census.gov.ph accessed on 14-09-2006.

Note: n.r., no records available.

[a] For 2002, the total number of farms using chemical fertilizers during January to June and during July to December for both irrigated and rain-fed farms; for 1991, no distinction was made between irrigated and rain-fed farms (see NSO, 1994), just the annual total number of farms using chemical fertilziers were listed.

north and the south directions in the late 1970s. The target service area of the irrigation scheme is 6600 ha (irrigated area: 2200 ha). Just north of this area is another irrigation scheme, that is, the Cabagan Irrigation Scheme, operational since 1985–1986 and covering a total area of 1273 ha (irrigated area: 652 ha).

Similar irrigation schemes were constructed throughout Northeast Luzon, leading to a substantial increase in the area used for irrigated crop cultivation (mainly rice). Table 17.2 shows changes in selected farm characteristics between 1960 and 2002, including increases of 256% and 155% for areas with irrigated farms in the Cagayan Valley and Isabela Province, respectively. The areas planted with rice show increases of 52% and 34%, respectively, but these figures include nonirrigated rice that covered about 25% of the total rice area in 1991. Instead of one rice harvest per year, rice is planted in most of these areas two times and in some lowland areas (near Roxas) three times per year. Average rice yields were 2,921 kg ha^{-1} for 1990 (ALMED, 1993) and 3,964 kg ha^{-1} for 2002 (total irrigated area: 394,857 ha; Bureau of Agricultural Statistics, 2006).

In the mid-1980s, high-yielding hybrid corn varieties were introduced in lowland areas and spread rapidly throughout the region. The total area planted with corn more than doubled between 1980 and 2002 (Table 17.2). Total corn yields increased from about 665,338 metric ton (or 1,606 kg ha^{-1}) in 1990 (ALMED, 1993) to 790,411 metric ton (3,204 kg ha^{-1}) in 2000 (Bureau of Agricultural Statistics, 2006). Nowadays, high-yielding yellow corn has replaced traditional white corn varieties as major crop in most nonirrigated agricultural fields.

The use of high-yielding rice and corn varieties requires adoption of modern technologies and intensive care, resulting in relatively high production costs associated with mechanical traction, chemical fertilizers, and irrigation fees. During the period 1970–1990, fertilizer consumption increased by an annual average of 11.5% nationwide. Domestic fertilizer sales were 1,447 metric ton in 1990, reached a maximum of 2,038 metric ton in 1997, and dropped to 1,864 metric ton in 1999. The import and national use of pesticides increased threefold from 4,725 metric ton in 1980 to 13,634 metric ton in 1992, with insecticides accounting for 29% of the total pesticide use (IBON, 2000). The 1991 census of agriculture in Northeast Luzon revealed that 66% of the total number of farms used high-yielding varieties, 71% used chemical fertilizers, 77% used pesticides, and 44% used irrigation pumps (Table 17.2). The use of tractors has grown dramatically over the years, from 508 in 1960 to 96,676 in 1991.

17.5 FOREST PATCHES: SPECIES RICHNESS AND MAJOR PRODUCTS

The vegetation studies show clear differences in species richness between gallery forests and woody patches. Various tree species that occur in the gallery forests are also present in the primary forests. These include *Diospyros pilosanthera*, *Ixora longistipula* (mayanman), *Antidesma pentandrum* (bignai-pogo), *Cynometra ramiflora* (balitbitan), *Litsea glutinosa*, *Semecarpus philippinensis*, and *Timonius* sp. The tree species in woody patches are typical for disturbed forests, forest edges, and woodland, such as, *Antidesma ilocanum* (arusip) and *Psidium guajava* (guava), *Mallotus philippinensis*, *Leea philippinensis* (kaliantan), *Trichospermum lanigerum*, and the shrub *Lantana camara*. Table 17.3 lists the most common tree and shrub species for the representative forest or woody patches in the lightly and intensely grazed areas and a gallery forest. The total number of woody species encountered in this study is lowest (25 in total) for the forest patches of the intensely grazed and burned catchment. Against expectation, the total number of species encountered in the gallery forest (51) is lower than that in the forest patches of the lightly grazed catchment (62). On the one hand, this is due to the smaller number of plots that were investigated in the gallery forest. On the other hand, the quadrat size used in gallery forest was adjusted to the ‘‘average’’ size of wood patches for comparative purposes and, hence, was not the optimal quadrat size for floristic descriptions in gallery forests. However, species–area–curve analysis was performed at this site, suggesting a total of 70 species for an optimal quadrat size of 3200 m^2. The high number of tree species for the small-sized woody patches is the result of the inclusion of both characteristic forest

TABLE 17.3

Common Tree and Shrub Species for Woody Patches on Sloping Grassland Subject to Light and Intensive Grazing and in a Protected Gallery Forest along a Creek near Cabagan in the Cagayan Valley, Philippines

Forest Patch	Total Number of Species	Woody Species
Woody patch in grassland with low grazing intensity ($n = 40$)	62	*Antidesma ilocanum* Merr. (93%; s and t)
		Blumea balsamifera (L.) DC. (59%; s)
		Casearia grewiaefolia Vent. (52%; t)
		Desmodium pulchellum (L.) Benth. (79%; s)
		Guioa koelreuteria (Blanco) Merr. (62%; t)
		Lantana camara L. (62%; s)
		Leea philippinensis Merr. (69%; t)
		Mallotus philippinensis Lamk. Müll. Arg. (59%; t)
		Psidium guajava L. (93%; t)
		Psychotria luzoniensis (Cham & Schlecht.) Rub. (83%; s)
		Trichospermum lanigerum (Merr.) Elm. (52%; t)
Woody patch in grassland with high grazing intensity ($n = 35$)	25	*Antidesma ilocanum* Merr. (96%; s and t)
		Desmodium pulchellum (L.) Benth. (61%; s)
		Psidium guajava L. (96%; t)
		Psychotria luzoniensis (Cham & Schlecht.) Rub. (71%; s)
Gallery forest ($n = 10$)	51	*Antidesma pentandrum* (Blanco) Merr. (80%; t)
		Aphananthe philippinensis Planch. (70%; t)
		Atalantia sp. (90%; s)
		Clerodendrum minahassae (Teijsm.) Q. Binn. (80%; s)
		Cynometra ramiflora L. (80%; t)
		Diospyros pilosanthera Blanco (100%; t)
		Guioa koelreuteria (Blanco) Merr. (70%; t)
		Ixora longistipula Merr. (90%; t)
		Leea philippinensis Merr. (60%; t)
		Litsea glutinosa (Lour.) C.B. Robinson (80%; t)
		Memecylon sp. (70%; t)
		Premna odorata Blanco (60%; t)
		Sapindus saponaria L. (60%; t)
		Semecarpus philippinensis Blanco (80%; t)
		Streblus asper Lour. (60%; t)
		Timonius sp. (Merr.) Elm. (80%; t)
		Triphasia trifolia (Burm. f.) P. Wilson (90%; s)

Source: From Snelder, D.J., *Agroforestry Syst*., 52, 207, 2001.

Note: Species with frequency >50% are listed. *n*, number of sample plots; t, tree; s, shrub or small tree.

and grassland (or forest edge) species. Species such as *L. camara* and *P. guajava* are typical for grassland and forest edges. They are both resistant to disturbance by cattle (by grazing and trampling) and fire. The seeds of *P. guajava* are spread by cattle that feed on its fruits, whereas *L. camara* invades at sites where light conditions get more favorable by canopy (Gentle and Duggin, 1997).

In addition to the forest patches and gallery forests, hillslope forests are present in both study areas. These forests are larger in size (up to 3 km^2) but similar in composition where they cover hillslope intersections between gallery forests. In other areas, most hillslope forests are planted or substantially enriched with timber species such as *Gmelina arborea* (yemane; paper tree), *Tectona grandis* (teak), *Pterocarpus indicus* (narra), *Swietenia* sp. (mahogany), *Acacia mangium*, *Eucalyptus* sp., *Samanea saman* (acacia), and *Leucaena leucocephala* (ipil-ipil). The species *G. arborea* is often found as single species in new plantations. It is a fast-growing timber tree promoted by the Department of Environment and Natural Resources.

The forest patches prove to be an important source of "free-access" products regularly tapped by low-income households during times of shortages and emergency needs. The major product extracted from forest patches is fuelwood. Tree species commonly used as firewood are *Antidesma ilocanum* (arusip), *A. pentandrum* (bignai-pogo), *M. philippinensis* (banato), and *D. pilosanthera* (bolong-eta). Furthermore, trees, preferably those with straight stems and branches, such as *D. pilosanthera*, are harvested for the construction of houses, bridges, and particularly fences around nearby ranches and to serve as posts. Guava and also the fruits of *A. pentandrum* (bignai-pogo) and *A. ilocanum* (arusip) are gathered for immediate consumption, supplementing local diets. Guava and the leaves of *Desmodium pulchellum* are also eaten by roaming cattle. A large number of species are of medicinal value, whereas different types of vines are used as fiber and rope. The forest patches further provide ant eggs, honey, fish, snails, shrimps, crabs, turtles, lizards, and birds that are caught for immediate consumption or sold along the national road. The shells of the snails are burned and used as lime to chew the nuts of the *Areca catechu* (betel palm). In some areas, it is believed that forest patches are protected by ghosts that live in trees such as *Ficus balete* (balete tree). The latter is a sacred tree and may not be cut.

17.6 FOREST PATCHES AND FUELWOOD EXPLOITATION

Gathering of fuelwood is the main activity in forest patches near Cabagan and Ilagan. The wood is gathered on a regular basis, whereas wood for timber, construction, fences, and posts is extracted only when orders are made.

17.6.1 Collection Methods

Wood for home consumption is mainly collected by women and children when, or after, working on nearby agricultural fields or attending cattle. Although in theory wood of any size serves as fuelwood, the size of collected wood is partly related to the type of transport. Wood transported at the back or on the head of people is not "too small and preferably straight," with a length of 75–150 cm and a diameter of 5–10 cm. Large and heavy pieces of wood are cut into appropriate sizes before transport. Wood transported by carabaos and jeeps show more variation in weight, curving, and dimensions. Professional wood gatherers, mainly men, supply wood for sale on local markets, at shops, and restaurants. They collect wood with a diameter of at least 7 cm and a length of 35 cm, that is, the size of the wood cuttings that are tied in bundles on the spot or, after transport, near their houses. Fuelwood bundles vary in size, with a diameter of 15–26 cm and a length of 35–80 cm. A standard bundle has an average diameter and length of 19 and 71 cm, respectively.

Most of the gathered fuelwood is dead plant material from fallen trees and branches. The availability of dead wood is highest during the typhoon season, from May to November. One typhoon passing the area is sufficient to significantly increase the quantity of dead wood, slowing down the fuelwood business for up to 1 year, as observed after the typhoons of 1997, 2003, and 2005. Fresh wood is used only when dead material is (temporarily) unavailable. Most fuelwood is gathered from forest patches on government land leased to individuals who prohibit the cutting of trees in accordance with the requirements of their lease agreement. However, control is difficult to exert, and some leaseholders sympathize with the needs of resource-poor farmers, allowing the cutting of tree branches. Fresh wood is usually left to dry on the spot for some time, after which it is lighter for transport. A machete and an axe are used for the gathering and cutting of wood.

17.6.2 SPECIES PREFERENCES

The interviews in this study reveal that gatherers have a clear preference for the type of wood to be used as fuelwood. Guava (*Psidium guajava*) is most often mentioned for its good burning quality: it gives "the strongest heat and burns longest" (Table 17.4). It is, however, a low-canopy tree not much affected by typhoons and relatively less readily available compared with tall fruit trees such as *Mangifera indica* (mango, mangga). Distance to wood source and species availability prove to be the controlling factors in decision-making for wood species in fuelwood-gathering activities. Of the 13 woody species most often encountered in the gallery forests near Cabagan, none was identified as preferred fuelwood species by the professional wood gatherers operating at those sites. Yet, the species frequency records for the wood bundles compiled by these gatherers corresponded to those made in the gallery forests (Table 17.5). Professional wood gatherers collect any species of wood that is readily available rather than spending extra time and energy in collecting preferred species. If available in small quantities, guava and other preferred species are just gathered for private use, whereas any species of wood serves as fuelwood for the local market, including *Antidesma ilocanum* (arusip), *A. pentandrum* (bignai-pogo), *Mangifera indica* (mango, mangga), *Macaranga grandiflora* (takip asin), *Mallotus philippinensis* (banato), *Melanolepis multiglandulosa*, *Nauclea orientalis* (bulala, mambog), *Nephelium philippinensis* (bulala), *S. saman* (raintree), *Diospyros pilosanthera* (bolong-eta), *Harpullia arborea* (uas), stoneapple, paper tree, acacia (check lapitling), and introduced species such as *L. leucocephala* (ipil-ipil) and *Gliricidia sepium*.

17.6.3 WOOD SOURCES AND MARKET SUPPLY

In the rural areas near Cabagan, with relatively low population density, fuelwood is gathered from different locations and types of forest patches. Fuelwood for private use is gathered mainly from forest patches on leased grassland (preferably close to villages and towns), trees along boundaries of cultivated fields, and homegardens. In addition, some private plantations are also used. Villagers living at a greater distance from major towns and adjacent to secondary forest gather fuelwood from both closed-canopy forest and forest patches. Table 17.6 gives an example of the

TABLE 17.4
Preference for a Specific Species to be Used as Fuelwood by Household Respondents in Isabela Province ($n = 30$)

Species	Common Name	Number of Times Reported
Psidium guajava	Guava	11
Macaranga grandiflora	Takip-asin	5
Samanea saman	Raintree	5
Leucaena leucocephala	Ipil-ipil	3
Mallotus philippinensis	Banato	3
Elaeocarpus parvilimbus	Lapitling	2
Antidesma ilocanum	Arosip	2
Nephelium philippinensis	Bulala	1
Syzygium cumini	Duhat	1
Ficus nota	Tuwig	1
Chrysophyllum cainito	Star apple	1
Gliricidia sepium	Madre de cacao	1
Acacia sp.	Acacia ausie	1
Any species		15

TABLE 17.5
Most Common Species Encountered in Fuelwood Bundles Gathered from the Forest Patches for Marketing in Isabela Province, Philippines

Species	Common Name	Number of Bundles[a] in Which Species Present
Leucaena leucocephala	Ipil-ipil	46
Macaranga bicolor	Hamindang	35
Psidium guajava	Guava	24
Canarium asperum	Pagsahingin	21
Macaranga grandiflora	Takip-asin	19
Streblus asper	Kalios	18
Gmelina arborea	Gmelina	17
Mallotus multiglandulosus	Alim	17
Mangifera altissima	Pahutan	15
No identification possible		29

[a] Standard bundles with diameter of 19 cm, length of 71 cm, and volume of 4294 cm^3.

different fuelwood sources tapped by households in two villages located in the municipality of Cabagan. Angancasilian is located close to ranches (i.e., leased grassland) with gallery forests as the major source of fuelwood. In addition to these forests, homegardens and a nearby sawmill provide fuelwood. More than a decade ago, sawmills and other wood-based businesses were very common in the area. They provided wood residues for both home consumption and local markets. After the logging moratorium in 1992, the wood-based businesses reduced considerably and, with it, the availability of leftover wood, increasing pressure on alternative wood sources. In the other village Guibang (Masipi West), households make less use of forest patches as a source of fuelwood. Both homegardens and cultivated trees serve as important sources. The village is located in the midst of cultivated fields, leased grassland, and some tree plantations. Forest patches are lower in number and areal extent and located at a greater distance (2–4 km) from the village than Angancasilian. Shortages of fuelwood are further covered by simply buying fuelwood (category ‘‘other’’).

TABLE 17.6
Different Sources for Fuelwood Used by Households in the Villages of Angancasilian and Guibang, Isabela Province, Philippines

	Number of Households	
Wood Source	Angancasilian	Guibang
Forest patches on ranch	21	4
Forest patches near village and homegardens	5	
Tree nursery and forest patches on ranch	7	
Sawmill wood residues and forest patches on ranch	5	
Homegardens		8
Corporate coconut plantation		2
Wood residues from furniture shop		2
Trees on private cultivated land outside village		2
Others	2	6
Total	40	24

Most fuelwood sold at the Cabagan market is gathered from woody patches and gallery forests at four locations near Angancasilian, Guingab, Dalena (municipality of Cabagan), and Ballacayu (municipality of San Pablo) and a few sites that fall under the Integrated Social Forestry Program.

Fuelwood gathered for private use mainly serves cooking purposes. For example, about 50% of the households in urban areas of Isabela Province use wood for cooking purposes, and this is >80% in rural areas and about 87% in the municipality in Cabagan (1991 data; NSO, 1994). About 10% and 2% of the households in urban and rural areas, respectively, use charcoal.

The fuelwood gatherers in this study stated, however, an overall decrease in the yearly amount of wood sold at local markets over the last two decades, although they experience fluctuations from year to year. For example, in the 1970s, fuelwood gatherers in Dalena just northeast of Cabagan, at any time, could easily sell wood in the Cabagan market. On average, 100 bundles of wood were supplied three times a week by carabao sled. Nowadays, wood is solely gathered upon order, and generally 300 bundles of wood are transported three times a month by carabao and jeep. Likewise, in the 1970s, fuelwood gatherers in Angancasilian could easily sell, in 1 day, 100 bundles of wood collected from areas at 1 h walking distance. In recent times, wood is collected from areas at 4–5 h distance and only 60 bundles a week are sold on average, and all of them are special orders. One major drawback in the fuelwood business is the increasing use of other fuel sources, particularly LPG. The main bakery in Cabagan, which used to buy in 1 week 10 carts with 100 wood bundles each, has started using gasoline.

In the rural areas near Ilagan, with a relatively high population density, the availability of fuelwood sources is rather limited. For example, most forests south of Ilagan (Gayong Gayong Sur, Surcoc) have turned into grassland or have been cut to lay out fields for cultivation, including a sugarcane plantation of a former hacienda (Solweta) established in 1950. At present, all wood, that is, mainly fuelwood both for private use and special orders, are collected on private land from homegardens, tree plantations, degraded woody patches, and trees along field boundaries and streams. Most professional wood gatherers have, however, stopped their activities since 1990 because of wood shortage and reduced demand. Although in the mid-1980s four bakeries in the major towns of Ilagan, Cauayan, and Cabanatuan were supplied with wood from Gayong Gayong Sur and Surcoc, nowadays two of the four bakeries have started using gasoline as their main source of fuel. Likewise, most restaurants along the tarmac road in Surcoc use gasoline these days. The use of gasoline in villages at long distances from tarmac roads is, however, still limited: >90% of the households depend on wood.

The fuelwood and charcoal business is increasingly driven by the yearly incidences of typhoons. After a typhoon has hit the area, large quantities of dead wood—varying from branches to complete trees—are temporarily available at particular spots in the landscape. Floating trees in the Cagayan river are pulled ashore during typhoons. Charcoal production sites spontaneously arise to process the readily available wood, and fuelwood bundles are constructed, partly stored, and sold at local markets. Wood prices go down with 1 peso per bundle (from 5 to 4 peso) and remain low for some time until wood becomes more scarce again. Minor fluctuations in fuelwood operations are caused by the alternating seasons, with the highest peak in the dry seasons when most collection sites are easily accessible. Moreover, farmers are less occupied with working in their fields and, hence, spend more time gathering wood to earn additional income.

17.7 DISCUSSION

The regional increase in agricultural land is partly attributed to the inflow of migrants. Three categories of migrants and arrival periods have been distinguished (Van den Top, 2003). Pioneer migrants from upland and lowland areas in North Luzon entered the Sierra Madre Mountain Region between 1950 and 1965, hoping to find unoccupied fertile land for agriculture. The arrival of large-scale, mechanized, and corporate logging companies in the late 1960s attracted skilled laborers, also from other regions, until 1990 when the companies started to close their gates. After 1990, migration was dominated by landless and jobless farmers and laborers pushed away from their area of origin because of the lack of livelihood opportunities.

Since the closure of various logging companies and the declaration of a moratorium on logging operations in 1992, various groups of landless laborers live as squatters in the hilly zone at the foot of the Sierra Madre Mountains. They have resorted to the cultivation of corn, and to a lesser extent banana, on hillslopes and irrigation of rice on flat terrain. They mainly occupy land officially covered by PLA, refraining from other areas because of the worsened peace-and-order situation in the forested uplands at times, the government embargo on cultivation of officially declared forestland, and the shortage of land in lowland areas.

The construction of irrigation channels, the enlargement of fields, and the lay out of new corn and rice fields have led to the removal of numerous natural forest patches. This is confirmed by the 1950 and 1980 data on land use near Cabagan and Ilagan and the data on woodland and forests presented in Table 17.1. The forest patches that remain are often botanically and ecologically poor: they consist of few pioneer tree species, are smaller in size, and are subject to unfavorable microclimatic conditions and soil erosion due to broken canopies and gaps. For example, the vegetation study near Cabagan reveals that 66% of the patches in the frequently burned and grazed grassland (with 25 woody species in total) covered areas of less than 50 m^2. In the lightly grazed and burned grassland of an adjacent catchment area with similar soils and substrates, most patches, i.e., 82% of the total number (with 62 woody species in toal) are more than 50 m^2 (Snelder, 2001b). They are also highest in organic matter content (organic C: $2.79\% \pm 0.69\%$) and, unlike young tree plantations, they survive relatively intense grassland fires. Woody patches south of Ilagan are small in size (<50 m^2) and nothing more than shrubby vegetation with low species richness (<25 tree species covering areas similar to near Cabagan) and very hard to penetrate because of prickly vines and bamboo. Some clear-cut parts are now covered with herbaceous vines and *Mimosaceae* species and newly planted trees (*Gmelina arborea*, *Eucalyptus* sp.; Brekelmans, 2000). Gallery and hillslope forests that are still in relatively good condition are favored by specific conditions: they are either located along steep slopes virtually inaccessible to humans and livestock or referred to as “sacred” land inhabited by ghost or protected by holders of lease agreements, or a combination of these.

Compensation of forest loss by tree-crop plantations is limited and is in sharp contrast with the efforts of the local furniture industry to find alternative timber production sites to continue their operation. Timber demands are mainly met through indiscriminate planting of *G. arborea* (Snelder, 2001b) and illegal logging operations. The planting of trees along field boundaries is often objected to because of reductions in yields of major cash crops. Land tenure insecurity is, at first sight, not a limiting factor in tree plantation. The 1991 agricultural census revealed that most farms are owned or partly owned (Table 17.2), a situation that has officially not changed much in the succeeding years. Since the implementation of the Comprehensive Agrarian Reform Program (CARP) in 1988, private landholdings in excess of 5 ha or more have been expropriated by the government for distribution among tenant farmers. However, the actual transfer of land ownership has not been a smooth process, as confirmed by recent field observations (Schuren, 2002). Resistance of landowners has caused much delay and, in various cases, the transfer is still being debated. Moreover, once turned over to tenants, the land ownership is not, per definition, of long duration. The emancipation of farmers from long-lasting feudal bondage proves to be a difficult process in which credit facilities are inadequate. For example, farmers turn to former landlords to obtain loans for the annual amortization payments and the purchase of inputs to cultivate their newly obtained land. Over the years, mounting debts increase farmers’ dependence, gradually expropriating their land and restoring the feudal system.

Smallholders nowadays dominate the agricultural sector. The average farm area has decreased since 1971 from 3.47 and 3.38 ha to 1.54 and 1.85 ha in 2002, for the Cagayan Valley and Isabela Province, respectively, which is primarily attributed to the land reform under PD70 and CARP (DA, 1998b). The smallholders reserve their farms for intensive cultivation of high-value, short-maturing cash crops rather than extensive, long-maturing tree plantations. Although forest patches have been established by spontaneous tree-planting activities in fields close to villages and towns, most efforts of reforestation have been directed at large-scale tree plantations. These plantations are of low species diversity and partly funded by international institutions like the World Bank (DENR, 1997) to combat future shortages of timber and fuelwood. Examples are found at a few privately owned sites close to major towns such as Ilagan where forest resources are scarce and at a number of government-sponsored project sites. Yet, the level of success of such plantation activities proves to vary substantially with project management schemes, site properties, and contractor characteristics (Pasicolan, 1996). Although most remaining upland forests are protected and not easily accessible, tree planting is not automatically perceived as an urgent activity in areas where mountains with closed-canopy forests fill up the distant horizon. In addition, fire contributes to substantial losses in newly established tree plantations every year (Masipiqueña et al., 2000).

Fuelwood collection is mainly directed at the gathering of dead wood—fresh wood being solely used when dead wood is not readily available—to meet daily household needs in rural areas. The wood is collected from forest patches close to villages and towns. These observations correspond to the statements made by Soussan et al. (1992). The closure of sawmills, and with it the loss of wood residues as a source of fuelwood and charcoal, made rural households even more dependent on forest patches for their daily fuelwood supply. However, the increasing shortage of wood and the increased walking distance to wood-gathering sites have affected the economic viability of fuelwood sales on local and urban markets, particularly where alternative fuel sources have become readily available. The present tendency of fuel diversification in urban areas has resulted in decreased wood demands for commercial purposes. Exceptions to this rule are the high demands for wood during fiesta time, when wood is used for the preparation of grilled meat, and during periods when prices for alternative fuels reached above-average values. Yet, most professional wood gatherers operating near Ilagan have stopped their activities since 1990 because the nearest, major wood sources disappeared. Another reason is the improved infrastructure linking remote towns with major marketing points, facilitating the transport of all sorts of products from, and also to, urban areas. The latter has resulted in all sorts of changes in lifestyle and living environment. An increasing number of rural households have changed from wood to gasoline as their main source of cooking fuel, the use of which is perceived as "easier, faster, and cleaner." More houses are made of concrete hollow blocks and corrugated metal sheets rather than local wood materials. People have become less dependent on wood and other forest products for daily life. The latter is not true for the low-income groups and people in isolated areas who still rely on free-access forest resources. For these groups, which still comprise about 43% of the total number of families in the region, homegardens and multipurpose tree plantations are likely to play an important role in the supply of all sorts of wood products in the near future.

The wood gatherers attribute the disappearance of woody vegetation at former gathering sites to the large-scale extension of (1) irrigated rice fields in the mid-1970s and (2) corn fields in the mid-1980s. Before the operation of major irrigation schemes such as the one near Tumauini established in 1976, rice was not cultivated on a large scale and was even imported from other provinces. The cash earned through wood gathering, often a full-time occupation, was used to buy rice and other products. With the establishment of irrigation schemes, woody patches in lowland areas were removed to create "wet" land for rice cultivation. These changes offered new livelihood opportunities. Full-time wood gatherers who had to turn to more distant fuelwood-gathering sites took up farming for at least part of their time. They could earn a higher income working as wage laborers and, at the same time, plant rice for home consumption and some cash. As a part-time wood gatherer stated "his present partial income from 1 ha of rice (with yearly two harvests) and 0.5 ha of corn is more or less comparable to his previous total income from full-time wood gathering."

The introduction of high-yielding hybrid corn varieties with significant market value in the mid-1980s further stimulated the extension of agricultural fields onto land with grass and tree covers. Moreover, it offered alternative livelihood to the wage laborers previously employed by logging companies. Yellow hybrid corn, almost exclusively used as raw material in the production of stock

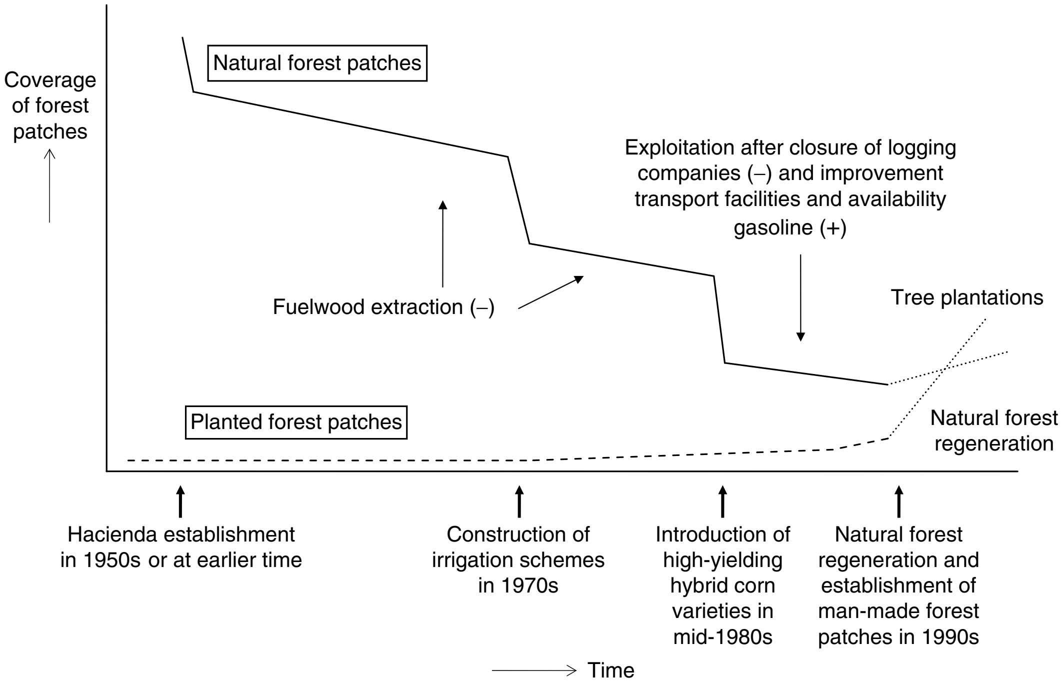

FIGURE 17.3 Schematic diagram of changes in forest-patch cover over time as related to human activities.

feed, has become a popular cash crop because of the flourishing fast-food business in Metro Manila and other urban areas. The high demand for chicken, eggs, and beef has stirred the poultry and livestock industry and led to a significant growth in stock feed production (Van den Top, 2003). Yellow corn accounts for about 70% of livestock mixed feeds produced throughout the Philippines (DA, 1998a). White corn traditionally serves as staple food and is still preferred as main staple by about 12 million Filipinos nationwide (DA, 1998a; personal communication). Northeast Luzon produced 21% of the total national corn production in 2000 (i.e., 952 metric ton corn).

Figure 17.3 summarizes in a schematic diagram the major impacts of fuelwood exploitation and agricultural events on forest-patch cover over time. Although fuelwood extraction has affected forest patches at a relatively low degree, with regular fluctuations due to alternating seasons and typhoon events, the effects of land clearing for irrigation schemes and intensive crop cultivation were sudden and led to irreversible changes. These land-use changes are not so much associated with a precondition of resource depletion but rather with specific conditions associated with the introduction of new technologies and market incentives.

17.8 PROSPECTS FOR FOREST-PATCH CONSERVATION

This study suggests that agricultural intensification leads to the removal of forest patches and the simplification of land-use systems in Northeast Luzon. The preservation of woody elements on grassland and cultivated fields clearly depends on the directions of future national and regional land-use strategies. The urgency of "keeping trees on the land" has been acknowledged by government institutions and acted upon in various forestry programs, however, with relatively limited success. Conservation highly depends on farmers' perceived value of forest patches and the possible integration of forest patches into local farming systems, without jeopardizing their ecological functions and services.

The management of natural forest patches may fit farmers' economic strategies in various ways. As opposed to tree plantations, it requires low labor and cash investment, whereas the patches show greater resistance to fire. The function of forest patches as providers of fuelwood and other multiple services and products with flexible harvesting times reduces risks and meets emergency needs; a benefit that needs, however, more attention.

The various types of forest patches demand a distinct management approach. The diverse gallery forests along streams demand a more conservative approach with a low level of exploitation and should be restored where fragmented by disturbance to form the foundation for a network of stable landscape elements. From a farmer's perspective, these forests may substantially increase in value if their function as reservoirs for natural enemies of agricultural pests can be proven. Likewise, forest patches conserve local populations of trees and are potential "stepping stones" for seed dispersal, both crucial functions for reforestation efforts. These and other ecological services need further investigation. Isolated woody patches can be linked and structured through enrichment planting and assisted natural regeneration, including indigenous multipurpose species, in such a way that they meet farmers' specific needs and form stable line elements across cultivated fields and grasslands, contributing to soil erosion control. Naturally vegetated strips on cultivated fields proved to be highly effective in soil conservation and are widely adopted as a low-labor and zero-cash-cost alternative by farmers in Claveria, Mindanao, Philippines (Garrity et al., 1999). Adequate credit facilities and transfer of knowledge among farmers and from farmers to (non)government institutions and vice versa are needed to encourage tree-planting activities and forest conservation. Reforestation programs directed at diversification of tree-planting activities including small-scale farm forestry are considered most promising, particularly among low-income groups. Some form of compensation for sharing the responsibility, and its associated costs, of the management and conservation of natural resources with local and regional government institution seems appropriate.

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18 Adoption, Profitability, Impacts, and Scaling up of Agroforestry Technologies in Southern African Countries

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CONTENTS

18.1 INTRODUCTION

Because of a mix of agroecological factors (incessant drought, low soil fertility, environmental degradation) and other man-made problems (illiteracy, unfavorable development policies), the southern African region faces several challenges including worsening poverty, food insecurity, low income base, and more recently HIV/AIDS pandemic. Low soil fertility is identified as one of the greatest biophysical constraints to increasing agricultural productivity (Bekunda et al., 1997; Sanchez, 1999). The degradation of soils is caused by a breakdown of the traditional production systems resulting from shortening of fallow periods due to population pressure (Kwesiga et al., 1999). With the collapse of the erstwhile government support for the use of mineral fertilizer (e.g., through subsidies and distribution channels), in the 1990s, the ability of most smallholder farmers to purchase the same level of mineral fertilizers was reduced because the input became unaffordable to them. In addition, many countries in southern Africa are landlocked, thus increasing the cost of transporting fertilizer from the ports. Howard and Mungoma (1996) estimated that the use of mineral fertilizer fell by 70% following an increase in the cost of the inputs. The subregion also faces a rapid degradation of the miombo woodland, shortage of fodder, and decreasing access to fuelwood supplies (Kwesiga and Beniest, 1998). For example, Chidumayo (1987) estimated that Zambia alone loses ~200,000 ha of forests per year. Some of the key avenues for overcoming food insecurity and rural poverty in southern Africa include reversing soil fertility depletion, intensifying and diversifying land use with introduction of high-value products, and facilitating an appropriate policy environment for the smallholder farming sector. Although mineral fertilizer is still one of the best options for overcoming land depletion and increasing food production, the majority of the smallholder farmers are unable to afford and apply the fertilizers at the recommended rates and at the appropriate time because of high cost and delivery delays (Kwesiga et al., 2003; Akinnifesi et al., 2006). Low-cost technologies are needed on a scale wide enough to improve the livelihood of these farmers. This will require the adoption of new approaches to agriculture and rural development. Agroforestry has proven to be one of such approaches. For the past 15 years, farmers and researchers from different national and international institutions led by the International Centre for Research in Agroforestry (ICRAF), otherwise known as the World Agroforestry Centre, have been combining their expertise and resources to develop agroforestry technologies and options to address some of these challenges facing smallholder agricultural production and the environment in the subregion. The different types of agroforestry technologies address specific human and environmental needs in southern Africa. These include *fertilizer tree systems* for replenishing soil fertility, *rotational woodlots* for solving fuelwood problems, *fodder banks* to supplement feed for livestock, and *indigenous fruit trees* for improving nutrition during the seasonal hunger periods and enhancing the preservation of indigenous plant genetic materials.

18.2 AGROFORESTRY TECHNOLOGY OPTIONS IN SOUTHERN AFRICA

The key agroforestry technologies that have been the focus of research and development efforts in the southern African region in the past 15 years are given below.

18.2.1 FERTILIZER TREE SYSTEMS

This system is one of the pioneer agroforestry technologies in the southern African region. Its development began in Zambia and it includes improved tree fallows (common in Zambia) and mixed intercropping technologies (popular in Malawi). The concept of intensifying land use for sustainable crop production by integrating nitrogen-fixing trees and crops for soil fertility replenishment requires careful selection of agroforestry technologies and judicious management of limited available resources. The expectations of households and their preferences were important considerations in designing technologies and choosing appropriate species. The mechanisms for improved soil fertility in fertilizer tree systems are explained by the capacity of certain leguminous trees to fix large amounts of nitrogen from the air through rhizobia contained in their root nodules and accumulate the fixed N together with the native soil nutrients they draw from different soil horizons in their roots, stems, branches, and leaves as they grow, and the nutrients accumulated in tree biomass during growth. The tree biomass is then cut and the biomass is incorporated into the soil during land preparation. When the tree biomass decomposes, it releases nutrients to crops grown in the subsequent 2–3 years without adding external fertilizer but relying simply on the residual effect of the increased soil fertility. Fertilizer tree systems do not produce a similar instantaneous effect on crop yields as mineral fertilizers; trees need time and resources on their own to be well established in the field. The plant species used in fertilizer tree systems to overcome soil fertility problems in southern Africa include improved fallows based on *Sesbania sesban, Tephrosia* spp., *Gliricidia sepium*, and *Cajanus cajan* and relay fallow cropping with short rotation shrub and tree species. Results showed that two year fallows with *Sesbania* can yield nitrogen biomass in the range of 70–100 kg ha^{-1} and can be applied as green manure. Field trials show that maize yields obtained from such fertilizer tree systems consistently reaches two or more times the yields from farmers' practice of continuous maize production without application of external mineral fertilizer inputs. Further details of fertilizer tree systems are described elsewhere (Mafongoya et al., 2003; Phiri et al., 2003).

18.2.2 BIOMASS TRANSFER

Farmers have been growing vegetables widely during the dry season in wetlands (known locally as *dambo*s) but declining soil fertility has posed a major challenge. Biomass transfer refers to cutting and carrying ("transferring") nutrient-rich leaves of agroforestry species (usually planted in the upland) to fertilize fields for the production of high-value vegetable crops and an extra maize crop in the dambos during the dry season. Biomass transfer offers smallholder farmers the opportunity to supplement their incomes by growing cash crops that fetch high prices in urban markets. In this system, nitrogen-fixing trees or shrubs are planted on a separate plot and the leaves are regularly cut and used to fertilize neighboring field plots in a cut-and-carry way, especially in the dambos. It simply involves transferring of leaves and twigs of fertilizer trees from one part of the farm to another. Farmers harvest trees planted at the upland to fertilize vegetables cultivated in the dambos during the dry season and use the coppices to fertilize their maize during the main season, thereby having two full crops in a year. In Eastern Zambia, *Gliricidia sepium* leaf mulches were used in combination with nitrogen fertilizers. In a given season, the responses to *G. sepium* leaf biomass were consistently higher than sole crop and mulch from other sources. It was estimated that yield of 3 ton ha^{-1} of maize could be achieved either through application of 52 kg ha^{-1} N or incorporation of 3.4 ton ha^{-1} (dry weight) or 15 ton ha^{-1} fresh weight of *Gliricidia* green manure.

18.2.3 INDIGENOUS FRUIT TREE CROP SYSTEM

Many miombo indigenous fruit trees are important for food and nutritional security, as well as a source of income for rural communities in southern Africa, with women and children being the main beneficiaries (Akinnifesi et al., 2004, 2006). It has been estimated that wild fruit trees represent ~20% of total woodland resource use by rural households in Zimbabwe (Campbell et al., 1997). Until recently, there has been little effort to cultivate, improve, or add value to these fruits. In complementing the earlier emphasis on soil fertility improvement, developing indigenous fruit and nut trees into tree crop systems continue to be an important strategy to reduce poverty and hunger and to create employment opportunities in rural areas (Akinnifesi et al., 2004, 2006). Domestication involves accelerated and human-induced evolution to bring species into wider cultivation through a farmer-driven and market-led process (ICRAF, 1997). The tree-domestication initiative aims at building on the desire of rural communities to cultivate indigenous fruits and nuts to meet their livelihood needs, especially food and nutritional security, increase household income, create employment, and diversify farming systems and the rural economy (Akinnifesi et al., 2006).

The domestication of indigenous fruit trees will increase their quality and productivity and can create opportunities for marketing their products, so empowering smallholder-farming communities to conserve and cultivate them. Tree crop development and commercialization of indigenous fruit trees from the miombo woodlands in southern Africa requires a long-term, iterative, and integrated strategy for tree selection and improvement, for the promotion, use, and marketing of selected germplasm and its integration into agroforestry practices (Akinnifesi et al., 2006). On the basis of household surveys to identify the important traits for improvement, the four priority indigenous fruit tree species that were identified in southern Africa are *Uapaca kirkiana*, *Strychnos cocculoides*, *Parinari curatellifolia*, and *Sclerocarya birrea*. More recently, the marketing and commercialization component of this program is receiving more emphasis. Rural entrepreneurs have been trained in fruit processing and business skills. The dissemination of these innovations have involved farmer-to-farmer exchanges where successful farmers pass on their skills and experience to new farmers entering the business, as well as formal courses to train trainers. This bottom-up approach has ensured community ownership of the implementation of the business and dissemination skills and a market driven tree-domestication initiative and promises to have a significant effect in raising rural incomes.

18.2.4 ROTATIONAL WOODLOTS

The problem of deforestation is high in the southern African region, particularly in intense tobacco-growing countries such as Tanzania and Mozambique where farmers require high quantities of fuelwood to cure the leaves. Rotational woodlots are meant primarily to provide high-quality wood biomass. Some of the woodlot species also helps to fertilize the soil and are therefore grown in rotation with food crops (Kwesiga et al., 2003). The main woodlot species that have been promoted in the subregion are Acacias especially *Acacia crassicarpa*, *Acacia polyacantha*, and *Acacia auriculiformis*.

18.2.5 FODDER BANKS

This involves the growing, harvesting, and preservation of browse of nutritious protein-rich leguminous tree leaves during the wet season and using them as protein supplement for ruminant animals during the dry season. Although commercial feed concentrate is available, smallholder farmers consider it expensive and many cannot afford it. The research and development of this agroforestry technology has been much more emphasized in Zimbabwe where livestock production is more predominant. This agroforestry technology helps to reduce the cost of formulated animal concentrate feeds for smallholder farmers.

18.3 ADOPTION OF AGROFORESTRY TECHNOLOGIES

18.3.1 FROM TECHNICAL FEASIBILITY TO FARMER ADOPTION

In the past one and half decades, the biophysical performance and the relevance of the agroforestry technologies in southern Africa have been well demonstrated (Kwesiga and Coe, 1994; Mafongoya et al., 2003; Kwesiga et al., 2003; Mithöfer and Waibel, 2003; Nyadzi et al., 2003; Kuntashula et al., 2004). As this chapter shows, gradually the focus of agroforestry research has changed from purely biophysical and field trials to the incorporation of socioeconomic and on-farm research to allow for studies of profitability and acceptability of the different agroforestry technologies to be carried out in a much more real-life context. Research and development activities on agroforestry have therefore expanded to include questions on farmer uptake, adoption, and impact of the technologies. Farmer adoption and the impact of new farm technologies on adopters (and nonadopters) are some of the key measures of the overall success or otherwise of such innovations.

In general, the uptake of agroforestry technologies is more complicated than of annual crops (Scherr and Müller, 1991; Mercer, 2004) because of the multicomponents and the multiyears through which testing, modification, and uptake of the technologies takes place. As a result, a precise definition of the "adoption" of agroforestry often poses a challenge. Some authors (e.g., Adesina et al., 2000; Franzel et al., 2002) distinguished between "testers," "experimenters," and "adopters." Other authors (e.g., Ajayi et al., 2003) regard the uptake of agroforestry technologies as a continuum and posit that farmers can be assigned positions in the continuum based on the extent of uptake of the different components of the technology. A recent study in Zambia (Ajayi, 2007) reveals that the key criteria that farmers use for assessing the level of "adoption" of agroforestry technologies are good management (timely weeding and pruning) of agroforestry fields, density and mix of trees species planted, number of years of continuous practice of agroforestry, and size of land area that a farmer cultivates to agroforestry. In a strict sense, therefore, different degrees of "adoption" of agroforestry technologies can be identified.

18.3.2 FACTORS AFFECTING THE ADOPTION OF AGROFORESTRY

Several empirical studies have been carried out to gain insights into the adoption of agroforestry in the southern African region. The specific studies investigated the types of farmers who adopt (do not adopt) agroforestry (Gladwin et al., 2002; Kuntashula et al., 2002; Phiri et al., 2004; Ajayi et al., 2006a). Other studies examined the factors that drive the adoption of agroforestry; why do some farmers continue to adopt more than others do (Franzel and Scherr, 2002; Place et al., 2002; Ajayi and Kwesiga, 2003; Ajayi et al., 2003; Thangata and Alavalapati, 2003; Keil et al., 2005; Ajayi, 2006; Jera et al., 2006).

Access to information on agroforestry, training opportunities, good quality seeds, property rights on land, size of available land, flexibility, and compatibility of agroforestry to existing farming systems among others are important factors affecting adoption of agroforestry (Place, 1995; Place and Dewees, 1999). The result of specific empirical studies to assess the factors influencing the adoption of agroforestry (fertility tree systems) in Zambia is summarized in Table 18.1. In general, the factors that influenced farmers' adoption decision about agroforestry technologies fall within four broad categories. These are those that exert (1) positive influence on farmers' adoption decisions, (2) negative impacts, (3) ambiguous or no direct effect, and (4) systemic influence on all types of households in a given community and spatial locations.

18.3.3 SOCIOANTHROPOLOGICAL PERSPECTIVE FOR UNDERSTANDING FARMERS' ADOPTION OF AGROFORESTRY

A number of surveys to investigate the actual and potential adoption of agroforestry technologies have focused primarily on the influence of different household and farm characteristics on the adoption by farmers. However, the inevitable implication that measuring the influence of household and farm characteristics in itself may provide insufficient explanations and thus there is need for different approaches. The process of adoption is complicated, dynamic and the various factors are likely to influence each other—hence they should not be treated in isolation, ignoring their mutual interdependencies and reducing the adoption decision to a zero-sum game, as is frequently done. If individual household and farm characteristics are singled out, where one study considers a certain characteristic to have a positive influence on adoption, another study may view the same characteristic as having a negative influence. The differences can sometimes very well be clarified from the institutional and social contexts of the specific respective study areas. Such qualitative research methodologies compliment quantitative research approaches, provide insights into farmers' adoption patterns, and improve the understanding of the process of adoption of agroforestry technologies from the perspective of farmers. The qualitative methodologies may enable the comprehension of the process of adoption on the basis of diversity as found among informants and generating the relevant variables in the course of interviewing and observation (see e.g., van Donge et al., 2001). This qualitative approach was used to study the history of interventions and the present-day consequences for agroforestry adoption in southern Malawi. Given the complex process of decision

TABLE 18.1
Factors Affecting Farmers' Decision to Adopt Fertilizer Tree Systems in Zambia

Study (and Number of Households Involved)	Wealth	Age	Sex	Education	Labor/ Household Size	Farm Size	Uncultivated Land	Use of Fertilizer	Off-farm Income	Oxen Ownership	Village Exposure to Improved Fallows
Factors affecting farmers' decision to plant fertilizer tree fallows for the first time											
Franzel (1999) (157 households)			N		N						
Phiri et al. (2004) (218 households)	+		N								+
Kuntashula et al. (2002) (218 households)	+	N		N		+	N		N	+	
Ajayi et al. (2006b) (305 households)			N		+, N	N		+			
Peterson et al. (1999) (320 households)	+					+				+	
Factors affecting farmers' decision to continue to plant fertilizer trees											
Keil (2001) (100 households)	+, −	N	N	N	+	+					
Place et al. (2002)		+	N	N	N	N					+

Note: +: positive association with planting improved fallows; −: negative association; N: no association; ±: positive or negative depending on the value; blank means the variable was not tested in the specific study.

making by farmers, an adjusted research methodology is necessary to gain a better understanding of the process of adoption, which needs to be contextualized, both within the socioeconomic context of the farm and family enterprise and in time.

18.4 FINANCIAL PROFITABILITY AND RETURNS TO INVESTMENT IN AGROFORESTRY TECHNOLOGIES

Profitability analyses that were carried out in the southern African region show that the various agroforestry technologies are profitable relative to conventional production practices where trees are not grown (Place et al., 2002; Franzel, 2004; Ajayi et al., 2006b). The results of a recent study in Zambia to assess the financial profitability of five soil fertility management technologies—*Sesbania sesban*, *Gliricidia sepium*, *Tephrosia vogelii*, continuous maize production with fertilizer, and continuous maize production without fertilizer—show that over a 5 year period, agroforestry-based soil fertility management technology (“fertilizer tree fallows”) are more profitable than farmers’ practices of continuous maize production without external inputs but, it is less profitable than full fertilizer application (Ajayi et al., 2006b). The 50% government subsidy on mineral fertilizer particularly enhanced its superior financial performance over agroforestry-based options. However, when valued at its market price, the magnitude of the differences in the profitability of agroforestry option and mineral fertilizer option decreases by 30%, and the net present value of fertilizer ($349) is very close to one of the agroforestry options (net present value (NPV) of $309). The mineral fertilizer option has a lower benefit cost ratio (BCR), implying that the higher net benefits obtained in mineral fertilizer option was achieved through a relatively higher investment cost.

Table 18.2 shows that that for every unit of money invested into maize production, the farmer gains an *extra* 1.65 units through mineral fertilizer option, an extra 1.91 units of money, if *Gliricidia* fallow option is used, an extra 2.13 units of money in *Sesbania sesban* fallow fields, an extra 1.74 units of money in *Tephrosia* fallow fields, and a 1.01 unit of money if farmers’ conventional maize production practice is followed. Due to the challenge of HIV/AIDS pandemic and its possible effect to degrade the quantity and quality of labor supply in farm households, it is hypothesized that the returns to labor will become an increasingly important factor in the acceptability of agricultural production technologies to farmers and the decision to adopt them in the future. Analysis shows that

TABLE 18.2

Profitability of Maize Production ha^{-1} Using Tree Fallows and *Subsidized* Fertilizer Options over a 5 Year Cycle in Zambia

Type of Production System	Description of Land-Use System	NPV (Zambian Kwacha)	NPV (US$)	BCR ($/$)
Continuous, no fertilizer	Continuous maize for 5 years	584,755	130	2.01
Continuous + fertilizer (subsidized at 50%)	Continuous maize for 5 years	2,243,341	499	2.65
Continuous + fertilizer (at nonsubsidized market price)	Continuous maize for 5 years	1,570,500	349	1.77
Gliricidia sepium	2 years of *Gliricidia* fallow followed by 3 years of crop	1,211,416	269	2.91
Sesbania sesban	2 years of *Sesbania* fallow followed by 3 years of crop	1,390,535	309	3.13
Tephrosia vogelii	2 years of *Tephrosia* fallow followed by 3 years of crop	1,048,901	233	2.77

Market price for fertilizer includes a 50% subsidy by the government.

Figures are on 1 ha basis, using prevailing costs and prices and an annual discount rate of 30%.

the returns to a person labor day are \$3.20 for mineral fertilizer option and \$2.50, \$2.40, and \$1.90, respectively, for the three agroforestry-based options that were investigated. By comparison, the return to labor for the unfertilized maize system was only \$1.10, whereas the daily agricultural wage is around \$0.50. Thus, although the recommended dose of fertilizer option is the highest performer at current subsidized rates, at the full economic cost, the tree fallow options are only slightly less economically attractive. In areas where transport costs of fertilizer are high, the tree fallow options may outperform the fertilizer option. Sensitivity analysis shows that different price and other policy scenarios affect the financial profitability of different production systems. In general, the prevailing price of the staple crop (maize), cost of capital (interest rate), cost of including subsidy on fertilizer, and the wage rate of labor are key determinants of the relative financial attractiveness and the potential adoptability of the production systems even when agronomic relationships between inputs and outputs remain the same.

18.4.1 Rotational Woodlots

The financial analysis carried out in Tanzania regarding rotational woodlots shows that despite higher costs and longer payoff, rotational woodlots generate an NPV of US\$388 ha^{-1}, which is six times higher than the net benefit obtained in conventional maize fallow systems (Franzel, 2004). Rotational woodlots consistently maintained its superior financial performance over conventional maize systems even when maize prices and labor cost changes up to 50%.

18.5 SCALING UP OF AGROFORESTRY TECHNOLOGIES

18.5.1 Approaches and Methods for Scaling up

Following the successful demonstration of the potential of agroforestry technologies to make positive impact on the livelihoods of smallholder farmers in southern Africa, various agroforestry research and development institutions have been focusing efforts in scaling up these technologies to reach a greater number of resource-poor smallholder farmers who could potentially benefit from the technologies. Scaling up is expected to bring more quality benefits to more people over a wider geographic area, more quickly, equitably, and lastingly. Because of the complexities of factors that affect scaling up, going to scale requires vertical and horizontal processes. The vertical process represents efforts to influence policy makers and donors and is generally institutional in nature. The horizontal process (also referred to as scaling out) refers to the spread across communities, institutions, and geographic boundaries (IIRR, 2000). Both processes characterize scaling up interventions of agroforestry. Agroforestry partners have focused efforts on a process of institutionalizing agroforestry in the research, extension, and development and education arenas to get policy makers, researchers, extension workers, development workers, educationalists, and farmers to forge their efforts jointly to address the factors that influence going to scale. At the policy level, each country has a National Agroforestry Steering Committee (NASCO) charged with the responsibility to facilitate the institutionalization of agroforestry in the relevant sectors. Specifically, the NASCOs' roles include identifying priority agroforestry research and development areas and guiding donor support accordingly.

Three major interrelated and mutually enforcing strategies employed in the scaling up of agroforestry technologies in southern Africa are capacity building, partnerships and networking, and promoting policies more conducive to adoption with the central focus being strengthening of local capacities to innovate as a way of ensuring sustainability of technological enhancement (Böhringer et al., 2003). Among the key interventions characterizing these strategies are the following: farmer-centered research and extension approaches, establishment of strategic partnerships, knowledge and information sharing, establishing viable seed systems, developing market options, local institutional capacity strengthening, diversification of agroforestry technologies, and influencing policy at different levels. In building farmer capacity and providing them with management and problem-solving skills through learning by experience in the field, a mixture of approaches are used to reach farmers and improve their lives through agroforestry. These approaches have been pursued within a framework of a scaling up concept initially comprising the following four prongs:

1. *Training of farmer trainers and local change teams:* This approach involves direct training of farmers as trainers with the ultimate goal being that the farmers trained will in turn provide training in agroforestry to fellow farmers in a given locality.
2. *Training of project partners:* This involves agroforestry research institutions making available training to the staff of development partners and NGO projects who work at the grassroots level. The major objective for this type of training is to enable partners to implement training for farmer trainers in their own project areas.
3. *Farmer-to-farmer exchange visits:* This approach involves exposing farmers to agroforestry by facilitating their visits to farmers in other locations who have been practicing agroforestry for some time and have started to get benefits from adoption of the technologies. As benefits accruing from agroforestry technologies take long, especially the soil fertility improvement options, exposure of farmers to benefits realized by those farmers who have adopted the technologies has proven to be a very effective way of promoting adoption.
4. *Support to national research and extension initiatives:* This involves support to existing government initiatives on sustainable farming, particularly extension work at the field level. One of the major challenges in implementing agroforestry has been underinvestment in the public research and extension systems, manifested in severe logistical as well as methodological limitations.

From 2004, other methodological approaches to scale up agroforestry have been developed. These include the use of existing local institutions (and consultants) to conduct training on agroforestry, providing technical and logistics support to agroforestry networks, the establishment or strengthening of school community links, and sensitizing policy makers about agroforestry benefits by producing policy briefs and use of public media channels and events (local radio, TV programs, documentaries, field days, agricultural shows, etc.). These policy shapers include parliamentarians, cabinet ministers, provincial and district administrators, and village councilors, traditional authorities that could help catalyze adoption of agroforestry or forestry in their respective constituencies.

18.5.2 NUMBER OF FARMERS REACHED THROUGH AGROFORESTRY TECHNOLOGIES

As a result of these scaling up efforts, the number of farmers who have been reached with different agroforestry technologies in the five southern African countries has increased from a few hundred farmers in the early 1990s to 417,000 farmers in 2005 (ZBAFP, 2005) (Table 18.3). Several factors contribute to the increases recorded in the number of farmers who have been reached through agroforestry technologies. First, it is the deliberate effort by several institutions to focus on the scaling up of the technologies using the different prongs described earlier. Several institutions that were interested in promoting natural resource management options provided added impetus to disseminate information on agroforestry innovations among farmers. Such institutions include the World Vision Integrated Agroforestry Project in Zambia (ZIAP), Soil Conservation and Agroforestry Extension (SCAFE) in Zambia, Malawi Agroforestry and Extension (MAFE), and the Eastern Province Development Women Association (EPDWA). These were complemented by interests in agroforestry technology through organizations such as Plan Zambia and Kehitysyhteistyon Palvelukeskus (KEPA), a Finnish-based Service Centre for Development Cooperation. In partnership with ICRAF, these institutions assisted in reaching a nucleus of farmers through direct

TABLE 18.3
Numbers of Farmers Reached through Different Agroforestry Technologies in Five Southern African Countries

	Methodological Approach Employed to Reach Farmers				
Country	Training of Farmer Trainers and Local Change Teams	Training of Partner Institutions	Support to National Research and Extension Initiatives	School–Community Linkages	Country Totals
Malawi	15,476	68,243	26,982	—	110,701
Mozambique	4,491	—	—	—	4,491
Tanzania	15,000	106,228	83,000	29,500	233,728
Zambia	15,387	37,838	8,358	—	61,583
Zimbabwe	—	—	—	—	7,000*
Prong totals	50,354	212,309	118,340	29,500	417,503

Source: *Zambezi Basin Agroforestry Project Annual Report 2004/2005*, Harare, Zimbabwe: International Centre for Research in Agroforestry (ICRAF), Southern Africa Regional Programme.

* The breakdown of the figure for Zimbabwe is not available.

training and provision of initial tree seed to farmers. The period coincided with the increasing emphasis by ICRAF on development programs aimed at accelerating the scaling up or scaling out agroforestry technologies trees among farmers in the subregion. Second, in the development of agroforestry technologies in the southern African region, a constructivist approach was actively encouraged, that is, farmers were encouraged to try the technologies, then modify and readapt them based on their experiences and desires to make them more acceptable to their circumstances. Third, some private-sector organizations found a niche in agroforestry to fulfill their goal for a responsible corporate citizenship by being responsive to the environmental and natural resource implications of their activities. Among these are tobacco companies who are training their contract farmers on the use of poles from fertilizer tree species to make sheds for curing tobacco to avoid further deforestation associated with tobacco curing operations.

18.5.3 CONSTRAINTS TO THE SCALING UP OF AGROFORESTRY

A recent global review of the adoption of agroforestry shows that the level of diffusion of agroforestry technologies has generally lagged behind scientific and technological advances attained in such technologies, thereby reducing their potential impacts (Mercer, 2004). The experience with regards to the adoption of agroforestry technologies in southern Africa has not been too different from the global trend. Although agroforestry is financially profitable and there has been an increasing trend in the uptake of the technologies by farmers, the widespread adoption of agroforestry technologies by many more smallholder farmers is nonetheless constrained by several challenges such as local customs, institutions, and policies at the national level. Some of the constraints are highlighted below.

Local and national policies: Some local customary practices and institutions prevailing in the subregion (especially incidence of bush fires and browsing by livestock during the dry season, and absence of perennial private right over land) limits the widespread uptake of some agroforestry technologies. The animals destroy the trees after planting either by browsing the leaves and removing the biomass or by physically trampling over the plants. Community's institutional regulations for fruit collection, land and tree tenure all affect individual farmer's decision to invest in establishing an indigenous fruit tree orchard. However, agroforestry institutions have been working in collaboration with traditional rulers, government officials, community-based organizations, NGOs, and national partners to resolve these institutional bottlenecks (Ajayi and Kwesiga, 2003).

Training: Agroforestry technologies are generally incipient technologies and relatively new phenomena compared with conventional agricultural practices that farmers have known, been used to, and received training for a much longer period. Unlike annual crop production technologies and conventional soil fertility management options, fertilizer tree systems require skills in terms of management of the trees. Capacity for doing this needs to be built at the national level. The costs of providing information greatly decrease over time, but they are critical when helping farmers get started with the practice.

Seed and germplasm: One of the greatest constraints of some agroforestry technologies is the lack of access to quality seeds. Unlike the seeds of annual crops in which established institutions exist to promote them and private sector organizations have been engaged in their multiplication and distribution, there is little or no institutional structure to make the seeds of agroforestry available "off the shelf."

Awareness: Over several years, there have been structural shifts toward "quick fixes" and technologies that render immediate benefits. The opportunity of agroforestry technologies to provide some medium and long-term benefits to individuals and the public simultaneously is not as yet well communicated to many stakeholders.

Human resource capacity: The human capacity, infrastructures, and institutional support for agroforestry are not as well developed as for annual crop technologies. Such missing support include well-developed input and output market to enhance access of smallholder farmers to ensure that they get the price premium for their crop produce.

18.6 IMPACT OF AGROFORESTRY TECHNOLOGIES

Agroforestry technologies have multiple impacts on both adopters and nonadopters, on food security, and on the environment as presented in Table 18.4. The impacts of agroforestry that are most pronounced are on improved soil fertility (hence food security), more income, and increased firewood supply. Some of the costs and benefits identified in Table 18.4 occur simultaneously in the same agroforestry field depending on the type of tree planted. Table 18.4 is based on the several studies that were carried out in the region coupled with personal observations of the authors over several years of research and development work on agroforestry in the region. Some of the costs and benefits require a more rigorous study to quantify their economic value. Despite the existence of negative spillovers of some aspects of agroforestry (e.g., incidence of fire and grazing), we expect the overall effect of the technologies to be positive.

18.6.1 IMPACTS ON YIELD AND FOOD SECURITY

Fertilizer tree systems increase the yield of maize (the staple food crop in the region) by two or more times compared with the usual smallholder farmers' practice of continuous maize without nutrient inputs (Kwesiga et al., 2003; Akinnifesi et al., 2006). One way to assess this impact is in terms of food security—by determining the number of days of additional food they provide to a household. Using the present average, fallow plot area is of 0.20 ha, and the per capita maize consumption the systems generate between 57 and 114 extra person days of maize consumption per year (Ajayi et al., 2006b). The total monetary value of the nitrogen fixed by fertilizer trees in the region is estimated at US\$5.7 million per annum. This is however assuming that all 150 kg N would be available to the maize (during the following production seasons), but it is likely that some will be lost due to leaching or gaseous emissions, thus reducing this figure accordingly. Depending on distance and condition of the roads, the cost of transportation of fertilizer bags from the shops in the major town or cities to farmers' village ranges between 10% and 25% of the purchase cost of fertilizer.

TABLE 18.4
Types of Impacts of Agroforestry Technologies in Southern Africa

	Individual	Public at Large
Cost	• Land • Labor • Tree seeds and nursery establishment • Pest control (some tree fallow species only) • Working equipments • Risk of uncontrolled fire outbreak	• Incidence of *Mesoplatys* beetle pest (restricted to specific species only) • Reduction of free grazing area during dry season • Risk of uncontrolled fire outbreak
Benefit	• Yield increase of subsequent crops • Opportunity for farm diversification (e.g., compatible with fish farming and growing of high-value vegetables) • Increase in fodder and maize stubble (for livestock) • Fuelwood available in field, and so reduces time spent searching for wood • Use of tree leaves (*Tephrosia vogelii)* as "pesticides" to remove ticks from livestock • Suppresses the growth of weeds • Potential to mitigate the effects of drought during maize season • Stakes for tobacco curing	• Carbon sequestration • Suppression of weeds • Improved soil infiltration and reduced runoff • Enhanced biodiversity • Serves as wind breaks • More fuelwood available to reduce deforestation

Source: Adapted from Ajayi, O.C. and P. Matakala. Food production and environment protection in developing countries: Bridging the policy disconnect. Paper presented at *International Workshop—Rural Development, the Roles of Food, Water and Biomass: Opportunities and Challenges*, held in Dakar, Senegal, 14–16 November 2005. ENDA Senegal and Plant Research International, the Netherlands.

18.6.2 IMPACT ON THE ENVIRONMENT

In addition to increase in food production, agroforestry has positive impacts on the conservation of the natural resource base and the protection of the environment. For example, the fertilizer tree systems can provide up to 10 tons of wood biomass ha^{-1} (Kwesiga and Coe, 1994). This greatly reduces the burden of carrying firewood over long distance and the time spent searching for wood energy (especially by women). Agroforestry provides "live barns" and alternative source of stakes for curing tobacco and thus has great potential to contribute to the reduction of the deforestation of the miombo woodland. Deforestation is an important economic problem in the southern African region. In Zambia alone, it is estimated that the miombo is being deforested for development purposes at a rapid rate of between 200,000 and 300,000 ha $year^{-1}$. To the extent to which farmers are able to source for fuel and other wood requirements for their households from improved fallow fields, cutting of wood from communally owned forests and hence deforestation may be reduced. A recent study in southern Africa shows that carbon stored in agroforestry fields varied between 2.5 and 3.6 tons ha^{-1} $year^{-1}$ (Paramu Mafongoya, 2005, personal communication). The trees have overall net positive impact on the soil invertebrates and perform important ecosystem functions that can affect plant growth (Sileshi and Mafongoya, 2006). In addition to improving soil fertility and food security, some agroforestry technologies improve the physical properties of the soils. This is because in agroforestry fields, the soil aggregation is higher and this enhances water infiltration and water-holding capacity (Phiri et al., 2003). The improvement in soil aggregation and water-holding capacity could contribute to minimizing the risk of productivity loss during drought years. Some fertilizer tree system species (e.g., *Sesbania sesban*) reduce the incidence of noxious weeds (*Striga* *hermonthica*), which generally thrive under conditions of low soil fertility in the region (Kwesiga et al., 1999). This aspect of fertilizer trees provides a big incentive for women who are often responsible for weeding the family's fields.

18.6.3 IMPACT ON INCOME GENERATION AND DIVERSIFICATION OF LIVELIHOOD

Several studies indicate that rural communities can increase their incomes by utilizing and marketing tree products from forests and horticultural tree crops grown on-farm (Campbell et al., 1997; Akinnifesi et al., 2006). Indigenous fruits contribute on average ~42% of the natural food basket that rural households rely on in southern Africa (Campbell et al., 1997). In 2002, a household food security survey found that ~60%–85% of rural households in the "Chinyanja Triangle" (i.e., Malawi, Zambia, and Mozambique), lacked access to food for three to 4 months per year, and 26%–50% of the respondents had relied on indigenous fruits for sustenance during this critical period (Akinnifesi et al., 2004). In Malawi, wild fruits are most important in the areas where rural people frequently face annual food shortages and the deforestation rate is high. The home consumption and marketing of these fruits contributed substantially to household livelihoods and cash income, and enabled households to live above the poverty line during the critical famine periods. An ex ante analysis in Zimbabwe showed that indigenous fruits contributed to a poverty reduction of up to 30% and an income above the poverty line throughout the year, and households marketing fresh fruits have been able to maintain income flows above the poverty line throughout the year (Mithöfer et al., 2006). The benefits from selling indigenous fruits come at a critical time when income is generally low and provides nutrition and food when agricultural labor demands are high.

Feasibility assessments and business plans have been completed for fruit enterprises in four countries (Malawi, Zambia, Zimbabwe, and Tanzania). These efforts, which have produced profits of 15%–28% for fruit concentrate, have provided women groups with a 40% of the market share. As a result of simple production processes, low capital investment, and low fixed costs, an average internal rate of return of 34% was attainable on inputs (Jordaan et al., 2007). In Tanzania, six women groups who received training and supports had increased their average income from less than US$ 200 to US$ 911 person^{-1} year^{-1} from processed fruit jam, juice, and wine. Rural communities in Zimbabwe, Zambia, and Tanzania were able to move from being "price takers" to "price givers," by increasing their income through processing and sale of indigenous fruits. These pilot activities are now at take-off point and have provided valuable lessons as a strong platform for scaling up to other tree products and areas.

18.6.4 OTHER IMPACTS

Biomass transfer technology enables farmers to grow high-value crops (ginger, garlic, cabbage, and onion) in the wetlands (dambos), thereby offering the opportunity to enhance food production and diversify farmers' income. Crop production takes place during off-season when farm produce attracts higher prices. The farm enterprise using biomass transfer is highly profitable and net profit ranges between $700 and $1000 ha^{-1} compared with $366 for control plots (Kuntashula et al., 2004). In Zimbabwe, farmers who adopt fertilizer tree fallows (*Sesbania sesban*) obtained an increase in cash which they use for discretionary spending (Muhdara et al., 2003). In Zambia, agroforestry helps farmers to purchase necessary daily commodities and clothing and use the additional money earned from increased crop yield to pay their children's school fees, attend to health care for their family members, and buy higher value food like meat and fish (Keil, 2001; Schüller et al., 2005).

The development and promotion of some agroforestry technologies, such as fertilizer tree systems, resulted in several unintended extra benefits (positive externalities) to the farmer and the society on one hand and some unintended problems (negative externalities) on the other. The negative externalities include *Mesoplatys* beetles attack on *Sesbania sesban*. Other social and institutional problems are browsing of fertilizer tree fallow species by free ranging animals and the incidence of bush fires destroying fertilizer tree fallow species especially during the dry season. In some cases, these incidents cause unintended social problems resulting from a conflict of economic interests among different sections of the community (Ajayi, 2001; Ajayi and Kwesiga, 2003). Collaborative efforts by traditional chiefs, village headmen, farmers, and research and development organizations and policy dialogues between the different stakeholders have resulted in various approaches to try and find ways of dealing with the problem of livestock browsing and fire.

18.7 DISCUSSION AND WAY FORWARD

This chapter describes the adoption, profitability, and impacts of agroforestry technologies with special reference to the southern African region. In doing this, a large amount of knowledge has been generated. A number of important lessons have also been learnt.

The first lesson is that although the technical characteristics of agroforestry technologies are important, they are neither the exclusive nor the sufficient conditions to guarantee their adoption by smallholder farmers. Therefore, in addition to technological fixes, the adoption of agroforestry can be increased considerably by also focusing on understanding and influencing the processes of innovation, intervention, and policy. Agroforestry technologies should therefore also be socially situated, not only in biophysical domains.

Second, adoption of agroforestry is not a simple direct relationship based on technology and farmers' characteristics alone, but it is a mix of several factors. These include household-specific factors (e.g., age, education), technological factors (e.g., "waiting period" before farmers obtain benefits, quantity, and distribution of labor inputs requirements to manage the technology), institutional and policy factors (e.g., land tenure system, agricultural subsidies, incidence of fire, and grazing), and geographical factors (e.g., type and characteristics of soils which determine the biophysical limits of technologies, access to roads and markets, location of a village relative to institutions promoting agroforestry).

Third, because of its long-term nature, the adoption of agroforestry may not take place in a policy vacuum. It is necessary that adoption of agroforestry be facilitated by a conducive policy and institutional framework at both local and national level (Haggblade et al., 2004). Recent profitability analysis (Ajayi et al., 2006a) shows that different "external" factors affect the financial attractiveness and potential adoptability of fertilizer tree fallows and other soil fertility management options even when technical relationships (e.g., yield coefficients) between inputs and outputs remain constant. Most households do not have direct control over the major factors that determine profitability and potential adoptability of soil technologies and therefore appropriate policies and institutions that are contributing to scaling up adoption of fertilizer tree fallows should be facilitated.

Fourth, the pattern of distribution of benefits (or costs) of agroforestry technologies among various sectors of a community are important factors that enhance (or inhibit) their widespread adoption (Ajayi and Kwesiga, 2003). Issues related to property rights are important in farm communities. As a result, national policies need be complemented by institutional support at the local level to reduce current constraints of property rights and other institutional constraints affecting the adoption of agroforestry.

Fifth, inadequacy of tree seeds, seedlings, and other planting material has repeatedly been identified as one of the most important constraints to the greater adoption of agroforestry. Successful scaling up is based on sustainable supply of germplasm of high physiological and genetic quality for a wide range of agroforestry species that can meet the needs and priorities of small-scale farmers. Adequate institutional and organizational mechanisms that are conducive to large-scale production and distribution of agroforestry seed are also needed. ICRAF and other organizations that support agroforestry activities have been supplying large quantities of free seeds to farmers. Although it is appreciated that free seeds should be part of the dissemination process in the initial stages of the program, continued free tree seed supplies make it difficult to determine the effective demand for agroforestry seed and undermine the establishment of a sustainable seed system

(Mitti et al., 2004). A more sustainable seed and germplasm supply for smallholder farmers should be developed.

Sixth, due to a mix of groups of variables that influence farmers adoption decision of agroforestry, a single cross-sectional data based exclusively on either of the groups of factors alone will most likely produce incomplete results at the best or even misleading results in some cases. As a result, beyond conventional adoption studies that are based exclusively on household surveys, more comprehensive georeferenced studies that integrate variables from the different groups of factors identified earlier will provide more accurate insights into the adoption of agroforestry in the subregion. An example of such studies is an ongoing multidisciplinary survey presently being implemented concurrently in five countries in the subregion (Malawi, Mozambique, Zambia, Zimbabwe, and Tanzania) to evaluate the influence of household-specific factors, community factors, and project-based scaling up activities on farmer adoption of agroforestry and to estimate the *number* and *proportion* of households that are ‘‘testers’’ or ‘‘adopters’’ of key agroforestry technologies.

Seventh, a number of the studies on the adoption of agroforestry technologies have often been limited by lack of common methodologies. This has rendered cross-site comparisons of the results difficult or sometimes impossible. Detailed characterization (much more comprehensive than hitherto available) of the causes, nature, and severity and extent of deforestation, soil fertility, and food security problems are required to provide information for diagnosis of the problems and evaluate opportunities through agroforestry options for solving them across temporal and spatial scales. Such detailed information will serve as a valuable resource for making informed decisions for targeting appropriate agroforestry technologies to suitable geographical locations and recommending appropriate policies for ensuring impact on the problems the subregion faces.

Eighth, although most agroforestry technologies are profitable over time (i.e., they record positive net present values), one constraint is that a number of such technologies attain break-even point ~2–3 years after initial investment. This implies that farmers must commit to initial investment in terms of land, capital, or labor and must absorb net losses for a couple of years before receiving profits from adoption. In low-income countries where smallholder farmers have a low capital and savings base, significant levels of adoption of the technologies may be limited to farm households who are sufficiently well-off to withstand these net losses. Other types of households may require targeted changes in policy and institutional framework that helps them cope through the ‘‘waiting period’’ for them to achieve a significant increase in the level of adoption.

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19 Pasture Production and Tree Growth in Agroforestry Systems of Northwest Spain

M.P. González-Hernández and M.J. Rozados Lorenzo

CONTENTS

19.1 INTRODUCTION

Agroforestry (AF) systems have been demonstrated as potentially more profitable than a combination of pure agricultural and forest plots (Dupraz et al., 1996). Although forest grazing has gained wide acceptance in the Mediterranean countries, a more recent approach has been to introduce trees to nonwooded grazing areas. The reasons to promote "trees on pastures" are varied and many: to maintain a fodder resource on areas where landowners would otherwise have planted forests and excluded rearing activities, to diversify farm incomes, provide shade and shelter to animals, and shift grass production into summer with the help of shade in dry climates (Dupraz and Newman, 1997).

Perhaps one of the most compelling reasons for including trees on farmlands that are prone to erosion hazards is the role of trees in soil conservation and erosion control (Nair, 1987). The importance of AF in soil conservation in its broader sense, meaning conservation of fertility and prevention of erosion, has been studied and discussed by several authors (Wiersum, 1984; Lundgren and Nair, 1985; Young, 1985, 1986). Nair (1987) reported that the inclusion of trees and woody perennials on farmlands can, in the long run, result in marked improvements in the physical conditions of the soil including: permeability, water-holding capacity, aggregate stability, and soil temperature regimes. Although the impact of water use by trees on water availability to crop plants, in different climatic conditions, is not yet very fully understood, there seems to be evidence that the hydrological characteristics of catchment areas are favorably influenced by the presence of trees (Nair, 1987).

Three centuries ago, many European landscapes were still agroforests, mainly because high-value trees from the original forests were kept when the land was cleared for cultivation (Dupraz and Newman, 1997). Some examples include oaks (*Quercus* spp.) for acorns, beeches (*Fagus* spp.) for mast, and ashes (*Fraxinus* spp.) for fodder.

In their review of surviving European AF techniques, Dupraz and Newman (1997) reported some remnants of these old practices, indicating that most of them occur in Mediterranean zones. In the Iberian Peninsula, the *dehesa* is still the largest agroforest in Europe, covering an area of 2 million hectares with widely spaced oak trees and fodder intercrops. This particular AF system has been broadly studied in Spain (Montoya, 1980; Gómez Gutiérrez, 1991; San Miguel, 1994).

The intensification, specialization, and mechanization of agriculture ultimately became key factors in the elimination of trees. Eventually, this led to some agricultural land in Europe laying abandoned, resulting in increased fire risk due to the accumulation of fuel biomass. Silvopastoral systems, or the production of good quality timber and pasture in combination with compatible livestock production, have been reported to be a way to prevent fires and increase income for the landowners (Silva-Pando and González-Hernández, 1992; Rigueiro et al., 1999).

Research on the development of techniques to produce high-quality hardwood timber while maintaining grazing animal production, began in some European countries such as Great Britain in the early 1980s (Dupraz and Newman, 1997) and France in 1988 (Montard et al., 1999). The need for providing more experimental data has been an incentive for ongoing silvopastoral research in Europe (Etienne, 1996; Koukoura and Papanastasis, 1996; Sibbald, 1996; Montard et al., 1999) and, more recently, studies have begun to be published on herbage production in association with broadleaved trees (Montard et al., 1999; Balandier et al., 2002).

Other studies on AF in Spain (San Miguel, 1995), and particularly in Galicia, have also contributed to a better understanding of the advantages of AF systems and the complexity inherent to these practices (Rigueiro, 1979; Rigueiro, 1985, 2000; Silva-Pando et al., 2002a).

19.2 GALICIA FOREST LAND AND TYPES OF SILVOPASTORAL SYSTEMS

Galicia covers around 30,000 km^2 with granite, schist, and slate parent soil materials that lead to ranker-like soils, litosols, or humic cambisols (FAO, 1988). Soils are typically acidic and sandy with medium contents of organic matter and low levels of nutrients. Land uses in this region are shown in Table 19.1.

In Galicia, forestland includes not only woodlands, but also riparian areas, shrublands, and natural grasslands. Only 3% of the forestland is public and natural. The remaining forestland is seminatural (antropic influenced) and interspersed with low-intensity farming systems, creating a landscape mosaic. These lands provide timber, fiber, firewood, fertilizers, medicinal plants, small fruits, mushrooms, as well as animal production through browsing and grazing (Silva-Pando et al., 2002a). Some of these traditional uses of natural resources have dropped significantly in the last decades as a result of emigration and new agricultural practices that focus on higher productivity (e.g., modern machinery, mineral fertilizers, and herbicides). Consequently, some agricultural lands have been abandoned and overtaken by shrubs as mentioned above. Studies on silvopastoral systems have shown that the consumption of this spontaneous vegetation by livestock can be a useful tool to prevent fires (Rigueiro, 1979; Silva-Pando and González-Hernández, 1992; Rigueiro et al., 1999). Although some of these silvopastoral practices are based on the use of shrublands by livestock, these plant communities have low nutritional value (González-Hernández, 1995; González-Hernández and Silva-Pando, 1999; González-Hernández et al., 1999; González-Hernández et al., 2003). The main groups of plants in these communities are gorse (*Ulex europaeus*, *Ulex gallii*, *Ulex minor*), broom (*Cytisus scoparius*, *Cytisus multiflorus*, *Cytisus striatus*), heathers (*Erica arborea*, *Erica cinerea*, *Erica aragonensis, Erica mackaiana*, *Erica umbellata*, *Calluna vulgaris, Daboecia cantabrica*), some of the Cistaceae family (*Halimium lasianthum*, *Halimium umbellatum*, *Cistus psilosepalus*, *Cistus salvifolius*), and grasses (*Agrostis* spp., *Avenula sulcata*, *Pseudarrhenatherum longifolium*).

Combination of timber and animal production has been considered an alternative for Galician land not only to yield economic benefit to the landowner and prevent fires, but also to bring ecological advantages (Rigueiro, 2000). With this intent, AF systems have been established on

TABLE 19.1
Land Use in Galicia Based on Data from the Third National Forest Inventory

Land Use	Area (ha)	Area (%)
Pinus pinaster	383,632	12.97
Quercus robur	187,789	6.35
Eucalyptus globulus	174,210	5.89
P. pinaster and *E. globulus*	159,414	5.39
Q. pyrenaica	100,504	3.40
P. sylvestris	63,196	2.14
P. radiata	59,198	2.00
Castanea sativa and other broadleaved	45,518	1.54
P. pinaster and broadleaved	37,982	1.28
P. pinaster, *E. globulus*, and *Q. robur*	33,782	1.14
Q. robur and other broadleaved	16,655	0.56
Q. robur, *E. globulus*, and other broadleaved	12,285	0.42
Scrubland	105,830	3.58
Riparian trees	25,456	0.86
Treeless forest land	634,123	21.44
Agricultural land	821,155	27.77
Unproductive land	52,746	1.78
Wetlands	6,001	0.20
Inland water	17,154	0.58
Total Galicia	**2,957,509**	**100.00**

Source: Adapted from Ministerio de Medio Ambiente, *Tercer inventario forestal nacional*, Direccion General Conservación de la Naturaleza, Madrid, Spain, 2000.

previously abandoned agriculture lands by planting trees and substituting shrubs of low nutritive value with grasses and legumes of high productivity and nutritional quality.

In Galicia, the most commonly used tree species for AF research have been *Eucalyptus globulus*, *Pinus pinaster*, *P. sylvestris*, and *P. radiata* (Rigueiro, 1985, 2000; Silva-Pando et al., 1998). Eucalyptus plantations are located on elevations below 500–600 m, and trees are mostly planted at densities of 800–1000 trees ha^{-1} (Silva-Pando et al., 1998). Pinewoods are located at elevations below 1400 m and the optimum tree density in silvopastoral systems with *P. pinaster* has been reported around 600–800 trees ha^{-1}, and for other pine species around 400–600 trees ha^{-1} (Silva-Pando et al., 2002a). Heathers, gorse, and hard grasses are the main groups of plants under these pine and eucalyptus stands. Many of those plants have limited nutritional quality and are unlikely to meet nitrogen and energy requirements for livestock (González-Hernández and Silva-Pando, 1996, 1999; González-Hernández et al., 2003). For this reason, AF systems that combine grasses and legumes of high productivity and nutritional quality are a good alternative to improve the food resources for grazing. Orchardgrass (*Dactylis glomerata* L.) and white clover (*Trifolium repens* L.) are known as shade tolerant species that have been successfully used in AF systems in Galicia, and they also represent a lower fire hazard than shrublands (Rigueiro, 1985; Piñeiro and Pérez, 1988; Silva-Pando et al., 2002b).

Understory productivity of Galician forests is related to tree density and canopy closure, as well as management practices (González-Hernández et al., 1998). Climate variables and light interception by the overstory also determine the pasture production in Atlantic AF systems (Silva-Pando et al., 2002b). We have monitored several experimental AF plots in pine stands since 1978 till the present that have focused on the effect of above and belowground factors on pasture production and tree growth. More recently, we have extended our research to other AF studies that examine the effect of tree spacing on pasture production and tree growth. Here we summarize some of our results that have contributed to a better understanding of the interactions that occur in AF systems as well as provided some guidelines for designing future AF plantations in temperate climates.

19.3 EFFECT OF ABOVE AND BELOWGROUND FACTORS ON PASTURE PRODUCTION AND TREE GROWTH

The study site was established in 1980 within a 25 year old *P. pinaster and P. sylvestris* stand located in Galicia (Monfero, A Coruña, Spain) at 650 m. The climate is Atlantic, with mild wet winters and warm summers. Annual precipitation is about 2000 mm, most of it occurring from September to May. Mean annual temperature is 10.6°C, and temperatures of the coldest and warmest month are 5.4°C in January and 17.4°C in August, respectively. The area included 11 ha divided into five plots with different tree species and tree canopy: *P. sylvestris* (PS), *P. pinaster* with homogeneous canopy (PP), *P. pinaster* stand adjacent to a natural gap (PPGA), and another two plots without trees (WT), and control (C) (Table 19.2). A mixture of orchardgrass (*D. glomerata* cv. "Artabro") and white clover (*T. repens* cv. "Huia") was sown at densities of 25 and 10 kg ha^{-1}, respectively. All sown plots were previously fertilized. The control plot was not sown or fertilized.

19.3.1 PASTURE PRODUCTION UNDER DIFFERENT TREE CANOPIES

Light, soil moisture, and soil nutrients are the most limited resources in plant communities (Harper, 1977). Sharrow (1999) reported on the competition between trees and ground vegetation for both aboveground (light) and belowground (soil moisture and nutrients) site resources and the beneficial as well as competitive effects that occur from the interactions between plants as they attempt to use the resources available to them.

Several researchers have found that pasture production is influenced by tree canopy and that understory biomass can be a linear function of intercepted radiation (Sibbald and Sinclair, 1990; Knowles et al., 1999). Light transmitted through tree canopies is influenced by tree species, age, and canopy architecture (Percival and Knowles, 1984; Ovalle et al., 1989; Sibbald and Sinclair, 1990; Armand and Etienne, 1996; González-Hernández et al., 1998). Our results concluded that tree canopy architecture impacts production of understory layers by producing different microclimatic conditions, which influence pasture yield and its seasonal pattern (Silva-Pando et al., 2002b). Significant decrease in pasture production occurred under tree canopy cover in comparison to open stands without trees. Annual pasture production ranged from 40% to 55% under *P. pinaster* (833 trees ha^{-1}) and from 10% to 18% under *P. sylvestris* (700 trees ha^{-1}), compared to the phytomass harvested in plots without trees. Light transmission through the canopy significantly

TABLE 19.2
Diameter (DBH) and Height (H) of Trees in the Different Plots Monitored

Plot	Overstory	Trees ha^{-1}	DBH (1992) (cm)	DBH (1996) (cm)	H (1996) (m)
PS	*Pinus sylvestris*	700	23.74 ± 0.72	25.64 ± 0.71	12.67 ± 0.18
PP	*Pinus pinaster*	833	25.34 ± 1.14	26.80 ± 1.21	12.90 ± 0.35
PPGA	*Pinus pinaster*	833	23.47 ± 0.78	25.46 ± 0.88	11.08 ± 0.20
WT	Without trees	0	0	0	0
C	Without trees	0	0	0	0

Source: From Silva-Pando, F.J., M.P. González-Hernández and M.J. Rozados, *Agroforestry Syst.*, 56, 203, 2002.

influenced pasture production. These results also suggested a linear relationship between relative annual production and light transmission ($R^2 = 0.90$, $P < 0.05$), both described as the percentage of herbage production and light transmission through the canopy relative to those under no tree canopy conditions. Other studies have reported similar conclusions, obtaining significantly higher herbage production under *P. pinaster* at densities of 300 trees ha^{-1} than in medium and high-density stands with 600 and 1200 trees ha^{-1} (Papanastasis et al., 1995). These tree–herbage production interactions are less significant in young plantations, and overstory may not influence understory pasture production until 10 years after the trees are planted (Platis et al., 1999).

Several studies have shown the seasonal influences of temperature on net productivity (Hawke, 1991) and the microclimatic effect of trees on herbaceous productivity and quality (Percival and Knowles, 1984; Vales and Burnell, 1988). Seasonal pasture production in our AF study was influenced by different tree canopy and fluctuations of pasture production became less apparent as the percentage of light intercepted by the tree canopy increased. Seasonal variation of pasture production was lower under *P. sylvestris* (light transmission 16%–21% of daily solar radiation) than under *P. pinaster* (light transmission 24%–36%).

Some authors have reported the tempering effect of tree canopy on sown pasture production under severe climatic conditions (Ovalle et al., 1989; Etienne, 1991). In our study, minimum temperatures during the night were tempered under the tree canopy and maximum temperatures during the day were higher in open stands than under tree canopy (Silva-Pando et al., 2002b). Fluctuations of temperature and photosynthetically active radiation (PAR) under tree canopies were less apparent than in "no tree canopy" conditions, and a similar pattern was reflected in the seasonal variation of production. In the early summer sampling, we obtained the maximum pasture yield, corresponding to spring production (April–June), when temperatures and radiation levels are increasing and precipitation is more moderate. The minimum pasture production was harvested from fall to spring. Understory herbage production during the autumn was higher than in the stand without trees, which is consistent with other observations of sown pastures in temperate climates (Sibbald, 1996). Conifers in pastures increase the minimum daily air temperatures and reduce soil temperatures, both of which could potentially benefit cool season forages (Sharrow, 1999). Clearly, a skilled management of tree canopy cover can play a role in shifting forage productivity toward strategic periods by counteracting seasonal limitations (Armand and Etienne, 1996).

In general, light availability becomes limited in silvopasture forage production when trees are sufficiently large that the canopy produces dense shade over a large portion of the pasture. Although light can contribute to the patterns of forage production around individual trees, these effects are minor compared with other factors, such as competition for soil moisture (Sharrow, 1999). Thus, the mechanisms underlying the linear relationships between radiation interception by tree canopy and understory biomass production should be analyzed further by taking into account all the physiological aspects of tree and pasture growth. Depending on the circumstances, other factors may also become limiting and influence the overstory or understory relationships. For example, it has been reported that plants rarely compete for light without concurrently competing for nutrients and water (Sharrow, 1999). In the following sections, soil moisture and the response of pastures and trees to fertilization will be analyzed.

19.3.2 SOIL WATER CONTENT UNDER DIFFERENT TREE CANOPIES

We studied soil water content in the upper 20 cm of soil for two consecutive years under different tree canopies. Soil water content varied under the different canopies and was significantly high in both *P. pinaster* stands (Figure 19.1). The highest *P. sylvestris* canopy closure interfered to the greatest extent with the precipitation able to reach the soil. This difference in canopy architecture could account for the lowest water content in the soil. The presence of shrubs in unsown plots could prevent the humidity loss in soils and soil water content values were higher than in open sown plots during the summer season (Figure 19.2). Minimum values during winter were found under

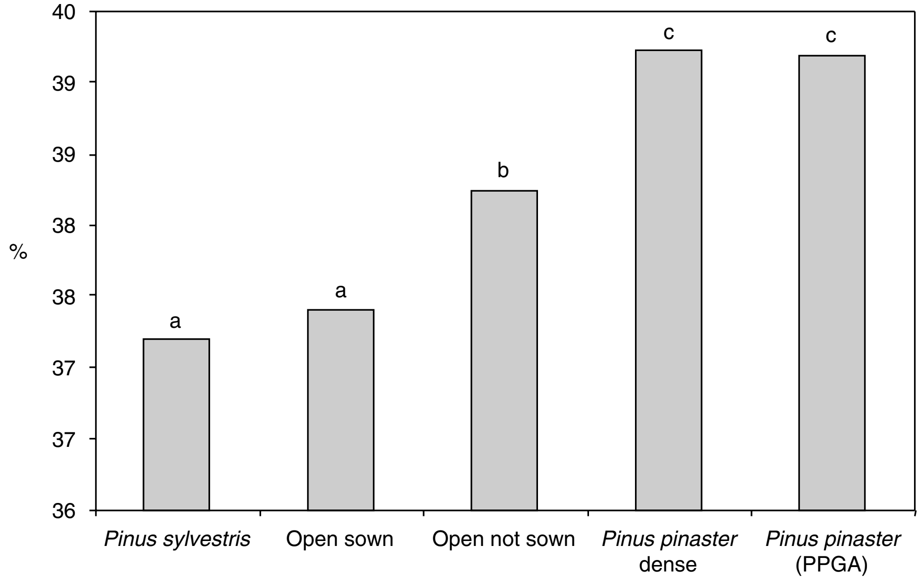

FIGURE 19.1 Soil water content under various canopy closures. Different letters indicate significant differences in soil water content applying Student's *t*-test.

P. sylvestris trees, whereas in summer, minimum values were reached in open sown plots (Figure 19.2).

Soil moisture content was correlated to our pasture production and tree growth results. Pasture production was higher under *P. pinaster* where light and soil water content were also the highest. Seasonal pattern of soil moisture corresponded to herbage yield and, consequently, understory herbage production harvested in the autumn was higher than in the stand without trees, correlating to less water availability during this time of the year in open stands (Figure 19.2). Soil water content under *P. sylvestris* was not significantly different when compared with that of open sown stand (Figure 19.1), and light seemed to be the most determinant factor for pasture production under *P. sylvestris*. Because both light and soil water content are significantly different under *P. pinaster* tree canopy and open stand, it is more difficult to know which factor is more determinant in herbage yield under *P. pinaster*. However, the linkage between plant moisture uptake, soil nutrient

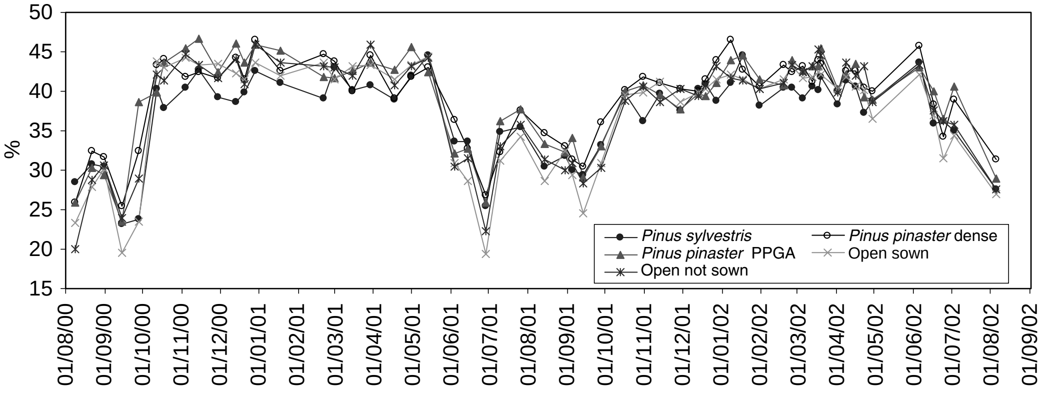

FIGURE 19.2 Monthly evolution of soil water content under different canopies.

extraction, and evapotranspiration from leaves makes separation of competition for nutrients, water, and solar radiation very difficult to evaluate (Sharrow, 1999).

19.3.3 EFFECT OF NITROGEN FERTILIZATION ON PASTURE PRODUCTION AND TREE GROWTH

Maintaining an optimum soil nutrient level with regular fertilizer application is one of the management practices that occur in AF systems. Pastures of high quality and productivity generally require that nutrients and pH of the soil are adequate and optimized. In Galicia, both inorganic and organic fertilization have shown positive effects on pasture production in silvopastoral systems (Rigueiro et al., 2000; Rozados Lorenzo et al., 2001; Mosquera-Losada et al., 2004). Predicting the response of pasture production to fertilizers is an important variable in determining the efficacy and cost effectiveness of soil management in AF systems. In general, soil analyses are not calibrated to predict the amount of N that is needed for a specific pasture production and this subject warrants further research.

We have quantified the seasonal pasture production under *P. pinaster* and *P. sylvestris* canopies using three different levels of nitrogen fertilizer: 50 kg N ha^{-1} (low), 100 kg N ha^{-1} (medium), and 200 kg N ha^{-1} (high). A mixture of orchardgrass (*D. glomerata* cv. "Artabro") and white clover (*T. repens* cv. "Huia") was sown at densities of 25 and 10 kg ha^{-1}, respectively (Rozados Lorenzo et al., 2001).

The highest pasture production was found in the fall and the lowest in the winter under both tree canopies. The pasture production response to the increasing levels of N only occurred in the first seasonal harvest (summer) and under *P. pinaster* canopy. The effect of increasing N fertilizer on pasture production was less consistent in the same season under *P. sylvestris*, with no differences found between the lowest and highest levels applied. Furthermore, we also found a higher pasture production in the control treatment than production yielded after the application of 100 kg N ha^{-1}. Lee and Yun (1985) have reported that optimum rates of fertilizer depend on shading intensity and suggest that forests with 40% or more shade offered no prospect of pasture improvement. On the basis of these findings, we could expect that having 80% shade under *P. sylvestris* would also limit the response of pasture production to fertilization. However, different results have been found in studies on *P. elliottii* silvopastures where herbage production was not entirely limited by light, even under relatively dense tree shade. These findings suggest that trees had no effect upon forage production until they were 7 years old and intercepted 84% of sunlight (Sharrow, 1999). Pasture production under these conditions were 56%–79% of open pasture yields and were approximately doubled by N and P fertilizer application. In the same study, the following year, forage yields were 19%–38% of open pasture yields and again were approximately doubled by fertilizer application, despite pastures under trees receiving only 7% radiation.

It is known that the response of pasture production to fertilization is affected by its botanical composition and that high levels of N in grass–legume mixtures will favor grass production, negatively affecting the legume component due to competition (Cosper et al., 1967). Our results showed no improvement of pasture production by increasing levels of N fertilizer, and herbage yield responded positively only to the lowest level (50 kg ha^{-1}). Medium and high levels did not improve production, and pasture production was higher in the control treatment than when medium levels of fertilizer were applied.

It has been observed that more than 90% of the nitrogen used for spring growth of clovers in clover–perennial grass silvopastures comes from biological fixation of atmospheric N (Sharrow, 1999). We observed a slightly positive effect on percentage cover of *D. glomerata* at increasing levels of N fertilizer, but a negative response to the N level increase in clover. Some studies have shown that N fertilizer increases grass production as a result of stimulating their root system expansion and consequently, supports a better extraction of soil moisture (Sharrow, 1999). In addition, caution has been advised when fertilizing legumes with N, because this can have a negative effect on the nitrogen symbiotic fixation. Barnes et al. (1995) have reported that there is no response of pasture to N fertilization when the percentage of clover reaches 50% in the grass–legume mixture. In mixtures, to obtain response to N fertilization, the maximum percentage of legumes has been estimated at roughly 25% (Hacket et al., 1969). Management guidelines from other studies suggest that when the legume proportion in the mixture is higher than 30%, the main objective should be to maintain the legume. After the legume component has decreased below 30%, then the objective of fertilization should be to focus on increasing the grass production (Aldrich et al., 1970). On the basis of these results, a mixture of 40% clover used in our study could be too high to have a positive response on pasture production. Furthermore, the slow initial establishment of orchardgrass, as well as its progressive dominance in the mixtures with clover, could reinforce this competitive effect (Piñeiro, 1993).

Sharrow (1999) reported that although direct transfer of biologically fixed nitrogen from legumes to associated plants does occur, the quantity is relatively small compared with that derived from the soil nitrogen pool. Most legume N becomes available when plant parts die and decompose. The same author reported that although subclover competition tended to depress newly planted *P. radiata* growth during the first 3 years, 7 years after planting soil nitrogen levels were 36% greater and tree diameters 14% larger compared with those of controls.

We studied the effect of those same different levels of N fertilization on tree growth. The response of tree diameter and height to fertilization was different for *P. sylvestris* and *P. pinaster*. All three levels of N fertilization positively influenced height and diameter growth in *P. sylvestris* trees whereas *P. pinaster* trees did not seem to benefit from any of the levels of N fertilization (Figure 19.3). This could be explained by pasture competition for soil nutrients because herbage yield was higher under *P. pinaster* and consequently had a higher soil nutrient intake. Competition for belowground resources appears to influence both tree and pasture growth more than competition for light, until the point at which pasture is excluded by tree canopy closure (Sharrow, 1999).

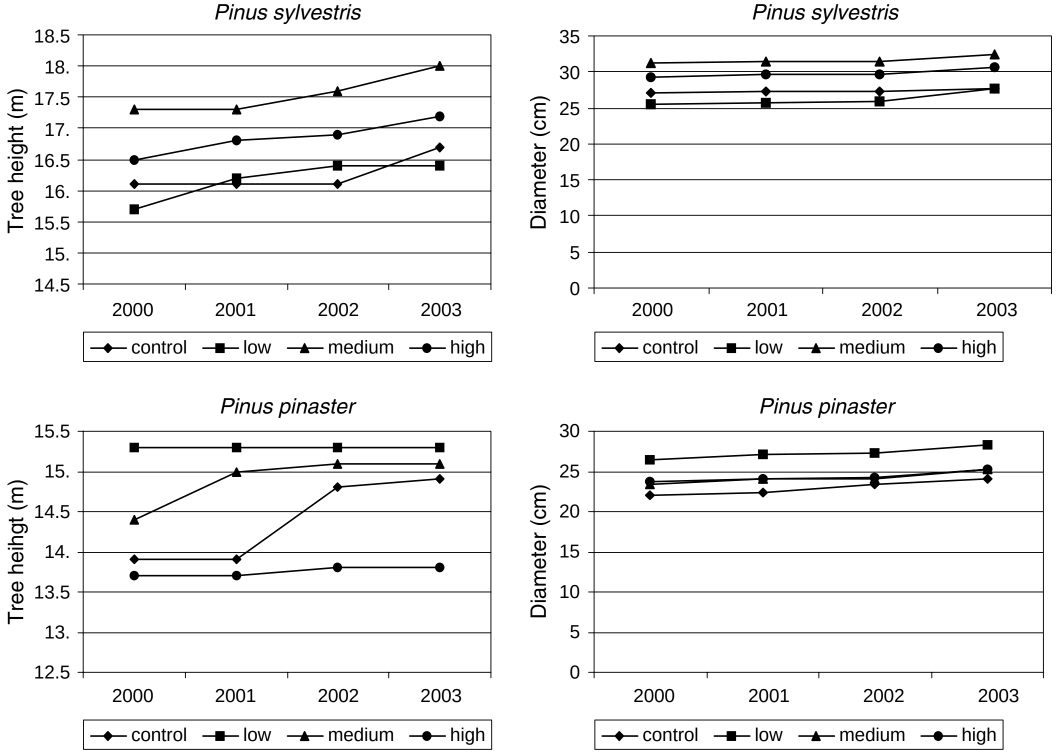

FIGURE 19.3 Diameter (cm) and height (m) response of trees to different doses of nitrogen fertilizer.

Presumably, herbaceous production under dense tree canopies is limited by light whereas that of younger or more open canopied forest is reduced by competition with the trees for soil resources. Under *P. sylvestris*, the pasture production was more limited by light and therefore there was more N fertilizer available for the tree component.

Large conifers and ground vegetation compete primarily for soil resources, and therefore understory forage plants often respond strongly to nitrogen fertilization, even under dense tree canopies (Sharrow, 1999). Belowground competition may vary seasonally, with soil nutrients being most important in the spring, whereas competition for soil moisture is dominant in the summer (Sharrow, 1995).

19.4 EFFECT OF TREE SPACING ON PASTURE PRODUCTION AND TREE GROWTH

To ensure the sustainability of the intercrop and reduce plantation costs tree spacing has proved to be an important factor (Balandier and Dupraz, 1999). To achieve these two goals the initial tree density should be as low as possible, consistent with a final harvest of a 50–80 mature trees (Balandier and Dupraz, 1999). There is, however, some controversy in using low tree densities because doing so can generate certain growth or stem form problems not found in higher tree planting densities. Protection against wind or excessive temperature is limited in widely spaced trees, and more open field conditions may increase frost risks (Friedrich and Dawson, 1984; Aussenac, 1986), thereby aggravating the competitive impact of the understory (Frochot and Levy, 1986; Frochot 1990; Baldy et al., 1994).

Conversely, to ensure pasture sustainability, trees have to be widely spaced to mitigate competition from the trees. Therefore, a crucial research question has been whether very widely spaced trees (50–400 stems ha^{-1}) in AF plantations produce quality timber. Balandier and Dupraz (1999) have reported satisfactory growth of widely spaced forest trees in AF plantations when the choice of tree species suited local conditions.

We monitored pasture production and tree growth using different tree species and spacing. This AF site is located at the coast in Pontecaldelas, Pontevedra, NW Spain (42°22′N, 8°29′W, 386 m a.s.l.). The climate is temperate and humid, with mild winters. Annual precipitation is about 2740 mm, most of it occurring from September to May. Mean annual temperature is 12.2°C, and temperatures of the coldest and warmest month are 6.3°C and 17.5°C in December–January and in July–August, respectively. The study site was established in 1996 and the area included 0.8 ha of acidic sandy brown soil on a granitic substrate. Five tree species were planted at different densities in a parallel row design with systematic sampling: *Pseudotsuga menziesii*, *Pinus pinaster*, *P. radiata*, *Quercus rubra*, *Castanea sativa*, and *Betula alba*. Densities were 222, 256, 424, 518, 1048, 2000, and 2500 stems ha^{-1}, corresponding to 7×8, 6×7, 5×6, 4×5, 3×4, 2×3, and 2×2 m tree spacing. Pasture yield monitored was a mixture of *D. glomerata* var. "Artabro" and *T. repens* var. "Huia."

Four years after plantation, the height of the trees ranged from 1.1 to 6.3 m (Figure 19.4). At various densities, tree growth patterns differed among species. Poor height growth was recorded at high densities of 2000–2500 trees ha^{-1} (2×3 and 2×2 m tree spacing) for *Q. rubra* and *B. alba*, whereas height of *C. sativa* and *P. radiata* were affected positively by these same high tree densities. Tree height increased progressively for *Q. rubra* and *C. sativa* at densities from 222 trees to 1048 (7×8, 6×7, 5×6, 4×5, 3×4 m tree spacing). *B. alba* height growth was benefited at the lowest tree densities (7×8, 6×7 m tree spacing) whereas *P. radiata* was the species for which height was positively influenced by high tree density.

Tree height growth under different tree densities was very inconsistent for *Pseudotsuga menziesii* and *Pinus pinaster*. Both tree species showed the highest value of height growth at the lowest tree density. Although *Pseudotsuga menziesii* had the second highest growth at 1048–2500 (3×4, 2×3, and 2×2 m tree spacing), and medium tree density of 518 trees ha^{-1} (4×5 m tree spacing) provided the second best growth in height for *Pinus pinaster*.

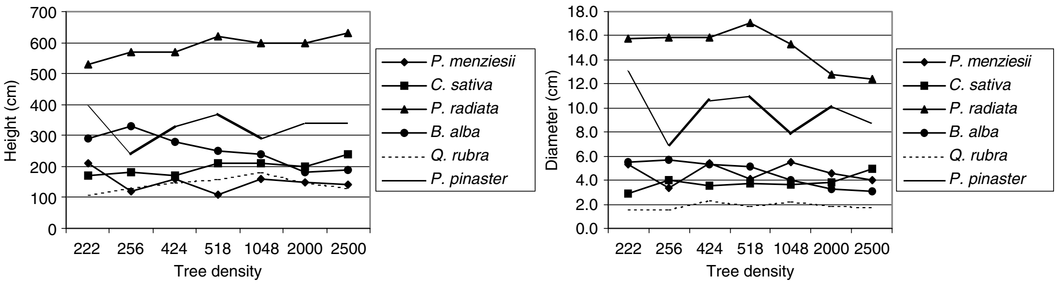

FIGURE 19.4 Height and diameter response of trees to different tree spacing.

Basal stem diameters ranged from 1.6 to 17.1 cm (Figure 19.4) according to species and tree density. The diameter followed the same pattern as height in response to tree spacing, with the exception of *P. radiata* in which basal diameter was lower at medium to high tree density. *C. sativa* basal diameter increased with tree density, *P. pinaster* and *Pseudotsuga menziesii* kept their inconsistent pattern, and *B. alba* basal diameter decreased with tree density.

The height/diameter ratio (*H*/*D*) is often used to quantify tree stability, although it is generally applied to older trees. For mature trees, *H*/*D* values below 100 generally indicate good stability (Balandier and Dupraz, 1999). In our study, different tree spacing resulted in differing ratios, although for all species, our results showed values under 100 (Figure 19.5).

Crops or swards compete for water and nutrients, and strongly influence the tree growth pattern (Anderson and Sinclair, 1993; Montard et al., 1999; Gakis et al., 2004). In AF, two features are combined: sustainability of the intercrop production throughout most of the tree life and a high value of timber crop. In their review of case studies representing current AF research in Europe, Dupraz and Newman (1997) reported that tree–pasture interactions were quite severe. With 100 stems ha^{-1} of *Fraxinus excelsior*, sward yield was computed to be 83% of the control at year 10, falling to zero at year 40 (Doyle et al., 1986).

Tree–pasture competition was most strongly reflected in our study with the lowest pasture production under *Pinus radiata*, the species that increased height at high tree density (Figure 19.6). Our results also showed that the effect of tree species on pasture production was varied and most significant at the highest densities (3×4, 2×3, and 2×2 m tree spacing) and the lowest densities (6×7, 7×8 m tree spacing). No significant differences in pasture production were found for intermediate tree density values. Under *Pseudotsuga menziesii* and *Pinus pinaster*, pasture

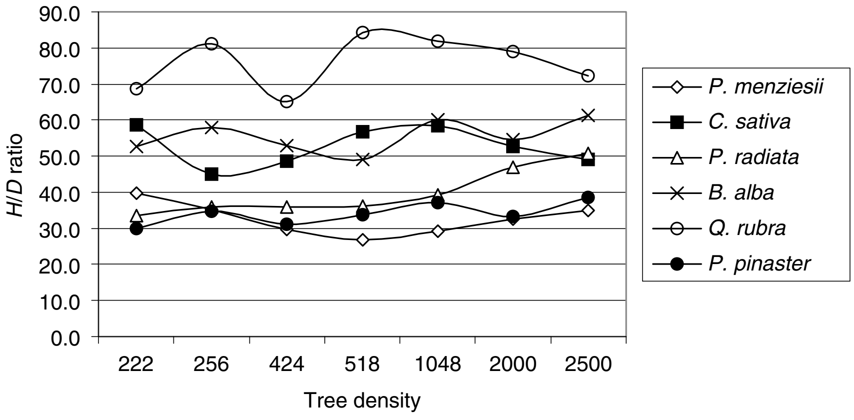

FIGURE 19.5 Response of the *H*/*D* ratio to different tree spacing.

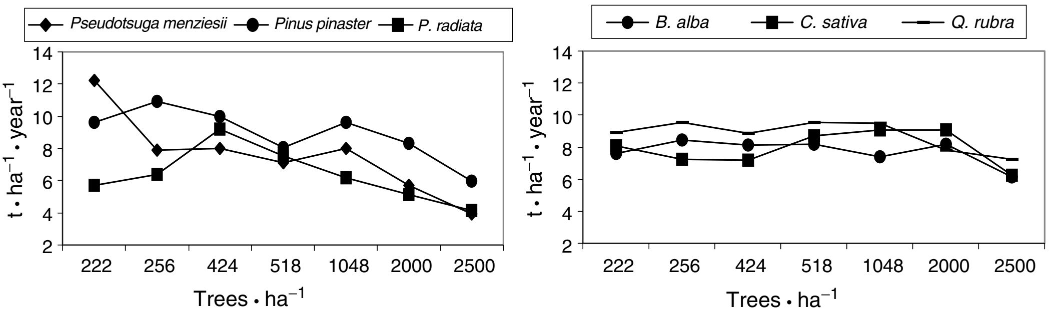

FIGURE 19.6 Pasture production under various conifer and broadleaved tree species at different tree spacing.

production decreases started from 1048 trees ha^{-1}, whereas under *P. radiata* decreases appeared at densities as low as 424 trees ha^{-1}. At the fifth to sixth growing season, Burner and Brauer (2003) reported that herbage productivity was negatively influenced by densities lower than 840 trees ha^{-1} under loblolly pine (*Pinus taeda*). However, Pearson et al. (1995) did not find significant decreases in herbage yield under loblolly pine densities of 4451 trees ha^{-1} 10 years after planting.

Production under deciduous species seemed to be slightly influenced by stand density, probably due to slower growth and smaller stems, relative to conifer species (Figures 19.6 and 19.7). Among broadleaved trees, only *B. alba* showed significant effect of tree spacing on pasture production. In general, for the three deciduous species studied, only density as high as 2000 trees ha^{-1} significantly influenced pasture production. Other AF studies with broadleaved tree species have shown that trees did not reduce sward productivity during the first 7 years when pasture production was evaluated under different conditions of tree density, even as high as 2500 stems ha^{-1} (Sibbald and Agnew, 1995).

When we analyze pasture production and tree development together, our results suggest that, spacing of 4×5 m (518 *P. radiata* trees ha^{-1}) and 5×6 m (424 *P. pinaster* and *Pseudotsuga menziesii* trees ha^{-1}) maintain the most consistent pasture production over time. This is consistent with other research conducted in temperate areas such as Great Britain (Adams, 1984) and France (Lemoine et al., 1983; Rapey et al., 1994) where trials to stimulate fodder production in coniferous plantations (*P. menziesii* and *Picea abies*) include a reduction of tree stocking to below 600 stems ha^{-1}.

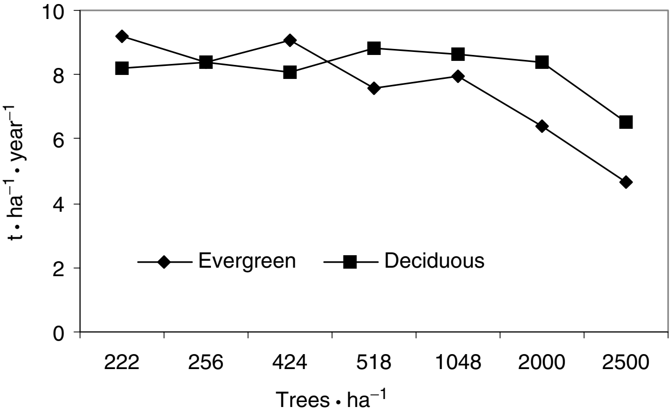

FIGURE 19.7 Pasture production under broadleaved and evergreen tree at different tree spacing.

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Index

ECOLOGY/FORESTRY

ECOLOGICAL BASIS OF AGROFORESTRY

Faced with the growing problems of climate change, ecosystem degradation, declining agricultural productivity, and uncertain food security, modern agricultural scientists look for potential relief in an ancient practice. Agroforestry, if properly designed, can mitigate greenhouse effects, maintain ecosystem health and biodiversity, provide food security, and reduce poverty. Poorly implemented agroforestry, however, may not only exacerbate existing problems, but also contribute in its own right to the overall negative effects of our depleted and failing ecosystems. With a diminishing margin for error, a thorough understanding of the ecological processes that govern these complex systems is, therefore, crucial.

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