2 An Ecological Foundation for Temperate Agroforestry

Richard K. Olson

Berea College
Berea, Kentucky

Michele M. Schoeneberger

USDA Forest Service
Lincoln, Nebraska

Stefanie G. Aschmann

USDA NRCS
Lincoln, Nebraska

> *When we have destroyed the forests and prairies to replace them with agriculture, we have never known what we were doing because we have never known what we were undoing.*
> Wendell Berry (as quoted by Jackson, 1987)

An *agroecosystem* is an ecosystem that is altered and managed to produce food and fiber for human use. Ecosystems are the foundation of our food system, and if we wish to keep that system functioning, we need to understand its ecological underpinnings.

The primary natural ecosystems of North America are dominated by either perennial grasses or woody perennials. Prior to European settlement, grasslands occupied 39% of the current USA, while forests and shrub-dominated systems covered most of the remainder (Sims, 1988). These highly diverse and dynamic assemblages of species evolved during millions of years in response to major changes in climate and physiography. Powered by solar energy only, they have sustained production and other ecosystem functions while generating a legacy of rich soils and other biological wealth.

Because of this legacy, most of these ecosystems have been converted to agroecosystems through the substitution of annual grasses [e.g., corn (*Zea mays* L.), wheat (*Triticum aestivum* L.)] and broadleaves [e.g., soybeans [*Glycine max* (L.) Merr.]] for the original perennial vegetation. Where climate or other conditions preclude the planting of row crops, native grassland is exploited through the substitution of domestic livestock for native herbivores, and forests are intensively managed for timber production. The conversion has been extensive and thorough; at

---

American Society of Agronomy, 677 S. Segoe Rd., Madison, WI 53711, USA. *North American Agroforestry: An Integrated Science and Practice.*

the extreme are states like Illinois with <0.05% of its land area retained in natural ecosystems (White, 1978). Excluding Alaska, 23% of the U.S. land area is now crop land (USDOC, 1990), 30% is rangeland (USDA, 1981), 25% is classified as commercial timberland (USDA, 1982), and 5% supports structures (USDA, 1995). Less than 2% is formally protected as wilderness (Reed, 1989). The landscape is now a seminatural matrix (Roberts, 1988) within which humans and all other species must survive.

The goal of this conversion has been to maximize the amount of net primary or secondary production from these systems that can be used by humans. In the short term, this goal has been met and a massive increase in food supplies has been generated. Other effects of conversion, however, are not so benign and bring into question the sustainability of this level of production. For example:

- In addition to solar energy, U.S. agroecosystems use large amounts of fossil fuels to power machinery or produce other inputs such as fertilizer and pesticides. Irrigated corn production in Nebraska requires more than 59 million kilojoules (kJ) (14 million kilocalories (kcal)) of fossil energy input per hectare (Pimentel & Burgess, 1980), or 1 kJ input for each 1.84 kJ harvested. When the entire American food system, field to table, is considered, as many as 42 kJ (10 kcal) of energy input is required for each kJ (kcal) consumed (Pimentel & Pimentel, 1996). This energy profligacy occurs in a country that imports more than 50% of its annual oil consumption (USDOC, 1996), and whose domestic oil reserves will be largely exhausted within 25 yr (Cleveland & Kaufman, 1991; Kaufman, 1991).
- Approximately 30% of U.S. crop land has been severely damaged because of erosion, salinization, or waterlogging (Pimentel et al., 1995). Erosion continues at a rate of 1.93 billion Mg (2.13 billion tons) of soil per year, with water erosion causing soil loss in excess of replacement levels on 55 million ha (136 million acres), and wind erosion causing soil loss in excess of replacement levels on 47 million ha (116 million acres; USDA, 1995).
- The genetic diversity of the original systems has been greatly reduced. Instead of the 250 to 300 plant species found in an equivalent area of tall-grass prairie (Steiger, 1930), or the 100 species in a similar area of oak–hickory (*Quercus* sp.–*Carya* sp.) forest, a typical mid-western corn–soybean farm attempts to maintain only two species. In addition, genetic diversity within the major U.S. crops is quite low. Farmers who plant several hybrids or varieties to increase their diversity are often planting essentially the same thing, under different names (NAS, 1972; Raeburn, 1995). These highly simplified systems are at increased risk from pest outbreak or environmental extremes.

Sustainability is "the ability of an agroecosystem to maintain production through time, in the face of long-term ecological constraints and socioeconomic pressures" (Altieri, 1987). Our current farming system faces declining domestic energy reserves, soil loss in excess of regeneration, and a rapidly increasing human population with a concomitant increase in demand for agricultural products. Although farmers have adopted practices such as contour planting, no-till, and precision application of chemicals in an attempt to reduce some of the negative effects of agriculture, farming systems based on monocultures or simple rotations of annuals are not sustainable.

Further diversification of crops offers many advantages. The addition of herbaceous perennials such as alfalfa (*Medicago sativa* L.) and grasses increases the perennialism that is such a dominant feature of native ecosystems (Fig. 2–1; Van Andel et al., 1993). Reduced erosion, fixation of atmospheric N by legumes, and reduced energy inputs (Heichel, 1978) are benefits of adding certain perennials to an agroecosystem. Livestock offer further diversification and a mechanism for converting forages into higher-value products (Bender, 1994).

Although forests are a major vegetation type in the USA, few farmers consciously integrate trees and other woody perennials into their farms as a way of increasing diversity and sustainability. This approach to farming is known as agroforestry and is defined as: intensive land management that optimizes the benefits (physical, biological, ecological, economic, social) from the biophysical interactions created when trees and/or shrubs are deliberately combined with crops and/or livestock (Garrett et al., 1994).

The key words in this definition are *interactions*, *benefits*, and *optimizes.* Biophysical interactions require a certain spatial and temporal proximity of the com-

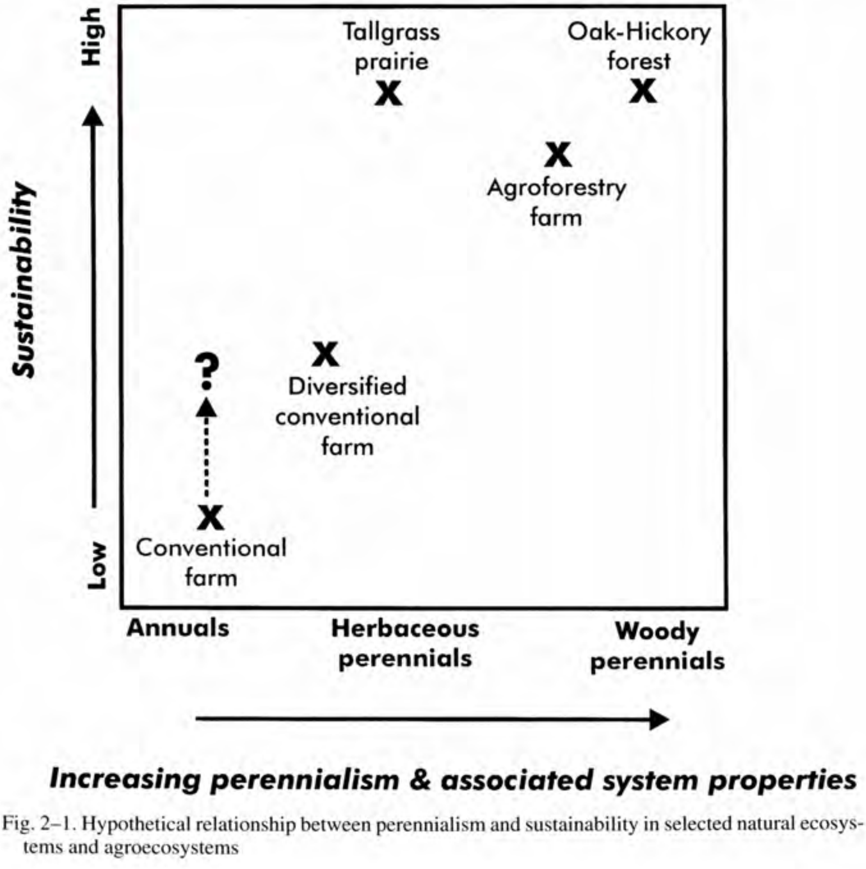

Fig. 2–1. Hypothetical relationship between perennialism and sustainability in selected natural ecosystems and agroecosystems

ponents. A woodlot on one corner of the farm, isolated from and not beneficially interacting with crops or livestock, does not constitute agroforestry by this definition. When trees, crops, and livestock are in close enough proximity to interact in a way that is significant to the farmer, the types of interactions depend on the species involved and their particular spatial and temporal relationships. Not all interactions are beneficial; for example, competition between trees and row crops for water, nutrients, and light can reduce row crop yields. Obtaining optimal benefits from agroforestry requires knowledgeable selection and placement of the woody and nonwoody components. A random jumble is unlikely to perform well.

Unfortunately, there is no single optimal design that can be disseminated by extension agents to interested farmers and ranchers. Differences in climate, topography, soils, crops, and livestock exist at scales from local to regional. Agroforestry practices must be designed to fit the particular ecological, social, and economic context of the farm in question. Component interactions in agroforestry practices have been investigated to a small extent (Ong & Huxley, 1996), but the emphasis has been on tropical systems. Whether we are considering temperate or tropical agroforestry, Muschler, in *An Introduction to Agroforestry* (Nair, 1993), points out "that the complexity and lifespan of agroforestry makes investigations of mechanisms and processes extremely difficult." Leaving consideration of socioeconomic issues for later, how can we obtain the ecological knowledge necessary for the optimal design of a wide variety of temperate agroforestry practices.

The answer lies, at least in part, in the native ecosystems upon which U.S. agriculture is built. Highly sustainable, these systems were locally adapted to the environmental conditions under which they evolved. Natural ecosystems can provide models for the design of sustainable agroecosystems (Woodmansee, 1984; Soule & Piper, 1992; Davies, 1994). We believe that it is possible to identify structural and functional characteristics of natural ecosystems that contribute to their sustainability, and retain or incorporate these into agroecosystems while maintaining production. Regional and local differences in natural ecosystems can serve as guides for tailoring agroforestry practices that best fit a particular farm's environmental conditions. Our goal in the remainder of this chapter is to illustrate some of the structural and functional relationships among woody and herbaceous vegetation in natural ecosystems of the USA, and to show how these relationships apply to agroforestry practices.

ECOLOGICAL INTERACTIONS IN MIXED TREE AND HERB SYSTEMS

Categories of Systems

Ecosystem function is determined not just by species composition, but also by the spatial and temporal arrangement of the component species. Figure 2–2A identifies eight main types of North American temperate ecosystems based on the spatial and temporal relationships of their woody and herbaceous components. Figure 2–2B uses the same framework to identify structurally analogous agroforestry practices found in this region.

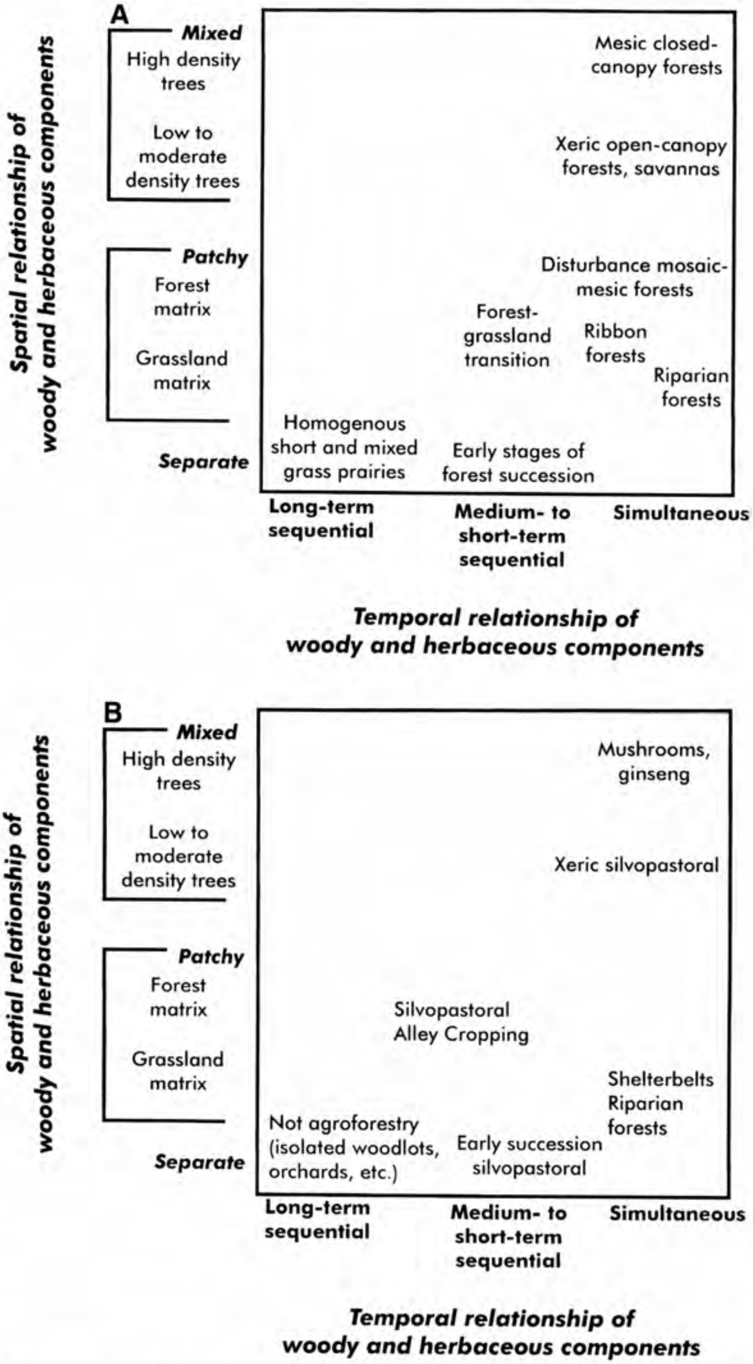

Fig. 2–2. Categorization of (A) ecosystems in terms of the spatial and temporal relationships of the woody and herbaceous components; (B) temperate agroforestry practices in terms of the spatial and temporal relationships of the woody and herbaceous components.

Table 2–1. Summary of the most important processes in interactions between woody and herbaceous species in natural ecosystems of the USA, and in the analogous agroforestry practices.

Natural system category	Key processes in interactions among woody and herbaceous species	Analogous agroforestry practices in which these processes are important
Mesic forest, closed canopy	• canopy interception of solar radiation and modification of microclimate	• mushroom production • ginseng production
Disturbance patchiness in forest landscape	• gap-creating disturbances • edge effects • landscape processes	• none (in tropical areas this would be swidden agriculture)
Early successional systems	• progressive modification of microclimate as tree canopy closes	• black walnut alley cropping • silvopastoral—grazing of early successional stages
Xeric forest, open canopy	• competition for water • localized interception of solar radiation	• silvopastoral practices
Mixture of forest and grass patches in transition zones	• topographic patterns often serve as template • chronic stress and disturbance	• silvopastoral practices
Ribbon forests	• windspeed reduction • snow distribution	• windbreaks
Riparian forests in grasslands	• corridors for movement of wildlife • specialized wildlife habitat • interception of sediment and nutrients	• riparian forests in crop land or pasture matrix
Isolated grasslands	• no interactions	• not agroforestry

The main concern in agroforestry practices is performance or the relationship between structure and function. In each of the natural systems, there are certain processes that are most important in determining interactions among the woody and nonwoody components and in turn the overall function of the system (Table 2–1). These same processes influence interactions among the components of analogous agroforestry practices.

Closed Canopy Mesic Forests

Water, nutrients, and light (energy) are the main resources for which plants compete. In more mesic (wetter) sites with adequate nutrients, light is the limiting factor. Trees and shrubs invest heavily in structural components to lift leaves above competitors and capture light before it reaches the ground. Forests often have two or three canopy layers as trees, saplings, and shrubs capture light at different levels, and this vertical stratification may increase total energy capture by the system. Total leaf area can be quantified as leaf area index (LAI), the ratio of total leaf surface area to ground surface area. Depending on leaf orientation, a canopy with an LAI of 3 to 4 can intercept 90% of incident solar radiation (Loomis & Connor, 1992). Mature mesic forests generally have an LAI of 8 to 10 (Odum, 1971), so competition for light is intense, with only 1% to 5% of incident solar radiation reaching the forest floor in closed canopy deciduous forests (Hicks & Chabot, 1985). Light penetration was 6% in a high elevation fir forest with full crown density, and 18% when crown density was 50% (Smith, 1985). Light quality as well as quantity is affected by tree canopies, with radiation below the canopy relatively enriched in red wavelengths (Atzet & Waring, 1970). The light environment under plant canopies is highly variable both spatially and temporally as sun flecks shift with changes in the angle of incident sunlight.

As a result of competition for light, mesic forests often have sparse ground-level vegetation. For example, in a tulip tree–oak (*Liriodendron tulipifera* L.) forest in the southern Appalachians, only 2 g kg^{-1} of the total aboveground biomass consisted of herbaceous species (Harris et al., 1975). Some deciduous forests, however, have rich herbaceous layers (Braun, 1967), and seasonal changes in LAI offer temporal niches for certain species. Spring ephemerals in forest understories are able to leaf out and capture substantial light energy before overstory canopy development occurs. Goldenseal (*Hydrastis canadensis* L.), an understory forb native to many eastern deciduous forests, produces >950 g kg^{-1} of its aboveground biomass within the first month of its growing season, well before the overstory canopy is developed (Eichenberger & Parker, 1976). Other species are adapted to full-shade conditions and experience peak development in late summer (Greller, 1988). The co-occurrence of these strategies results in increased capture of light energy as well as more efficient use of other resources. Nutrient uptake by spring ephemerals may sequester up to 900 g kg^{-1} of the N and K that could potentially be leached during the spring from some Midwest forests (Blank et al., 1980; Peterson & Roelf, 1982). Thus, temporal as well as spatial stratification plays a role in system function.

Forest canopies modify other aspects of microclimate in addition to radiation. During the day, interception of solar radiation by the tree canopy creates a temperature maximum at the height of maximum foliage density (Oke, 1987). This creates a temperature inversion that increases the atmospheric stability in the canopy relative to open terrain, partially decoupling the local atmosphere from the external environment. Windspeed decreases rapidly with distance into the canopy, while daytime humidity increases and CO_2 concentration decreases due to transpiration and photosynthesis by the foliage. At the forest floor, this altered environment affects seed germination, plant establishment, litter decomposition, and the population dynamics of microorganisms, insects, and other organisms (Belsky, 1994; Jackson et al., 1990; Tiedemann & Klemmedson, 1973; Vetaas, 1992).

Agroforestry options for closed canopy forests are limited to crops that are adapted to a low-light environment. Shiitake mushrooms [*Lentinula edodes* (Berkeley) Pegler; Harris, 1986] and ginseng (*Panax quinquefolius* L.; Duke, 1989) fit perfectly in this situation, both requiring the protected environment of the forest floor. Shiitake is grown by inoculating oak logs with mushroom spawn and then stacking the logs under a hardwood or conifer canopy. If the site is a deciduous forest, shade cloth is used to provide protection during leafless months. Ginseng, a medicinal herb, is cultivated in a variety of temperate deciduous forests, although most often associated with maple (*Acer saccharum* Marsh.) and beech (*Fagus grandifolia* Ehrh.). It grows well at light intensities from 5 to 30%, and is sometimes intercropped with goldenseal to deter root rot (Duke, 1989).

Disturbance Patches and Early Successional Systems

Forest canopies are heterogenous. In regions where the climate is mesic enough to support closed-canopy forests, disturbances such as wind, avalanches or fire create gaps that support herbaceous vegetation for a brief period of time. In old-growth forests of the eastern USA, 9.5% of the land area was in small gaps (created by the death of one to several trees) (Runkle, 1982). New gaps formed on 1% of the land each year while an equal area of gaps closed due to sapling growth. Less frequently, larger areas are disturbed by hurricanes, fires, insect outbreaks (e.g., gypsy moth) and other large-scale events (Spies & Franklin, 1989). Since European settlement, most U.S. forests have been logged at least once.

The smaller the gap, the more important are edge effects such as competition for water and nutrients, shading, and reduction of windspeed. Edges also are zones of increased diversity and activity for many species of insects, birds, and mammals, and at the landscape scale, the size and distribution of gaps is an important determinant of many forest functions. Swidden or slash-and-burn agriculture mimics the process of gap formation and succession in many tropical forests and has been called the most sustainable form of agriculture when practiced appropriately (Kleinman et al., 1995). An analogous form of temperate shifting agriculture was practiced by Native Americans in the New England region (Davies, 1994). In some temperate U.S. forests, logging of small patches to mimic natural processes of gap formation offers a more sustainable alternative to large-scale clear-cuts (Maser, 1994).

Grasses and forbs dominate a gap immediately following disturbance, but are replaced soon by trees or shrubs. This transition is known as succession, "an orderly process of community development that involves changes in species structure and community processes with time, and results from modification of the physical environment by the community" (Odum, 1971). Keever (1950) described a typical succession pattern for abandoned farmland in the North Carolina Piedmont with crabgrass (*Digitaria* sp.), asters (*Aster* sp.), and ragweed (*Ambrosia artemisiifolia* L.) dominating the first 2 yr, followed by broomsedge (*Agropogon virginicus* L.), which was gradually replaced in 10 to 15 yr by shortleaf (*Pinus echinata* Mill.) or loblolly pines (*Pinus taeda* L.). A hardwood understory develops by 60 yr and forms the climax oak–hickory forest by 150 yr.

Succession results from the gradual modification of microclimate as the expanding canopy intercepts increased amounts of solar radiation each year. Enough light reaches the ground early in the successional process to support significant forage production. With proper management to limit damage to young trees, livestock can be grazed as part of a silvopastoral agroforestry practice. Stocking rates are reduced as tree growth reduces light levels until canopy closure eliminates forage production. Successional silvopastoral practices are particularly well developed in New Zealand and Australia (e.g., Anderson et al., 1988).

Successional principles also can be applied in cropping systems. Perhaps the best known temperate example is alley cropping with black walnut (Garrett & Kurtz, 1983; Williams & Gordon, 1992). Black walnut (*Juglans nigra* L.) is planted at wide spacings (e.g., 12 m), and row crops are grown in the alleys for up to 10 yr. When shading reduces row crop yields, forage crops are substituted either for harvest or direct grazing. By the time canopy closure ends profitable forage production, nut production provides income until the trees are cut for timber, and the process is reinitiated.

Xeric and Transitional Forests

In more xeric (drier) sites, moisture is limiting and competition for resources is greater belowground than aboveground. Forest canopies become more open as trees become more widely spaced, and a greater proportion of light reaches the ground. Higher light levels may allow the development of significant amounts of ground-level vegetation. Ponderosa pine (*P. ponderosa* Laws.) forests throughout the Rocky Mountains and longleaf pine (*P. palustris* Mill.) forests in the southeastern USA frequently have dense grass understories that are maintained in part by periodic fires (Daubenmire, 1978). In still drier areas, tree density decreases until scattered individuals in a grassland matrix form a savanna such as the blackjack oak (*Q. marilandica* Muenchh.)–post oak (*Q. stellata* Wangenh.) savanna in eastern Texas, pinyon–juniper (*Pinus* sp.–*Juniperus* sp.) savanna in the southwestern USA, and the oak–hickory savanna in western Missouri. The oak savanna, characterized by a sparse overstory of oaks and an understory of herbs and grasses, is a transitional zone between the eastern forest and the grasslands (Packard, 1988). Oak savanna was once a major community across the Midwest, although it was severely diminished after the Euro-American settlement of the 1800s. Prior to settlement and overgrazing, large areas of sagebrush steppe in the intermountain West also showed a codominance of shrubs (*Artemisia*) and perennial bunchgrasses (West, 1988).

As the preceding examples suggest, disturbance (e.g., grazing, browsing, drought, fire) is a critical mediator of the competition between trees and grasses (Belsky, 1994; Hamerlynck & Knapp, 1996; Jeltsch et al., 1996). In the southeastern coastal plain, longleaf pine forests with a grassy understory are maintained by fires of 3 to 10 yr frequency that allow regeneration of the pines but prevent establishment of hardwoods, which have denser canopies than the pines and would inhibit grasses (Daubenmire, 1978). Most savannas are maintained by fire, and if fire is prevented or overgrazing leaves insufficient fuel to carry a fire, succession proceeds to a denser forest. Grasses are physiologically and morphologically adapted to burning, and some observers liken the relationship between grasses and fire to calling in air strikes on your own (dug in) position to prevent it from being overrun. The ecological message is that a particular balance between grasses and trees can often be maintained only through regular disturbance.

As water becomes more limiting, trees disappear and grasses or shrubs dominate. Grasses have a high root to shoot mass ratio, which provides an advantage in competing for water and nutrients. A shrub such as mesquite, however, also has an extensive lateral and vertical root system that allows it to compete effectively for water as well as nutrients against grasses in the arid grasslands in which it occurs (Tiedemann & Klemmedson, 1973). In other cases, competition for belowground resources is reduced by the exploitation of different soil layers by different species. In the blue oak (*Q. douglasii* Hook. & Arn.) savanna, competition for water between oaks and grasses is reduced by vertical stratification of the two root systems, with grasses occupying mainly the top meter of soil and oak roots penetrating >25 m (Jackson et al., 1990). This stratification also promotes more efficient cycling and retention of N in the ecosystem.

In addition to competition for resources, trees and grasses in these mixed systems may compete through direct interference; for example, the allelopathic suppression of understory plants in oak forests in Oklahoma (McPherson & Thompson, 1972). Alternatively, some interactions may be positive. Survival of grass seedlings was three times greater within a California blue oak savanna than in an adjacent open grassland (Jackson et al., 1990) due to the more favorable environment for seedling establishment (e.g., higher relative humidities, decreased evaporation, and increased near-surface soil moisture and nutrient levels).

Within a particular climatic region, topographic and soil patterns may have a strong influence on spatial patterns and interactions of woody and nonwoody species. Throughout much of the Great Plains grasslands, trees and shrubs are restricted to riparian areas, rocky escarpments, mesic north-facing slopes, and other sites offering increased moisture availability and protection from fire. Rockier soils also provide better opportunity for tree seedling establishment in competition with the thick root mass of grasses (Wells, 1965). At the northern edge of the prairie, grasses on the uplands form a mosaic with groves of poplar (*Populus* sp.) located in depressions or on protected slopes (Daubenmire, 1978).

Significant grass production in a forest matrix allows timber production and grazing to co-exist on more than 69 million ha (170 million acres) in the USA (USDA, 1981). The dual functions of these silvopastoral practices can be enhanced by management based on ecological principles. On mesic sites, thinning and pruning of trees maintains forage production while promoting high quality timber. Prescribed burns can prevent invasion by undesirable species while maintaining an open and productive understory. In semiarid areas, avoiding overgrazing is the most effective means of preventing the replacement of grasses by shrubs.

Ribbon Forests

When wind encounters the edge of a forest, some of the air is deflected over the canopy for a distance of up to 20 tree heights (Cionco, 1985; Fritschen, 1985). If the forest occurs as a narrow strip, this deflection of air creates a protected zone to the leeward in which wind speed is reduced, wind-related stresses such as desiccation are decreased, and snow deposition may increase.

This modification of microclimate is essential to the maintenance of ribbon forests (Billings, 1969; Peet, 1988), a fascinating feature of subalpine regions in the Rocky Mountains. Ribbon forests are arranged as alternating parallel strips of forest and moist alpine meadow oriented perpendicular to the prevailing winds. Snow accumulation to the lee of each forest strip inhibits seedling establishment, while tree growth rates at the far edge of each drift are increased by water from snowmelt and protection from desiccation by winter winds. Thus, the pattern and spacing of forest strips is determined by the effect of tree canopy structure on windspeed and snow deposition.

Ribbon forests are a classic model for one of the most common temperate agroforestry practices, windbreaks. Farm windbreaks are linear groups of trees that provide a sheltered microclimate for leeward fields. The extent and degree of shelter depends on structural characteristics of the windbreak such as height, density, and orientation, and these can be manipulated to meet particular management goals. For example, dense windbreaks result in deposition of snow in drifts close to the leeward edge and act as living snowfences. More porous windbreaks cause snow to be distributed more evenly across the leeward field, a preferable situation if soil moisture conservation or protection of winter wheat from desiccation is the goal (Brandle & Finch, 1991).

Riparian Forests

Particularly in arid and semiarid regions, riparian forests are often the only mesic vegetation type and serve a critical role as wildlife habitat. In Arizona and New Mexico, an estimated 80% of all vertebrates are dependent upon riparian habitat for at least part of their life cycle (Johnson, 1989). As linear features in the landscape, riparian forests may serve as corridors for the movement of many species between otherwise isolated patches of habitat (Forman & Godron, 1986). Woody vegetation also plays an integral role in the stabilization of streambanks (Smith, 1976), shading of streams reduces water temperature, and detritus inputs to the stream from the forest provide an energy source as well as habitat structure for aquatic organisms.

Because they occupy low spots in the landscape, riparian forests receive water and water-borne nutrients and sediment from upland areas, filtering and trapping many of these inputs before they reach the stream bed (Lowrance et al., 1984). These forests interact not just with adjacent fields, but with systems throughout the landscape, linked through the hydrologic pathways of the watershed. In agricultural regions, this landscape-level water quality function is particularly important. For example, despite large applications of N fertilizer to corn, peanuts (*Arachis glabrata* Benth.), and other crop land in a Georgia piedmont watershed, very little N left the watershed in streamflow, due in part to accretion of N in riparian forest biomass, and denitrification in the saturated riparian soils (Lowrance et al., 1985). Maintenance of a young-age forest through selective logging can improve the water quality function of the stand by maintaining plant nutrient uptake at a high rate (Welsch, 1991).

Isolated Grasslands

In large areas in the Great Plains, particularly in the more xeric short- and mixed-grass prairies, grasses and forbs exist largely independently of any woody species. The same situation exists in some of the larger high-elevation meadows in the Rockies and, presettlement, the grasslands of the Central Valley of California and the Palouse Prairie of eastern Washington. How small a grassland can be before adjacent woodlands have a significant effect is, of course, a key question in terms of agroforestry design. Long-term—hundreds or thousands of years—these grasslands did not remain isolated as evidenced, for example, by the presence of conifers throughout the Great Plains during the various glacial periods (Axelrod, 1985).

General Principles

We can highlight four ecological principles that will be of particular use in designing and evaluating agroforestry practices:

1. Ecosystems are Distinguished by Spatial and Temporal Heterogeneity.

An ecosystem or landscape consists of a mosaic of patches and linear components. The boundaries between patches are often the site of increased rates of processes such as nutrient and energy exchange, competition, water flow, and movement of organisms (Raney et al., 1981; Holland et al., 1991a). For example, most of the removal of NO_3^- from water entering a Georgia riparian forest occurred in the first 10 m of a 55-m-wide forest (Lowrance, 1992). Designers of agroforestry practices should pay particular attention to the interfaces of woody and nonwoody components within their systems (Dix et al., 1995).

Temporal variability also is important. Some variability, such as diurnal and seasonal environmental change or longer-term successional change, is predictable and can easily be considered in designing practices; for example, a winter wheat (*Triticum aestivum* L.) alley cropping practice in which the wheat completes most of its growth before the tree crop leafs out each spring (Chirko et al., 1996). Other sources of variability (e.g., drought) are less predictable but no less important to practice design and function.

2. Disturbance is a Primary Determinant of Ecosystem Structure and Function.

Ecosystems have adapted to various degrees and combinations of fire, drought, wind, flood, pest outbreaks, and other disturbances. Much of the heterogeneity in landscape structure is due to patterns of disturbance. Removal of a critical disturbance, for instance, fire from an oak savanna, is a major disruption of system function and may trigger a structural shift to a closed forest (Bragg et al., 1993).

Management of agroforestry practices requires the appropriate application of disturbance to maintain the state that best meets management goals. In conventional row crop agriculture, tillage, a type of disturbance rarely seen in natural ecosystems, is required to prevent normal successional processes. Agroforestry managers need to consider the use of fire, grazing, and selective cutting to mimic natural disturbance patterns and maintain the agroecosystem at a later stage of succession.

Agroforestry practices must also be designed to handle sporadic, though inevitable, environmental stresses such as drought, high wind, intense rain, and extreme cold. Windbreaks, riparian forests, silvopasture, and other agroforestry practices that add perennial crops and groundcover to the farm will generally increase the system's resilience and resistance to these stresses.

3. Perennialism is the Most Common Condition in Natural Ecosystems.

Annual plants dominate only after certain disturbances and are quickly replaced in the successional process by perennials. Disturbances severe enough to provide an opening for annuals also provide a window for accelerated loss of soil and nutrients from the system. These windows are generally short in natural systems, but if the disturbance is repeated regularly as in row crop agroecosystems, the cumulative losses of soil and nutrients can greatly reduce the productive capacity of the system.

Agroforestry practices provide one means of adding perennials to a conventional row crop farming system. There are many nonwoody perennials such as grasses, alfalfa (*Medicago sativa* L.), and clover (*Trifolium* sp.), however, that can also provide benefits. An optimal agroecosystem design will consider all potential perennial crops as well as appropriate annual crops.

4. Structural and Functional Diversity are Important to Ecosystem Performance, but are Difficult to Quantify.

If an ecosystem includes species whose roots exploit different soil depths or whose leaves capture sunlight at different heights in the canopy, this structural and functional diversity may increase the efficiency of the system in using resources and maintaining production. Many species function similarly, however, so species diversity alone is a poor measure of functional diversity (Olson & Francis, 1995). Agroforestry provides an obvious way to increase the structural diversity of a row crop farm. While this change in diversity will undoubtedly have functional effects, the farm manager needs to carefully consider the relationship between structure and function as it applies to his/her management goals. Random additions of woody perennials will increase species diversity but are unlikely to produce an optimal result.

OPTIMIZING AGROECOSYSTEM PERFORMANCE WITH AGROFORESTRY

We continue our discussion of the ecological principles underlying temperate agroforestry by examining the tall-grass prairie–oak–hickory forest transition of the Midwest, with emphasis on the Western Corn Belt ecoregion (Omernik, 1987). This ecoregion is one of the premier agricultural areas in the USA. Encompassing most of Iowa, southern Minnesota, eastern Nebraska, northeastern Kansas and northwestern Missouri, the Western Cornbelt is a major producer of corn, soybeans, and livestock and represents a region where agroforestry has tremendous potential.

Prior to European settlement, an east-west transect through the region crossed ecosystems dominated by perennial grasses, trees, or various mixtures of the two. Many of the tree–grass combinations described previously in this chapter were present in the native ecosystems of this region, providing an extensive ecological foundation for the adoption of agroforestry. An understanding of these native systems is the first step in designing agroforestry practices appropriate to the region.

General Overview of Native Systems and Farming Systems

Tallgrass Prairie

The tallgrass or bluestem (*Andropogon* sp.) prairie (Küchler, 1975) occupied the eastern third of Kansas and Nebraska and the northern one-half of Iowa, with extensions northward into southwestern Minnesota and along the eastern borders of the Dakotas. To the east and southeast, the tallgrass prairie merged as an extensive savanna-like ecotone with the oak–hickory forest. Dominant species included big bluestem (*A. gerardii* Vitman), little bluestem (*Schizachyrium scoparium* (Michx.) Nash), switchgrass (*Panicum virgatum* L.), and indiangrass [*Sorghastrum nutans* (L.) Nash]. Reaching heights of 1 to 3.3 m, these grasses formed a deep carpet extending across a landscape of moderately undulating terrain. Although three or four grass species often comprised more than one-half of the biomass in the prairie, 77 or more plant species, with forbs in the majority, might occupy a square kilometer (200 species per square mile). Trees and shrubs occurred primarily as bands along streams or as occasional patches or scattered individuals (Kucera, 1992).

The climate of the tall-grass prairie is characterized by dry, cold winters and hot summers with maximum precipitation during the spring and summer. Potential evapotranspiration during the growing season generally exceeds precipitation, resulting in depletion of soil moisture and imposition of water stress on vegetation. Severe droughts are common, and these dry extremes along with fire and grazing prevented the encroachment of trees into the native prairie. Grasses have morphological and physiological adaptations to fire, grazing, and drought that make them superior competitors to trees in this environment (Risser, 1985). In fact, productivity of the tallgrass prairie may decline without periodic fire and grazing (Knapp & Seastedt, 1986).

The climate and grassland vegetation of this region, acting upon a variety of parent materials, led to the development of typical Mollisols (Buol et al., 1989). These soils are characterized by high concentrations of organic matter and nutrients, and low to moderate leaching. Within the tallgrass ecosystem, the majority of living biomass, total organic matter, and nutrients is belowground. This is a key reason why this ecosystem is very resilient to surface disturbances such as fire and grazing—most of the ecosystem is not affected directly. Mollisols, however, are excellent agricultural soils, and virtually all of the tallgrass prairie has been converted to cropland. Tillage has a direct impact on the predominant source of biomass in the ecosystem and often leads to significant and long-term degradation (Dormaar & Smoliak, 1985).

Oak–Hickory Forest

This region represents the western-most part of the Eastern Deciduous Forest, and extends from Texas to Canada (Braun, 1967). The central portion is the best developed and, especially in the Western Cornbelt, contains some of the most highly productive agricultural and hardwood forest lands. Many oak–hickory stands, however, occupy sites that were severely disturbed (e.g., by clearcutting, fires, grazing, or cultivation) during the 1800s, and then abandoned, especially in the Depression years, due in part to reduced soil quality.

The region has a continental climate with warm summers and cold winters. More than 50% of the total precipitation falls during the growing season, and the overall water balance is favorable for tree growth. Oak–hickory forest soils include relatively young, highly productive agricultural soils that developed from loess deposits over glacial and rock parent materials (Mollisols and Alfisols), and older, more highly weathered soils that escaped glaciation (Ultisols) (Parker & Merritt, 1994). Both the oaks and hickories are widespread ecologically and diverse genetically (Barnes, 1991). Oaks, with their deep-penetrating taproot and fire-resistant bark, are especially well-suited to the conditions in the tree-grassland interface.

Farm Characteristics in the Western Corn Belt

The average farm size in Iowa is 133 ha (328 acres; Bultena et al., 1995) with a trend toward larger farms in the west. Since 1940, the average farm size has increased 88% while the number of farms has decreased 51% (Stauber et al., 1995).

More than one-half the farmland in Iowa is rented, and the average age of an Iowa farmer is more than 50 yr, both important considerations in the adoption of a long-term practice like agroforestry. Approximately 85% of Iowa crop land is planted to either corn or soybeans, most often in rotation, and most crop land is not irrigated.

In 1994, >95% of Iowa row crop hectares were treated with herbicides, while 70% of the corn hectares and 50% of soybean hectares were cultivated to control weeds (Weber, 1996). Total weed control costs averaged about \$74 per hectare (\$30 per acre; Keeney, 1996). Corn crop land usually receives >112 kg ha^{-1} (100 pounds acre^{-1}) of N per year in the form of chemical fertilizers. More than 50% of the original topsoil has been eroded in Iowa (Pimentel et al., 1995), and crop land erosion continues at an average of 15 Mg ha^{-1} (6.7 tons acre^{-1}; USDA, 1995). Average yearly net income per farm is about \$22,500 (Watt, 1995).

Comparing the Ecological Condition of Native Systems and Farming Systems in the Western Corn Belt

Indicators of Ecosystem Condition

The questions to be addressed are how has conversion to farming changed the structure and function of the native ecosystems of this region; and how can agroforestry be used to restore to Corn Belt agroecosystems some of the beneficial ecosystem characteristics that have been lost? To answer these questions, we must first consider how to compare the condition of different ecosystems.

A tall-grass prairie, oak–hickory forest, and corn–soybean farm are characterized in Appendix 2–1A. Each site is assumed to be 263 ha (650 acres) and broadly representative of these systems in the western Corn Belt ecoregion. These are not actual sites, but represent a synthesis of data from many different sources (e.g., for the prairie, Sims & Singh, 1978; Webb et al., 1978; Risser & Parton, 1982; and for the oak–hickory forest, Harris et al., 1975; Rochow, 1975). This approach is necessary because no single prairie or oak–hickory site within this region has been fully characterized.

A quick review of the fundamental environmental (physiography, climate, soils), structural (physiognomy, species composition, biomass distribution), and functional (energy flow, nutrient cycling, hydrology) characteristics of the three ecosystems (Appendix 2–1A) suggests that conversion of either prairie or oak–hickory forest to a grain farm results in major reductions in species diversity, a decrease in the relative importance of belowground biomass, an increase in the flux of nutrients through the system, and export of significant amounts of net primary production.

To evaluate these structural and functional changes, trends expected in stressed ecosystems (Odum, 1985) were selected as indicators of ecosystem condition. Most of these trends are apparent in the conversion of prairie and forest to corn–soybean farming (Table 2–2). Expressed in familiar agronomic terms, grain farms in the western Corn Belt suffer from excessive soil erosion and loss of soil organic matter, require large amounts of cultural energy to maintain fertility and control pests, and contribute to off-site environmental degradation through pollution of surface- and groundwater.

Many of the negative effects accompanying the conversion of native ecosystems to agriculture are the result of the replacement of perennials by annuals and the corresponding maintenance of the system at an early rather than a late stage of succession. Agroforestry by definition is a perennial practice, and its inclusion in a farming system will increase the perennialism of the system with likely concurrent improvements in function.

In the remainder of this chapter, we will examine the system-level effects of adding agroforestry practices to a conventional farm. To fully judge the effects, however, indicators of economic and social sustainability as well as of ecological sustainability must be considered. To be sustainable, an agroecosystem has to make a profit and it has to meet societal demands for food and fiber. If changes to a farm are made solely to improve the ecological trends in Table 2–2, the effect on the overall system may be negative.

Indicators of Agroecosystem Sustainability

The issue of sustainability and the choice of indicators of agroecosystem condition have been considered frequently (e.g., Harrington, 1992; Lefroy & Hobbs, 1992; Stockle et al., 1994; Campbell et al., 1995). Although the debate continues about which group of indicators is most appropriate, there has been considerable convergence among the choices. We have compiled a suite of indicators (Table 2–3) based on our examination of Appendix 2–1A, Odum's trends (Table 2–2), and the work of Francis et al. (1997) on indicators of functional sustainability of farms. This group of indicators reflects our summary view of agroecosystem sustainability, i.e., in an increasingly resource-poor world, farms that maintain a high rate of conversion of solar energy into marketable crops, minimize ancillary energy and material inputs, and preserve their natural capital (e.g., soil) will be the most sustainable.

Although it is fairly easy to determine which trend in an indicator favors sustainability, it is more difficult to quantify the particular values of an indicator that represent high or low sustainability. As indicated in the footnotes to Table 2–3, we set upper and lower bounds for our indicators based on benchmark farming systems in the region such as irrigated continuous corn (e.g., high energy inputs), on the properties of a particular soil [e.g., 11 Mg soil erosion ha^{-1} (5 tons $acre^{-1}$) is the tolerance limit for a Sharpsburg silty clay loam, 4–6% slope; fine, montmorillonitic mesic Typic Argiudoll], or on economic benchmarks (e.g., the poverty level for a family of four). The goal is to ground the evaluations in a realistic assessment of the range of conditions in the region of interest.

Once indicators of sustainability have been defined, they can then be used to evaluate the effect of agroforestry practices on the sustainability of a western Corn Belt farming system. We have taken two approaches to demonstrate this process: a qualitative narrative based on interviews with two farmers in the region who practice agroforestry (see Case Studies 1 and 2) and a quantitative comparison of two synthetic farms modeled from regional data (Table 2–4). One of the synthetic farms is the conventional corn–soybean farm described in Appendix 2–1A, while

Table 2–2. Trends expected in stressed ecosystems (Odum, 1985) and the evidence for these trends in a corn–soybean farm relative to a prairie or oak-hickory ecosystem (drawn from Appendix 2–2A).

Trend	Farm characteristics that support Odum's trend relative to native ecosystems
Energetics	
1. Community respiration increases	Tillage increases decomposition of soil organic matter
2. P–R (production–respiration) becomes unbalanced (< or > 1)	System production exceeds respiration due to export of NPP from system
3. P–B and R–B (maintenance/biomass structure) ratios increase	Data not available
4. Importance of auxiliary energy increases	17.3×10^3 MJ ha^{-1} (1670 Mcal $acre^{-1}$) input (as fertilizer, fuel, labor, and more)
5. Exported or unused primary production increases	450 g kg^{-1} (45%) of NPP exported as grain
Nutrient cycling	
6. Nutrient turnover increases	see # 7
7. Horizontal transport increases and vertical cycling of nutrients decreases	Internal N cycling decreases from 960 to 560 g kg^{-1} (96 to 56 %) of total N flows
8. Nutrient loss increases (system becomes more "leaky')	Loss of N from farm is 7 to 50 times greater than from natural ecosystems
Community structure	
9. Proportion of *r*-strategists increases	Annual crops replace perennials
10. Size of organisms decreases	Corn smaller than oaks and soybeans smaller than tall grasses
11. Lifespans of organisms or parts (e.g., leaves) decrease	Crops are annuals
12. Food chains shorten	Not shortened, but food web complexity probably reduced as one consumer (humans) co-opts almost one-half of NPP
13. Species diversity decreases and dominance increases	Two species dominate
General system-level trends	
14. Ecosystem becomes more open (i.e., input and output environments become more important as internal cycling is reduced)	Inputs of cultural energy and chemicals, and export of harvested crops are essential to system maintenance
15. Autogenic successional trends reverse (succession reverts to earlier stages)	System maintained at first year of secondary succession by annual tillage
16. Efficiency of resource use decreases	Annual NPP reduced despite large inputs of external materials and energy
17. Parasitism and other negative interactions increase, and mutualism and other positive interactions decrease	Chemical and energy inputs required to reduce specific pest populations on specific hosts
18. Functional properties (such as community metabolism) are more robust (homeostatic-resistant to stressors) than are species composition and other structural properties	Despite drastic reduction in biodiversity and simplification of structure, system continues to be productive

the other is a more diversified farm that incorporates windbreaks, an herbaceous perennial crop, and two woody perennial crops in block plantings.

The size and machinery complement of each synthetic farm was determined from a survey and analysis of Nebraska farms (Bernhardt, 1994), and a schedule of operations was developed for each farm based on best management practices for

Table 2–3. Selected indicators of sustainability for agroecosystems, and the indicator values for the conventional and agroforestry farms described in Table 2–4.

Indicator	Definition	Value indicating high sustainability	Value indicating low sustainability	Conventional farm	Agroforestry farm
Harvest†	weight of harvested crops and livestock in kg ha^{-1} (lb A^{-1}), dry weight	7952 (7100)	0	3805 (3397)	3923 (3503)
Cultural energy input‡	Total nonsolar energy inputs MJ ha^{-1} (MJ A^{-1})	0	59 259 (24 000)	17 264 (6992)	14 091 (5707)
Energy output–input§	Ratio of energy in harvested crops to cultural energy inputs	5	< 1	3.9	4.5
Energy capture efficiency¶	Energy in harvested crops as % of growing season PAR	1.0	0	0.38	0.35
Water use efficiency#	Harvested biomass (g m^{-2}) divided by AET (mm)	1.15	0	0.61	0.61
Imported fertilizer††	N + P in kg ha^{-1} (lb A^{-1})	0	151 (135)	44 (39)	26 (23)
N losses‡‡	N losses in kg ha^{-1} (lb A^{-1}) (erosion and leaching)	0	45 (40)	28 (25)	20 (18)
Soil erosion§§	Wind + water in Mg ha^{-1} (tons A^{-1})	0	11 (5)	11 (5)	7.8 (3.5)
N balance¶¶	N inputs/N outputs	1	<0 .8 >1.2	0.6	0.64
P balance##	P inputs/P outputs	1	<0.8 >1.2	0.7	0.46
Crop diversity†††	Number of crops per farm	12	1	2	7
Hired labor‡‡‡	Hours ha^{-1} (acre)	0	5 (2)	1 (0.4)	5 (2.0)
Net income§§§	Dollars ha^{-1} (acre)	235 (95)	89 (36)	99 (40)	249 (101)
Capital borrowing¶¶¶	Debt/variable income	0	1	0.63	0.46
Farmer knowledge###	Total skills and knowledge held by farm family	high	low	medium	high

† High value is dry weight of grain from Nebraska irrigated corn (9406 kg ha^{-1} (150 bu A^{-1})).

‡ The value indicating low sustainability is the energy input per hectare to produce irrigated corn in Nebraska (Pimentel, 1980).

§ From Pimentel (1980), energy output/input ratio for U.S. soybean production is 4.15:1; Ohio alfalfa is 6.17:1; corn and wheat are around 2.5:1. So 5:1 is a reasonable upper end to scale.

¶ Loomis & Connor (1992) show that the theoretical maximum daily energy capture efficiency of a crop is 12% PAR (photosynthetically active radiation). Tivy (1990, p. 109), however, writes that only in exceptional cases do crop efficiencies exceed 2% PAR for an entire growing season, and efficiency in terms of economic yields is only 0.3 to 0.4%. If 2% capture of PAR is a high efficiency, then 1% PAR in harvest (50% of total NPP harvested) is a high upper bound for energy capture efficiency.

1.15 is the water use efficiency for corn (grain only) on a central Iowa farm (Loomis & Connor, 1992).

†† Irrigated corn yielding 9406 kg ha^{-1} (150 bu A^{-1}) would export 128 kg ha^{-1} (114 lb A^{-1}) N and 22 kg ha^{-1} (20 lb A^{-1}) P.

‡‡ High value (45 kg ha^{-1}(40 lb A^{-1})) is two times the estimated N losses for corn on a central Iowa farm (Loomis & Connor, 1992).

§§ 11.2 Mg ha^{-1} (5 tons A^{-1}) is *T*-value for Sharpsburg silty clay loam, 4-6% slope.

¶¶ System outputs (harvest and losses) within ±20% of inputs (imported and N-fixation) is considered close to balance (inputs/outputs = 1). Values ≥1would indicate potential environmental problems or depletion of fertility.

System outputs (harvest and losses) within ±20% of inputs (imported P) is considered close to balance (inputs/outputs = 1). Values ≥1 would indicate potential environmental problems or a depletion of fertility.

††† Bender (1994) grows 12 crops on his eastern Nebraska organic farm. Diversity of this magnitude is required to implement flexible rotations for weed control and fertility, and provide sod and pasture crops for grazing and erosion control.

‡‡‡ Irrigated corn in Nebraska requires 5 h labor ha^{-1} (2 h labor A^{-1}) (Selley, 1996).

§§§ A 172-ha (425-A) farm would have to generate \$89 ha^{-1} (\$36 A^{-1}) in net income to keep a four-person family above the official poverty line (\$15,141; USDOC, 1996; Table 732). An average size Nebraska cash grain farm (255 ha (630 acres)) generating \$235 ha^{-1} (\$95 A^{-1}) would be in the 90% of net farm income for that type of farm. (B. Johnson, 1995, unpublished data.

¶¶¶ A value of 1 indicates that the income remaining after fixed costs are covered is just sufficient to repay operating loans plus interest.

This is very difficult to quantify, but it is assumed to be positively correlated with the number of crops and enterprises on the farm.

Table 2–4. Characteristics of two model farms in eastern Nebraska representing a conventional cash grain operation and an agroforestry alternative.

Characteristic	Conventional farm	Agroforestry farm
Size in ha (A)	264 (650)	172 (425)
Rented land (%)	55	0
Crops in ha (A)		
corn	132 (325)	34 (83)
soybeans	132 (325)	61 (151)
grain sorghum		34 (83)
alfalfa		24 (60)
Christmas trees		4 (9)
hazel nut production		6 (16)
windbreaks		9 (23)
Area in perennials (%)	0	25
Soil type	Sharpsburg silty clay loam, 4–6% slope	Sharpsburg silty clay loam, 4–6% slope

east-central Nebraska. Economic performance of the two systems was then quantified with a model developed by Olson (1998), and erosion and nutrient losses were evaluated with PLANETOR, a farm-scale environmental and economic model (Center for Farm Financial Management, University of Minnesota). Energy and nutrient budgets for each farm were compiled from published values of the embodied energy of farm inputs (Pimentel, 1980) and crop nutrient and energy contents (Church, 1984; Holland et al., 1991b).

The values of each indicator for the two farms are given in Table 2–3. We also present these data in the form of cobweb polygons (Gomez et al., 1996) which provide a simple visual comparison of the whole system (Fig. 2–3). The high and low sustainability values for each indicator (Table 2–3) are standardized to form 100-point scales along the spokes of the polygon. Because the low sustainability values are located at the center of the diagram, a larger-area polygon implies a more sustainable system.

A rapid appraisal of the polygons (Fig. 2–3A, B) and Table 2–3 suggests that the agroforestry farm is more sustainable than the conventional corn–soybean farm. Although the systems perform similarly as measured by production indica-

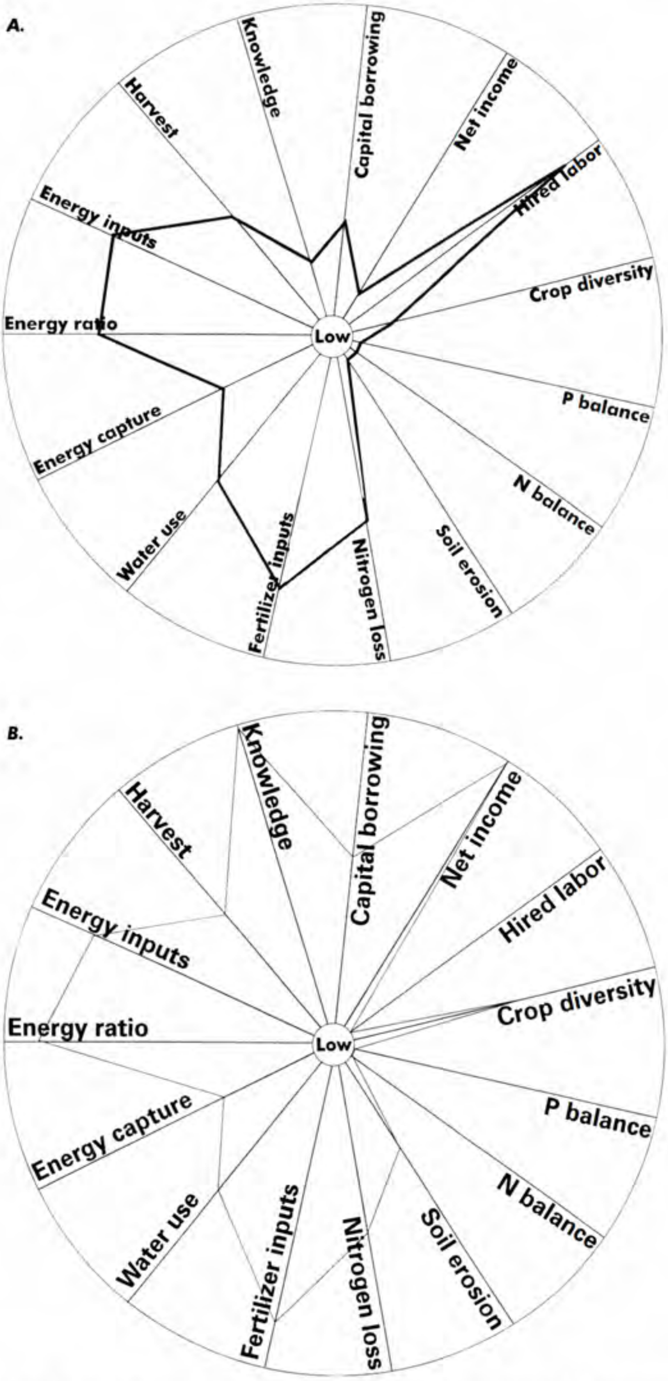

Fig. 2–3. Cobweb polygon presentation of standardized values for sustainability indicators. (A) Conventional corn–soybean farm and (B) diversified agroforestry farm. See text for details.

tors (e.g., harvest, energy capture efficiency, water use efficiency), the agroforestry farm does better economically (net income, capital borrowing) and in some measures of resource conservation (e.g., erosion, N loss). Neither system has a sustainable nutrient balance in that each exports considerably more N and P than it imports.

Of course, there is no way to tell from system-level indicators how much of the improvement in the performance of the agroforestry farm is due to its woody perennial components. The underlying performance data (not shown) indicate that the tree components had a major impact on economic returns. Christmas trees and hazel nuts (*Corylus* sp.) were very profitable, and windbreaks increased crop yields more than enough to compensate for the land taken out of production. Tree crops (with grassed alleys) eliminated water erosion on the land they occupied, although for the whole farm, alfalfa was equally important in reducing water erosion. Windbreaks provided no benefit in reducing wind erosion because soil loss by wind is insignificant on these soils when adequate residue is left each fall.

A final observation concerns the definition of agroforestry. The windbreaks on this model farm, by interacting with the field crops (biologically and physically), clearly meet the definition of agroforestry. The Christmas trees and hazel shrubs, although woody perennials, are planted in blocks and may have only minimal biophysical interaction with other components of the farming system. Does the inclusion of block plantings of trees on a farm necessarily constitute agroforestry? Not by the definition given earlier in this chapter, the derivation of which is fully explored in Gold et al. (2000, this publication).

Without question, when one considers the distribution of labor (data not shown) on these two farms, there are advantages to having the woody perennials. The conventional farmer is very busy in the spring and early fall with much less to do in-between. On the agroforestry farm, the hazels require a great deal of labor for harvest in late July and early August, and Christmas tree sales provide work in late November and December. The inclusion of block plantings of tree crops represents both an economic and a social interaction with other components of the farm (see Kurtz, 2000, this publication; Rule et al., 2000, this publication) but not necessarily one of a biophysical nature. Agroforestry, in North America, is currently defined in terms of five individual practices (see Gold et al., 2000, this publication). As it continues to evolve, however, a broader definition at farm and landscape scales may become appropriate.

Case Study 1: Dan Shepherd; Randolf County, Missouri

Farm Description

Dan Shepherd owns a 1215-ha (3000-acre) farm in north-central Missouri, approximately 72 km northwest of Columbia, on which he raises nuts, bison (*Bison bison*), eastern gammagrass [*Tripsacum dactyloides* (L.) L.] seed, and other crops. The farm contains both uplands and bottomlands. Historically the uplands supported prairie grasses while the bottomlands supported trees. Today the bottomlands are mostly protected from flooding by levees, though in the past the area flooded regularly. Most of his nut trees are planted in the protected bottomlands, although approximately 4 ha (10 acres) of walnuts have been planted on bottomlands outside the levee, and 2 ha (5 acres) of butternuts (*Juglans cinerea* L.) and heart nuts (*Juglans sieboldiana* var. *cordiformis* L.) are on upland soils.

The farm lies in a frost pocket, which limits the crops that can be produced. Fruit trees that thrive a few miles away sustain frost damage in this area. The bottomland soils on Dan's farm are poorly drained. Wetness is the major limitation of these soils though they are suitable for both nut tree and row crop production. The upland soils also are generally poorly drained. They can produce row crops and trees, but are often best suited to pasture (Randolf County Soil Survey).

Enterprises

Dan has three main enterprises: pecan [*Carya illinoensis* (Wangenh.) K. Koch] production, bison meat, and eastern gammagrass seed production. Secondary crops produced include walnuts, butternuts, heart nuts, forage grasses, corn, and soybeans. Dan has been using the bottomlands to grow pecans and walnuts intercropped with corn and soybeans and the uplands to pasture his bison and grow eastern gammagrass for seed production. With 446 ha (1100 acres) planted to eastern gammagrass, Dan is the largest eastern gammagrass seed producer in the world.

Ecological Interactions

Succession: Dan has been alley cropping between his nut trees for more than 10 yr. His pecans are planted in a 12 m × 9 m (40 × 30 ft) grid pattern. Between pecan rows he now plants corn that he eventually feeds to the bison. When the trees were smaller he grew a corn–soybean–wheat rotation, but now that they are larger he is limited to six rows of corn between tree rows. As the pecan yields have increased, the crop yields have decreased. Dan expects that within 2 yr he will probably have to switch from row crops to grass, probably bluegrass (*Poa pratensis* L.), because it grows well in the bottomlands and doesn't compete with the pecans. Dan's butternut and heart nut trees are growing on upland soils. He grows tall fescue (*Festuca arundinacea* Schreb.) under these trees for erosion control and harvests it as feed for the bison.

Competition: The trees and crops in Dan's alley cropping practice compete for nutrients and light. The bottomlands are wet enough, however, that there is little problem with competition for water except with soybeans in very dry years. Light is the primary factor of concern. As the trees have grown taller and developed fuller crowns they have gradually shaded the row crops to the point where only a few rows will thrive. Most farm equipment is designed for wide passes. When the trees were small there was ample room, but now Dan has trouble maneuvering equipment in the alleys without damaging the trees. Also, the herbicides he uses on his row crops are not necessarily good for the trees. One advantage of the alley cropping, though, is that the trees no longer need individual fertilizer applications. They can get the nutrients they need from the crop land applications. Another advantage is that, especially on the higher ground, the trees provide needed shade for the grass species growing beneath them, a benefit whose effects are noticeable in the hottest part of the summer.

Insects and other Pests: Dan has not noticed either a decrease or an increase in pests due to the interactions among the woody and herbaceous crops. The trees tend to be plagued by walnut caterpillars (*Datana integerrima* Grote & Robinson), shoot borers (*Acrobasis demotella* Grote), and fungi (scab) with or without the crop present. The corn is susceptible to root worm with or without the trees.

Wildlife: The trees on Dan's farm contribute significantly to wildlife. An avid hunter, Dan sees the wildlife (except the crows) as beneficial additions to his farm. Species that use the trees for cover or nesting include turkey (*Meleagris gallopavo*), deer (*Odocoileus* sp.), quail (*Colinus virginianus*), and dove (*Zenaida macroura*).

Other: Due to concerns with soil compaction and contamination of cracked nuts with feces, the bison are not grazed among the trees, nor does Dan grow eastern gammagrass between the trees. Eastern gammagrass is a bunch grass that is difficult to mow evenly, a necessary condition for nut harvest.

Dan sees the trees as providing significant intangible benefits to his farm. Not only do they provide recreation in the form of added hunting opportunities, they also provide aesthetic value and are excellent advertising for the farm.

Case Study 2: Calvin Kissick; Reno County, Kansas

Farm Description

Calvin Kissick owns a 49-ha (120-acre) farm in Reno County, Kansas, near the town of Haven. The farm is bounded on the north by the Arkansas River, and Gar Creek runs east–west through the property. Row crops, primarily wheat, sorghum [*Sorghum bicolor* (L.) Moench.], and occasionally soybeans are grown on a 14-ha (35-acre) parcel south of the creek. North of the creek are approximately 34 ha (85 acres) of native grass that Calvin leases out for grazing. Four to five percent of the land is in trees.

The soils on the farm are primarily sandy loam flood plain soils susceptible to both wind and water erosion. They are moderately fertile but crop production is frequently limited by flooding. The climate is continental with an average annual precipitation of 762 mm (30 inches). Prevailing winds are from the south–southwest in summer and from the north in winter.

Agroforestry Practices

Gar Creek is bordered on the north by an established riparian forest 24 to 61 m (80 to 200 ft) wide that is separated by a fence from the grazing land. A 14-m (45-ft) well with a solar-powered pump provides water for the cattle. On the south side of Gar Creek, Calvin is planting a second riparian forest buffer consisting of two rows of trees [black walnut, bur oak (*Quercus macrocarpa* Michx.), and green ash (*Fraxinus pennsylvanica* Marsh.)] 4.5 m (15 ft) apart. The southern border of the farm is lined with a windbreak planted in the 1930s as part of the Prairie States Forestry Project. The windbreak consists of two rows of cedars (*Juniperus* sp.), four rows of green ash, and a one-row Osage orange [*Maclura pomifera* (Raf.) Schneid.] hedge. The northern and eastern edges of the farm also are lined with Osage orange hedges planted in the 1800s to mark the boundaries.

Ecological Interactions

Soil protection and microclimate modification: The southern windbreak and perhaps to some extent the northern riparian forest provide wind protection for the crop land in summer and winter. Calvin is convinced that had the windbreak not been planted on the south side of the farm, the soil would have been lost to wind erosion. The windbreak also moderates temperature, protects the crop from wind damage, and helps retain moisture. The benefits to the crops of these microclimate modifications are visually apparent in a 30-m (100-ft) -wide band of healthier crops that starts 15 to 30 m (50–100 ft) north of the windbreak.

Since it was fenced from grazing, the north bank of Gar Creek has begun to stabilize. Grass and other herbaceous vegetation have increased on the banks, and the water is clearer. Calvin is confident that the new plantings will eventually stabilize the south bank and further improve water quality.

Succession: The 1930s windbreaks are showing their age. Wind damage is extensive, some trees have died, and most need trimming. Some natural regeneration of hackberry (*Celtis occidentalis* L.) and other trees is occurring, but soon the windbreak will need to be replaced. At that time, Calvin plans to plant two rows rather than seven rows of trees. In the pasture north of the riparian area, tree invasion is an ever-present problem. Calvin mows the pasture annually to control unwanted trees and other weeds.

Competition: Competition between crops and trees is noticeable along the edge of the windbreak where the first few rows of crops show signs of moisture stress. Competition for light and nutrients also may occur. Calvin is planning to experiment with a root plow to reduce belowground competition.

Riparian buffer establishment: Development of a riparian forest along the south side of Gar Creek has been difficult. During the past 3 yr, Calvin has had to replant about 25% of the trees. Bur oak and black walnut have performed the best and green ash the worst.

Calvin installed weed barrier fabric at planting and mows between the seedlings two to three times a year to control weeds. Black walnut growth was initially slowed by a smut, but has since fully recovered. Insect damage to the seedlings has been minimal, but deer browsing has been a problem.

Wildlife: In the riparian area north of Gar Creek, deer, wild turkey (*Meleagris fallopavo*), and song birds have increased since the cattle were excluded. The windbreak harbors rabbits (*Sylvilagus floridanus*), an occasional deer or racoon (*Procyon lotor*), and a variety of birds. Calvin's brother owns 57 ha (140 acres) of forested land adjacent to Calvin's farm. Deer use this larger woodland for shelter, but follow the riparian corridor to Calvin's property to feed on pasture and crops.

CONCLUSIONS

Agroforestry offers a means of regaining some of the structural and functional characteristics that contribute to the sustainability of natural ecosystems, but were lost during the conversion of those ecosystems to agroecosystems. An understanding of the structure and function of natural ecosystems, of "what we are undoing," is essential to the successful implementation of agroforestry.

A complete knowledge of the many ecological processes and interactions responsible for a natural system's sustainability will always elude us; an ecosystem is just too complex. Perennialism and a high proportion of area in mid- to late-successional states, however, is the usual condition of natural ecosystems and an obvious goal in designing a sustainable agroecosystem. We are making progress in determining how to meet that goal, but there is a great deal left to be learned, both in basic and applied ecology. Some of the areas warranting further research include:

- belowground interactions such as microbial ecology, root competition, and mycorrhizal associations,
- beneficial interactions among trees, shrubs, grasses, and forbs,
- system-level experiments and on-farm demonstrations including economic analyses, and
- conservation biology for natural ecosystems—how can we preserve as fully functional ecosystems (and as models for agroecosystems), the few remnants of natural systems that remain?

We close with a cautionary note. Natural ecosystems are amazingly tough and resilient, and when converted to agroecosystems can be incredibly productive; however, they have their limits. Millions of hectares of desertified and abandoned farmland are testimony to the ability of humans to abuse an ecosystem beyond its capacity to recover. Greatly increased inputs of energy and chemicals combined with some major genetic manipulations have allowed production from the remaining agroecosystems to keep pace so far with most human needs for food and fiber.

The oil is running out, however, the human population increases by 90 million per year, and we continue to pave, erode, and otherwise destroy farmland while demanding more and more material goods. Agroforestry can help make our farming systems more efficient and sustainable, but it cannot extend the natural limits of ecosystems to capture and convert sunlight to human use.

Appendix 2–1A

Comparative characteristics of a native tallgrass prairie, conventional corn–soybean farm, and oak–hickory forest.

Characteristic	Native tallgrass prairie	Conventional farm	Oak–hickory forest
Physiography	Central lowlands of level to undulating terrain	Central lowlands of level to undulating terrain	Central lowlands and Ozark Plateaus
Climate	eastern Kansas	east central Nebraska	north central Missouri
Mean annual temperature	12.8°C	10.0°C	12.2°C
Mean January temperature	−2.3°C	−6.2°C	−2.5°C
Mean July temperature	26.7°C	25.0°C	25.5°C
Average frost-free period, d	190	167	194
Mean annual ppt.	800 mm	695 mm	1015 mm
Quarterly ppt., % of annual (Jan.–Mar./Apr.–June/...)	12/38/32/18	12/40/33/15	18/32/27/23

(continued on next page)

Characteristic	Native tallgrass prairie	Conventional farm	Oak–hickory forest
Annual ppt. variability (1961–1990)	no. yr < 75% of ave. = 4 no. yr > 125% of ave. = 4	no. yr < 75% of ave. = 5 no. yr > 125% of ave. = 7	no. yr < 75% of ave. = 3 no. yr > 125% of ave. = 2
Hydrology			
Actual ET	760 mm	649 mm	747 mm
Potential ET	780 mm	704 mm	842 mm
Percolation/runoff	40 mm	46 mm	268 mm
Soil			
Predominant soil order	Mollisol	Mollisol	Alfisol
Suborder	Udoll	Ustoll	Udalf
Total soil organic matter, 0–100 cm	31600 g m^{-2}	19640 g m^{-2}	7027–13 403 g m^{-2}
Soil N, 0–100 cm			
Inorganic N	4.5 g N m–2	4.6–11.0 g N m^{-2}	3.2–6.1 g N m^{-2}
Organic N	1550–1580 g N m^{-2}	982 g N m^{-2}	410–782 g N m^{-2}
Community, Composition, and Structure			
Physiognomy	Grasses dominant with a closed canopy varying from 0.5 to 3.3 m	Maize or soybean monocultures with closed canopies of uniform height (2.5–3 m maize; 0.7–1 m beans).	Broadleaf deciduous forest in which 50% or more of the trees are oaks or hickories with a closed canopy varying from 5.5–8.3 m (ave. 6.3).
Dominant floristic type	Perennial grasses	Annual grass (maize) and annual broadleaf (soybean)	Perennial broadleaf trees
Species richness, 250 ha	200–300 vascular plant species	dominance by two crop species	approx. 94 vascular plant species
Species eveness	>50% of aboveground biomass in 3 to 4 species	>90% of aboveground biomass in two species	>90% of aboveground biomass in 3 to 4 species
Average annual maximum aboveground biomass (includes litter)	650–1075 g m^{-2}	870 g m^{-2} (live only)	10 934 g m^{-2}
Average annual maximum belowground biomass	1869–1985 g m^{-2}	160 g m^{-2}	2592 g m^{-2}
Belowground OM as % of total OM (includes living and SOM)	>98%	96%	47–59%
Leaf area index	4 m^{2} m^{-2}	4–4.5 m^{2} m^{-2}	2.5–4.5 m^{2} m^{-2}
Root depth distribution	90% of root biomass in top 50 cm	>90% of root biomass in top 75 cm	>50% in upper 30 cm
Energy Flow			
Solar radiation, growing season PAR	2227 × 10^{3} kJ m^{-2}	1772 × 10^{3} kJ m^{-2}	2017 × 10^{3} kJ m^{-2}
Nonsolar energy inputs	0	17.3 × 10^{3} MJ ha^{-1} (1670 Mcal/A)	0
Annual aboveground NPP (net primary production)\	300–700 g m^{-2}	983 g m^{-2}	597.6 g m^{-2}
Annual belowground NPP	500–1200 g m^{-2}	177 g m^{-2}	950 g m^{-2}

(continued on next page)

Characteristic	Native tallgrass prairie	Conventional farm	Oak–hickory forest
%total NPP as seed–grain	6%	45%	2%
Energy capture efficiency, %	0.72–1.09	0.74	1.08
Water use efficiency	1.8–2.18	1.87	2.07
Partitioning of aboveground NPP			
Herbivores	13–45%	5%	<0.2%
Detritivores	55–87%	50%	99%
Exported from the system	0%	45%	0%
Nutrient cycling			
Ave. annual N inputs	1.0 g N m^{-2}	8.2 g N m^{-2}	2 g N m^{-2}
Ave. annual N losses	0.26–0.6 g N m^{-2}	2.2 g N m^{-2}	2 g N m^{-2}
Ave. annual N export in harvested crop	0	10.9 g N m^{-2}	0
Belowground N (% of total)	>98%	>98%	92%
Soil erosion	minimal	11 200 kg ha^{-1} yr^{-1}	minimal
Changes in soil SOM pool	stable	Loss of 30% or more during first 30 yr of cultivation	stable

REFERENCES

Altieri, M.A. 1987. Agroecology: The scientific basis of alternative agriculture. Westview Press, Boulder, CO.

Anderson, G.W., R.W. Moore, and P.J. Jenkins. 1988. The integration of pasture, livestock and widely-spaced pine in southwest Western Australia. Agrofor. Syst. 6:195–211.

Atzet, R., and R.H. Waring. 1970. Selective filtering of light by coniferous forests and minimum light energy requirements for regeneration. Can. J. Botany 48:2163–2167.

Axelrod, D.I. 1985. Rise of the grassland biome, Central North America. Bot. Rev. 51:163–201.

Barnes, B.V. 1991. Deciduous forests of North America. p. 219–344. *In* E. Rohrig and B. Ulrich (ed.) Temperate deciduous forests. Ecosystems of the World 7. Elsevier, New York.

Belsky, A.J. 1994. Influences of trees on savanna productivity: Tests of shade, nutrients, and tree-grass competition. Ecology 75:922–932.

Bender, J. 1994. Future harvest. Univ. of Nebraska Press, Lincoln.

Bernhardt, K.J. 1994. Sustainable agriculture impact study. 1994 ACE Project Annual Rep. Dep. of Agricultural Economics, Univ. of Nebraska, Lincoln.

Billings, W.D. 1969. Vegetational pattern near alpine timberline as affected by fire–snowdrift interactions. Vegetatio 19:192–207.

Blank, J.L., R.K. Olson, and P.M. Vitousek. 1980. Nutrient uptake by a diverse spring ephemeral community. Oecologia 47:96–98.

Bragg, W.K., A.K. Knapp, and J.M. Briggs. 1993. Comparative water relations of seedling and adult *Quercus* species during gallery forest expansion in tallgrass prairie. For. Ecol. Manage. 56:29–41.

Brandle, J.R., and S. Finch. 1991. How windbreaks work. Ext. EC 91-1763-B. Univ. of Nebraska, Lincoln.

Braun, E.L. 1967. Deciduous forests of eastern North America. Hafner Publ. Co., New York.

Bultena, G.L., E.O. Hoiberg, and J. Dansingburg. 1995. Impact of sustainable farming on the structure of American agriculture. p. 67–82. *In* A.R. Bird et al. (ed.) Planting the future. Iowa State Univ. Press, Ames.

Buol, S.W., F.D. Hole, and R.J. McCracken. 1989. Soil genesis and classification. Iowa State Univ. Press, Ames.

Campbell, C., W. Heck, D. Neher, M. Munster, and D. Hoag. 1995. Biophysical measurement of the sustainability of temperate agriculture. p. 251–276. *In* M. Munasinghe and W. Shearer (ed.) Defining and measuring sustainability: The biophysical foundations. United Nations Univ., New York.

Chirko, C.P., M.A. Gold, P.V. Nguyen, and J.P. Jiang. 1996. Influence of direction and distance from trees on wheat yield and photosynthetic photon flux density (*Qp*) in a *Paulownia* and wheat intercropping system. For. Ecol. Manage. 83:171–180.

Church, D.C. 1984. Livestock feeds and feeding. O&B Books, Corvallis, OR.

Cionco, R.M. 1985. Modeling windfields and surface layer wind profiles over complex terrain and within vegetative canopies. p. 501–520. *In* B.A. Hutchinson and B.B. Hicks (ed.) The forest–atmosphere interaction. D. Reidel Publ., Boston.

Cleveland, C.J., and R.K. Kaufmann. 1991. Forecasting ultimate oil recovery and its rate of production: Incorporating economic forces into the models of M. King Hubbert. Energy J. 12:17–46.

Daubenmire, R. 1978. Plant geography. Academic Press, New York.

Davies, K.M., Jr. 1994. Some ecological aspects of northeastern American Indian agroforestry practices. Northern Nut Growers Assoc. Ann. Rep. 85:25–37.

Dix, M.E., R.J. Johnson, M.O. Harrell, R.M. Case, R.J. Wright, L. Hodges, J.R. Brandle, M.M. Schoeneberger, N.J. Sunderman, R.L. Fitzmaurice, L.J. Young, and K.G. Hubbard. 1995. Influences of trees on abundance of natural enemies of insect pests: A review. Agrofor. Syst. 29:303–311.

Dormaar, J.F., and S. Smoliak. 1985. Recovery of vegetative cover and soil organic matter during revegetation of abandoned farmland in a semiarid climate. J. Range Manage. 38:487–491.

Duke, J.A. 1989. Ginseng: A concise handbook. Reference Publ., Algonac, MI.

Eichenberger, M.D., and G.R. Parker. 1976. Goldenseal (*Hydrastis canadensis* L.) distribution, phenology and biomass in an oak–hickory forest. Ohio J. Sci. 76:204–210.

Forman, R.T.T., and M. Godron. 1986. Landscape ecology. John Wiley & Sons, New York.

Francis, C.A., S.G. Aschmann, and R.K. Olson. 1997. Indicators of functional sustainability of farms and watersheds. p. 64–66. *In* J.L. Steiner et al. (ed.) Proc. of Conf. on Interactions: Investigating ecosystem dynamics at a watershed level, Athens, GA. 13–16 April 1997. USDA-ARS, Southern Piedmont Conserv. Res. Ctr., Watkinsville, GA.

Fritschen, L.J. 1985. Characterization of boundary conditions affecting forest environmental phenomena. p. 3–23. *In* B.A. Hutchinson and B.B. Hicks (ed.) The forest–atmosphere interaction. D. Reidel Publ., Boston.

Garrett, H.E., and W.B. Kurtz. 1983. Silvicultural and economic relationships of integrated forestry-farming with black walnut. Agrofor. Syst. 1:245–256.

Garrett, H.E., W.B. Kurtz, L.E. Buck, J.P. Lassoie, M.H. Gold, H.A. Pearson, L.H. Hardesty, and J.P. Slusher. 1994. Agroforestry: An integrated land-use management system for production and farmland conservation. Resource Conservation Act appraisal document. Nat. Resour. Conserv. Serv., Washington, DC.

Gold, M.A., W.J. Rietveld, and H.E. Garrett. 2000. Agroforestry nomenclature, concepts, and practices for the USA. p. 63–77. *In* H.E. Garrett et al. (ed.) North American Agroforestry: An integrated science and practice. ASA, Madison, WI (this publication).

Gomez, A.A., D.E.S. Kelly, J.K. Syers, and K.J. Coughlan. 1996. Measuring sustainability of agricultural systems at the farm level. p. 401–410. *In* J.W. Doran and A.J. Jones (ed.) Methods for assessing soil quality. SSSA Spec. Publ. 49. SSSA, Madison, WI.

Greller, A.M. 1988. Deciduous forest. p. 287–316. *In* M.G. Barbour and W.D. Billings (ed.) North American Terrestrial Vegetation. Cambridge Univ. Press, New York.

Hamerlynck, E.P, and A.K. Knapp. 1996. Environmental and physiological factors influencing the distribution of oaks near the edge of their range. p. 17–20. *In* D.H. Harnett (ed.) 14th North American Prairie Conf., Prairie Biodiversity, Manhattan, KS. 12–16 July 1994. Kansas State Univ., Manhattan.

Harrington, L. 1992. Measuring sustainability: Issues and alternatives. J. Farming Syst. Res.–Ext. 3:1–19.

Harris, B. 1986. Growing shiitake commercially. Science Tech Publ., Madison, WI.

Harris, W.F., P. Sollins, N.T. Edwards, B.E. Dinger, and H.H. Shugart. 1975. Analysis of carbon flow and productivity in a temperate deciduous forest ecosystem. p. 116–122. *In* Productivity of World Ecosystems. Natl. Academy of Science, Washington, DC.

Heichel, G.H. 1978. Stabilizing agricultural energy needs: Role of forages, rotations, and nitrogen fixation. J. Soil Water Conserv. (Nov/Dec):279–282.

Hicks, D.J., and B.F. Chabot. 1985. Deciduous forest. p. 257–277. *In* B.F. Chabot and H.A. Mooney (ed.) Physiological ecology of North American plant communities. Chapman & Hall, New York.

Holland, M.M., P.G. Risser, and R.J. Naiman (ed.) 1991a. Ecotones: The role of landscape boundaries in the management and restoration of changing environments. Chapman & Hall, New York.

Holland, B., A.A. Welch, I.D. Unwin, D.H. Buss, A.A. Paul, and D.A.T. Southgate. 1991b. The Composition of foods. Royal Soc. of Chemistry, Cambridge.

Jackson, L.E., R.B. Strauss, M.K. Firestone, and J.W. Bartolome. 1990. Influence of tree canopies on grassland productivity and nitrogen dynamics in deciduous oak savanna. Agric. Ecosyst. Environ. 32:89–105.

Jackson, W. 1987. Altars of unhewn stone. North Point Press, San Francisco.

Jeltsch, F.S. Milton, W.D.J. Dean, and N. Van Rooyen. 1996. Tree spacing and coexistence in semiarid savannas. J. Ecol. 84:583–595.

Johnson, A.S. 1989. The thin green line: Riparian corridors and endangered species in Arizona and New Mexico. p. 35–46. *In* G. Mackintosh (ed.) Preserving communities and corridors. Defenders of Wildlife, Washington, DC.

Kaufmann, R.K. 1991. Oil production in the lower 48 states: Reconciling curve fitting and econometric models. Resour. Energy 13:111–127.

Keever, C. 1950. Causes of succession on old fields of the Piedmont, North Carolina. Ecol. Monogr. 20:231–250.

Keeney, D. 1996. Biological control: Its role in sustainable agriculture. Leopold Letter 8(4):3,6.

Kleinman, P.J.A., D. Pimentel, and R.B. Bryant. 1995. The ecological sustainability of slash-and-burn agriculture. Agric. Ecosyst. Environ. 52:235–249.

Knapp, A.K., and T.R. Seastedt. 1986. Detritus accumulation limits productivity of tallgrass prairie. BioScience 36:662–668.

Kucera, C.L. 1992. Tall-grass prairie. p. 227–268. *In* R.T. Coupland (ed.) Natural grasslands — introduction and western hemisphere. Ecosystems of the World 8A. Elsevier, Amsterdam.

Küchler, A.W. 1975. Potential natural vegetation of the conterminous United States. Map (scale 1:3,168,000). American Geographical Soc., New York.

Kurtz, W.B. 2000. Economics and policy of agroforestry. p. 321–360. *In* H.E. Garrett et al. (ed.) North American Agroforestry: An integrated science and practice. ASA, Madison, WI (this publication).

Lefroy, E., and R. Hobbs. 1992. Ecological indicators for sustainable agriculture. Aust. J. Soil Water Conserv. 5:22–28.

Loomis, R.S., and D.J. Connor. 1992. Crop ecology. Cambridge Univ. Press, Cambridge.

Lowrance, R. 1992. Groundwater nitrate and denitrification in a coastal plain riparian forest. J. Environ. Qual. 21:401–405.

Lowrance, R.R., R.A. Leonard, L.E. Asmussen, and R.L. Todd. 1985. Nutrient budgets for agricultural watersheds in the southeastern coastal plain. Ecology 66:287–296.

Lowrance, R., R. Todd, J. Fail, Jr., O. Hendrickson, Jr., R. Leonard, and L. Asmussen. 1984. Riparian forests as nutrient filters in agricultural watersheds. BioScience 34:374–377.

Maser, C. 1994. Sustainable forestry. St. Lucie Press, Delray Beach, FL.

McPherson, J.K., and G.L. Thompson. 1972. Competitive and allelopathic suppression of understory by Oklahoma oak forests. Bull. Torrey Bot. Club 99:293–300.

Nair, P.K.R. 1993. An introduction to agroforestry. Kluwer Academic Publ., Dordrecht, the Netherlands.

NAS. 1972. Genetic vulnerability of major crops. Natl. Academy of Science, Washington, DC.

Odum, E.P. 1971. Fundamentals of ecology. W.B. Saunders, Philadelphia.

Odum, E.P. 1985. Trends expected in stressed ecosystems. BioScience 35:419–422.

Oke, T.R. 1987. Boundary layer climates. Methuen & Co., London.

Olson, R.K. 1998. Procedures for evaluating alternative farming systems: A case study for eastern Nebraska. Vol. 8. Extension and education materials for sustainable agriculture. Ctr. for Sustainable Agric. Syst. Univ. of Nebraska, Lincoln.

Olson, R.K., and C.A. Francis. 1995. A hierarchical framework for evaluating diversity in agroecosystems. p. 5–34. *In* R.K. Olson et al. (ed.) Exploring the role of diversity in sustainable agriculture. ASA, CSSA and SSSA, Madison, WI.

Omernik, J. 1987. Ecoregions of the conterminous United States. Map (scale 1:7,500,000). Ann. Assoc. Am. Geograph. 77(1):118–125.

Ong, C.K., and P. Huxley (ed.) 1996. Tree–crop interactions: A physiological approach. CAB Int., Willingford, England.

Packard, S. 1988. Just a few oddball species: restoration and the rediscovery of the tallgrass savanna. Restor. Manage. Notes 6:13–20.

Parker, G.R., and C.Merritt. 1994. The central region. p. 129–171. *In* J.W. Barrett (ed.) Regional silviculture of the United States. John Wiley & Sons, New York.

Peet, R.K. 1988. Forests of the Rocky Mountains. p. 63–101. *In* M.G. Barbour and W.D. Billings (ed.) North American terrestrial vegetation. Cambridge Univ. Press, New York.

Peterson, D.L., and G.L. Roelf. 1982. Nutrient dynamics of herbaceous vegetation in upland and flood plain forest communities. Am. Midland Nat. 107:325–339.

Pimentel, D. (ed.) 1980. Handbook of energy utilization in agriculture. CRC Press, Boca Raton, FL.

Pimentel, D., and M. Burgess. 1980. Energy inputs in corn production. p. 67–84. *In* D. Pimentel (ed.) Handbook of energy utilization in agriculture. CRC Press, Boca Raton, FL.

Pimentel, D., C. Harvey, P. Resosudarmo, K. Sinclair, D. Kurz, M. McNair, S. Crist, L. Shpritz, L. Fitton, R. Saffouri, and R. Blair. 1995. Environmental and economic costs of soil erosion and conservation benefits. Science (Washington, DC) 267:1117–1123.

Pimentel, D., and M. Pimentel (ed.) 1996. Food, energy, and society. University Press of Colorado, Niwot.

Raeburn, P. 1995. The last harvest. Simon & Schuster, New York.

Raney, J.W., M.C. Bruner, and J.B. Levenson. 1981. The importance of edge in the structure and dynamics of forest islands. p. 67–96. *In* R.L. Burgess and D.M. Sharp (ed.) Forest Island dynamics in man-dominated landscapes. Springer-Verlag, New York.

Reed, P.C. 1989. The national wilderness preservation system: The first 23 years and beyond. p. 2–20. *In* Proc. of the National Wilerness Colloquium, Wilderness Benchmark 1988, Tampa, FL. 13–14 Jan. 1988. Gen. Tech. Rep. SE-51. USDA-For. Serv., Southeastern Forest Exp. Stn., Asheville, NC.

Risser, P.G. 1985. Grasslands. p. 232–256. *In* B.F. Chabot and H.A. Mooney (ed.) Physiological ecology of North American plant communities. Chapman & Hall, New York.

Risser, P.G., and W.J. Parton. 1982. Ecosystem analysis of the tallgrass prairie: Nitrogen cycle. Ecology 63:1342–1351.

Roberts, L. 1988. Hard choices ahead on biodiversity. Science (Washington, DC) 241:1759–1761.

Rochow, J.J. 1975. Mineral nutrient pool and cycling in a Missouri forest. J. Ecol. 63:985–994.

Rule, L.C., C.B. Flora, and S.S. Hodge. Social dimensions of agroforestry. p. 361–386. *In* H.E. Garrett et al. (ed.) North American Agroforestry: An integrated science and practice. ASA, Madison, WI (this publication).

Runkle, J.R. 1982. Patterns of disturbance in some old-growth mesic forests of eastern North America. Ecology 63:1522–1546.

Selley, R.A. (ed.) 1996. Nebraska crop budgets. Nebraska Coop. Ext. EC96-872-S. Univ. of Nebraska, Lincoln.

Sims, P.L. 1988. Grasslands. p. 265–286. *In* M.G. Barbour and W.D. Billings (ed.) North American terrestrial vegetation. Cambridge Univ. Press, New York.

Sims, P.L., and J.S. Singh. 1978. The structure and function of ten western North American grasslands: IV. Compartmental transfers and energy flow within the ecosystem. J. Ecol. 66:983–1009.

Smith, D.G. 1976. Effect of vegetation on lateral migration of anastomosed channels of a glacier meltwater river. Geol. Soc. Am. Bull. 87:857–860.

Smith, W.K. 1985. Western montane forests. p. 95–126. *In* B.F. Chabot and H.A. Mooney (ed.) Physiological ecology of North American plant communities. Chapman & Hall, New York.

Soule, J.D., and J.K. Piper. 1992. Farming in nature's image. Island Press, Washington, DC.

Spies, T.A., and J.F. Franklin. 1989. Gap characteristics and vegetation response in coniferous forests of the Pacific Northwest. Ecology 70:543–545.

Stauber, K.N., C. Hassebrook, E.A.R. Bird, G.L. Bultena, E.O. Hoiberg, H. MacCormack, and D. Menanteau-Horta. 1995. The promise of sustainable agriculture. p. 3–15. *In* A.R. Bird et al. (ed.) Planting the future. Iowa State Univ. Press, Ames.

Steiger, T.L. 1930. Structure of prairie vegetation. Ecology 11:170–217.

Stockle, C., R. Papendick, K. Saxton, G. Campbell, and F. van Evert. 1994. A framework for evaluating the sustainability of agricultural production systems. Am. J. Altern. Agric. 9:45–50.

Tiedemann, A.R., and J.O. Klemmedson. 1973. Effect of mesquite on physical and chemical properties of the soil. J. Range Manage. 26:27–29.

Tivy, J. 1990. Agricultural ecology. Longman Scientific & Technical, New York.

USDA. 1981. An assessment of the forest and range land situation in the United States. Forest Resour. Rep. 22. USDA-For. Serv., Washington, DC.

USDA. 1982. An analysis of the timber situation in the United States, 1952–2030. Forest Resour. Rep. 23. USDA-For. Serv., Washington, DC.

USDA. 1995. Summary report: 1992 national resources inventory. USDA-NRCS, Washington, DC.

USDOC. 1990. 1987 Census of agriculture. U.S. Dep. of Commerce, Bureau of the Census. U.S. Gov. Print. Office, Washington, DC.

USDOC. 1996. Statistical abstract of the United States 1996. 116th ed. U.S. Dep. of Commerce, Bureau of the Census, Washington, D.C.

Van Andel J., J.P. Bakker, and A.P. Grootjians. 1993. Mechanisms of vegetation succession: A review of concepts and perspectives. Acta Bot. Neerland. 42:413–433.

Vetaas, O.R. 1992. Micro-site effects of trees and shrubs in dry savannas. J. Veg. Sci. 3:337–344.

Watt, D.L. 1995. Economic position and performance of sustainable farms. p. 93–130. *In* A.R. Bird et al. (ed.) Planting the future. Iowa State Univ. Press, Ames.

Webb, W., S. Szarek, W. Lauenroth, R. Kinerson, and M. Smith. 1978. Primary productivity and water use in native forest, grassland, and desert ecosystems. Ecology 59:1239–1247.

Weber, E. 1996. No-till study offers new incentive to cultivate. Leopold Letter 8(4):1,7.

Wells, P.V. 1965. Scarp woodlands, transported grassland soils, and concept of grassland climate in the Great Plains region. Science (Washington, DC) 148:246–249.

Welsch, D.J. 1991. Riparian forest buffers. NA-PR-07-91. USDA-For. Serv., Radnor, PA.

West, N.E. 1988. Intermountain deserts, shrub steppes, and woodlands. p. 209–230. *In* M.G. Barbour and W.D. Billings (ed.) North American terrestrial vegetation. Cambridge Univ. Press, New York.

White, J. 1978. Illinois natural areas inventory technical report. Vol. I. Survey methods and results. Illinois Nat. Areas Inventory, Urbana.

Williams, P.A., and A.M. Gordon. 1992. The potential of intercropping as an alternative land use system in temperate North America. Agrofor. Syst. 19:253–263.

Woodmansee, R.G. 1984. Comparative nutrient cycles of natural and agricultural ecosystems: A step towards principles. p. 145–156. *In* R. Lowrance et al. (ed.) Agricultural ecosystems: Unifying concepts. John Wiley & Sons, New York.