Ecological Benefits of the Conservation Reserve Program

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Abstract: *The Conservation Reserve Program was initiated in 1985 to reduce soil loss on highly erodible agricultural land. This stated objective of the program has been quite successful. However, there are other unintentional yet significant ecological benefits to the program that merit evaluation. These benefits include the reversal of landscape fragmentation, maintenance of regional biodiversity, creation of wildlife habitat, and favorable changes in regional carbon flux. These and other benefits should be used by policy makers and federal officials to maintain the program even after enrollment expectations have been achieved.*

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Beneficios ecológicos del Programa de Conservación de Reservas

Resumen: *El Programa de Conservación de Reservas fue iniciado en 1985 para reducir la pérdida de suelo en tierras agrícolas altamente erosionables. Este preciso objetivo del programa ha sido bastante exitoso. Sin embargo, existen otros beneficios ecológicos, nointencionales pero significativos, del programa que merecen una evaluación. Estos beneficios incluyen la reversión de la fragmentación del paisaje, el mantenimiento de la biodiversidad, la creación de hábitats para la vida silvestre, y cambios favorables en el flujo regional del carbono. Estos y otros beneficios deberían ser utilizados por formuladores do planes y por organismos gubernamentales para mantener el programa aun después del cumplimiento de las expectativas de enrolamiento planteadas.*

Introduction

The Conservation Reserve Program (CRP) was enacted in the United States as part of the 1985 Food Security Act to reduce soil loss on highly erodible agricultural land. In the 1990 farm bill (Food, Agriculture, Conservation and Trade Act), these CRP provisions were extended to 1995. The goal of the U.S. Department of Agriculture (USDA) is to enroll 18 million hectares (ha) of such land. Nearly 14 million ha have been enrolled, with a resulting average reduction in soil lost by erosion of about 42,000 kg per ha (17 tons per acre), or a total reduction of about 800 million tons per year (Young & Osborn 1990). At present, about 40% of the land enrolled in the CRP has potentially severe soil erosion problems—land capability classes IV–VIII—with much of that amount in classes IV (28% of the total) and VI (11%) (Anonymous 1990*a*). The remaining 60%, which is in classes I–III (Anonymous, 1990*a*), has less severe or unusual erosion problems, represents less erodible portions of eligible erodible fields, or is in filter strips (Napier 1990*a*).

Landowners who enroll in the CRP sign 10-year contracts with the USDA. In return for enrolling their erodible land, landowners receive annual rent per acre plus half the cost of establishing and maintaining permanent vegetative land cover, preferably trees or shrubs (Napier 1990*a*). The CRP will likely prove to be economically beneficial, particularly from the point of view of landowners. Young and Osborn (1990) have calculated that the present value of net income to farmers could increase between $9.2 billion and $20.3 billion owing to lower production costs, rental payments, and increased market prices, with much of that increase occurring after 1992. A CRP program of 18 million ha could cost consumers between $13 billion and $25 billion over the life of the program because of lower agricultural production and higher market prices, assuming that consumer food costs increase by less than one percent per year (Young & Osborn 1990). Other costs would include establishing vegetative cover and technical assistance. However, these overall program costs would be more than offset by benefits from increased net farm income, soil productivity improvements, and revenue from hunting, fishing, and other recreational activities. Thus, the present value of the national net program benefit, in terms of total national income, could be between $3.4 billion and $11 billion (see Young and Osborn [1990] for assumptions).

Benefits and costs of the CRP are usually expressed in monetary terms; by retiring erodible land, however, other ecologically desirable objectives can be achieved, including (1) reduced sedimentation in lakes, rivers, and streams, (2) reduced nonpoint-source agricultural runoff, (3) improved water quality and retention, and (4) the development of wildlife habitat (Huang et al. 1990; Young & Osborn 1990; Kinsinger 1991). To date, the CRP enrollments have been allocated to the soil conservation cover types listed in Table 1. The greatest planted area is in forage and native grass (12 million ha). Trees and wildlife plantings comprise the other major land cover types. A smaller area occurs in such linear features as windbreaks, waterways, and riparian filter strips, whose ecologic impact exceeds their cumulative area. Thus, the CRP has moved large areas toward more natural vegetation, the ecologic significance of which warrants examination.

Our purpose is to discuss these secondary, unintentional ecological benefits of the CRP that have not been given the attention they deserve and that should be given more weight in policy formulation. These benefits include mitigation of landscape and habitat fragmentation, maintenance of regional biodiversity, establishment of wildlife habitat, and changes in regional carbon flux. We focus on forested regions, although there are similar benefits to be gained in other areas (such as grasslands). As a case study for some of these ecological benefits, we will present possible scenarios for Cadiz Township, a southern Wisconsin agricultural landscape (Sharpe et al. 1987; Dunn et al. 1991). Examples of other benefits will be drawn from the ecological literature.

Global CO_2

Reforestation in the humid midlatitudes has been proposed as an antidote to combustion of fossil fuels and destruction of tropical forests (Schroeder 1991). Past and present deforestation in the midlatitudes has been viewed as contributing to the global CO_2 problem. As a landscape is cleared of natural vegetation, especially of forests, the region changes from a carbon sink to a carbon source (Delcourt & Harris 1980; Sharpe & Johnson 1981). For instance, much of the southeastern United States was cleared of forest from 1750 to 1880 and converted to agricultural use (Johnson & Sharpe 1976; Sharpe & Johnson 1981; Alig et al. 1988). By 1980, virgin forests in the southeastern United States covered only 1% of their former area (Delcourt & Harris 1980), and the region experienced an average net carbon loss

Table 1. Allocation of total enrolled Conservation Reserve Program land to conservation cover types.

Cover Type	*Area (hectares)*
Forage and Native Grass	12,022,420
Trees	882,609
Wildlife Plantings	793,967
Windbreaks and Riparian Filter Strips	22,529
Water Bodies	11,026
Other	49,166

*Data from Anonymous (1990*b*).*

of 0.13 Gton per year despite significant land conversion to second-growth forest. Commercial forest land biomass storage increased dramatically, however, from 53.2 Mg/ha in 1952 to 72.2 Mg/ha in 1977. As a result, the region has functioned as a carbon sink rather than a source since at least 1960, reversing the negative carbon flux. As of 1980, only 3% of the total carbon lost had been recovered (Delcourt & Harris 1980), a result similar to that shown for the Georgia Piedmont (Sharpe & Johnson 1981). Although this trend is discouraging, there is opportunity for much greater carbon storage if presettlement forests are considered the norm. If commercial forest area were reduced by only 10%, the region would return to being a carbon source. Thus, the carbon balance is highly sensitive to land use changes.

The potential role of the CRP in converting landscapes to carbon sinks has not been investigated. Currently, 882,000 ha of land are under contract for tree planting under the CRP (Table 1). Most of this area (748,000 ha) is in the Atlantic and Gulf coastal states (Anonymous 1990*b*). Using Schroeder's (1991) mean estimate of carbon sink strength of forests (5 Mg C/ha/yr), we calculate that these CRP lands will have a sink strength of 4.4×10^6 Mg C/yr for the next few decades. By contrast, the contribution of the United States to atmospheric carbon via fossil fuel combustion and cement manufacturing is estimated to be 1.22 Gt/yr (Hammond 1990), or nearly 300 times as great as the sink strength of the CRP land. Thus, the tree planting program in the CRP can make only a small contribution toward countering the impact on atmospheric CO_2 of fossil fuel use in the United States. However, recent rates of land use change involving deforestation for urban and agricultural development (that do not account for the CRP) are estimated to contribute 6×10^6 Mg C/yr. The tree planting program of the CRP would balance the impact on atmospheric carbon of these other land use changes but would not address the larger global issue. Increases in the biomass on other CRP land (grasses and other vegetation) would contribute modestly to the CO_2 sink strength of CRP lands.

In Cadiz Township (Green County), Wisconsin, deforestation began around 1830 and proceeded rapidly. By 1882, forest had declined to about 30% of the land area and has remained at about 5% since the turn of the century (Sharpe et al. 1987). We have no direct estimates of carbon flux in Cadiz Township. Nevertheless, if enrollment of land in the CRP were appreciable and the percentage of land cover in forest were increased from the present 5% (Fig. 1A), it is likely that a positive effect on the carbon balance would result.

Landscape Fragmentation

In many landscapes of eastern North America, natural vegetation (primarily forest) has been replaced by agriculture (Curtis 1956; Auclair 1976; Burgess & Sharpe 1981; Moss & Hosking 1983; Whitney & Somerlot 1985; Sharpe et al. 1987). In some areas, especially in portions of the eastern United States, agriculture did not prove to be economically viable and natural vegetation became reestablished following farm abandonment (Johnson & Sharpe 1976; Turner 1987; Turner & Ruscher 1988). In the upper Midwest to date, land use appears to be much more stable, in part as a result of more fertile soils. As noted earlier for Cadiz Township, the percentage of land area in forest has remained near 5% since the turn of the century.

At present, the Cadiz Township landscape contains 84 woodlots that average 5.6 ha in area (Fig. 1A). Only 3 of the 84 woodlots are large enough to contain interior forest, with the remaining woodlots containing mostly edge habitat. Because the Cadiz data were stored in a

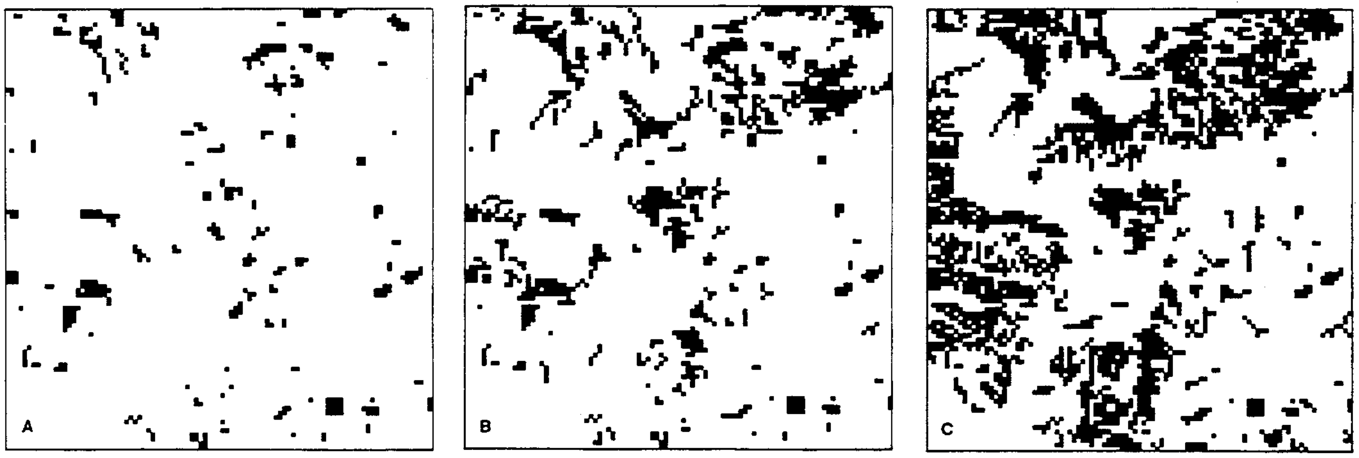

Figure 1. Potential changes in forest cover (dark areas) and spatial pattern under Conservation Reserve Program scenarios. (A) Present forest cover, (B) projected forest cover under scenario 1, (C) projected forest cover under scenario 2.

grid cell (raster) data base, interior forest is defined here as any 1-ha cell surrounded by filled or partially filled forested cells (Dunn et al. 1991). The average distance between woodlots is 439 m. The pattern of this landscape is typical of much of the Midwest in that it is highly fragmented. As early as 1904, Shriner and Copeland (1904) expressed concern about the effects of deforestation in Green County, including increased sedimentation, decreased streamflow, and loss of some native plants.

When the distance between woodlots is great, dispersal of plant propagules and certain animals is impeded. In southeastern Wisconsin, the dispersal distances of *Tilia americana* L. (basswood), *Acer saccharum* Marsh. (sugar maple), and *Fraxinus pennsylvanica* Marsh. (green ash) are about 30 m, 90 m, and 280 m, respectively (Johnson 1988); therefore, even wind-dispersed *Acer* and *Fraxinus* seeds would rarely be transported from one woodlot to another in the Cadiz landscape. There are interesting consequences of this population isolation for genetic diversity voiced decades ago by J. T. Curtis (1956). Guntenspergen et al. (unpublished data) have found evidence of significant differences in gene frequencies among ten *A. saccharum* populations in Cadiz Township. By contrast, the heavier fruits of *Quercus* spp. (especially *Q. rubra* L.) are transported 1 km or more by blue jays (*Cyanocitta cristata*) in suburban Virginia (Darley-Hill & Johnson 1981) and up to 4 km in rural southeastern Wisconsin (Johnson & Adkisson 1985). Clearly, the present Cadiz landscape is fragmented with respect to the dispersibility of at least four tree species (Dunn et al. 1991).

According to Grumbine (1990), wholesale landscape alteration could be difficult to mitigate in the absence of a major policy initiative. The CRP unintentionally provides such an initiative. We will pursue this idea by considering two CRP scenarios in Cadiz Township (see Dunn et al. [1991] for data sources and methodology). Under the first scenario (Fig. 1B), only the farmland with the most severe erosion problems (capability classes VI–VIII) would be enrolled in the CRP and converted to natural vegetation. Under the second scenario (Fig. 1C), all eligible land in capability classes IV–VIII would be enrolled.

If the more modest assumption of scenario one were realized, several changes in landscape pattern, and thus in process, should eventually result (Table 2). First, the number of woodlots in the landscape would increase from 84 to 97 as some fields reverted to natural vegetation. Second, the average woodlot size would increase to 12.8 ha as other fields adjacent to existing woodlots are converted to natural vegetation. Third, the number of woodlots with interior forest would increase from 3 to 17 (a total of 109 ha of forest interior across the landscape). Finally, average distance between woodlots would decrease to 351 m, and forest cover in the landscape would increase from about 5% to about 13%.

Under scenario two, 103 woodlots would be expected, with an average size of 25.5 ha. The number of woodlots with interior would decline to 13 as enrolled farmland developed into forest and ultimately connected existing woodlots. However, the amount of interior forest would increase to 359 ha. Under this scenario, the landscape would become 27% forested, and woodlots would average 288 m apart (Dunn et al. 1991).

Enrollment of eligible Cadiz Township farmland in the CRP would allow old-field succession to take place over large areas. As these areas undergo vegetation change, formerly isolated seed sources could contribute seeds to newly available portions of the landscape. Furthermore, woodlots that currently are isolated would be brought closer together, in the sense that some landscape connectivity would be reestablished. As a result, the isolation of woodlots containing *A. saccharum*, *Fraxinus americana* L. (white ash), *T. americana*, or *Q. rubra* (in other words, woodlots from which no dispersal can occur) would decrease markedly (Fig. 1A–C). The Cadiz landscape then would become unfragmented in terms of *Q. rubra* under either CRP scenario. In addition, as succession proceeds, the vegetation structure and composition at individual sites would

Table 2. Historic and potential changes in forest characteristics of Cadiz Township, Green County, Wisconsin.

Attribute	*Year* 1882	*Year* 1978	*CRP Scenario 1*	*CRP Scenario 2*
Total forest area (ha)	3339	473	1223	2623
Number of woodlots	47	84	97	103
Average woodlot size (ha)	58	6	13	26
Average distance between woodlots (m)	153	439	351	288
Percentage of landscape in forest	38	5	13	27
Total edge (km)	242	94	222	366
Edge/forest area (m/ha)	72	199	178	139
Area of interior forest (ha)	1295	11	109	359
Number of woodlots with interior	44	3	17	13

Data from Dunn et al. (1991).

change. This could have major implications for landscape energy flux and water cycling (see Ryszkowski & Kedziora 1987), with greater water retention expected as a result of larger areas in old-field and forest.

It is unlikely that all eligible land will be enrolled in the CRP, and some will not be abandoned but will be treated under conservation plans using changes in crop rotation or strip cropping. In fact, as of November 1990 only 210.4 ha in Cadiz Township had been entered in the CRP by 13 landowners (F. C. Dillon, personal communication).

There are several reasons why many farmers have not enrolled in the CRP (Camboni et al. 1990; Napier 1990*b*; Napier & Napier 1991), but four reasons seem to be of overriding importance. First, many U.S. farmers are not convinced that their production is presently suffering as a result of erosion (Napier 1990*b*). Second, 45–50% of farmers in the U.S. do not participate in government farm programs. Thus, threats of losing government subsidies as a result of noncompliance are immaterial (Napier 1990*b*). Third, a number of farmers apparently resent government intervention in farm-level decision making (Napier & Napier 1991). Finally, many farmers seem to be uninformed about the economic benefits of the CRP (Camboni et al. 1990), overestimating the program's costs and underestimating its benefits (Napier & Napier 1991); Long et al. (1991) cite numerous ways in which they have effectively educated Arkansas farmers. However, the CRP does hold promise for reversing some of the negative ecological consequences of deforestation, habitat fragmentation, and loss of regional biodiversity.

Although old-field succession often proceeds slowly, the process could be hastened by planting tree species native to the region. Such plantings would not only increase the structural complexity of the old fields, but would serve as recruitment foci for bird-dispersed seeds. For example, two-year-old fields in New Jersey containing artificial saplings had more seed input via birds than did control fields of the same age (McDonnell & Stiles 1983).

Wildlife Habitat

If reforestation were strongly encouraged under the CRP, major benefits to wildlife could result. For decades, it was assumed that "edge" benefited wildlife by increasing habitat heterogeneity. More recently, the benefits of edge have been treated with some skepticism (see Harris 1984; Reese & Ratti 1988; Robinson 1988). Although the creation of edge habitat can increase the species richness of a patch both in terms of plants (Ranney et al. 1981; Dunn & Loehle 1988) and birds (Reese & Ratti 1988), the edge can attract predators and brood parasites, resulting in the decline or local extirpation of bird species of the forest interior (Kendeigh 1944; Harris 1984; Robinson, 1988).

If regional landscape fragmentation were to continue, fewer and smaller woodlots could result (Sharpe & Johnson 1981; Sharpe et al. 1987; Forman & Godron 1986). Small woodlots are unsuitable for large wide-ranging forest-dwelling wildlife species, harbor a disproportionate number of predators, and lack enough forest interior to support a suite of nongame wildlife species (Robinson 1988). In a region of southern Illinois, forest fragmentation has had negative effects on nongame bird populations, including threats to habitat specialists such as the Ovenbird (*Seiurus aurocapillus*) and the Worm-Eating Warbler (*Helmitheros vermivorus*) (Robinson 1988). Lowered breeding success of many nongame populations has been attributed to nest predation and brood parasitism (Robinson 1988). Thus, the destruction of habitat patches and the creation of more edge could threaten regional species diversity.

An irony posed by implementation of the CRP is that it will result in both more edge and more interior habitat. Edge habitat will develop on enrolled fields, at least during the early stages of succession. Also, enrolled fields adjacent to existing woodlots could create a greater proportion of edge relative to the size of the woodlot. As succession proceeds, both effects will diminish. The existing forest edge, in particular, could develop into forest interior in terms of composition and structure, with the successional field becoming the new edge.

The Cadiz landscape can again serve as an example. From 1882 to 1978, the total length of edge (forest perimeter) decreased from 242 km to 94 km (Table 2) as forested areas were converted to agriculture (Sharpe et al. 1987; Dunn et al. 1991). Under either CRP scenario, total edge would increase dramatically in part because of the early successional character of the enrolled farmland and because more and larger patches will necessarily have greater total perimeter. However, the ratio of edge to woodlot area will actually decrease as total forest interior increases, a favorable outcome for the preservation of faunal diversity at the regional landscape scale. Furthermore, the number of woodlots with forest interior will increase from current levels (Table 2; Dunn et al. 1991).

Although larger patches of forest might be desirable, area alone does not guarantee greater species richness of either flora (see Weaver & Kellman 1981; Dunn & Loehle 1988) or fauna (Miller & Harris 1977; Usher 1985). Within the context of a particular landscape, however, larger woodlots should have several benefits. For example, they have more interior forest and thus are more likely to support viable populations. In addition, as farmland returns to forest, distance between woodlots will decrease and connectivity will increase. Consequently, there should be fewer dispersal barriers to forest-dwelling wildlife (Merriam 1988).

Conclusions

Many pieces of legislation pertain to land and resource use, including the National Forest Management Act, the Endangered Species Act, the Clean Air and Clean Water Acts, and the Resource Conservation and Recovery Act. In fact, numerous authors have cited the important cumulative consequences of the many pieces of legislation with biodiversity and land management implications (see Jahn & Schenck 1991; Long et al. 1991). However, Grumbine (1990) believes that the preservation of biodiversity is threatened by a lack of focus and coherence on the part of U.S. federal agencies with land management responsibilities. Among other solutions, he has called for new legislative initiatives to resolve large-scale management problems. Although this is a noble goal, the many laws already enacted could, if adequately enforced, go far toward resolving the biodiversity issue.

The Conservation Reserve Program has numerous hidden benefits that hold promise for the restoration of landscapes in terms of both pattern and process. These include improved connectivity among landscape elements, enhanced dispersal of plant species among woodlots, development of valuable wildlife habitat, maintenance or restoration of regional biodiversity, improvements in carbon flux, and enhanced aesthetics. Farmers need to be encouraged to allow areas to undergo succession or to plant them with native species, rather than to plant cover crops.

The enhancement of succession by native tree planting would shorten the time in which fields contain "weeds," a fact that might be important in farmers' decision making with respect to participation in the CRP (Nassauer 1989). A weed-free appearance is important to many farmers and to urban visitors to rural areas (Nassauer 1989). The CRP could have a major impact on changing (for the better) the appearance of rural America, which might be one indication of landscape "health" (Simpson 1989).

Naveh (1987) compares the biocybernetic and thermodynamic characteristics of natural and altered (deforested) landscapes and makes the perhaps obvious point that as human interference in natural landscape processes increases, the landscape system often becomes less stable (higher entropy, requiring increased management). Furthermore, he stresses that attempts to stabilize such systems by capturing gaseous and solid waste products and by harnessing destabilizing energy could fail because they, too, could require even further entropy-producing and polluting processes. This is not to say that natural and agricultural systems are mutually exclusive. In many instances, both systems can coexist without drastically affecting each other (Hobbs & Saunders 1991, but see Carroll 1990). As Kinsinger (1991) points out, farmers have long provided good wildlife habitat by periodically idling land, by planting shelterbelts, and by maintaining hedgerows. To achieve some restoration of altered landscapes, low energy means should be used, especially those depending on solar energy (such as increased biological production), including establishment of vegetated shelterbelts and riparian strips and the conservation of remnant vegetation. The relevance of the CRP to these objectives is clear.

Most of these benefits merit further quantitative study. In any event, the CRP should not be promoted solely in terms of reducing soil erosion, but also in terms of these additional values and efforts made to encourage permanent adoption of changes.

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Literature Cited

Alig, R. J., F. C. White, and B. C. Murray. 1988. Economic factors influencing land use changes in the south-central United States. USDA Forest Service Research Paper SE-272. Asheville, North Carolina.

Anonymous. 1990*a*. USDA divides CRP acres by land class. Journal of Soil and Water Conservation 45:288.

Anonymous. 1990*b*. Conservation Reserve Program ninth signup results. USDA Agricultural Stabilization and Conservation Service, Conservation and Environmental Protection Division. Washington, D.C.

Auclair, A. N. 1976. Ecological factors in the development of intensive management ecosystems in the midwestern United States. Ecology 57:431–444.

Burgess, R. L., and D. M. Sharpe, editors. 1981. Forest island dynamics in man-dominated landscapes. Springer-Verlag, New York, New York.

Camboni, S. M., T. L. Napier, and S. B. Lovejoy. 1990. Factors affecting knowledge of and participation in the Conservation Reserve Program in a microtargeted area of Ohio. Pages 205–222 in T. L. Napier, editor. Implementing the Conservation Title of the Food Security Act of 1985. Soil and Water Conservation Society, Ankeny, Iowa.

Carroll, C. R. 1990. The interface between natural areas and agroecosystems. Pages 365–383 in J. H. Vandermeer and P. M. Rosset, editors. Agroecology. McGraw Hill, New York, New York.

Curtis, J. T. 1956. The modification of mid-latitude grasslands and forests by man. Pages 721–736 in W. L. Thomas, editor. Man's role in changing the face of the Earth. University of Chicago Press, Chicago, Illinois.

Darley-Hill, S., and W. C. Johnson. 1981. Acorn dispersal by the blue jay (*Cyanocitta cristata*). Oecologia **50**:231–232.

Delcourt, H. R., and W. F. Harris. 1980. Carbon budget of the southeastern U.S. biota: analysis of historical change in trend from source to sink. Science **210**:321–323.

Dunn, C. P., and C. Loehle. 1988. Species-area parameter estimation: testing the null model of lack of relationship. Journal of Biogeography **15**:721–728.

Dunn, C. P., D. M. Sharpe, G. R. Guntenspergen, F. Stearns, and Z. Yang. 1991. Methods for analyzing temporal changes in landscape pattern. Pages 173–198 in M. G. Turner and R. H. Gardner, editors. Quantitative methods in landscape ecology. Springer-Verlag, New York, New York.

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

Grumbine, E. D. 1990. Viable populations, reserve size, and federal lands management: a critique. Conservation Biology **4**:127–134.

Hammond, A. L., editor. 1990. World resources 1990–91: A report by the World Resources Institute. Oxford University Press, New York, New York.

Harris, L. 1984. The fragmented forest. University of Chicago Press, Chicago, Illinois.

Hobbs, R. J., and D. A. Saunders. 1991. Re-integrating fragmented landscapes—a preliminary framework for the Western Australian wheatbelt. Journal of Environmental Management **33**:161–167.

Huang, W.-Y., K. Algozin, D. Ervin, and T. Hickenbotham. 1990. Using the Conservation Reserve Program to protect groundwater quality. Journal of Soil and Water Conservation **45**:341–346.

Jahn, L. R., and E. W. Schenck. 1991. What sustainable agriculture means for fish and wildlife. Journal of Soil and Water Conservation **46**:251–254.

Johnson, W. C. 1988. Estimating dispersibility of *Acer, Fraxinus,* and *Tilia* in fragmented landscapes from patterns of seedling establishment. Landscape Ecology **1**:175–187.

Johnson, W. C., and C. S. Adkisson. 1985. Dispersal of beech nuts by Blue Jays in fragmented landscapes. American Midland Naturalist **113**:319–324.

Johnson, W. C., and D. M. Sharpe. 1976. An analysis of forest dynamics in the northern Georgia Piedmont. Forest Science **22**:307–322.

Kendeigh, S. C. 1944. Measurement of bird populations. Ecological Monographs **14**:67–106.

Kinsinger, A. E. 1991. The promise of the 1990 farm bill for fish and wildlife. Journal of Soil and Water Conservation **46**:255.

Long, J. D., D. Akers, and S. N. Wilson. 1991. The Arkansas response to federal farm program opportunities. Journal of Soil and Water Conservation **46**:272–275.

McDonnell, M. J., and E. W. Stiles. 1983. The structural complexity of old field vegetation and the recruitment of bird-dispersed plant species. Oecologia **56**:109–116.

Merriam, G. 1988. Landscape dynamics in farmland. Trends in Ecology and Evolution **3**:16–20.

Miller, R. I., and L. D. Harris. 1977. Isolation and extirpations in wildlife reserves. Biological Conservation **12**:311–315.

Moss, M. R., and P. L. Hosking. 1983. Forest associations in extreme southern Ontario ca. 1817: a biogeographical analysis of Gourlay's Statistical Account. Canadian Geographer **27**:184–193.

Napier, T. L. 1990*a.* The Conservation Title of the Food Security Act of 1985: an overview. Pages 3–10 in T. L. Napier, editor. Implementing the Conservation Title of the Food Security Act of 1985. Soil and Water Conservation Society, Ankeny, Iowa.

Napier, T. L. 1990*b.* The evolution of US soil-conservation policy: from voluntary adoption to coercion. Pages 627–644 in J. Boardman, I. D. L. Foster, and J. A. Dearing, editors. Soil erosion on agricultural land. John Wiley & Sons, New York, New York.

Napier, T. L., and A. S. Napier. 1991. Perceptions of conservation compliance among farmers in a highly erodible area of Ohio. Journal of Soil and Water Conservation **46**:220–224.

Nassauer, J. I. 1989. Agricultural policy and aesthetic objectives. Journal of Soil and Water Conservation **44**:384–387.

Naveh, Z. 1987. Biocybernetic and thermodynamic perspectives of landscape functions and land use patterns. Landscape Ecology **1**:75–83.

Ranney, J. W., M. C. Bruner, and J. B. Levenson. 1981. The importance of edge in the structure and dynamics of forest islands. Pages 67–93 in R. L. Burgess and D. M. Sharpe, editors. Forest island dynamics in man-dominated landscapes. Springer-Verlag, Berlin, Germany.

Reese, K. P., and J. T. Ratti. 1988. Edge effect: a concept under scrutiny. Transactions of the North American Wildlife and Natural Resources Conference **53**:127–136.

Robinson, S. K. 1988. Reappraisal of the costs and benefits of habitat heterogeneity for nongame wildlife. Transactions of the North American Wildlife and Natural Resources Conference **53**:145–155.

Ryszkowski, L., and A. Kedziora. 1987. Impact of agricultural landscape structure on energy flow and water cycling. Landscape Ecology **1**:85–94.

Schroeder, P. 1991. Can intensive management increase carbon storage in forests? Environmental Management **15**:475–481.

Sharpe, D. M., and W. C. Johnson. 1981. Land use and carbon storage in Georgia forests. Journal of Environmental Management **12**:221–233.

Sharpe, D. M., G. R. Guntenspergen, C. P. Dunn, L. A. Leitner, and F. Stearns. 1987. Vegetation dynamics in a southern Wisconsin agricultural landscape. Pages 137–155 in M. G. Turner, editor. Landscape heterogeneity and disturbance. Springer-Verlag, New York, New York.

Shriner, F. A., and E. B. Copeland. 1904. Deforestation and creek flow about Monroe, Wisconsin. Botanical Gazette **37**:139–143.

Simpson, J. W. 1989. Landscape medicine: a timely treatment. Journal of Soil and Water Conservation **44**:577–579.

Turner, M. G. 1987. Spatial simulation of landscape changes in Georgia: a comparison of 3 transition models. Landscape Ecology **1**:29–36.

Turner, M. G., and C. L. Ruscher. 1988. Changes in landscape patterns in Georgia, USA. Landscape Ecology **1**:241–251.

Usher, M. B. 1985. Implications of species-area relationships for wildlife conservation. Journal of Environmental Management **21**:181–191.

Whitney, G. G., and W. J. Somerlot. 1985. A case study of woodland continuity and change in the American Midwest. Biological Conservation **31**:265–287.

Young, E. C., and C. T. Osborn. 1990. Costs and benefits of the Conservation Reserve Program. Journal of Soil and Water Conservation **45**:370–373.