ELSEVIER

Landscape and Urban Planning 33 (1995) 179–193

LANDSCAPE
AND
URBAN PLANNING

Greenway planning: developing a landscape ecological network approach

John Linehan*,[a], Meir Gross[a], John Finn[b]

[a]Department of Landscape Architecture and Regional Planning, University of Massachusetts, 109 Hills North, Amherst, MA 01003, USA
[b]Department of Forestry and Wildlife, University of Massachusetts, Amherst, MA 01003, USA

Abstract

Greenway planning has steadily grown in popularity in the planning and design professions as an efficient and socially desirable approach to open space planning. The purpose of this paper is to present a theoretical and methodological approach to greenway planning that accounts for regional biodiversity and systematizes the selection of greenway links. The approach used in this paper is based on the premise that a network of wildlife reserves and corridors should serve as the skeletal framework of a comprehensive greenway system. The paper draws from the knowledge bases of landscape ecology, conservation biology, network theory, and landscape planning. A case study is presented to demonstrate the approach using a forested region of Central New England.

Keywords: Greenways; Networks; Landscape ecology; Wildlife corridors

1. Introduction

In his book, *Greenways for America*, Little (1990) defined greenways as protected linear corridors that improve environmental quality and provide for outdoor recreation. Although much attention has been drawn to greenways recently, they have been a component of landscape planning for over a century (Fabos, 1991). Only recently, however, have greenways been considered systematically as integral to the protection of ecological structure and function, and central to the open space planning process (Ahern, 1991a). There is still a need, however, for methods that make the link between ecological structure and function at broad spatial and temporal scales in both basic and applied research (Turner and Gardner, 1991). Greenways provide an excellent opportunity in this regard.

This lack of a firm theoretical and methodological understanding of landscape ecology within the planning and design professions is partly responsible for the failure to address adequately the role of regional biodiversity within the planning community. By linking ecological structure and function, a regional greenway system may be able to protect biodiversity, provide present and future open space needs, and allow for economic growth and development (Ahern, 1991b). A wildlife corridor system that protects regional diversity should be at the forefront of the greenway planning process and could serve as the skeletal framework of a regional greenway system. Such a system could then go on to provide recreational opportunities, help control community development patterns, guide overall growth management efforts, protect the character of a

* Corresponding author: Telephone: (413) 545-2255. Fax: (413) 545-1772.

SSDI 0169-2046(94)02017-5

region, and protect the health, safety, and welfare of society.

One of the attributes of greenways is that they provide an approach to open space planning that better responds to the growing awareness of and concern for the interconnectedness of ecological systems. This represents a significant improvement over approaches that determine open space on a residual basis of undevelopable areas (Ahern, 1989). This approach generally includes floodplains, wetlands, steep slopes, and water resources, as well as agricultural, visual, and historical resources into an open space plan, but often fails to consider the configuration, juxtaposition, and functional relationships between these as an essential component of an ecologically sound approach to development (Forman and Godron, 1986).

Part of the problem is that conventional zoning regulations in the USA are designed a priori to open space planning, and often dictate the layout of development patterns that effectively mandate sprawl and hence fragmentation (Yaro et al., 1989). This ad hoc approach fails to address the disintegration of unprotected connected open spaces which once provided a sense of identity to communities and provided the basis for local biodiversity. The erosion of such de facto greenways is a direct contributor to the lost sense of region and place in many areas, and is one of the causes for the current interest of the greenway movement (Hiss, 1991).

The purpose of this paper is to present a theoretical and methodological approach to greenway planning that better accounts for regional biodiversity and systematizes the selection of greenway links. The approach used in this paper is based on the premise that a network of wildlife reserves and corridors should serve as the skeletal framework of a comprehensive greenway system. The paper draws from the knowledge bases of landscape ecology, conservation biology, network theory, and landscape planning. First, we will discuss the potential and importance of greenways as a tool to combat habitat fragmentation and potential species loss. Second, we will present an overview of the methods used. Third, we will present a case study to demonstrate the approach using a forested region of Central New England. Finally, we will close by discussing the advantages and limitations of the methods as well as how these can fit within the contexts of both the biodiversity and greenway movements.

2. An ecological basis for greenway planning

2.1. Habitat protection and fragmentation

Greenways provide an opportunity to reduce the impacts of habitat fragmentation. Habitat fragmentation is considered one of the most serious threats to biological diversity and is a primary cause of the extinction crisis (Harris, 1984; Wilcox and Murphy, 1985; Brown et al., 1991; World Resource Institute et al., 1992). The two major effects of fragmentation are loss of habitat and habitat isolation. Habitat loss decreases population sizes and increases extinction rates, and isolation decreases the likelihood of recolonization of otherwise productive habitat (MacArthur and Wilson, 1967; Burgess and Sharpe, 1981; Wilcove et al., 1986; Opdam, 1991).

The protection of connectivity of a forest matrix can function both as habitat and corridor for wide-ranging forest interior species that may otherwise be locally extirpated because of fragmentation. A greenway plan that addresses the needs of fragmentation-sensitive species may be able to approximate the natural landscape pattern required by these species and thereby prevent the loss of species that would otherwise be expected from fragmentation (Noss, 1987a). Whereas current suburban development patterns are largely responsible for fragmentation in eastern North America, other anthropogenic disturbances, including forestry practices, chemical applications, pollution, and agriculture (diCastri and Hadley, 1988), and other more natural disturbances such as hurricanes and fires, can change the structure of a landscape matrix (Heinselman, 1981). Understanding the potential of other disturbances is as important as controlling development, as disturbances are closely tied to habitat availability and distribution.

2.2. Conservation schemes

Habitat protection has resulted in successes at conserving some endangered species habitats, but these often fail because they are based more upon property lines, economics, and ad hoc acquisition strategies than on ecological function (Stolenburg, 1991). Protection of endangered species should remain a conservation priority; however, a larger challenge is to insure the integrity of existing natural communities and ecosys-

Potential Advantages

1- Increased immigration, which could
   A- Increase or maintain species richness and diversity
   B- Increase population sizes of particular species
   C- Decrease probability of extinction
   D- Permit species re-establishment
   E- Prevent inbreeding depression/maintain genetic diversity
2- Increased foraging area for wide ranging species
3- Provide escape cover for movement between patches
4- Increase accessibility to a mix of habitats
5- Provide alternative refuge from large disturbances
6- Provide greenbelts to:
   A-limit urban growth
   B-abate pollution
   C- provide recreational opportunities
   D-enhance and protect scenery
   E-improve land values

Potential Disadvantages

1- Increased immigration, which could
   A- facilitate the spread of diseases, pests, ect.
   B- decrease the level of genetic variation between populations (outbreeding depression)
2- Facilitate spread of fire and other contagious catasrophies
3- Increase exposure to hunters, poachers, and predators
4- May not funtion for species not specifically studied
5- Cost and conflicts with conventional conservation direction of preserving endangered species

Fig. 1. Pros and cons of wildlife corridors (Noss, 1987a).

tems, thereby minimizing the number of species that become endangered.

Once we change our focus from rescuing isolated critical habitat areas to insuring overall ecological integrity, the connection between patches becomes as important a parameter as patch size, shape, and type. Although a corridor network should not be seen as the end-all solution to conservation problems, it can be a cost-effective complement to the strategy of large multiple reserve systems. The pros and cons of corridors have been the subject of debate, and are summarized in Fig. 1. Although it is uncertain if wildlife corridors will function as theorized (Simberloff and Cox, 1987), Noss (1987b) stated that perhaps the best argument for corridors is that the landscape was connected before settlement, and that fragmentation has been largely an anthropogenic impact that has reduced connectivity, and that many of the disadvantages of corridors could be avoided or mitigated by enlarging corridor width or applying ecologically sound zoning regulations. At the center of this debate is the issue of scale. Ecological processes, physical characteristics, and the presence or absence of resources are all interrelated, and they each possess a scale at which they function. It is this scale that is responsible for the dynamic relationship between landscape configuration and function (Carlile et al., 1989), so that neither corridors nor large reserves will work as a stand-alone solution to our wildlife protection problems; an integrative solution based on functional scales of operation may represent a more balanced approach.

The issue of large versus smaller, numerous reserves is currently a contested topic in conservation biology. Too few reserves may lead to long-term failure as surely as having reserves that are too small (Quinn and Hastings, 1987; Gilpin, 1988). A basic premise is that a minimal viable population needs a minimum area and resource base to survive, and it appears that the current reserve system in the USA is grossly inadequate. In other words, there is no agreement on what an adequately large reserve is or what the full implications of fragmentation are (Grumbine, 1990). It is from this uncertainty that the question of maintaining overall versus selective regional diversity and the role of corridors has arisen (Walker, 1992).

3. Overview of methods

The major steps for delineating corridors discussed in this paper are: (1) land cover assessment; (2) wildlife assessment; (3) habitat assessment; (4) node analysis; (5) connectivity analysis; 6) network generation; (7) evaluation. The overall procedure is shown in Fig. 2.

3.1. Land cover assessment

The first step involves aggregating land use data into habitat types, and then adding ancillary data that can better differentiate the required habitats based on vegetation, hydrology, patch size, and the degree of urban-

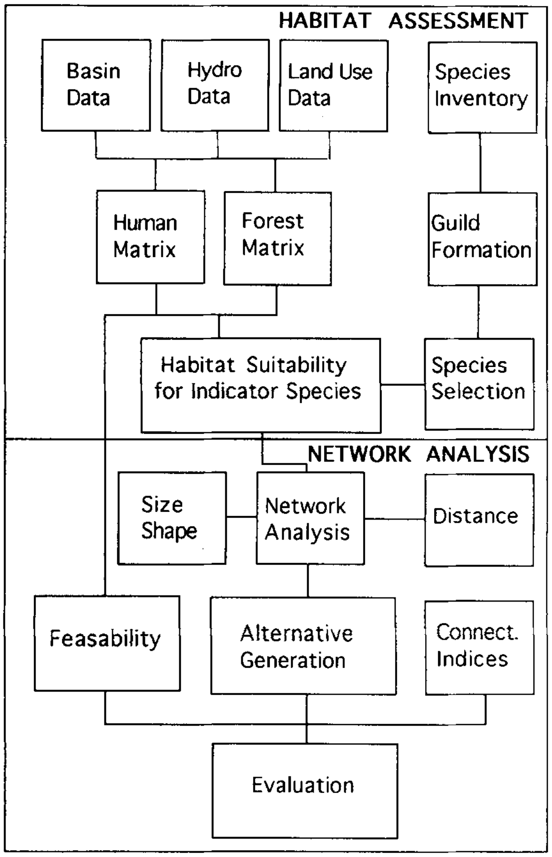

Fig. 2. General overview of procedures: a habitat-based approach to greenway network assessment.

ization. These vegetative and hydrological complexes can then be incorporated into a geographic information system (GIS).

3.2. Wildlife assessment: species inventory, guild formation, and indicator species selection

Once the life histories and habitat needs of indigenous wildlife species are incorporated into a database, guilds can be linked to vegetative and hydrological complexes, as plant communities are a major determinant of the movement of organisms and habitat suitability (Burley, 1989; Schwabe, 1989). This can be achieved by sorting the data into guilds based on the habitat types. A conceptual overview of this process is shown in Fig. 3.

Once the relevant species have been assigned to a habitat type, species can be selected that can serve as indicators of local diversity (Scott et al., 1993). The selection of indicator species is one of the most significant decisions in the overall corridor planning process for several reasons (Soule, 1986). Indicator species are often selected based on their overall role in the food web, so that if the chosen species disappears, exirpation of functionally dependent species may not be far behind. Another approach is to select species that are dependent on a number of ecosystems for their survival, so that adequate protection of the indicator species represents protection of the other species dependent on each separate habitat type. A third approach is based on the ability of the species to represent the needs of a larger species assemblage. Such an approach may be unsuitable at the micro- and meso-scales, as a given habitat change may affect species differently, but may be useful at the macro-scale, where data resolution would not permit the analysis of these subtle changes (Starfield and Bleloch, 1986).

3.3. Habitat assessment and suitability analysis

Habitat can be viewed as an integration of the spatial component, area or quantity, and the essential resources of food, water, and cover found therein. If either adequate expanse or resources are lacking, the area can be considered as non-habitat for a given target species. An optimal habitat, therefore, is an area that is both attractive to and highly productive for the species in question (Weller, 1985). Given this definition, in most cases suitability can be derived as a function of patch size, shape, cover type, accessability, and quality, although the parameters that determine quality will vary based on the species selected. The relationship between landscape structure and function can now be assessed. The basis for analysis of landscape structure is based on the landscape ecological taxonomy described by Forman and Godron (1986). From this perspective, the landscape can be interpreted as being composed of patches and corridors that exist within a landscape matrix. The matrix of a landscape has a greater relative area than any of the patch types within it and is the most connected part of the landscape, and plays a predominant role in the dynamics of the landscape. Patches are nonlinear areas that differ in appearance from the surrounding background landscape matrix. Corridors are linear areas or elements that differ from the surrounding matrix that may be isolated, but generally connect

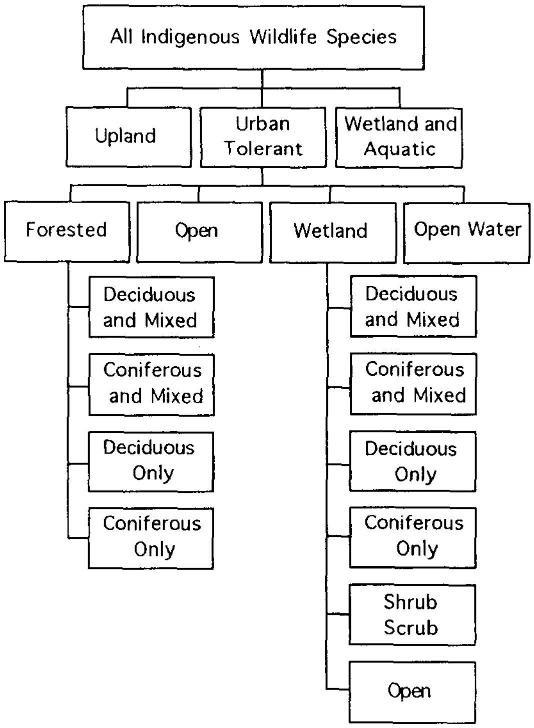

Fig. 3. Conceptual overview of habitat clustering procedure used for guilding.

patches of a similar vegetative cover. A network then, in a landscape ecological sense, can be understood to be the system of patches and corridors existing within and abstracted from the landscape matrix.

3.4. Node analysis

The purpose of this step is to determine the relative significance of each of the nodes. As a greenway is a network, it can be assessed with existing network analysis theories and methods. Graph theory provides a useful approach for analyzing networks, as it allows the analyst to optimize a given flow-related objective. Since its emergence, graph theory has been used in numerous professions including economics, management, sociology, anthropology, electronics, neurology, linguistic analysis, and transportation planning (Haggett et al., 1977). In graph theory, the degree to which all nodes in a system are linked is known as network connectivity. The parameters that determine network connectivity are (1) the number of separate networks within the region, (2) the number of links within the network, and (3) the number of nodes within the network. Nodes are generally nonlinear elements that can be considered to be a place or an event. Within the wildlife conservation framework, these locations or nodes can refer to patches, habitats, protected areas, or corridor intersections. Links and routes can be seen as synonymous with corridors (Lowe and Morydas, 1975), and are defined as linear elements that facilitate the flow of energy, matter or species. Within the context of greenway planning, nodes could be any discrete nonlinear resource that are to be included in the greenway. This could include such things as protected open spaces, critical habitat areas, historic buildings or districts, farms, recreation areas, overlooks, and waterbodies. As the ecological function used in this study is habitat for fragmentation-sensitive wildlife, nodes were defined as currently protected forested open spaces no smaller than 50 ha. Although this value is somewhat arbitrary, it was used as it corresponds to the data produced by the US Forest Service wildlife inventory (DeGraaf and Rudis, 1983). Once the nodes were determined based on this rule, they were rated based on size, shape, and habitat value.

3.5. Connectivity analysis

The most common method for assessing the interaction between pairs of nodes is the gravity model (Sklar and Constanza, 1991). The level of interaction between the nodes can be used to represent the significance of potential greenway links. In general, the greater the size (or other quality attribute), closer the distance, and the less degree of 'friction', the greater the level of interaction. This can not only aid in determining efficient greenway networks, but it can also be used to determine the probable accessibility to certain suitable habitat areas. A simple application of the gravity model is shown in Fig. 4. In its simplest form, the gravity model calculates the interaction between each pair of nodes a and b using the following formula:

$$G_{ab} = (N_a \times N_b) / (D_{ab})^2 \quad (1)$$

where G_{ab} is the interaction between nodes a and b, N_a is node weight of node a, N_b is node weight of node b, and D_{ab} is distance between the centroids of nodes a and b.

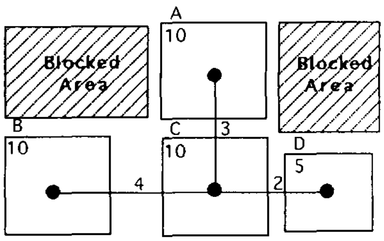

Fig. 4. Application of the gravity model where the level of interaction (G) of each pair of nodes can be assessed as follows from Eq. (1): $G_{(AB)}=(10\times10)/7^2=2.04$; $G_{(AC)}=(10\times10)/3^2=11.11$; $G_{(AD)}=(10\times5)/5^2=2.00$; $G_{(BC)}=(10\times10)/4^2=6.25$; $G_{(BD)}=(10\times5)/6^2=1.39$; $G_{(CD)}=(10\times5)/2^2=12.50$ (Forman and Godron, 1986).

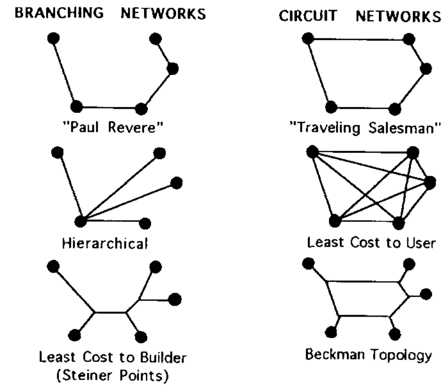

Fig. 5. Examples of common network typologies (Hellmund, 1989).

3.6. Network generation

The next step is to generate schemes that connect the nodes based on the information gained from the model. One of the primary concerns in network analysis is efficiency, which is generally defined in terms of 'cost to user' and 'cost to builder' (Haggett and Chorley, 1972). This cost balancing is a useful framework for determining the spatial patterning of networks. The case of 'least cost to user' is one where the costs of moving between any two points are kept to a minimum. In an ideal situation, this would be represented by a network in which all points are directly connected (see Fig. 5). When the 'cost to builder' is to be minimized, the network will be a branched network, or a minimum spanning tree (MST) (see Fig. 5).

Two basic forms of MSTs are the 'Paul Revere' and the 'Stiener point' typologies. Paul Revere networks are MSTs in which all nodes are visited once and there are no extraneous segments. In contrast, a Stiener point network is an MST in which all the nodes are terminal and are served by only single links that converge on floating points, as is shown in Fig. 5. Conversely, Stiener point networks require floating nodes from where dispersion and concentration of flow both occur. The final open type described here are hierarchical networks, which are a subset of the least cost to user network where flow is directed through a centralized point of redistribution. Riverine patterns can also be interpreted as hierarchical.

As networks become more complex, they take on the form of closed loops. In its most basic form of a minimal loop network, such as the 'traveling salesman' example, the network is computable from a multitude of readily available algorithms. The other extreme, as already stated, is the least cost to user typology where all nodes are directly connected. A third loop typology are 'Beckman' networks that attempt to balance cost to user and cost to builder by incorporating loops that vary from all points connected to a Stiener point network. When the cost to the user is the dominating concern, an open triangular polygon results. When a low cost of implementation is the main objective, a network in which all points are linked at a central point results. Beckman alternatives are based on the fact that intermediate solutions are common, and the ideal solution will vary depending on the relative importance between builder and user costs between each pair of nodes.

3.7. Evaluation

The alternative networks can then be evaluated through the use of connectivity indices; these are statistical measures that are useful in calculating network efficiency. The indices selected for this study were the Gamma, Beta, and Cost Ratio indices. The Gamma index is calculated by dividing the number of links in the network by the maximum number of possible links. Gamma can be interpreted as per cent connectiveness. The Gamma index can be adjusted so that the values better correspond to the regional conditions, so that links that are either not feasible or not desirable can be factored out. (The issue of why to adjust the values is discussed in more detail in the case study section of the paper.) In the case study, adjusted values were calculated by replacing the term representing the maximum

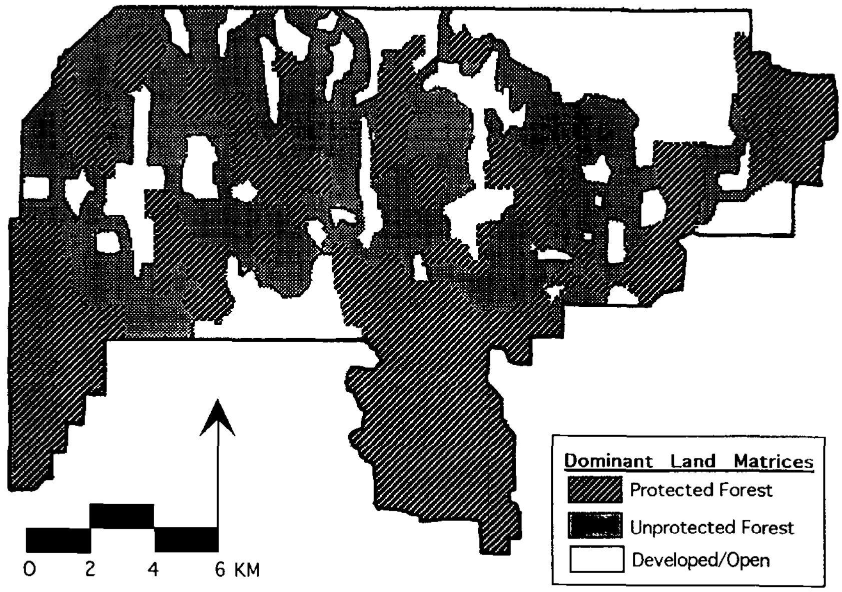

Fig. 6. Map of dominant matrix types and open space identification numbers.

number of links (which is based on all nodes connected) with a value that represents the number of nodes that could be connected based on site conditions. The Beta index is calculated by dividing the number of links in the network by the number of nodes. Beta values of less than one signify a dendogrammatic pattern, a value of one signifies a single circuit, and values greater than one indicate more complex levels of connectivity (Haggett and Chorley, 1972). Whereas the Gamma and Beta indices are designed to measure fairly abstract attributes, the Cost Ratio is based on the landscape conditions and socioeconomic realities where a measure of efficiency will inevitably have to return to concrete measures to account for cost differentials of alternative greenway networks and links. The Cost Ratio is calculated by dividing the number of links in the network by their total distance, resulting in a value per total distance. The formulae for these indices (adapted from Dalton et al. (1973, pp. 7–12) and Forman and Godron (1986, pp. 417–419)) are listed below:

$$\text{Gamma} = \text{no. of links/maximum no. of links} \quad (2)$$

$$\text{Gamma(adjusted)} = \text{Gamma/Gamma for maximum network based on constraints} \quad (3)$$

$$\text{Beta} = \text{no. of links/no. of nodes} \quad (4)$$

$$\text{Cost Ratio} = 1 - (\text{no. of links/distance of links}) \quad (5)$$

4. Case study

4.1. Regional overview

The study area is of 140 000 ha, located in Central Massachusetts, extending from Leominster State Forest in the east to the Quabbin Reservoir to the west (see Fig. 6). The area can be characterized as consisting of two matrix types: (1) a forested matrix consisting of mixed hard- and softwoods with a fairly evenly dispersed density of contiguous hydrologic systems; (2) a human-dominated matrix characterized by older village development, newer subdivision and strip development, pasture, and a large amount of fragmented remnant forest patches. Forested areas within the human-dominated matrix were excluded from consideration for routing a wildlife corridor, as the ability to route a wide enough corridor for fragmentation-sensitive species in these areas is overly constrained by the degree of subdivision, the cost of land, and the density

Table 1
Distribution of land covers in the study region

Land use	Hectares	%
Forest	130040.68	92.50
Intensive agriculture	1477.92	1.05
Extensive agriculture	1117.36	0.80
Nonforested wetland	1435.84	1.02
Mining	97.68	0.07
Open land	722.76	0.51
Recreation	212.80	0.15
Residential	1887.64	1.34
Commercial	47.32	0.03
Industrial	83.64	0.06
Urban open land	136.28	0.10
Transportation	366.24	0.26
Waste disposal	62.04	0.04
Water	2756.16	1.96
Woody perennial	17.88	0.01
Total	140462.24	

of roads. Whereas these areas are important in the planning of a comprehensive multiple use greenway system, their utility for wildlife species with large home ranges is limited. As shown in Table 1, the study region is largely forested. In fact, forest, water, and open wetland combines to account for over 95% of the study area. However, as Fig. 6 shows, the configuration of the land uses creates a situation where many of these 'natural' areas are isolated, highly fragmented remnant patches that contain less than adequate forest interior, and hence in all likelihood, fail to function as habitat for fragmentation-sensitive species. Fig. 6 also shows the configuration of protected forested areas that are greater than or equal to 50 ha.

4.2. Indicator species selection

The indigenous wildlife species of the region were incorporated into a database and sorted to correspond to the vegetative complexes as shown schematically in Fig. 3. The data on wildlife species were taken from 'New England Wildlife: Habitat, Natural History, and Distribution' (DeGraaf and Rudis, 1983). The species were sorted so as to fulfill the following objectives: (1) to determine any natural species associations; (2) to aid in the selection of appropriate indicator species; (3) to cover the habitat needs of a large number of other species; (4) to select species requiring large ranges so that species requiring smaller ranges may be adequately protected; (5) to select species that are uncommon enough to warrant attention, yet not rare enough to be unrepresentative of the guild (Linehan, 1992). The purpose of the sorting was not to assume actual species associations, but to obtain a general overview of what species may potentially occur within these habitat types at the landscape scale. Many species require specific landscape characteristics not addressed in this study; many other species require multiple habitats as part of their life history. Any efforts at protecting a specific species should be made on its specific habitat needs and on a more site-specific basis.

For the purpose of this study, it was important to select an indicator species that was sensitive to fragmentation and whose protection will buffer the effects of fragmentation upon other species. For this reason, the basis for selection of such a species was its sensitivity to fragmentation and disturbance, its habitat needs, its range, and its representational value in connectivity analysis. The two species selected for this study were the river otter (*Lutra canadensis*), and the fisher (*Martes pennanti*). As the needs of the two species differ greatly, adequate habitat and corridor protection of these two species may effectively protect the habitats of, and provide corridors for, a wide range of wildlife species (DeGraaf and Rudis, 1983; Organ, 1989; Ahern, 1991b; Linehan, 1992). Together, these two species cover a range of both terrestrial and aquatic habitats, are highly sensitive to fragmentation, are at the top of the food chain, and serve to indicate an intact tree canopy and good water quality. If adequate protection can be provided for these two species, it is theorized that there is a good chance that adequate protection will be implicitly provided to a wide range of species. Although the role of the otter is incorporated in the research, for the purposes of this paper the approach described will be based on the habitat needs of the fisher, so as to simplify the paper.

4.3. Habitat needs of the fisher

The fisher is the largest species of the weasel family found exclusively in North America. Indigenous and reintroduced populations can at present be found in northern New England, in portions of the Appalachians south to Virginia, in the Northern Tier states, in eastern portions of Oregon and Northern California, and along the western border of California (Allen, 1983). The fisher is currently reoccupying areas of Massachusetts because of the shift in the landscape matrix from abandoned agriculture to forest (DeGraaf and Rudis, 1983).

Dense, closed-canopy coniferous and mixed forests are the preferred habitat for the fisher (Powell, 1982; Allen, 1983; Arthur et al., 1989), and all forms of urbanized land uses are considered to be unsuitable. Areas that were marginally suitable or have the potential to become suitable (such as young forests and abandoned fields) were determined to have a low suitability value. This includes the agricultural and woody perennial uses, as they have a strong propensity to revert to forest. It also includes areas that were shown by the literature to have only secondary values for fisher habitat. These uses include both water and open wetland areas, as the boundaries of these areas are highly productive in terms of prey value, and fishers were found to hunt on these edges. Kelly (1977) found that yearly overlap and mutual occupancy of habitats was common. Kelly also found that ranges parallelel valleys and nearly always ended at or coincided with streams, and that home ranges varied from 650 ha to 4000 ha, with an overall average of 1920 ha.

4.4. Node analysis

The existing protected open spaces were considered to be the nodes of the network, as future large-scale land acquisitions are unlikely within this particular landscape; linkage of the existing conservation and recreation lands coupled with a program of rounding out existing nodes to improve their function has become a conservation policy objective (Massachusetts Department of Fisheries, Wildlife and Law Enforcement, 1990). The node weight was determined by dividing the area of the node by the minimal required area for fisher of 650 ha and then multiplying by ten; the values for each node are shown in Table 2. This normalized the node weights so that weights less than ten were unsuitable for the selected species, values of ten were adequate to support a minimal viable population, and numbers greater than ten exceeded the minimum. Additionally, although eight nodes exceed the minima, only two nodes exceed the average. As no population data were available, this value was used as an indication of 'fisher potential'. It is not assumed that this is a direct indication of carrying capacity, as range overlap has been shown to occur.

Table 2

Sizeand node weights of protected open spaces shown in Fig. 6, where node weight$\frac{15}{16}$10× (are of the node 650) and 650 is the minimal required are for fishers (Node 1 has been crooped in trhe maps, so that the weight does not correspond to the soze as shwon)

Node no.	Hectares	%	Node wt.
1	7071	37.88	110
2	3521	18.86	110
3	1507	8.07	23
4	1183	6.34	18
5	1094	5.86	17
6	875	4.69	13
7	807	4.32	12
8	674	3.61	10
9	358	1.92	6
10	349	1.87	5
11	168	0.90	3
12	100	0.54	2
13	84	0.45	1
14	78	0.42	1
15	74	0.40	1
16	73	0.39	1
17	72	0.39	1
Other	579	0.03	
Total	18669		

4.5. Node interaction

The level of interaction between the nodes was determined by using a modification of the gravity model presented in Eq. (1). This was done for the sake of improving the handling and display of decimal points. The formula as used can be expressed as

$$G_{ab} = [(N_a \times N_b)/(D_{ab})^2] \times 100 \qquad (6)$$

The calculation of node weights (N_a, N_b) is described in the previous section. The matrix of interactions (G) between the 17 nodes is shown in Table 3. This matrix is useful for comparing the relative significance of greenway links. An example of the utility of this matrix will be demonstrated in Section 5. Also, as $(G_{ab}) = G_{ba}$), Table 3 is symmetrical; hence it is unnecessary to calculate both values.

4.6. Network delineation and assessment

The links between each pair of nodes were tested against the site conditions to determine which were

Table 3
Node interacetion (G) based on the gravity model

Node	Node															
	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17
1	11.64	8.27	1.37	0.73	12.12	22.88	0.69	2.45	1.58	0.34	0.14	1.24	0.29	0.21	0.22	0.31
2	0.00	12.18	8.26	2.53	2.63	10.02	5.03	1.08	2.32	1.48	1.40	0.37	0.38	1.41	1.84	0.49
3		0.00	1.17	0.55	3.71	8.22	0.87	1.84	5.26	1.35	0.09	0.78	0.99	0.56	0.43	0.41
4			0.00	3.83	0.36	0.47	2.84	0.17	0.17	0.17	1.69	0.04	0.06	0.17	0.20	0.12
5				0.00	0.20	0.24	0.67	0.10	0.07	0.08	0.17	0.02	0.04	0.06	0.07	0.06
6					0.00	2.87	0.20	6.15	0.36	0.12	0.04	1.77	0.18	0.07	0.07	0.08
7						0.00	0.28	0.86	3.71	0.21	0.05	1.22	0.15	0.12	0.11	0.09
8							0.00	0.10	0.11	0.20	0.32	0.03	0.05	0.22	0.36	0.11
9								0.00	0.17	0.07	0.02	0.31	0.15	0.04	0.03	0.05
10									0.00	0.11	0.02	0.06	0.05	0.05	0.05	0.03
11										0.00	0.02	0.02	0.05	0.64	0.34	0.26
12											0.00	0.00	0.01	0.02	0.03	0.01
13												0.00	0.03	0.01	0.01	0.01
14													0.00	0.02	0.02	0.04
15														0.00	0.84	0.04
16															0.00	0.02
17																0.00

feasible. Links were determined to be unfeasible if (1) they were blocked by unsuitable areas, (2) they were redundant owing to diversion around unsuitable areas or 3) they converged upon another node. The remaining links were mapped out in graph format as a representation of the greatest level of connectivity possible based on these constraints. Graph A in Fig. 7 represents the solution with the highest level of connectivity based on these constraints, and was used as the basis for computing the adjusted Gamma index. An example of the utility of these indices will be demonstrated in Section 5. The connectivity indices previously described

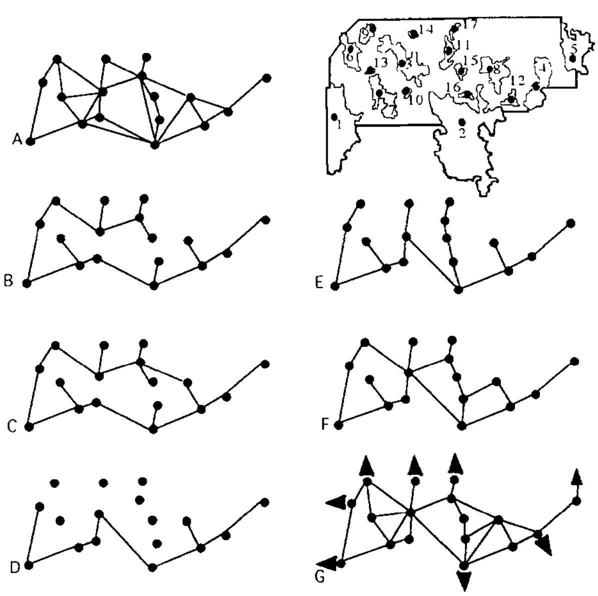

Fig. 7. Network alternatives generated for connectivity analysis.

Table 4
Connectivity indices for the seven network configuraions shown in Fig. 7

	Network	Nodes	Links	Gamma		Beta	Cost ratio
				Raw	Adjusted		
	Theory max	17	136	1.00	N/A	8.00	
A	Project max.	17	27	0.20	1.00	1.59	0.78
B	Paul Revere	17	16	0.12	0.59	0.94	0.80
C	Single loop	17	17	0.13	0.63	1.00	0.80
D	Major nodes	17	9	0.07	0.33	0.53	0.82
E	Alt. 1	17	16	0.12	0.59	0.94	0.68
F	Alt. 2	17	19	0.14	0.70	1.12	0.75
G	Alt. 3	17	25	0.18	0.93	1.47	0.75

N/A, not applicable.

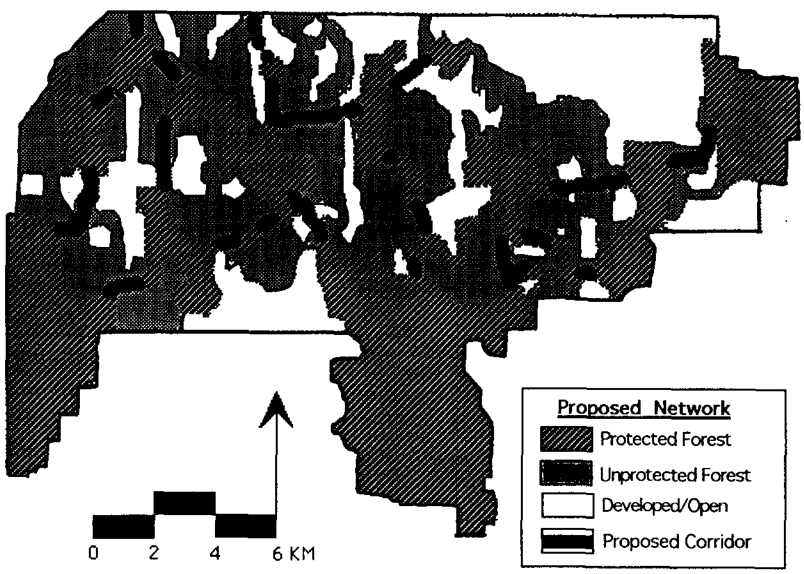

Fig. 8. Map of final network as applied to the study region.

in Eqs. (2)–(5) were calculated for all of the alternative networks discussed below, and the calculated values are shown in Table 4.

Network alternatives were generated based on the typologies previously discussed. Some of the networks that were generated are shown in Fig. 7. As previously stated, Network A in Fig. 7 represents the maximum network that is possible based on the constraints discussed above. Networks that represent the nearest approximations of the Paul Revere and the Traveling Salesman are shown in Fig. 7 (Networks B and C, respectively). It was found that (because of the degree of blockage and redundancy, as well as the levels of interaction between the node pairs shown in Table 3) none of the typologies previously discussed could be effectively implemented in their pure form, but rather served as rough guides. Networks D–G in Fig. 7 represent a continuum of increasingly complex networks, with each being a subset of the next, where D links only the major nodes, E links all 17 nodes in an MST, F forms three small circuits that correspond to the region's sub-basins, and G represents a complex multi-circuit network that approaches the least cost to user scenario.

Network D in Fig. 7 represents a network that concentrates on linking just those nodes that are greater than 650 ha, the minimum range size for fishers (Kelly, 1977), and where some smaller nodes are passed through in the process. Network E in Fig. 7 represents an MST that connects all 17 nodes efficiently and where the north–south links correspond to the region's physiography. Network F in Fig. 7 represents a network consisting of three simple loops. Because of the configuration of the open space, the topography, and interdigitation of the human matrix, this solution was altered to consist of three loops where Nodes 2 and 3 serve as pivot points. Network G in Fig. 7 represents an even more complex network, which provides additional loops for the purposes of redundancy and increased access. This alternative (also shown as a map in Fig. 8) represents a multiple loop alternative with higher levels of connectivity than the previous alternative, yet is slightly more realistic than the least cost to user alternative (Network A, Fig. 7). This alternative includes links to areas outside the region. These links are important, as linkage through these areas is already difficult owing to highway development that is quickly fragmenting the study areas to the north.

5. Discussion

The connectivity indices were found to be useful measures for describing the degree of connectivity. The adjusted Gamma index is very useful in that it can calculate the extent to which various degrees of greenway development correspond to either the maximum greenway possible or to an overall greenway master plan. In other words, if the maximum network based on the constraints (Network A, Fig. 7) was the desired plan, the adjusted Gamma indices represents the degree to which the plan has been achieved in terms of the number of links implemented. If Network G were the desired end, the adjusted Gamma shown in Eq. (3) could be altered so that the number of links in this alternative could be used in calculating the adjusted Gamma.

The gravity model was useful in selecting between alternatives. For example, one of the obvious differences between Networks B and C and Networks D–G is that Node 10 rather than Node 3 was used as the major path. Although this looks strange on the graphs, this is a function of the location of the centroid and the amount of node interaction. The actual distances between Node 2 and Node 10 are closer than they appear on the graph. The routing through Node 3 was done to maximize the amount of interaction within the networks because (as shown in Table 3) the amount of interaction between Nodes 2 and 3 (which has a value of 12.18) is significantly greater than the interaction between Nodes 2 and 10 (which has a value of 2.32).

The Adjusted Gamma and Cost Ratio were also found to be important indices in network evaluation, especially in adjudicating between networks with similar distances. An example of this can be seen by comparing Networks C and F. It would intuitively appear from the graphs in Fig. 6 that Network C is significantly more efficient from a least cost to implement perspective. Indeed, if one examined the Gamma and Beta indices, Network C would appear more efficient (cost effective), as it contains 17 links, and links all nodes with the simplest single loop solution, with an Adjusted Gamma value of 0.63 and a Beta of 1.0 (a single loop), whereas Network F contains 19 links and has an Adjusted Gamma value of 0.70, and a Beta of 1.12. However, the difference in cost effectivity as measured by the Cost Ratio is $0.80 - 0.78$, or a mere 2%, with a difference in total network difference of $86.25 - 85.10$ km, or 1.15 km. The planner could then substantively argue that the advantage of three loops in Network F that correspond to the region's sub-basins is as effective as the single loop in Network C, and that the social and ecological benefits of the former make F a better decision than C.

This type of analysis could then be continued so that the effects of the various links can be systematically tested in terms of link efficiency as measured by the amount of connectivity achieved per unit distance. It should also be noted that the Cost Ratio will favor simple and efficient networks, which may be considered less desirable for wildlife corridor networks because simple networks are more sensitive to future disturbances than are more complex networks with higher connectivities. Conversely, a comparison of the summation of interactions within a given network will favor complex networks. Both sets of comparisons are therefore useful when comparing networks with similar Gamma values, i.e. networks with similar levels of connectivity. The point of these examples is to demonstrate that (1) intuition alone may not yield the most efficient results, (2) the information gained from the indices and gravity model when used together can help make more informed decisions, and (3) a single index used alone can be misleading, and should be used in conjunction with the other measures as well as with reference back to the landscape conditions.

6. Conclusion

Network analysis is an appropriate approach to greenway planning, as it provides a method of systematizing the relationship between elements that can serve as greenway nodes as well as accounting for the conditions of the potential links. This approach also fits well within landscape ecology and landscape planning approaches, as the determination of node and link qualities can be directly obtained from standard overlay methods. Although the methods used were applied to the objective of maintaining suitable interior forest habitat for the fisher, the alternatives could well have been generated for and evaluated against other recreational, esthetic, economic, and cultural criteria. Regardless of the specific structure and function, any network can be defined in terms of nodes and links, and this allows for the reduction of otherwise complex flow patterns so that connectivity and efficiency can be readily assessed.

Randomly connecting open spaces is inefficient at best, and, given two or more complex alternatives, it is difficult simply to guess which networks are more efficient, which links are most significant, which networks have higher levels of connectivity, and which networks would be the most cost effective. The value of these techniques lies in the ability to support the qualitative ideas of greenways with an approach that allows for systematic assessment. The use of landscape ecology as a theoretical and scientific basis, and graph theory as part of the methodological approach, could serve to help systematize greenway planning and, in turn, help give it additional credence as an important land use strategy.

The alternative greenway networks presented here have been delineated with the single purpose of protecting adequate interior habitat to insure the viability of wide-ranging fragmentation-sensitive species. It is argued that regional biodiversity should be treated as a central function of greenways. Clearly, this should not be the only basis of greenway delineation; a comprehensive greenway should also evaluate for additional economic and cultural criteria. The point is that regional biodiversity protection can and should serve as the backbone of a more balanced greenway plan, as the resource needs of wildlife are generally more extensive and less flexible than the incorporation of cultural and recreational resources. As is evident in that the Gamma index shown in Table 4, the maximum network in the region is only 20% of the maximum amount of connectivity.

There is as of yet little consensus on whether a greenway designed for regional biodiversity would work as designed. Wildlife corridor networks are still largely untested, and it can be debated whether or not such an approach would work as planned. What is being hypothesized is that a well-planned greenway may be able to approximate the natural landscape pattern and thereby prevent the loss of species that would otherwise be expected from fragmentation. It cannot be overemphasized that greenways should not be seen as the end-all solution to wildlife conservation problems, but can be a cost-effective complement to an existing open space reserve system. It is our argument that current land use zoning schemes fail to account for biological diversity and may actually encourage fragmentation. To wait and debate the issue may result in a lost opportunity. As stated previously, the best argument for corridors is that the landscape was highly connected before settlement, and that fragmentation has been largely an anthropogenic impact that has reduced connectivity (Noss, 1987b).

The design of greenways that contribute to the protection of wildlife while providing the recreational, esthetic, and other human benefits is not a trivial task. First, a greenway designed as a conduit for one species will most likely serve as a habitat for others and as a barrier for still others. Second, the potential of viewing wildlife is a significant attraction for outdoor recreation (Mendelsohn, 1987). Landscape preference studies have demonstrated that wildlife has a positive, statistically significant, impact on scenic quality; the sheer expectation of seeing wildlife significantly increases landscape quality assessments (Hull and McCarthy, 1988). This often leads to a conflict between human use and wildlife, and this problem seems to be growing. If the two are to coexist, a better understanding of the conflicts and compatabilities between the two must be achieved (Strutin, 1991). It also leads to the conclusion that it would be erroneous to assume that any particular greenway can meet all of the human and wildlife needs of a given area.

As previously stated, fragmentation is the most serious threat to biodiversity. Given that the primary function of greenways is to provide linkages, they represent one of the most effective tools in preventing fragmentation and perhaps species loss at the regional level. Planning efforts to date have not been responsive to the needs of wildlife and have continually been obscured by economic objectives. From this perspective, greenways can be seen not only as an approach to linking open spaces, but, more importantly, as a tool to realize the relationship between ecological structure and function in an economically viable and socially desirable way.

Acknowledgments

Special thanks are due to Bruce MacDougall, Jack Ahern, and Todd Fuller of the committee for the original thesis, which served as the basis of this paper, and to the Massachusetts Agricultural Experiment Station, which provided funding for this research.

References

Ahern, J., 1989. Planning and design for sustainability in a changing New England landscape. Proc. Landscape/Land Use Planning Committee of the American Society of Landscape Architecture's 1989 Annual Meeting, Orlando, FL. American Society of Landscape Architecture, Washington, DC, pp. 1–12.

Ahern, J., 1991a. Planning and design for an extensive open space system: linking landscape structure to function. Landscape Urban Plann., 21: 131–145.

Ahern, J., 1991b. Greenways and ecology. Proc. Landscape/Land Use Planning Committee of the American Society of Landscape Architecture's 1991 Annual Meeting, Washington, DC. American Society of Landscape Architecture, Washington, DC, pp. 75–87.

Allen, A.W., 1983. Habitat suitability index model: fisher. US Department of the Interior, Fish and Wildlife Service, FWS/OBS-82/10.45, 19 pp.

Arthur, S., Gilbert, J. and Krohn, W., 1989. Home range characteristics of adult fishers. J. Wildl. Manage., 53(3): 674–679.

Brown, L., Flavin, C. and Postel, S., 1991. Vision of a sustainable world. In: L. Brown (Editor), The Worldwatch Reader on Global Environmental Issues. Norton, New York, pp. 299–316.

Burgess, R.L. and Sharpe, D.M., 1981. Forest Island Dynamics in Man Dominated Landscapes. Springer, New York.

Burley, J.B., 1989. Landscape for wildlife. Landscape Res., 14: 23–36.

Carlile, D.W., Skalski, J.R., Batker, J.E., Thomas, J.M. and Cullinan, V.I., 1989. Determination of ecological scale. Landscape Ecol., 2: 203–213.

Dalton, R., Garlick, J., Minshull, R. and Robinson, A., 1973. Networks in Geography. Phillip, London.

DeGraaf, R. and Rudis, D., 1983. New England Wildlife: Habitat, Natural History, and Distribution. US Department of Agriculture, National Forest Service, Washington, DC, TRNE-108, 491 pp.

diCastri, F. and Hadley, M., 1988. Enhancing the credibility of ecology: interacting along and across hierarchical scales. GeoJournal, 17: 5–35.

Fabos, J.G., 1991. From parks to greenways in the 21st century. Proc. Landscape/Land Use Planning Committee of the American Society of Landscape Architecture's 1991 Annual Meeting, Washington, DC. American Society of Landscape Architecture, Washington, DC, pp. 1–13.

Forman, R.T.T. and Godron, M., 1986. Landscape Ecology. Wiley, New York, 619 pp.

Gilpin, M.E., 1988. A comment on Quinn and Hastings: extinction in subdivided habitats. Conserv. Biol., 2: 290–296.

Grumbine, R.E., 1990. Viable populations, reserve size, and federal lands management: a critique. Conserv. Biol., 4: 127–134.

Haggett, P. and Chorley, R.J., 1972. Network Analysis in Geography. Edward Arnold, London, 348 pp.

Haggett, P., Cliff, A. and Firey, A., 1977. Locational Analysis in Human Geography. Wiley, New York, p. 32.

Harris, L., 1984. The Fragmented Forest. University of Chicago Press, Chicago, IL.

Heinselman, M.L., 1981. Fire and succession in conifer forests of Northern North America. In: H.H. Shugart and B. Botkins (Editors), Forest Succession: Concepts and Application. Springer, New York, pp. 374–405.

Hellmund, P., 1989. Quabbin to Wachusett Wildlife Corridor Study. Harvard Graduate School of Design, Cambridge, MA.

Hiss, T., 1991. The Experience of Place. Vintage Books, New York, 233 pp.

Hull, R.B. and McCarthy, M.M., 1988. Change in the landscape. Landscape Urban Plann., 15: 265–278.

Kelly, G.M., 1977. Fisher biology in the White Mountain National Forest and adjacent areas. Ph.D. Thesis, University of Massachusetts, Amherst.

Linehan, J.R., 1992. Wildlife corridor delineation for fisher and otter in Central Massachusetts: developing a network approach. Masters Thesis, Department of Landscape Architecture and Regional Planning, University of Massachusetts, Amherst, 141 pp.

Little, C., 1990. Greenways for America. Johns Hopkins University Press, Baltimore, MD, 237 pp.

Lowe, J.C. and Morydas, S., 1975. The Geography of Movement. Houghton–Mifflin, Boston, MA, 333 pp.

MacArthur, R.H. and Wilson, E.O., 1967. The Theory of Island Biogeography. Princeton University Press, Princeton, NJ, 203 pp.

Massachusetts Department of Fisheries, Wildlife and Law Enforcement, 1990. An Atlas of Massachusetts River Systems: Environmental Designs for the Future, University of Massachusettes Press, Amherst, MA.

Mendelsohn, R., 1987. Modeling the demand for outdoor recreation. Water Resour. Res., 23: 961–967.

Noss, R.F., 1987a. Corridors in real landscapes: a reply to Simberloff and Cox. Conserv. Biol., 1: 159–164.

Noss, R.F., 1987b. Protecting natural areas in fragmented landscapes. Nat. Areas J., 7: 2–13.

Opdam, P., 1991. Metapopulation theory and habitat fragmentation. Landscape Ecol., 5(2): 93–106.

Organ, J.F., 1989. Mercury and PCB residues in Massachusetts river otters: comparisons on a watershed basis. University of Massachusetts, Amherst.

Powell, R.A., 1982. The Fisher: Life History, Ecology, and Behavior. University of Minnesota Press, Minneapolis, 217 pp.

Quinn, J.F. and Hastings, A., 1987. Extinction in subdivided habitats. Conserv. Biol., 1: 198–208.

Schwabe, A., 1989. Vegetation complexes of flowing water habitats and their importance for the differentiation of landscape units. Landscape Ecol., 2: 237–253.

Scott, J.M., Davis, F., Csuti, B., Noss, R., Butterfield, G., Groves, C., Anderson, H., Caicco, S., D'Erchis, F., Edwards, T., Ullman, J. and Wright, R.G., 1993. Gap analysis: a geographic approach to protection of biological diversity. Wildl. Monogr., 123, 41 pp.

Simberloff, D. and Cox, J., 1987. Consequences and costs of conservation corridors. Conserv. Biol., 1: 63–71.

Sklar, F. and Constanza, R., 1991. The development of dynamic spatial models for landscape ecology: a review and prognosis. In: M. Turner and R. Gardner (Editors), Quantitative Methods in Landscape Ecology. Springer, New York, pp. 239–287.

Soule, M.E., 1986. What do genetics tell us about the design of nature reserves? Biol. Conserv., 35: 19–40.

Starfield, A.M. and Bleloch, A.L., 1986. Building Models for Conservation and Wildlife Management. Burgess, Edina, MN, 253 pp.

Stolenburg, W., 1991. Wildlife corridors: the fragment connection. Nat. Conserv. Mag., July–Aug.: 19–25.

Strutin, M. 1991. Somebody's got to try this stuff: can sustainable design save the Grand Canyon? Landscape Archit. Mag., 81(3): 50–59.

Turner, M. and Gardner, R., 1991. Quantitative Methods in Landscape Ecology. Springer, New York, pp. 3–14.

Walker, B.H., 1992. Biodiversity and ecological redundancy. Conserv. Biol., 6: 18–34.

Weller, M.W., 1985. The influence of hydrologic maxima and minima on wildlife habitat and production values of wetlands. Texas A&M University, College Station.

Wilcove, D.S., McLellan, C.H. and Dobson, P., 1986. Habitat fragmentation in the temperate zone. In: M.E. Soule (Editor), Conservation Biology: the Science of Scarcity and Diversity. Sinauer, Sunderland, MA, pp. 237–256.

Wilcox, B.A. and Murphy, D.D., 1985. Conservation strategy: the effects of fragmentation on extinction. Am. Nat., 125: 879–887.

World Resource Institute, The World Conservation Union (IUCN), United Nations Environmental Programme, 1992. Global Biodiversity Strategy, UNEP, NY.

Yaro, R.D., Arendt, R.G., Dodson, H.L. and Brabec, E.A., 1989. Dealing with Change in the Connecticut River Valley: a Design Manual for Conservation and Development. Lincoln Institute of Land Policy, Cambridge, MA, pp. 8–10.