

Spatially Heterogeneous Multi-Scale Response of Landscapes to Fire Suppression

Author(s): William L. Baker

Source: *Oikos*, Jan., 1993, Vol. 66, No. 1 (Jan., 1993), pp. 66-71

Published by: Wiley on behalf of Nordic Society Oikos

Stable URL: <https://www.jstor.org/stable/3545196>

REFERENCES

Linked references are available on JSTOR for this article:

https://www.jstor.org/stable/3545196?seq=1&cid=pdf-reference#references_tab_contents

You may need to log in to JSTOR to access the linked references.

JSTOR is a not-for-profit service that helps scholars, researchers, and students discover, use, and build upon a wide range of content in a trusted digital archive. We use information technology and tools to increase productivity and facilitate new forms of scholarship. For more information about JSTOR, please contact support@jstor.org.

Your use of the JSTOR archive indicates your acceptance of the Terms & Conditions of Use, available at <https://about.jstor.org/terms>



Wiley and JSTOR are collaborating with JSTOR to digitize, preserve and extend access to *Oikos*

JSTOR

Spatially heterogeneous multi-scale response of landscapes to fire suppression

William L. Baker

Baker, W. L. 1993. Spatially heterogeneous multi-scale response of landscapes to fire suppression. – *Oikos* 66: 66–71.

Large natural disturbances produce a patchy structure in landscapes. External changes, such as climatic change or the intentional suppression of disturbances, may alter this structure. Species sensitive to this structure may thus experience changes in available habitat, but species use the landscape at different spatial scales. I used a GIS-based spatial model to simulate the long-term effects of fire suppression on landscape structure in the Boundary Waters Canoe Area, Minnesota, USA. I analyzed changes in seven measures of landscape structure at three spatial scales during the simulation runs. The timing of a suppression effect varied with scale. At the coarsest scale there was an immediate effect in the case of some measures, but a delay for more than 250 years in the case of other measures. At the finest scale effects were spatially heterogeneous, with some parts of the landscape responding immediately and other parts not responding for hundreds of years. There are several implications. First, attributing vegetation changes to fire suppression requires identification of fire suppression actions in the area of vegetation change. Second, spatially explicit prescribed burning programs are needed if landscapes altered by fire suppression are to be restored. Third, species that use the landscape at different scales will observe different patterns of change in landscapes subject to fire suppression. Finally, landscapes can be expected to respond in a spatially heterogeneous manner to other kinds of changes (e.g., climatic change). Parks and wilderness areas with unaltered disturbance regimes are essential if we are to understand how and why landscapes change and the role that this change plays in the population dynamics of species.

W. L. Baker, Dept of Geography, Univ. of Wyoming, Laramie, WY 82071, USA.

Landscapes have a dynamic structure that is the result of the spatial arrangement and other attributes of landscape elements (e.g., patches, corridors, and matrix). Since the German geographer Carl Troll first articulated the importance of landscape structure to humans and other species that live in landscapes (Troll 1950), there has been an acceleration of interest in landscape ecology (e.g., Forman and Godron 1986, Turner 1987, Moss 1988, Schreiber 1988, Zonneveld and Forman 1990). In developing the field, landscape ecologists have borrowed some quantitative techniques from the spatial analytical tradition in geography (e.g. Haggett et al. 1977), but have also adapted newer techniques such as fractal analysis (e.g. Krummel et al. 1987). Although there have been substantial advances in understanding and quantifying how landscapes are structured (Turner

and Gardner 1991), the dynamics of landscape change remain poorly understood.

Organisms can be considered to interact with or “observe” the landscape at different scales, and this scale determines what patterns of landscape change are apparent. An organism can be considered to have different “ecological neighborhoods,” or scales of use, for different ecological processes (e.g., foraging, reproduction), and thus interacts with the landscape at different scales (Addicott et al. 1987). Variation in the scale of use among species is part of the reason for an emerging systematic analysis of scale in landscapes (Meentemeyer 1989).

Landscapes themselves have scaled phenomena with many sources, but in landscapes subject to large natural disturbances (e.g., fires, floods) the size distribution of

Accepted 21 April 1992

© OIKOS

patches produced by disturbance imposes a fluctuating scaling on landscape structure (Baker 1989a). When these disturbance landscapes are observed at multiple spatial scales, scale-dependent patterns of landscape change are apparent. In general, as the scale of observation becomes finer there appears to be greater spatial heterogeneity in the temporal pattern of change (Baker 1989a). It is unclear how this spatiotemporal pattern in disturbance landscapes will be affected by external changes that alter the pattern of disturbance, such as climatic change or the intentional suppression of disturbances.

The intentional suppression of fires in the landscapes of the western United States, beginning in the early 1900's, is an external change that has altered the disturbance process. Fire suppression may have allowed unnatural fuel buildups that have promoted larger fires (Minnich 1983), and may be responsible for tree invasions into grasslands as well as a variety of other post-1900 changes in vegetation (e.g., Arno and Gruell 1986). Although fire and fire suppression are both spatial processes in landscapes, the spatial aspects of disturbances and disturbance suppression have received little attention.

Experiments to analyze fires and fire suppression effects in landscapes at multiple scales are not generally feasible because of the large land area and long time scale needed for such experiments, but new tools such as geographical information systems (GIS) and spatial models of landscapes have made simulation experiments feasible (Baker 1989b). GIS has been used in a variety of applications, but these generally have been limited to static analyses of geographic phenomena (e.g., Ripple 1989). The potential utility of GIS in modelling dynamic phenomena is only beginning to be explored (Baker 1989b).

The purpose of the research reported here was to use a dynamic GIS-based simulation model to quantitatively analyze the long-term response of a landscape to fire suppression at several spatial scales. The working hypothesis was that landscapes subjected to fire suppression would respond in a spatially heterogeneous manner, with the heterogeneity increasing as the scale of observation became finer.

Methods

The model

The GIS model used here is modified from the model DISPATCH (Baker et al. 1991), and consists of four major components. The first component of the model is a disturbance generator written in SIMSCRIPT II.5 (CACI 1987). This disturbance generator consists of two sets of probability density functions that are used to generate a sequence of disturbances varying in timing and size. These distributions are derived from empirical

reconstructions of actual fires in the Boundary Waters Canoe Area, Minnesota, USA (BWCA) during the last few hundred years (Heinselman 1973, Baker 1989a).

The second component consists of spatial data on patch ages stored and manipulated in a GIS called GRASS (USA-CERL 1988). The GRASS GIS is a raster format public-domain software package that is particularly valuable for modelling because its source code is provided. The simulation code can also make direct use of GRASS commands and programming functions without going through a menu interface.

The third component is a set of landscape structure analysis programs, called the *r.le* programs, that were written specifically for this simulation research (Cai 1991, Baker and Cai in press). The *r.le* programs allow calculation of a variety of measures of landscape structure, using sampling units at several spatial scales, as the simulation runs. The last component consists of external statistical and graphical software for analyzing the simulation results.

The model is a logical model (Loehle 1983), not a predictive tool. The goal of the model is to enable the testing of hypotheses about how the overall structure of a mosaic of fire patches changes, not to precisely predict when and where a fire will occur in the BWCA. Verification with field data is thus not appropriate, but the model has been validated using a variety of standard methods (Sargent 1987).

The modelling experiment

The BWCA area is simulated by a GIS landscape consisting of a 150×360 cell (54 000 total grid cells or "pixels") rectangular area. Use of this simulated area, rather than the area inside the irregular BWCA boundary, should not change the general results as the model parameters have been rescaled to fit this somewhat larger area.

In this simulation I assume that the probability of disturbance (fire) initiation and spread is a linear function of the age (time since last disturbance) of each fire patch in the landscape. This disturbance probability is adjusted to a 0–1 scale by dividing by the maximum age that can be attained during the simulation. Each potential disturbance is initiated in the landscape and allowed to spread probabilistically based upon this map of adjusted disturbance probabilities. The map of patch ages is updated on an annual basis to account for changes in age produced by disturbance and by the passage of time.

The experiment consists of 10 replicates of each of two treatments, representing two different sequences of fire regimes. There were three periods with relatively constant fire regime in the BWCA (Heinselman 1973): (1) the presettlement period (until 1867 A.D.), (2) the settlement period (1868–1910 A.D.), and (3) the suppression period (from 1911 A.D.). The HISTORICAL

Table 1. Measures of landscape structure calculated as the model runs.

Weighted mean age = average age of all the pixels in the landscape.
Mean patch size = total number of pixels/number of patches.
Patch shape = $(0.282 \times \text{perimeter})/\text{area}^{1/2}$ (Patton 1975).
Fractal dimension – derived from the perimeter-area relation (Krummel et al. 1987). Values close to 1.0 indicate rather simple and geometric perimeters; values close to 2.0 indicate complex perimeters.
Shannon index = $\sum p_i \times \ln(p_i)$, where the sum is from $i = 1$ to the number of ages and p_i is the fraction of the landscape occupied by patch age i (Peet 1974).
Richness = total number of patch ages.
Angular second moment = an index of the fine-scale texture of the landscape (Haralick et al. 1973). Coarser textures have smaller values.

treatment recreates this historical pattern of fire regimes. The PRESETtlement treatment has a continuous presettlement fire regime, and acts as a control for assessing the effects of the HISTORICAL treatment. The experiment begins in 1368 A.D. with an age map initialized with random states. Preliminary testing indicated that the initial random states in the landscape had been completely burned by about 1668 A.D. eras-

ing the initial conditions. Both treatments run for 500 yr, until 1868 A.D., with the same presettlement fire regime. The experiment continues until 2368 A.D., 500 yr after the treatments begin to diverge.

Seven indices of landscape structure were used to determine when and how the landscape affected by the two treatments differed (Table 1). Landscape structure was measured at three spatial scales simultaneously as the model ran. Scale 1 is the whole simulated landscape, scale 2 consists of a west half and an east half, and scale 4 consists of the four quarters.

The null hypothesis of no treatment effect was not rejected if the 95% confidence interval of the difference in means (for the 10 replicates), for a particular measure (Table 1) at a particular time, overlapped zero (Bratley et al. 1987).

Results

At scale 1 (whole simulated landscape) there was a significant effect of settlement and fire suppression in some, but not all cases (Fig. 1). Settlement significantly decreased mean age and angular second moment, and significantly increased mean shape, Shannon diversity, and richness, but there was no significant effect on either mean size or fractal dimension.

At scale 1 there was an immediate response to suppression only in the case of Shannon diversity, richness, and mean shape. The effect of suppression was delayed for several decades in the case of mean age and fractal dimension, and for more than 250 yr in the case of mean size and angular second moment. The reasons for these differential responses are complex and are explained elsewhere (Baker in press).

As the same landscape is analyzed at finer scales of observation (scales 2 and 4) the same general trends are apparent, but the details of the trends differ among scales and are quite varied temporally (Fig. 1). For example, in 1968 A.D. in the case of mean size (Fig. 1b) there was a significant suppression effect only in the southeast quarter at scale 4, and this effect apparently was strong enough to produce a significant effect on the whole east half at scale 2, but the effect was apparently not strong enough to produce a significant effect on the

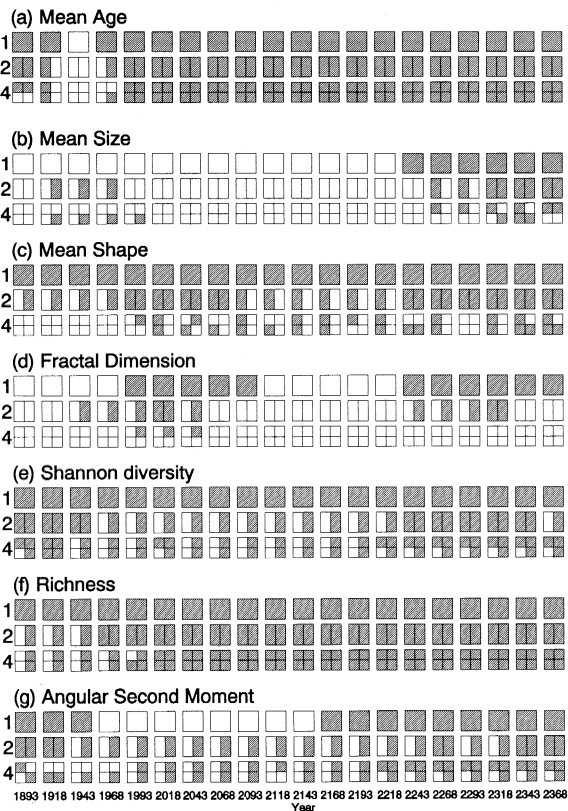


Fig. 1. Significant effects of settlement (1868–1910) and fire suppression (from 1911) over the course of the simulation at three spatial scales. Scales are: 1 = whole simulated landscape, 2 = the two halves of the simulated landscape, 4 = the four quarters of the simulated landscape. Shading indicates that the means for the two treatments were significantly different, based on the 95% confidence interval of the difference in means, for the portion of the simulated area that is shaded. The seven measures (a)–(g) are explained in Table 1.

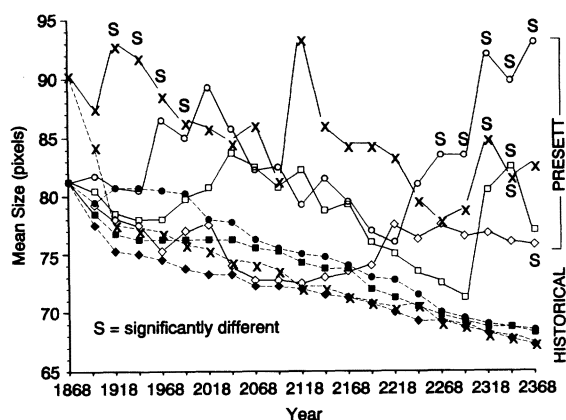


Fig. 2. Mean patch size at scale 4 during the simulation run in the four quarters of the PRESETT treatment (open symbols and solid lines) and the four quarters of the HISTORICAL treatment (closed symbols and dashed lines). An "S" indicates that the mean size differed significantly (based on the 95% confidence interval of the difference) in that year in the corresponding quarters of the two treatments.

whole landscape (scale 1). As another example, the settlement effect on mean age (Fig. 1a) was significant only on the north half at scale 4 in 1893 A.D., but the location of a significant effect shifted to the west half in 1918 A.D.

Changes in mean size in the four quarters at scale 4 during the simulation run illustrate several aspects of the complexity of response (Fig. 2). First, the four quarters did not begin in 1868 A.D. with the same mean size. There were large fires just before this time, that were concentrated in the southeast quarter ("x" in Fig. 2), and that produced a larger mean size in that quarter in 1868 A.D. It was only in this quarter that there was a significant effect of settlement, apparently because the settlement regime was able to produce a rapid decline in mean size without a rapid decline in mean size in the corresponding presettlement landscape. Thus the historical effect of fires before 1868 A.D. controlled the location of a significant settlement effect in the early 1900's. Second, there was comparatively little variation in the trajectory of change among the four quarters subjected to suppression, but substantial variation in trajectory among the quarters with the ongoing presettlement regime (Fig. 2). This is because the suppression regime was relatively inactive, producing comparatively few patches and small patches in a landscape containing many patches. The presettlement regime, in contrast, was more active, producing larger fires that could effect more substantial change. Third, the variation in the presettlement trajectories, rather than the settlement/suppression trajectories, appears to have controlled the moment when a significant difference between treatments occurred, although the likelihood of a significant difference increased in any event as mean size declined in the suppression landscapes. For example, in the

southeast quarter ("x" in Fig. 2) the difference became significant in 2318 A.D., not before, because it was then that there was a sudden rise in mean size due to a large disturbance in the landscape with the presettlement regime. The implication of this is that as the scale of observation becomes finer, the time of a treatment effect becomes more and more a matter of chance, because on smaller land areas individual disturbances affect a larger proportion of the observation unit and thus have more substantial effect. At the finest scale of observation the location of a treatment effect in essence tracks the occurrence of individual disturbances.

One consequence of this is that as the scale of observation becomes finer, there are more substantial differences (lags) in the timing of a suppression effect when parts of the same landscape are compared. At scale 2 a suppression effect on mean size is lagged 50 yr between the two halves, but at scale 4 the effect is lagged as much as 100 yr between the quarters (Fig. 1). In the case of Shannon diversity the lag among halves at scale 2 is 300 yr, while among quarters at scale 4 it may be more than 450 yr. The reason for the spatial heterogeneity in the timing of a treatment effect is that fires of the same size do not occur everywhere at the same time. Fires instead occur in a spatially and temporally heterogeneous manner producing a spatially heterogeneous effect.

In a single landscape, as opposed to two landscapes subjected to different treatments, the location of a suppression effect tracks the location of fires that are sup-

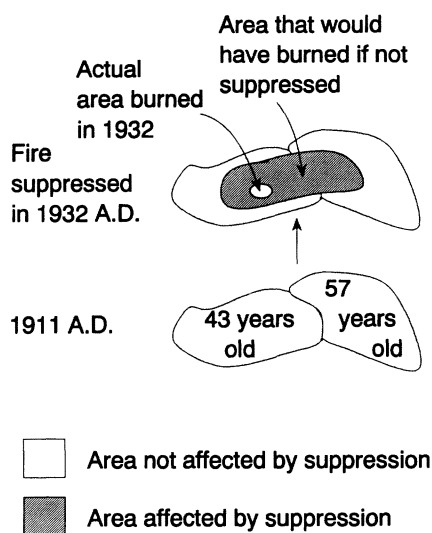


Fig. 3. Areas affected and not affected by suppression in a hypothetical landscape. In this landscape fire suppression began in 1911 A.D. when the landscape contained two patches that were burned 43 and 57 years earlier. The first fire that was suppressed occurred in 1932 A.D. There was, as a result, no effect from fire suppression in any part of the landscape until 1932, and in 1932 the only part of the landscape that was affected by suppression was the area that would have burned had the fire not been suppressed.

pressed (Fig. 3). Ignitions cannot generally be prevented; usually only the spread of a fire can be prevented. Thus, an ignition must occur before there can be a suppression effect at a particular location in the landscape. There is no suppression effect in a particular part of the landscape until an ignition occurs that is followed by a suppression action. Moreover, the only part of the landscape that is directly affected by suppression is the area that would have burned if the fire had not been suppressed. The remainder of the landscape and the actual area burned are not directly affected by suppression. The timing and spatial pattern of ignitions thus is the template that limits the timing and spatial location of a suppression effect. The landscape, at any instant after the initiation of a general policy of fire suppression, must then consist of a mosaic of patches that are not affected by suppression, and that have been affected by suppression for differing periods of time.

Discussion and conclusions

The effects of fire suppression and settlement observed in this study have some implications for how disturbance landscapes change in general. First, some characteristics of landscapes may respond immediately (e.g., Shannon diversity in this case), while other characteristics exhibit inertia (e.g., mean size in this case). Second, any source of alteration in the disturbance regime (e.g., climatic change) can be expected to produce a spatially heterogeneous response, with some parts of the landscape responding immediately and other parts responding only after a time lag. Third, the magnitude of the lags, when different parts of the landscape are compared, becomes greater as the scale of observation becomes finer. Fourth, the condition of the landscape at the time the disturbance regime is altered can affect the location and timing of a response. There is a historical legacy that affects the response. Finally, even when a disturbance alteration is applied uniformly (e.g., all fires in the BWCA suppressed) there will be a spatially heterogeneous effect. These five generalities undoubtedly can take quite different forms in different places, depending upon the characteristics of the disturbance regime (e.g., disturbance size, timing, and spatial distribution), the manner in which this disturbance regime is altered, and the state of the landscape itself (Turner 1987).

One ramification of increasing lags as scale becomes finer is that species may observe a different pattern of change, depending upon the size (scale) of their ecological neighborhoods (Addicott et al. 1987). Species whose neighborhood size is small and who are adapted to current landscape structure would experience large spatial variation in the pattern of change in landscapes undergoing an alteration in the disturbance regime (Fig. 2). However, because these species operate at the finest

scale where lags are greatest, they should be able to locate unaltered habitat somewhere in the landscape long after the disturbance regime begins to change. For these species there may be habitat "refugia" in the parts of the landscape exhibiting a lagged response to change. Species with larger neighborhood sizes have less chance to locate similar refugia, but the landscape may change less rapidly on coarser scales. The implication of this is that species that are good dispersers and that have small neighborhoods may be able to persist longer in landscapes experiencing change, than can species with larger neighborhoods and poorer dispersal capabilities.

Although there is little doubt that fire suppression does affect landscapes, the spatially heterogeneous character of the response to fire suppression means that it may be difficult to definitively attribute certain vegetation changes to fire suppression. The surest evidence that fire suppression is a cause of an observed change in a part of a landscape is to provide evidence that there was an ignition that was suppressed in that part of the landscape (Fig. 3). Evidence that suppression was initiated in the landscape as a whole at a particular time (e.g., Bonnicksen and Stone 1982) is insufficient evidence that fire suppression is necessarily the cause of observed changes in a particular part of a landscape. Separation of possible causes of vegetation change in landscapes is difficult because several causes may have acted nearly synchronously (e.g. livestock grazing and fire suppression) (Taylor 1990). Historical sources have proven valuable in separating competing causes, because it has been possible to reconstruct when land use changes occurred in relation to when vegetation change occurred (Vale 1982). It should be possible to more definitely separate competing causes of change if accurate records of the location of fire suppression actions can be obtained. In lieu of these records it may be possible to locate post-suppression fires, and thus the vicinity of suppression effects, in the field using tree-ring dating of fire scars and stand origins (Arno and Sneek 1977). Reconstructing which part of the landscape is in the suppression-affected area is more difficult, as it requires some estimate of the area that would have burned (Fig. 3). It might be possible to use fire-spread models such as BEHAVE (Burgan and Rothermel 1984) to estimate the area that would have burned without suppression, and thus estimate the area affected by suppression.

Many landscapes in parks and wilderness areas have experienced fire suppression, and prescribed burning has been widely used to remedy the effects of fire suppression (e.g., Parsons et al. 1986). My results suggest that is not possible to use prescribed burning to precisely correct the effects of suppression unless it is possible to determine where in the landscape the suppression effect is located (Fig. 3). On the finest scale prescribed fires that are incorrectly placed will simply alter landscape structure in a new way, rather than restore the landscape. Parts of the landscape that have un-

natural fuel buildups due to fire suppression will remain locations where unusual catastrophic fires can originate (Bonnicksen and Stone 1982) unless these locations can be specifically located and restored. A spatially sensitive prescribed burning program is thus a necessity if prescribed burning is to be used effectively in landscape restoration. It may be premature to try to implement such a spatially sensitive restoration program, or even any restoration program, at the present time given the limits of current burning technology and the need for further research on the response of other landscapes to fire suppression. Clearly, parks and wilderness areas that have relatively unaltered disturbance regimes are of inestimable value in understanding and quantifying the response of landscapes, and species living in landscapes, to past and impending global change.

Acknowledgements – This research was completed with funds from the Theoretical Ecology Program of the Ecological Research Division, Office of Health and Environmental Research, U.S. Dept of Energy (Grant No. DE-FG02-90ER60977). This support does not constitute an endorsement by DOE of the views expressed in this article. I appreciate the assistance of Y. Cai.

References

- Addicott, J. F., Aho, J. M., Antolin, M. F., Padilla, D. K., Richardson, J. S. and Soluk, D. A. 1987. Ecological neighborhoods: scaling environmental patterns. – *Oikos* 49: 340–346.
- Arno, S. F. and Sneek, K. M. 1977. A method for determining fire history in coniferous forests of the mountain west. – USDA Forest Service General Technical Report INT-42, Ogden, UT.
- and Gruell, G. E. 1986. Douglas-fir encroachment into mountain grasslands in southwestern Montana. – *J. Range Manage.* 39: 272–276.
- Baker, W. L. 1989a. Landscape ecology and nature reserve design in the Boundary Waters Canoe Area, Minnesota. – *Ecology* 70: 23–35.
- 1989b. A review of models of landscape change. – *Landscape Ecol.* 2: 111–133.
- in press. Effects of settlement and fire suppression on landscape structure. – *Ecology*, in press.
- and Cai, Y., in press. The r.le programs for multiscale analysis of landscape structure using the GRASS geographical information system. – *Landscape Ecol.*, in press.
- , Egbert, S. L. and Frazier, G. F. 1991. A spatial model for studying the effects of climatic change on the structure of landscapes subject to large disturbances. – *Ecol. Modell.* 56: 109–125.
- Bonnicksen, T. M. and Stone, E. C. 1982. Managing vegetation within U.S. National Parks: a policy analysis. – *Environ. Manage.* 6: 109–122.
- Bratley, P., Fox, B. L. and Schrage, L. E. 1987. A guide to simulation. – Springer, New York.
- Burgan, R. E. and Rothermel, R. C. 1984. BEHAVE: fire behavior prediction and fuel modeling system-BURN subsystem, Part 1. – USDA Forest Service General Technical Report INT-194, Ogden, UT.
- CACI, Inc. 1987. SIMSCRIPT II.5 programming language. – CACI Products Co, La Jolla, CA.
- Cai, Y. 1991. GLE-landscape ecological analysis software using the GIS GRASS. – Master's thesis, Univ. of Wyoming, Laramie, WY.
- Forman, R. T. T. and Godron, M. 1986. Landscape ecology. – Wiley, New York.
- Haggett, P., Chorley, R. J. and Frey, A. 1977. Locational analysis in human geography. 2nd ed. – Arnold, London.
- Haralick, R. M., Shanmugam, K. and Dinstein, I. 1973. Textural features for image classification. – *IEEE Transactions on Systems, Man, and Cybernetics* 3: 610–621.
- Heinselman, M. L. 1973. Fire in the virgin forests of the Boundary Waters Canoe Area, Minnesota. – *Quat. Res.* 3: 329–382.
- Krummel, J. R., Gardner, R. H., Sugihara, G., O'Neill, R. V. and Coleman, P. R. 1987. Landscape patterns in a disturbed environment. – *Oikos* 48: 321–324.
- Loehle, C. 1983. Evaluation of theories and calculation tools in ecology. – *Ecol. Modell.* 19: 239–247.
- Meentemeyer, V. 1989. Geographical perspectives of space, time, and scale. – *Landscape Ecol.* 3: 163–173.
- Minnich, R. A. 1983. Fire mosaics in southern California and northern Baja California. – *Science* 219: 1287–1294.
- Moss, M. R. (ed.) 1988. Landscape ecology and management. – Polyscience Publ., Montreal.
- Parsons, D. J., Graber, D. M., Agee, J. K. and Van Wagtendonk, J. W. 1986. Natural fire management in national parks. – *Environ. Manage.* 10: 21–24.
- Patton, D. R. 1975. A diversity index for quantifying habitat edge. – *Wildl. Soc. Bull.* 3: 171–173.
- Peet, R. K. 1974. The measurement of species diversity. – *Annu. Rev. Ecol. Syst.* 5: 285–307.
- Ripple, W. J. (ed.) 1989. Fundamentals of geographic information systems: a compendium. – American Society for Photogrammetry and Remote Sensing and American Congress on Surveying and Mapping, Bethesda, MD.
- Sargent, R. G. 1987. An overview of verification and validation of simulation models. – In: Thesen, A., Grant, H. and Kelton, W. D. (eds). Proceedings of the 1987 Winter simulation conference. Society for Computer Simulation, Washington, DC, pp. 33–39.
- Schreiber, K.-F. (ed.) 1988. Connectivity in landscape ecology. – Ferdinand Schöningh, Paderborn, Germany.
- Taylor, A. H. 1990. Tree invasion in meadows of Lassen Volcanic National Park. – *Professional Geographer* 42: 457–470.
- Troll, C. 1950. Die geographische Landschaft und ihre Erforschung. – *Studium Generale (Heidelberg)* 3: 163–181.
- Turner, M. G. (ed.) 1987. Landscape heterogeneity and disturbance. – Springer, New York.
- and Gardner, R. H. (eds) 1991. Quantitative methods in landscape ecology. – Springer, New York.
- USA-CERL. 1988. Grass 3.0 user's manual. – United States Army Corps of Engineers Construction Engineering Research Laboratory, Champaign, IL.
- Vale, T. R. 1982. Plants and people: vegetation change in North America. – Assoc. of American Geographers, Resource Publ. in Geography, Washington, DC.
- Zonneveld, I. S. and Forman, R. T. T. (eds) 1990. Changing landscapes: an ecological perspective. – Springer, New York.